Cumulative Effects Assessment and Management
for
Northeast British Columbia

Volume 2
Cumulative Effects Indicators, Thresholds, and Case Studies

Appendix 1
Cumulative Effects: Sources, Indicators, and Thresholds

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GLOSSARY

Access corridor – a linear feature created by humans (road, trail, pipeline, powerline, railway line, cutline) that may be used by pedestrians, vehicles, hunters, anglers, or animal predators.

ALCES – 'A Landscape Cumulative Effects Simulator' computer model used to forecast changes in landscape patterns and wildlife habitat. This simulator integrates four submodels (habitat availability, population, land use, and natural disturbance) and considers all land use activities likely to occur in the region. It includes both aquatic and terrestrial indicators and is able to incorporate random events such as fire.

Barrier – a barrier is present when it is not possible for animals to move across a corridor, and the habitat on either side of the corridor becomes isolated (e.g., a busy highway is a barrier for small ground-dwelling insects)

Biodiversity – the diversity of plants, animals, and other living organisms in all their forms and levels of organization (BCF and BCE 1995a). The basic goal of biodiversity conservation is to maintain naturally occurring ecosystems, communities, and native species (CEQ 1993).

Biogeoclimatic zone – a geographic area having similar patterns of energy flow, vegetation and soils as a result of a broadly homogeneous macro-climate (BCF and BCE 1995a).

Cautionary threshold – a threshold established to indicate that additional or more intensive monitoring is required to provide sufficient local data to confirm scientific predictions of both target and critical thresholds.

CEAM – Cumulative Effects Assessment and Management framework consisting of an overall strategy for identifying, scoping, assessing, and managing cumulative effects in northeast British Columbia.

Core area – an area with minimal human impacts. Core areas are relatively undisturbed, 'unroaded' areas; they are often source areas for plant and animal populations or metapopulations.

Corridor – a reasonably uniform, linear feature that differs from its surrounding landscape. Corridors can occur naturally (e.g., river valley; windrow, aeolian ridge) or as the result of human disturbance (e.g., roads, cutlines).

Critical threshold – a science-based target reflecting the continuous maximum amount of stress that a sensitive ecosystem or species can support without sustaining long-term harm. This may incorporate economic and social values to determine the acceptable magnitude of change, risk of long-term damage, or level of protection required.

Cumulative effects – changes to the environment caused by collective past, present, and future human actions; most result from the combined effects of simple, routine activities and projects.

DC – Disturbance Coefficient; an index assigned to each feature or activity type that rates the degree to which the disturbance zone of influence remains effectively useable by the species. Ratings are on a scale of 0 to 1 and are used to calculate habitat effectiveness.

Density-dependent – factors that affect population growth and parameters in relation to as animal abundance. These depress population growth as animal abundance increases, and increase growth as animal abundance decreases. Examples include food availability and quality. Predation may be density-dependent or –independent.

Density-independent – factors that affect population growth and parameters regardless of animal abundance. Examples include natural environmental disturbances such as fire, floods, or severe weather. Predation may be density-dependent or –independent.

Disturbance – a natural or human action that affects physical, chemical, or biological conditions.

Disturbance feature – a corridor or patch created by natural random events (e.g., burn or flood) or human action (e.g., cutblock, facility, community, road).

Disturbance trajectory – the calculated or predicted rate of natural or human disturbance.

Drainage - a subset of a watershed, generally less than 1,000 km^2 in size.

Early seral – forest that are younger than 40 years old (BCF and MELP 1999b).

Ecological resilience – the ability of a system or species to absorb natural and human disturbance without altering its fundamental structure (Weaver et al. 1996).

Ecological sink – an area with degraded habitat that has lower survival (or higher mortality) rates, causing local population declines. Although individual animals may continue to use this area, it creates a net loss to the population that may not be detectable for several generations.

Edge area - the area bordering patches and corridors where abiotic conditions (e.g., moisture, light, temperature, wind regimes) and biotic conditions (e.g., predation, mortality, competition, vegetation diversity and structure, species diversity and abundance) may be altered. Examples include the intersection between a cutblock and forest or a trail and native grassland.

Filter or porous barrier – a type of corridor across which some movement occurs but the rate of movement is less than through intact habitat (e.g., a busy highway is a porous barrier for large mobile wildlife like deer).

Fragmentation – the process of losing habitat continuity through temporary or permanent conversion of lands for human use (e.g., clearcutting forest, tilling native prairie for agriculture). Three general effects result from habitat fragmentation: (1) original habitat is lost, (2) remaining habitat patches decrease in size, and, (3) patches become increasingly isolated from one another.

Habitat – the environment in which an organism or biological population lives or grows.

Habitat alteration – habitat alteration occurs with disturbance of original habitat. *Temporary habitat alteration* occurs when pre-disturbance conditions are allowed to re-establish (e.g., forest regrowth after harvest). *Permanent terrestrial habitat alteration* occurs when different vegetation becomes established on the disturbed area (e.g., converting mixedwood forest to domestic grasses for hay production; introducing non-native species). *Permanent aquatic habitat alteration* occurs when substrate or channel conditions are modified, or where flow and sediment transport patterns are modified by upstream activities.

Habitat availability – the amount of usable habitat accessible to a particular species.

Habitat capability – the ability of a habitat unit to provide the life requisites of a species under optimum natural (seral) conditions, regardless of the current condition of the habitat, or the numbers of that species that are currently using the habitat (RIC 1999).

Habitat effectiveness – habitat quality, as perceived by a particular species. For instance, when a species uses the area around a man-made facility less than nearby areas of identical habitat, there has been a decrease in habitat effectiveness for that species.

Habitat loss – loss of habitat can occur in either terrestrial or aquatic ecosystems. *Terrestrial habitat loss* occurs when human activities disturb the soil or remove vegetation and regrowth is not allowed to occur (e.g., construction of a city, highway or industrial facility). *Aquatic habitat loss* occurs when water is removed, chemistry is substantially altered, or the structure of the waterbody is substantially altered.

Habitat suitability – the ability of a habitat unit, in its current condition, to provide the life requisites of a species. This rating is irrespective of the numbers of that species that are currently using the habitat (RIC 1999).

Habitat unit – a defined terrestrial or aquatic unit with consistent abiotic and biotic conditions.

Hazard level – the risk of an adverse cumulative hydrological effect defined by the British Columbia Interior Watershed Assessment Procedure (IWAP; BCF and BCE 1995b, BCF 1999).

Human activity – all forms of human actions including land conversion and disturbance, damming, water withdrawal, pedestrian, vehicle and aircraft movements, harassment, harvest, and contaminant input.

Index or metric – a numerical value used to represent or monitor the condition of an abiotic or biotic resource.

Indicator – a surrogate measure used to represent or monitor the condition of an abiotic or biotic resource. May be a representative species, an outcome, or an input.

Interior area – also referred to as core area in this report. Interior areas are those beyond the influence of edge effects.

IWAP – Interior Watershed Assessment Procedure; a method developed by the provincial government to help forest managers understand the type and extent of current water-related problems in a watershed, and to recognize the implications of proposed activity in that watershed (BCF 1999).

Juvenile – an individual age 1 or older that has not reached maturity.

Landscape – an area of tens to hundreds of square kilometres that includes one dominant background ecosystem. Northeast British Columbia consists of a number of landscape types including the forest landscape, the agricultural landscape, and the alpine/subalpine landscape.

Local population - A breeding group or stock with distinct genetic or life history attributes that interact on a regular basis. May also represent a component of a metapopulation or population found in a discrete or isolated area (Hanski et al. 1996).

Lowest observed effect level – concept from the field of ecotoxicology that represents the lowest concentration of a material used in a toxicity test that has a statistically significant adverse effect on the exposed population of test organisms compared to the controls. Also called lowest observed adverse effect level (LOAEL). This concept is also applicable to behavioural, physiological, and population response.

LU – Landscape Unit; an area of land and water used for long-term planning of resource management activities according to the *Forest Practices Code of British Columbia Act*. It may be used for designed strategies and patterns for landscape level biodiversity and for managing other forest resources (BCF and MELP 1999b).

LRMP – Land Resource Management Plan; a plan approved by the provincial government that establishes direction for land use and specifies broad resource management objectives and strategies.

Matrix – the dominant background ecosystem or land-use type within a habitat mosaic. Within the matrix, patches and corridors are reasonably uniform areas and linear features that differ from their surroundings.

Mature growth/mature seral – forests greater than a specified age (generally >80 years) based on biogeoclimatic zone and dominant species (BCF and MELP 1999b).

Metapopulation - a population of populations. This represents an abstraction of the population concept to a higher level. Metapopulations generally consist of a group of interacting but spatially discrete or isolated populations, subpopulations, or stocks. These subunits are linked by drainage networks but movement between subunits is infrequent and typically takes place across unsuitable habitat or over great distances (Hanski and Gilpin 1991; Dunham and Rieman 1999). An example of a fish metapopulation is a group of isolated headwaters populations found in the same watershed.

MSRM - British Columbia Ministry of Sustainable Resource Management

Natural disturbance type – an area characterized by a natural disturbance regime. The provincial government has established five natural disturbance types for managing biological diversity according to the *Forest Practices Code of British Columbia Act* (BCF and BCE 1995a; BCF and MELP 1999b).

OGC – British Columbia Oil and Gas Commission.

Old growth/old seral – forests greater than a specified age (generally >100 years) based on biogeoclimatic zone and dominant species (BCF and MELP 1999b).

Patch – a reasonably uniform area that differs from its surrounding landscape. Patches can occur naturally (e.g., sloughs, burns) or as the result of human disturbance (e.g., farmyards, wellsites, clearcuts).

Population – a group of interacting individuals of the same species in a defined area distinguished by a distinct gene pool or distinct physical characteristics.

Population limiting factors – processes that quantifiably affect the population rate-of-increase. Responsible for inducing year-to-year changes in animal abundance (Messier 1991).

Population regulating factors – density-dependent processes that ultimately keep populations within normal density ranges. These are a subset of limiting factors and predictably depress population growth as animal abundance increases (Messier 1991).

Reach – a defined watercourse channel section, tens to thousands of meters in length, with relatively consistent channel morphology, hydrology, and water chemistry.

Regional - an area more than hundreds of square kilometres that incorporates several landscapes.

Riparian - the banks and slopes next to streams, lakes and wetlands that are affected by elevated soil moisture levels for at least part of the year. These riparian areas protect water quality, stabilize banks, provide a continuous source of woody debris, nutrients, and food organisms, and regulate stream temperature (BCF and BCE 1995c; BCF and MELP 1999b).

Riparian clearings - cleared areas within 15 m of a waterbody, including linear corridors, communities and residences, industrial and commercial facilities, cutblocks, and agricultural fields.

Riparian roads - roads and trails within 100 m of a waterbody.

RMZ – Resource Management Zone; defined subdivisions of an approved LRMP that have unique sets of resource values, objectives to maintain or enhance those values, and a number of strategies to be implemented to achieve those objectives. These provide geographically-focused, strategic direction (Fort St. John LRMP 1997).

Seral stage – the stages of natural ecological succession of a plant community, for example from young through mature to old stage (BCF and BCE 1995a).

Stream crossing - a road, trail, pipeline, powerline, railway line, or cutline crossing of a watercourse.

Subpopulation - a breeding group or stock with distinct genetic or life history attributes that interact on a regular basis. May also represent a component of a metapopulation or population found in a discrete or isolated area (Hanski et al. 1996).

Subwatershed - a subset of a watershed, generally less than 1,000 km^2 in size.

Target threshold – a politically-defined goal reflecting the optimum amount of stress on the system. This threshold is more protective than the critical threshold to provide a margin of safety. A target threshold can be characterized as the level that is politically and practically achievable and provides adequate long-term protection to the environment or resource of interest.

Threshold – a point at which a resource changes to an unacceptable condition, with acceptability defined either from an ecological or social perspective.

Viable population – a self-sustaining population with a high probability of survival despite the foreseeable effects of demographic, environmental, and genetic stochasticity and of natural catastrophes (BCF and BCE 1995a).

Waterbody – a specific aquatic basins or channel (lake, pond, wetland, river, or stream).

Watercourse – a specific flowing channel (river or stream).

Watershed – a large drainage area, generally 1,000 to 10,000 km^2 in size, which flows directly into a large river such as the Peace River.

WHA – Wildlife Habitat Area contain critical habitat elements for one or more species of Identified Wildlife. These areas are mapped and approved by the Chief Forester and WLAP according to the *Forest Practices Code of British Columbia Act.*

WLAP – British Columbia Ministry of Water, Land and Air Protection.

ZOI – Zone of Influence; the distance to which a species is affected by an activity or disturbance.

1. INTRODUCTION

The British Columbia Oil and Gas Commission (OGC) established an environmental fund to address environmental effects associated with oil and gas development. In 2001, cumulative effects was one of three research envelopes for which funding was provided. Salmo Consulting Inc. (Salmo), Diversified Environmental Services (Diversified), GAIA Consultants Inc. (Gaia), and AXYS Environmental Consulting Ltd. (Axys), received funding under the cumulative effects research envelope to undertake two Case Studies in northeast British Columbia. The Case Studies were designed to test and develop approaches for assessing and managing cumulative effects in northeast British Columbia.

This report includes one of the components of the Case Studies, a literature review of ecological indicators and thresholds relevant to fish and wildlife management. Four main types of indicators and thresholds can be identified: Physical and Chemical, Ecological, Land Use, and Social. This review is largely restricted to ecological and land use indicators and thresholds, the classes of most direct relevance to fish and wildlife management. Physical and chemical thresholds are briefly discussed, but a review of social indicators is beyond the scope of this project.

As follows, this report is divided into six main sections with the detailed Case Studies provided in subsequent appendices.

Section 2. Cumulative Environmental Effects

- explains key concepts relevant to cumulative effects assessment and management, and
- describes the four main types of cumulative effects: habitat alteration and loss, barriers to movement, direct and indirect mortality, and disturbance.

Section 3. Management Thresholds

- explains key concepts relevant to thresholds.

Section 4. Ecological Indicators and Thresholds

- summarizes literature on four types of ecological indicators: habitat, population, biodiversity, and risk-based.

Section 5. Land Use Indicators and Thresholds

- summarizes literature on five types of land use indicators: human activity, human-caused mortality, access density, cleared or disturbed area, and watershed assessments.

Section 6. Species-Specific Indicators and Thresholds

- summarizes literature for three focus species: bull trout, grizzly bear, and woodland caribou.

Section 7. Indicators for Northeast British Columbia

- identifies recommended indicators for cumulative effects assessment and management in the region.

Section 8. References

2. CUMULATIVE ENVIRONMENTAL EFFECTS

Cumulative effects are environmental changes caused by combined past, present, and future human actions. Over the last twenty years, cumulative environmental effects have received increasing interest; it is now recognized that the combined effects of unrelated individual activities can be different in nature or extent from the sum of the effects of each individual activity (Contant and Wiggins 1991; FEARO 1994; Riffell et al. 1996).

2.1 LANDSCAPE ECOLOGY

Concepts and terms developed for the fields of landscape ecology and conservation biology provide a foundation for evaluating cumulative effects on fish and wildlife habitat and populations. The following discussion is adapted from Jalkotzy et al. (1998).

2.1.1 Terrestrial Habitat

The land we live on is a **mosaic**, or a pattern comprised of three basic units: patches, corridors, and the matrix. These three landscape elements may be natural or of human origin. The **matrix** is the dominant background ecosystem or land use type in the mosaic. Within this matrix, **patches** and **corridors** are reasonably uniform areas and linear features that differ from their surroundings. The matrix and the type and arrangement of patches and corridors in that matrix determine the suitability of the landscape mosaic for different species.

Scale is an important consideration for cumulative effect evaluations. A **landscape** represents an area of tens to hundreds of square kilometres that includes a particular mosaic. A **region** may contain many landscapes but may not necessarily have a repeating pattern of landscapes or landscape elements. At this larger scale, the type and spatial arrangement of landscapes determines the region's suitability for different species.

There are two main landscape types in northeast British Columbia. In the **forest landscape** (Figure 1), there are treed patches of variable size, age, structure, and species composition. Unforested openings associated with waterbodies, bedrock, and low-growing vegetative cover are also present. Within this matrix, there are human-disturbed patches such as wellsites and cut blocks, as well as disturbance corridors like roads, trails, and powerline and pipeline rights-of-way. The landscape outside forested areas (the **agricultural landscape**, Figure 2) is a human-dominated matrix with remnant areas of native trees, shrubs, and grasses in patches and along corridors (e.g., windrows or along road ditches).

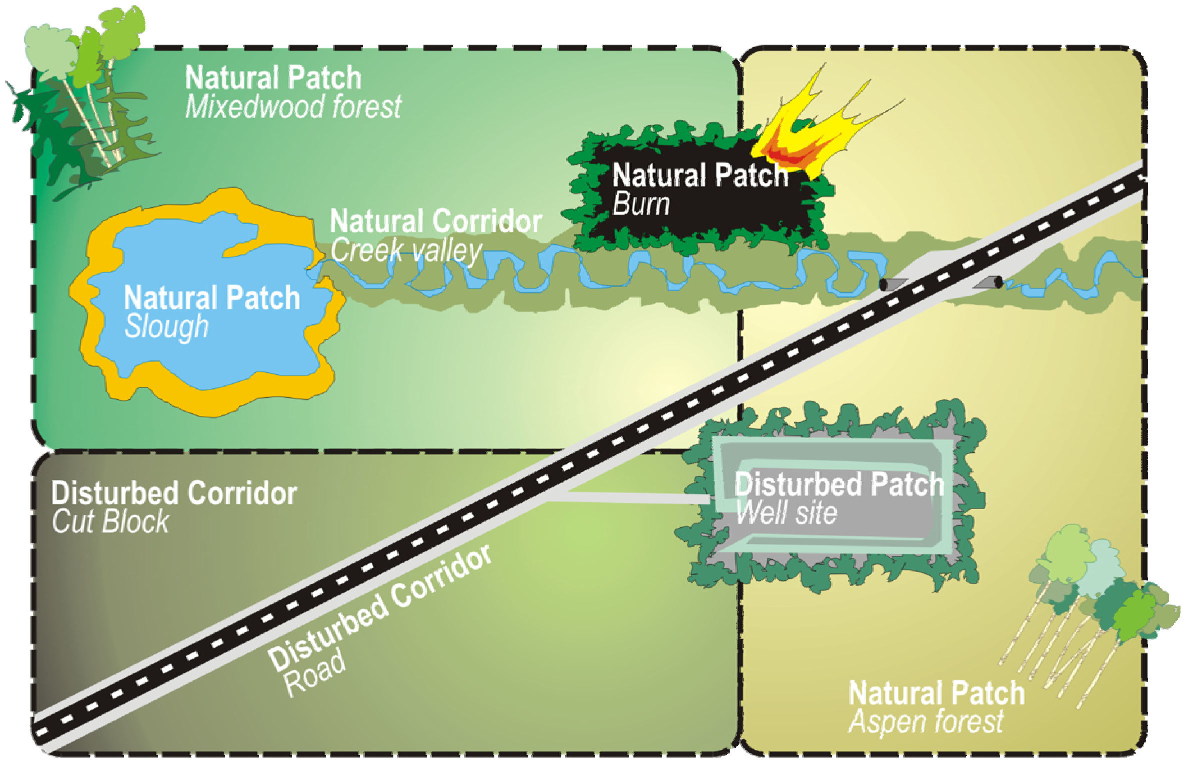

Figure 1. Elements of a forest landscape.

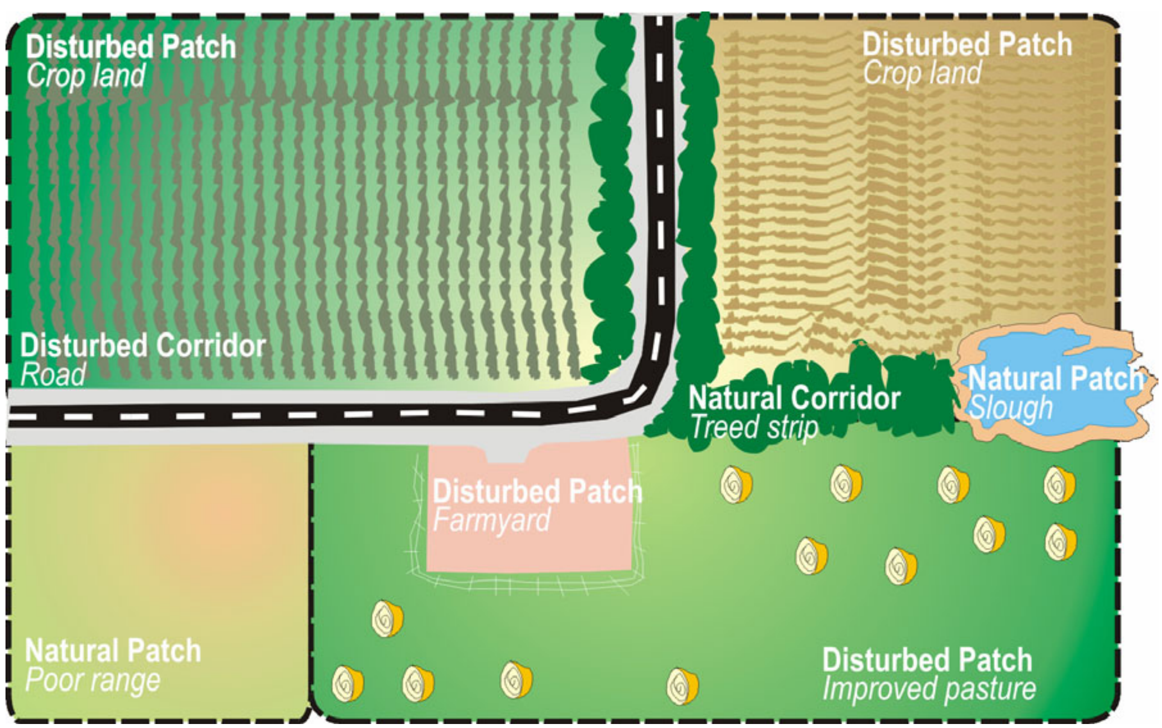

Figure 2. Elements of an agricultural landscape.

2.1.2 Aquatic Habitat

Landscape ecology can also be applied to aquatic systems. Watersheds are topographically defined areas within which surface water runoff drains to a lake or specific point on a stream. Watersheds are generally accepted to be an appropriate study unit because the quantity and quality of water and associated habitat at a specific point reflects the aggregate physical, chemical, and biotic factors upstream of this point (Omernik and Bailey 1997). Watersheds have provided the spatial boundaries for scientific study of the effects of natural and human factors on hydrology, water quality, and aquatic biota.

These factors display considerable variability over time and space and can be considered to occur along four dimensions. The longitudinal dimension integrates upstream-downstream linkages. Exchanges of matter and energy take place between waterbodies and the riparian/floodplain system along the lateral dimension. Interactions between waterbodies and groundwater occur along the vertical dimension. These pathways are integrated over time, the fourth dimension (Ward 1989).

Aquatic habitat can be divided into three main components: lentic, lotic, and riparian.

- **Lentic**, or standing water, habitat is found in wetlands, ponds, and lakes. The aquatic organisms found in a particular waterbody depend on its depth, size, substrate (bottom type), water quality, and the abundance and type of littoral (shallow nearshore) habitat, among other factors (Wetzel 1975).
- **Lotic**, or running water, habitat is found in creeks, streams, and rivers. Lotic systems normally consist of a pattern of tributaries joining one another and ultimately forming the 'mainstem' channel (Hynes 1970).
- **Riparian** areas include the banks and slopes next to streams, lakes and wetlands that are affected by elevated soil moisture levels for at least part of the year. These riparian areas protect water quality, stabilize banks, provide a continuous source of woody debris, nutrients, and food organisms, and regulate stream temperature (BCF and BCE 1995a, BCF 1999).

2.2 NATURAL VARIABILITY

The concepts of natural range of variability and ecological resilience are important for cumulative effects assessment and management. An underlying assumption of ecosystem management is that using natural disturbance patterns to guide management actions is one of the best possible means of achieving ecological sustainability in the absence of information on alternatives (Frissell and Bayles 1996; Andison 1999, 2000). A number of predictive modelling techniques incorporating random (stochastic) events have been developed to estimate the range of natural variability (Salmo et al. 2001).

2.2.1 Habitat

2.2.1.1 Terrestrial

Environmental conditions, habitat, and populations display natural variability that result from long-term climatic trends, short-term weather fluctuations, natural succession, and random events. The **natural range of variability** is determined by these factors as well as physiographic and watershed characteristics. The frequency and magnitude of disturbance events affect the composition, productivity, and successional state of terrestrial and riparian vegetation and stream channel characteristics (Ward 1989; Regier and Meisner 1990). Thus, the distribution, pattern, abundance, and connections between habitat units change in both time and space.

The primary natural disturbances influencing terrestrial habitat in the boreal forest are fire, insect and disease outbreaks, floods, and beaver (*Castor canadensis*) dams. Fire and insect outbreaks are by far the most important, and natural range of variability is often related to fire cycle (e.g., BCF and BCE 1995a). Fire cycles in the boreal forest have changed at least three times in the past 300 years, probably due to climate changes. Small fires (< 4 ha) have been most common, but infrequent large fires account for most of the area burned on a cumulative basis. Because fire cycles have been shorter than the lifespan of the dominant trees, old growth forest generally comprises a small percentage of the landscape in the boreal forest (Andison 1999, 2000; Johnson et al. 2001).

Most of northeast British Columbia is included within Natural Disturbance Type 3 (NDT3): ecosystems with frequent stand-initiating events. Historically, these forests experienced frequent fires that ranged in size from a few hectares to hundreds of square kilometres. Average fire size was likely 300 ha in some parts of the Boreal White and Black Spruce biogeoclimatic zone (BWBS; *Picea glauca* and *P. mariana*), but went as high as 6,000 ha in areas where topographic features did not limit fire spread. There were also frequent outbreaks of defoliating insects. Riparian areas provided special habitat characteristics not found in upland areas (BCF and BCE 1995a,c).

The mean disturbance event interval in NDT3 is 100 years for BWBS deciduous stands, 125 years for BWBS coniferous stands and Sub-boreal Spruce (SBS) stands, and 150 years for Engelmann Spruce-Subalpine Fir (ESSF; *Picea engelmannii-Abies lasiocarpa*) stands. Old growth stands are considered to be >100 years for BWBS deciduous stands, and >140 years for BWBS coniferous stands, SBS stands, and ESSF stands (BCF and BCE 1995a, BCF and MELP 1999b).

2.2.1.2 Aquatic

Disturbance events such as fires and windstorms that dramatically shape terrestrial systems may have relatively subdued long-term effects in aquatic systems (Minshall et al. 1997). Aquatic ecosystems respond more significantly to floods, acceleration of erosion, and channel barriers that have rather subtle or spatially restricted effects on terrestrial systems (Frissell and Bayles 1996).

The frequency and magnitude of flood events affect the composition, productivity, and successional state of riparian vegetation as well as the connectedness, depth, and productivity of floodplain channels (Ward 1989; Regier and Meisner 1990). In the boreal forest, the spring peak flow that results from melting snow is the largest runoff event of the year in 9 years out of 10. In one year out of 10 or 15, the largest peak event of the year is produced by a summer rain event; these rain-caused peak flows are by far the largest events (Alke 1995). Extreme changes in the amount of precipitation may cause floods or droughts that cause heavy sedimentation, alter channel morphology, and restrict fish movements (Meehan 1991).

Beaver dams, forest fires, strong winds, and landslides, can alter water quality, channel morphology, mobilize coarse woody debris that provides cover for fish, and create barriers to movement (Meehan 1991; Young 1994; Magnan and St-Onge 2000). The location and persistence of natural barriers are particularly important for migratory species such as Arctic grayling (*Thymallus arcticus*) and bull trout (*Salvelinus confluentus*) undertake seasonal movements between overwintering, rearing, and spawning areas.

2.2.2 Populations

Ecological resilience refers to the ability of a system or species to absorb natural and human disturbance without altering its fundamental structure (Weaver et al. 1996). Although fish and wildlife species present in northeast British Columbia have adapted to natural disturbances, individual species differ in their inherent ability to absorb disturbance and still persist as viable populations.

At the regional scale, ecological resilience is enhanced among groups of small, extinction-prone local populations that are linked by the movement of individuals between them (Hanski and Gilpin 1991). These **metapopulations** increase persistence by spreading the risk of disturbance occurring at the local population and individual levels over time and space. Species that are spatially restricted to a single subpopulation or a few individuals have lower ecological resilience and are less likely to persist over time (Weaver et al. 1996).

At more local scales, **population**-level responses such as increased reproduction and survival (generally referred to as compensation); offset increased rates of juvenile and adult mortality and thereby minimizing population fluctuations over time. These **density-dependent** effects are related to population size. Dispersal (immigration and emigration) by juveniles and adults is the mechanism by which vanishing local populations are rescued from extirpation and connectivity of metapopulations is maintained through time. In addition, local populations may exhibit a variety of life forms that increase the overall resilience of the metapopulation (Frissell and Bayles 1996; Mayhood 2001).

Finally, behavioural flexibility allows **individuals** to respond to local changes in habitat availability and quality and meet their energetic and reproductive needs (Weaver et al. 1996). This behavioural flexibility also complicates assessments of species response to both natural and human disturbance and is particularly relevant to the discussion of disturbance included in Section 2.3.4.

Species that mature early, have high reproductive rates, and are able to exploit a wide variety of food sources have higher ecological resilience because they are more able to compensate for natural and human disturbance. Other factors that affect species resilience include annual and seasonal home range size, population density and distribution, life expectancy, dispersal rates, ability to habituate, size, and mode of locomotion (Weaver et al. 1996).

At any point in time, population abundance and distribution reflects habitat availability and quality, as well as biological factors such as immigration, emigration, and juvenile and adult mortality that are dictated by weather, predation rate, and life history patterns. Population variability appears to increase with the length of time over which it is monitored or calculated, and this is assumed to reflect the influence of long-term positive or negative trends in abundance (Pimm and Redfearn 1988). Except for species with generation times longer than about 6 years, the range of variability observed after 50 years is assumed to correspond to normal variability excluding trends and major natural catastrophes (Thomas 1990). Populations occupying marginal or sub-optimal habitat should be expected to experience higher year-to-year fluctuations in abundance (Boulinier et al. 1998).

Environmental conditions during the winter and spring may have dramatic effects on fish and wildlife populations in this region that are not related to population size (**density-independent**). Winter snow depth is critical for many resident wildlife populations. Snow depths greater than 40 cm for deer (*Odocoileus* spp.), 50 cm for elk (*Cervus elaphus*), and 65 cm for moose (*Alces alces*) may result in significant overwintering mortality (Nietfeld et al. 1985). Weather conditions during the spring birthing period also affect neonate survival. Overwintering habitat may limit fish populations (Cunjak 1996). Water temperatures, current velocity, and water levels during the spring spawning, incubation, or emergence period often regulate year-class strength (Hubert et al. 1985; Ford et al. 1995; Cattaneo et al. 2002).

Natural disturbances, like a serious spruce beetle (*Dendroctonus rufipennis*) infestation of a coniferous forest, will have a variety of effects on bird populations. Bird responses will vary depending on their ecological requirements (especially nesting affinities), their ability to exploit altered habitats, the specific forest components experiencing change, and the time elapsed since the disturbance event. For example, a bird species requiring coniferous habitat for nesting and foraging had much lower abundance in heavily beetle-impacted boreal spruce forests, while understory-nesting birds, as a group, had significantly higher abundance in the same areas (Matsuoka et al. 2001).

2.3 CUMULATIVE EFFECTS IN NORTHEAST BRITISH COLUMBIA

In northeast British Columbia, cumulative effects result from direct loss and alteration of habitat due to petroleum exploration and production, forest harvest, agriculture and cattle grazing, subsistence and recreational use, municipal, urban, and residential development, and other industrial activities. Indirect disturbance and wildlife harvest also occurs due to use of roads and trails by recreational and subsistence users (e.g., hunters, fishermen). Cumulative effects can be classified into four types:

1. Habitat alteration, loss, and fragmentation,
2. Barriers to movement,
3. Direct and indirect mortality, and
4. Disturbance.

These four types will be discussed in greater detail in the sections below.

2.3.1 Habitat Alteration and Loss

Terrestrial **habitat loss** occurs when human activities disturb the soil or remove vegetation and regrowth does not occur (e.g., construction of a highway, secondary road, or industrial facility). This reduces available habitat for all species as long as the feature persists. Aquatic habitat loss occurs when water is removed, chemistry is substantially altered, or bed and banks are substantially altered.

Temporary **habitat alteration** occurs until natural conditions are restored to disturbed areas and waterbodies. Long-term or permanent terrestrial habitat alteration occurs when different vegetation regrows on the disturbed area (e.g., converting mixedwood forest to domestic grasses for hay production or reseeding a pipeline right-of-way with non-native agronomic species). Permanent aquatic habitat alteration occurs when substrate or channel conditions are modified, or where flow and sediment transport patterns are modified by upstream activities.

Permanent habitat alteration can also result from the introduction of non-native or exotic species (Douglas et al. 1990) and changes in grazing patterns and intensity (Platts 1991). The spread of non-native plants is affected by their dispersal capacity and the quality of immediately adjacent habitat. Cattle and linear disturbance corridors such as roads and trails can facilitate the spread of these species (Fleischner 1994; van Dorp et al. 1997; Hobbs 2001).

2.3.1.1 Habitat Fragmentation

Both dramatic and minor changes in habitat can benefit some species and hinder others. **Fragmentation** of the landscape occurs as continuity of the original matrix habitat is disrupted (Collinge 1996). Throughout history, fragmentation has naturally occurred through disturbances like glaciation, floods, and fires. In recent times, fragmentation has also occurred through conversion of lands for human use (e.g., harvesting and converting forest; tilling native prairie for use as crop-land).

Three general effects result from habitat fragmentation (Figure 3):

1. original habitat is lost,
2. remaining habitat patches decrease in size, and

3. patches become increasingly isolated from one another.

Figure 3. Representation of the habitat fragmentation process (adapted from Collinge 1996).

As original landscapes and aquatic habitat units become smaller and isolated, ecosystem function can be disrupted. With fragmentation, patches become more vulnerable to natural and human disturbances such as windstorms, fires, flooding, and non-native species invasion. This may affect pollination, seed dispersal, predation, and nutrient cycling (Lord and Norton 1990), exclude certain species, or increase their extinction probability over time (Andrén 1994; Collinge 1996).

As described further in Section 5.4, there are certain critical thresholds in the process of habitat fragmentation where rapid changes in the size and isolation of patches occur. A similar process can occur at the regional scale as original landscapes become smaller and isolated (Andrén 1994; With and Crist 1995; Mönkkönen and Reunanen 1999).

2.3.1.2 'Edge' Effects

Habitat alteration may occur indirectly through **'edge' effects** at the boundaries where a new patch or corridor joins the matrix. As expanses of continuous vegetation are fragmented, edges between the patches and surrounding habitat increase in length. Edges are particularly noticeable in forested landscapes, because of their dominant vertical structure.

Moisture, light, temperature, and wind regimes may change significantly at the fragment edge, and effects may extend into the patch for tens of metres. Edge orientation will affect the amount of environmental change; windward and/or south-facing edges will be warmer, drier and wider than leeward or north-facing edges. With the altered environmental conditions, the structure and composition of edge plant communities generally differ from those in interior (hereafter referred to as core) habitat. Other observed 'edge' effects include increased predation, mortality, competition, and brood parasitism; changes in vegetation diversity and structure; and changes in species composition and abundance (van der Zande et al. 1980; Wilcove et al. 1986; Laurance and Yensen 1991; Collinge 1996; Reed et al. 1996a; ESGBP 1998).

Edge effects are more pronounced in small or thin patches, since the ratio of edge to interior increases geometrically with decreasing patch size (Temple and Cary 1988; Collinge 1996; Culling and Anderson 2001). In very fragmented forests, virtually all remaining habitat may be so close to an edge that no functional core habitat remains (Temple and Cary 1988). For instance, if edge effects are assumed to extend 50 m from a patch edge, then a 1-ha square patch will contain no core habitat whatsoever while a 100-ha square patch will contain 81% core habitat (Figure 4; Collinge 1996). In contrast, a narrow 100 ha rectangular patch will contain only 69% core (Figure 5; Collinge 1996).

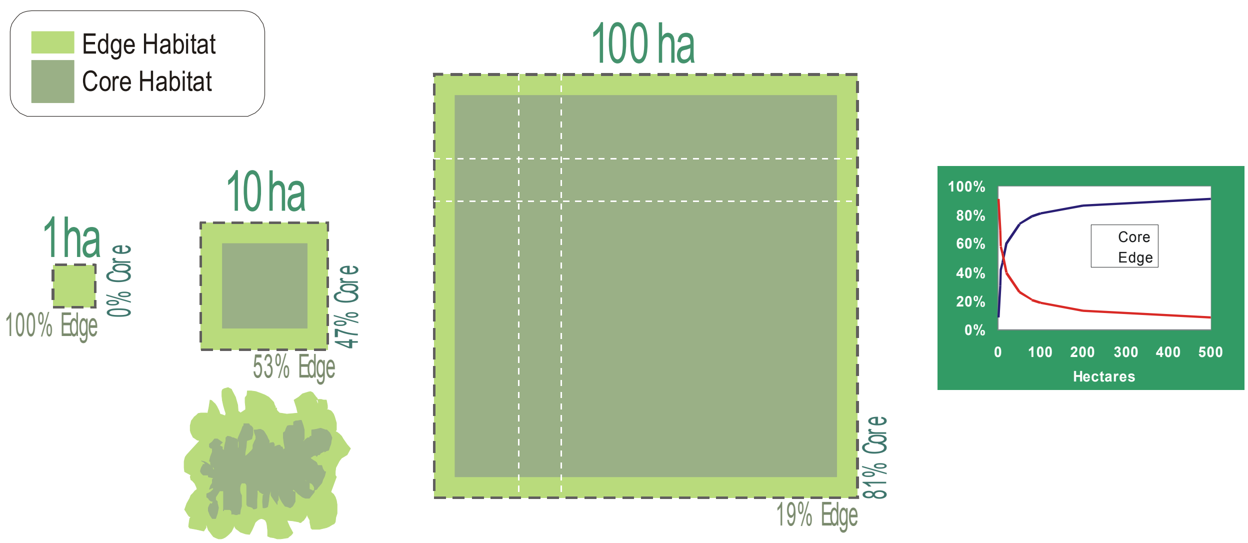

Figure 4. Relationship between patch size and edge area assuming 50 m edge width (adapted from Collinge 1996).

Roads create substantially more edge than clearings of equivalent area, due to patch geometry (Figure 5). In a foothills area, roads created 1.5 to 2 times more edge habitat than clearcuts. Taken together, roads and clearcuts influenced 2.5 to 3.5 times the land area directly disturbed by these uses (Reed et al. 1996). Roads will impact immediate and adjacent habitat quite differently than clearcuts; they can be expected to alter soil density, surface temperature, soil water content, light, dust, surface-water flow, pattern of run-off, and sedimentation (Trombulak and Frissell 2000).

The biological significance of edge effects differs between organisms and may need to be quantified by identifying the amount of habitat usable by a particular species (Andrén 1994; Fahrig 1997; Pasitschniak-Arts and Messier 1998; Mönkkönen and Reunanen 1999).

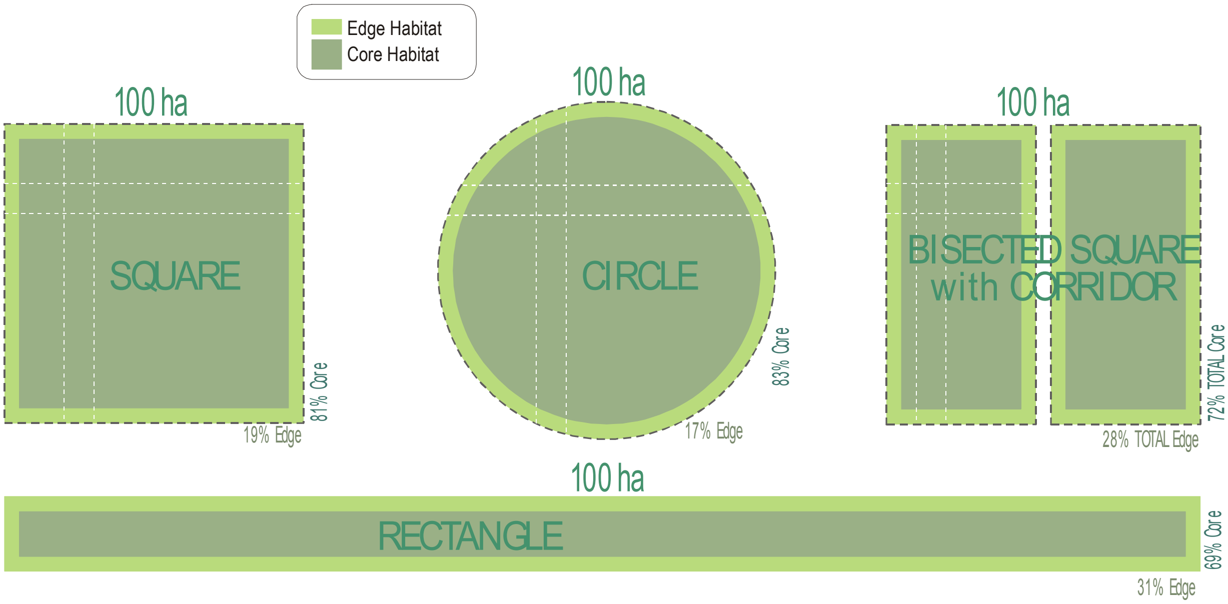

Figure 5. Relationship between patch geometry and edge area assuming 50 m edge width (adapted from Collinge 1996).

Overall, at any given scale, fragmentation is more likely to affect habitat specialists than habitat generalists (Lord and Norton 1990; With and Crist 1995; Collinge 1996; Kolozsvary and Swihart 1999), and core-adapted species than edge-adapted species (Figure 6). Edge effects are discussed further in Section 3.4.1.3.

Figure 6. Contrast between habitat loss for edge versus core species (adapted from Fahrig 1997).

2.3.1.3 Synergistic Effects: Livestock Grazing

Cumulative effects of habitat alteration and loss result from the total changes associated with existing, proposed, and likely future activities. For example, the combination of livestock grazing and human development appears to create a synergistic cumulative effect – one that is greater than that expected by adding the individual effects.

Forest ecosystems have evolved with grazing and browsing by native herbivores such as moose, elk, and deer. Under natural conditions, grazing pressure and intensity was variable and within the carrying capacity of the range. If forage production decreased, herbivores either moved to more favourable range or perished, bringing populations into balance with habitat capacity. Native vegetation became adapted to intermittent cropping by variable numbers of animals followed by years of rest.

In contrast, grazing and browsing by domestic livestock, along with associated agricultural practices, have decreased the density of native plant species and the diversity of native plant communities throughout western North America. A key difference is that livestock are confined within barriers, continually grazing the same range, sometimes at a rate higher than the area's carrying capacity. Restricted movement and higher stocking rates result in trampling and compaction of the soil, encouraging a vegetation change to shallow-rooted annual species or taprooted perennials that can grow in areas with lowered water tables (Epp and Townley-Smith 1980; Platts 1991; Fleischner 1994).

Livestock are attracted to riparian areas bordering waterbodies and these areas appear to have suffered greater habitat loss and alteration than upland areas in western North America (Platts 1991; Fleischner 1994). In forested areas, they may also create edge effects by attracting cowbirds (*Molothrus ater*), a nest parasite that reduces the nesting success of forest songbirds.

Non-working areas of wellsites are normally revegetated following construction in pasture. However, cattle appear to be attracted to these structures and sites, leading to nearly constant trampling of these areas (Epp and Townley-Smith 1980; ERIN 2000), and increasing the spread of weeds and non-native species. The resulting bare areas exceed those that would result from either petroleum development or cattle grazing alone. Grazing can increase nesting habitat quality for some non-game upland bird species (e.g., shorebirds) while others have lower nesting frequency in active grazing areas (e.g., sharp tailed grouse [*Tympanuchus phasianellus*]; Kantrud and Higgins 1992).

Thus, the net cumulative effect on habitat reflects habitat alteration and loss from all sources, as well as other types of effects such as barriers to movement and increased mortality.

2.3.2 Barriers to Movement

Animals may be reluctant or unable to cross corridors, although the actual effect depends on the species, the width of the corridor, the corridor characteristics, the surrounding habitat, and the frequency and volume of human use. In general, the likelihood and magnitude of barrier effects increases as corridor width increases, the corridor becomes more dissimilar from surrounding habitat, and vehicular and human use of the corridor increases.

A **barrier** is present when no movement occurs across a corridor and the habitat on either side of the corridor becomes fragmented at the individual animal or local scale. An example is an improperly constructed road culvert that completely blocks fish movement (Harper and Quigley 2000).

If some movement occurs but the rate of movement is less than through intact habitat, then the corridor is considered a **filter** or **porous barrier** (Jalkotzy et al. 1998). The busy Alaska Highway would be considered a filter for large mobile wildlife such as deer but might represent a barrier for small ground-dwelling insect species. In the forest landscape, most trails and roads are assumed to be filters for key wildlife species since they are comparatively narrow, low activity features that do not dramatically differ from the surrounding habitat.

2.3.3 Direct and Indirect Mortality

Activities associated with corridors may cause direct mortality, or increase the risk of mortality. This includes wildlife-vehicle collisions, powerline strikes or electrocutions, and fish mortality due to instream blasting or water withdrawal. Increased mortality risk is associated with hunter or predator use of corridors and facilities, and results from increased harvest facilitated by access. Both direct and indirect mortality affect the numbers of animals present in an area (Jalkotzy et al. 1998).

Hunting, fishing, and trapping can alter behaviour, population structure, and distribution patterns of fish and wildlife populations. In some cases, harvest may be partially if not completely additive with natural mortality (density-independent); immigration from unexploited areas may be necessary to sustain harvested populations (Knight and Cole 1995a; Kitchen et al. 1999).

Cumulative effects frequently occur because access corridors created for an individual project increase the level of human activity from all users. This can lead to increased harvest effort and success, increased vehicle mortality, and increased animal removal/ mortality for wildlife management (e.g., problem bears). These indirect mortality effects are difficult to manage (Mychasiw and Hoefs 1988; Trombulak and Frissell 2000).

2.3.4 Disturbance

Man-made facilities and corridors and the activities associated with them can directly and indirectly affect animals. The type and magnitude of impacts varies with the type of human activity. Consumptive activities such as hunting and fishing have a different impact than non-consumptive activities such as nature viewing. Motorized and non-motorized activities also have different effects (Knight and Cole 1995b). In most cases, the mere presence of a structure or right-of-way in grassland areas does not appear to disturb wildlife. Responses in forested areas are less clear and vary by species, season, and disturbance width, size, and geometry, among other factors.

Most published studies have focussed on short-term responses to human activity (Knight and Cole 1995a; Hill et al. 1997). In general, impacts are inversely related to the level and predictability of human disturbance. Continuous, frequent or unpredictable high intensity activities (e.g., motorized snow machines, powerboats, gun shots) cause greater response than low intensity or infrequent activities (e.g., generator motor noise). Human activity usually generates greater response than vehicle or equipment activity (e.g., Perry and Overly 1977; Holton 1982; Murphy and Curatolo 1987; Henson and Grant 1991; Hockin et al. 1992; Andersen et al. 1996; Hill et al. 1997; Webster 1997; Kitchen et al. 1999). Response is also affected by the type of activity, the timing and location of the activity, the age and sex of individuals, as well as the group size (Knight and Cole 1995b).

Animals may display a waning response to repeated or predictable disturbance that is perceived to be nonthreatening. This 'habituation' is most likely to occur in protected areas, or areas with consistently high levels of human activity (Knight and Temple 1995).

Disturbance may affect animals at the individual, population, and community level. Individual responses include *physiological changes* (e.g., increased heart rate), *behavioural changes* (e.g., reduced use of disturbed areas), and *ecological changes* (e.g., altered reproductive success). These individual responses may combine to produce effects at the population level such as altered abundance, spatial distribution, mortality rate, and reproductive rate. Differences in the responses of individual species can lead to community level changes in relative abundance, competition, or other species interactions (Karr and Freemark 1985; Knight and Cole 1995a).

2.3.4.1 Individual and Behavioural Response

A single disturbance event usually has only a short-term impact on an individual animal. For example, a deer will run over a ridge to get out of sight of a person walking down a trail, but may choose to return and resume its earlier activities once the person passes. This response may carry a cost in terms of energy expenditure and lost feeding. It will vary between animals, though; hunted populations typically exhibit stronger disturbance reactions to people along roads than do wildlife in protected areas.

Repeated human activity can cause cumulative impacts when areas adjacent to human-disturbed patches and corridors are used less than nearby areas with identical habitat. This loss of '*habitat effectiveness*' can accumulate over time and lead to progressive declines in species diversity and abundance (van der Zande et al. 1980; Riffell et al. 1996; Richardson and Miller 1997; Rodgers and Smith 1997; Gutzwiller et al. 1998; Jalkotzy et al. 1998).

Figure 7 depicts road-habitat effectiveness models developed for a hunted elk population; habitat effectiveness is inversely related to road density. This figure also demonstrates the effect of disturbance intensity on habitat effectiveness. Infrequently used access features like 'Primitive Roads' affect elk habitat use considerably less than more regularly used 'Primary Roads' and 'Secondary Roads' with moderate traffic volumes (Thomas et al. 1979; Lyon 1984; Thomas et al. 1988).

It is important to note that behavioural responses to disturbance and reduced reproductive success may not translate into population-level effects such as reduced animal numbers described below. This is particularly true for species with high reproductive rates and those that habituate to repeated disturbance (e.g., Geist 1978; Bergerud et al. 1984; Hockin et al. 1992).

2.3.4.2 Population or Ecological Response

Repeated disturbance can affect habitat use, reproduction, and survival; it can ultimately change population or community dynamics (Geist 1978; Yarmoloy et al. 1988). Responses vary considerably between and within species, but documented examples for birds include changes in feeding patterns, compromised nest defence, and reduced hatching and fledging success (Safina and Burger 1983 *in* Riffell et al. 1996, Tuite et al. 1984, Keller 1989 *in* Riffell et al. 1996; Skagen et al. 1991; Holmes et al. 1993, Riffell et al. 1996; several authors cited in Richardson and Miller 1997; Rodgers and Smith 1997; Gutzwiller et al. 1998). This is a potential concern for rare species with low abundances where displacement could affect their continued presence in an area (Hockin et al. 1992; Riffell et al. 1996).

While the importance of disturbance on animals is acknowledged, the understanding of population-level response is incomplete because populations respond simultaneously to both natural and human-caused influences (Hill et al. 1997).

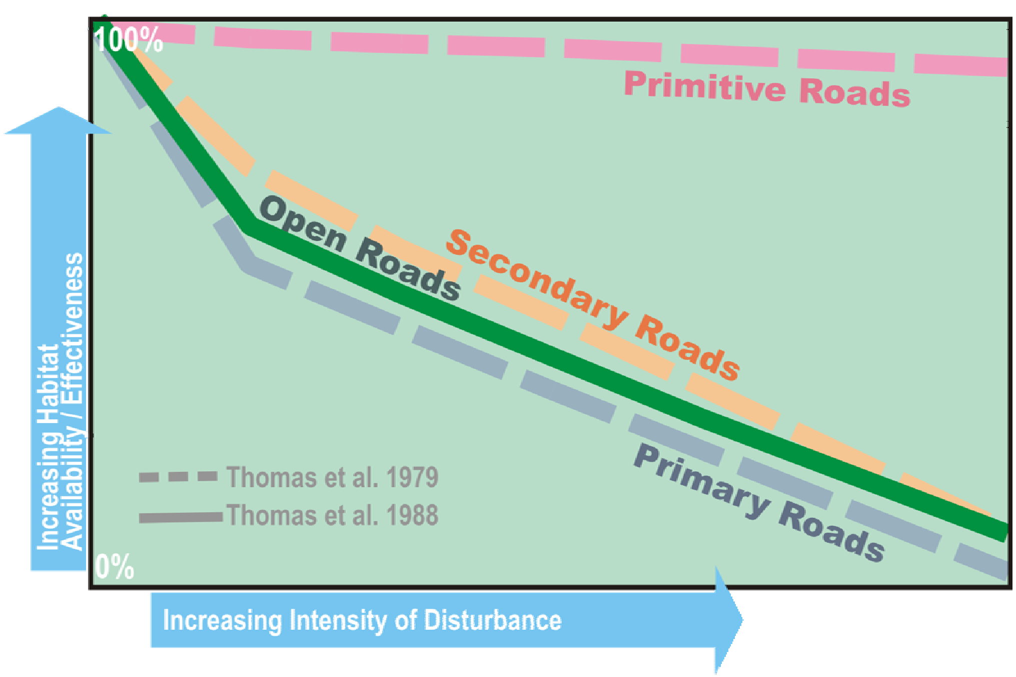

Figure 7. Impact of roads on predicted habitat effectiveness for elk.

3. MANAGEMENT THRESHOLDS

A **threshold** is an objective, science-based standard used to evaluate the acceptability of project-specific and cumulative effects (Ziemer 1994); acceptability can be defined from an ecological or social perspective. Thresholds should be based on measurable, attainable, and applicable attributes of ecosystems, communities, assemblages, or species (Merigliano et al. 1997; Hughes et al. 1998; Axys 2001a).

The threshold concept presumes that ecological or social capacity exists to appropriately accommodate human disturbance, that we are capable of identifying this capacity, and that we will implement activities that will not violate it. It also assumes that ecosystems and watersheds possess inherent resilience that allows recovery from past and continued human disturbance (Frissell and Bayles 1996).

A perceived regulatory advantage of thresholds is that they allow development activities to proceed without detailed review until the defined threshold is reached. Thus, if the incremental effects of a proposed activity do not cause threshold exceedance, then the effects are concluded to be insignificant and the proposal proceeds. However, once the threshold is reached, extra review or regulation is necessary (Ziemer 1994; Axys 2001a).

Thresholds may be based on outcomes (desired habitat or social conditions) or inputs (activity intensity). Outcome-based thresholds are preferred, since outcomes can be influenced by more than one input, and it is the outcome that is important from a management perspective. Nevertheless, thresholds based on acceptable inputs are required when desired outcomes cannot be practically defined (Merigliano et al. 1997). Examples of thresholds identified by Merigliano et al. (1997), Axys (2001a), and others are provided in Table 1.

Table 1. Examples of input-and output-based thresholds.

Output-Based Thresholds	Input-Based Thresholds
• Minimum amount of suitable habitat within a specified geographic area or watershed (e.g., greater than 60% moderate and high suitability habitat).	• Maximum human disturbance rate (e.g., maximum allowable backcountry trail use in persons/month).
• Number of animals in a specific geographic area or waterbody (e.g., 1,800 moose in a specified wildlife Management Unit).	• Maximum level of land use activity or development within a specified geographic area or watershed (e.g., cleared area, length of access, access density).
• Maximum body burden (e.g., tissue mercury content).	• Maximum pollutant ambient concentration or deposition rate.
• Non-native species presence (e.g., designated exotic weeds present on no more than 2% of area).	• Lowest pollutant concentration at which an adverse health or ecological effect is observed.

Four main types of indicators and thresholds can be identified: Physical and Chemical, Ecological, Land Use, and Social. The review provided here emphasizes ecological and land use indicators and thresholds; social indicators are not discussed. Physical and chemical thresholds are well established and widely applied, and they are briefly discussed as background to the discussion of indicators and thresholds of most direct relevance to fish and wildlife management. Additional information specific to each of these types is discussed as follows:

- Physical and Chemical Indicators and Thresholds (Section 3.2)
- Ecological Indicators and Thresholds (Section 4)
- Land Use Indicators and Thresholds (Section 5)

3.1 PHYSICAL/CHEMICAL INDICATORS AND THRESHOLDS

The best examples of science-based thresholds are air and water quality criteria developed by the provincial and federal governments. These are based on observed or modelled dose-response relationships; an example of which is shown in Figure 8.

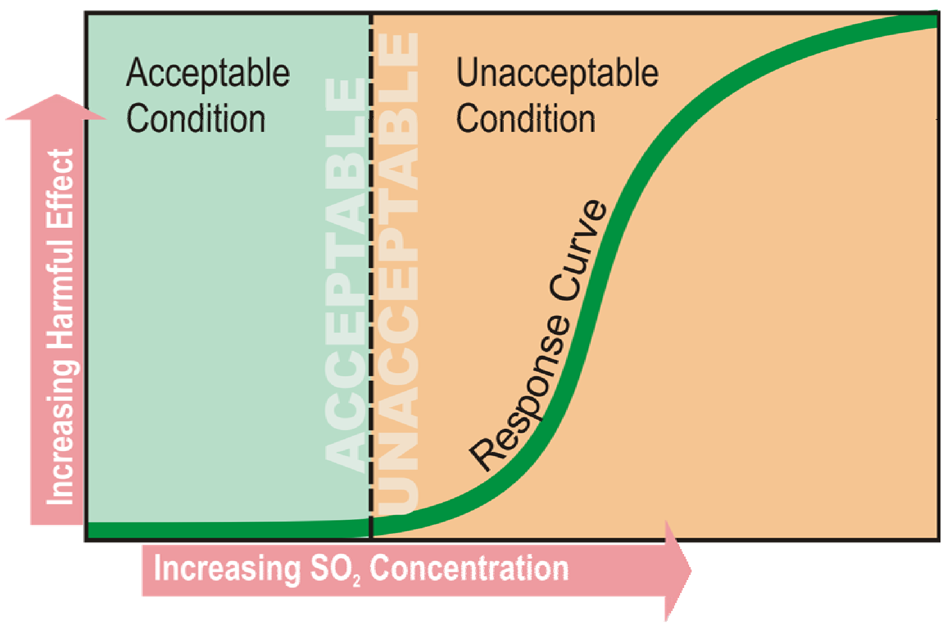

Figure 8. Theoretical dose-response curve showing an effect threshold.

In this example, the air quality threshold represents the concentration that causes detectable effects on a sensitive receptor (e.g., reduced growth of lichen). Identification of specific thresholds may be difficult because effects differ with the averaging period used, type of exposure (e.g., chemical form, dose), the ecological setting, and the life stage of a species. With this inherent uncertainty, regulators may build in a safety margin by choosing a threshold below the lowest detected effect concentration (Bull 1991).

In northeast British Columbia, federal and provincial air and water quality thresholds are routinely applied to evaluate proposed activities. These thresholds are designed to provide long-term protection to the environment (CCME 1996; MELP 2001).

3.2 TIERED THRESHOLDS

Tiered thresholds were originally developed to manage deposition of acidic air pollutants (Bull 1991, 1992). This approach provides an integrated framework that uses two or more quantitative thresholds to identify appropriate management and regulatory responses.

Figure 9 illustrates the tiered threshold model defined by the Clean Air Strategic Alliance (CASA; AENV and CASA 1999) for management of potential acidification input. This model identifies three receptor-based management tiers: **Critical**, **Target**, and **Cautionary Thresholds**.

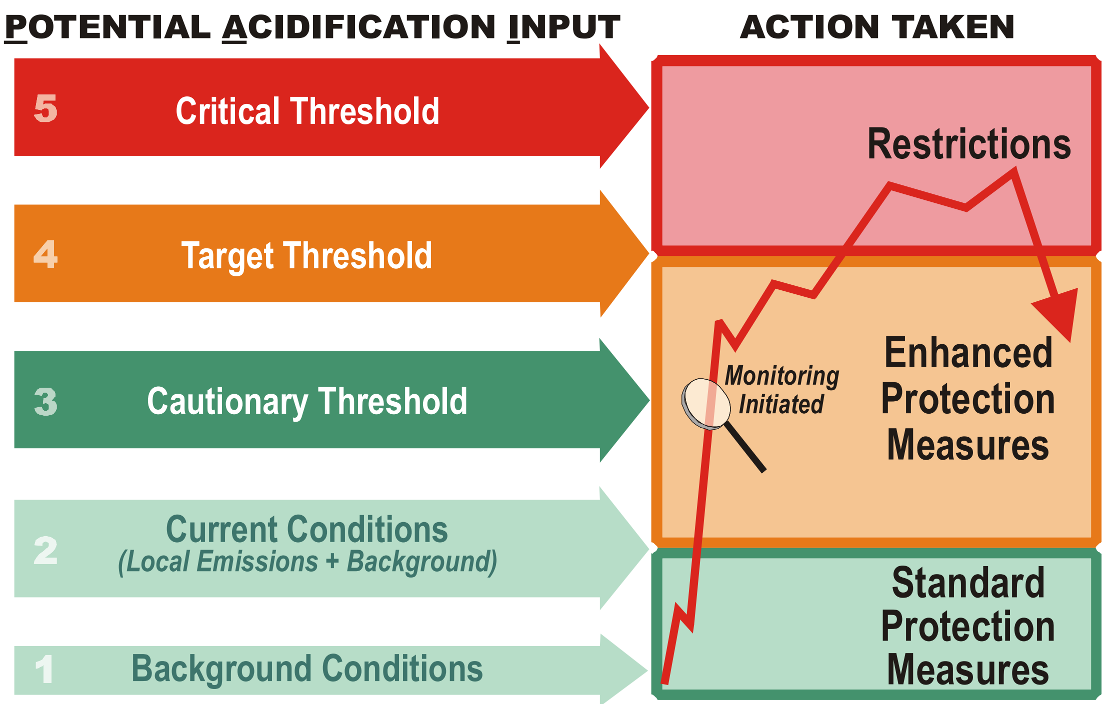

Figure 9. Application of tiered thresholds to management actions.

3.2.1 Critical Thresholds

A critical threshold is the continuous maximum amount of stress that an ecosystem or species can support without sustaining long-term harm (Bull 1992). To establish a critical threshold, an acceptable magnitude of change or level of protection must be defined. This definition should account for societal values on environmental quality and natural resource protection, which is difficult since there are many viewpoints on what constitutes an unacceptable ‘adverse’ effect. Once the acceptable protection level is defined, a science-based threshold is calculated from the best available information. This calculation is based on known cause and effect relationships like that depicted in Figure 8 (Bull 1991, 1992).

Acceptable change has been defined in a variety of ways. For example, the critical threshold may be based on predicted risk of population extinction (Lande 1987), predicted probability of population survival (Lamberson et al. 1992), or the probability and severity of an undesirable effect (Francis and Shotton 1997). CASA defined threshold loads from ‘levels of protection’, where a 100% level of protection meant protection for all ecosystems, while a 90% level of protection meant that 10% of ecosystems or species might experience stress above their critical load (AENV and CASA 1999). CASA defined Critical Thresholds as providing a 95% level of protection for sensitive, moderately sensitive, and low sensitivity soils.

When the Critical Threshold load is reached or approached, restrictive management practices are formally adopted. These can include pre-defined protection and recovery measures mandated through the review and approvals process. Examples include implementation of economic instruments that discourage emissions use, retrofitting of ‘Best Available Technology’, and use of predefined recovery responses like activity restrictions (AENV and CASA 1999).

3.2.2 Target Thresholds

A Target Threshold reflects a politically defined goal for the amount of stress on a system. It incorporates economic, social, and technological considerations; it should ideally be below the critical threshold to provide a margin of safety. A Target Threshold can be characterized as the level that is politically and practically achievable and provides adequate long-term protection to the environment or resource of interest.

The Target Threshold may reflect a precautionary management philosophy or uncertainty associated with scientific predictions of the Critical Threshold. In Alberta, CASA defined the target acid deposition threshold loads at approximately 90% of the critical load (AENV and CASA 1999; Figure 9).

When this threshold is reached, enhanced management practices are formally adopted. These can include expanded environmental monitoring and applied research, voluntary use of ‘Best Available Demonstrated Technology’, and implementation of enhanced protection or recovery methods like ‘No Net Habitat Loss’ or restrictive harvest regulations (AENV and CASA 1999). Where existing disturbance levels exceed the critical threshold, the target threshold may be set at or above the critical threshold load, or a series of diminishing target threshold loads may be applied over time to progressively reduce stress to levels below the critical threshold (AENV and CASA 1999).

3.2.3 Cautionary Thresholds

A Cautionary Threshold is established to indicate when additional or more intensive monitoring is required. This concept was established by CASA (AENV and CASA 1999) to ensure that sufficient local data existed to confirm scientific predictions of both target and critical thresholds.

When this threshold is reached, issue-specific monitoring is initiated to document environmental conditions or responses. No other management or mitigation actions are required, but activities must comply with established regulatory guidelines and best industry management practices. Routine environmental and activity monitoring is also conducted to confirm that best management practices are being applied (AENV and CASA 1999).

Where there is not enough information to determine how much stress a system can sustain, a decision can be made to define an interim threshold between the cautionary and target thresholds. Final thresholds would be established only after further monitoring, research, and stakeholder consultation.

4. ECOLOGICAL INDICATORS AND THRESHOLDS

Unlike chemical criteria, ecological thresholds are not well established for management of fish and wildlife populations, habitat, and biodiversity. There is relatively little published research detailing their practical derivation and implementation.

Ecological indicators take several forms:

- **Habitat Conditions**: the predicted availability or quality of habitat for selected animal species or guilds based on mapped characteristics (Section 4.1,
- **Species, Communities, or Guilds**: the presence, relative abundance, or perceived 'health' of a single plant or animal or defined group of plants or animals. Generally selected based on: economic, social, or ecological importance; sensitivity to change; or special conservation status (Section 4.2),
- **Biodiversity**: the number of species, habitat units, or ecosystems present in a defined area (Section 4.3), and
- **Risk-based:** the predicted probability of species or population loss over a specified time period (Section 4.4).

4.1 HABITAT INDICATORS AND THRESHOLDS

Studies in both temperate and tropical areas have shown a positive relationship between the number of species and the area of contiguous suitable habitat (e.g., Forman and Godron 1986; Seagle 1986; Wilcove et al. 1986; Flather and Sauer 1996). Thresholds based on protection of habitat are often the most practical tool for managing cumulative effects on fish and wildlife species (Axys 2001a).

Figure 10 provides a theoretical example of how habitat-based ecological thresholds can be developed and applied using the road density-habitat effectiveness response provided in Figure 7. The green line displays the observed or predicted effect on habitat availability or effectiveness as disturbance intensity increases. No single point represents a transition from an acceptable to unacceptable state. Thus, biological thresholds are most reasonably represented as a range to better reflect natural variability in environmental conditions and population parameters (Axys 2001a). In this example, 30% habitat availability/effectiveness defines the transition point from acceptable to unacceptable habitat condition. This range may be established arbitrarily or based on a calculated degree of risk.

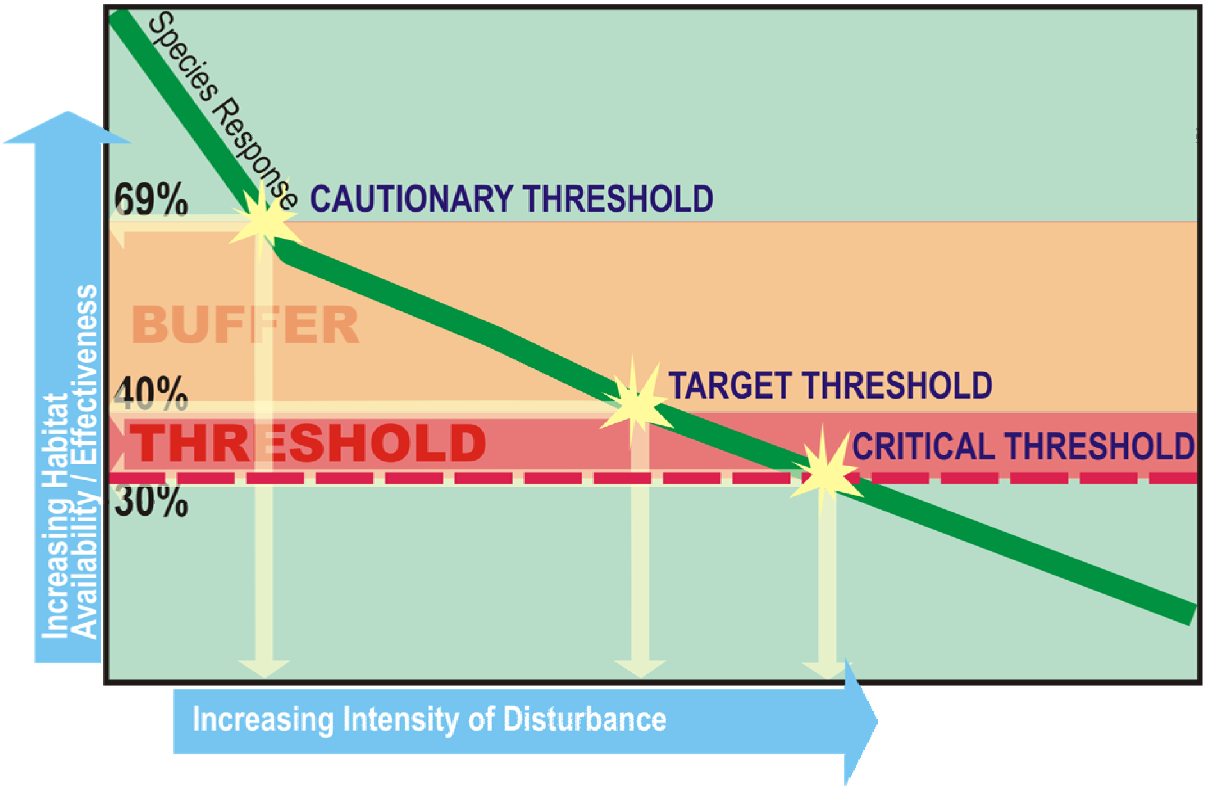

Figure 10. Ecological response curve and tiered habitat thresholds.

4.1.1 Habitat Availability

Habitat availability reflects an area's predicted wildlife capability or suitability by vegetation or ecological unit. These values estimate the area's ability to support and maintain individuals of an identified species (but see Van Horne 1983). Consideration of habitat availability allows terrestrial vegetation or ecological units and associated values (e.g., wildlife suitability, rare plant potential) to be quantified and compared with proposed habitat modification or loss at a variety of scales. Habitat suitability models frequently consider specific season and life requisites (Salmo et al. 2001).

Habitat thresholds are generally outcome-based. An outcome-based target threshold may be established at points where rapid changes in the size and isolation of habitat patches occur (e.g., change at 69% habitat indicator in Figure 10). Generalized habitat availability thresholds (e.g., >40% habitat availability) have been applied for landscape-level cumulative effects evaluations. Species-specific evaluations normally also consider the quality of available habitat (Table 2).

When more than 10% to 30% functional habitat remains in a region, species are affected by habitat loss, but strong habitat fragmentation effects are not common (Andrén 1994; Fahrig 1997). In a North America wide study of four species of breeding tanagers, Rosenberg et al. (1999) found that tanagers (as well as nest predators and parasites) exhibited the strongest negative response to fragmentation in areas of their range that were the most fragmented. Responses to fragmentation were significant but less severe in less fragmented areas (reviewed by Andrén 1994; Rosenberg et al. 1999). In a boreal landscape with only 11% of the forest harvested, habitat configuration explained significant variation in species presence/absence data for only 20% of bird species examined (Hannon 2000).

Table 2. Habitat availability guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Habitat Availability	▪ >10% to 30% of landscape in suitable habitat for birds and mammals (Andrén 1994).	▪ Threshold based on review of published studies and theoretical modelling.
	▪ >40% of landscape in suitable habitat for habitat specialists (Wilcove et al. 1986; Lee and Gosslink 1988; Laurance and Yensen 1991; With and Crist 1995).	▪ Theoretical threshold based on modelling.
	▪ >20% to 35% of landscape in suitable habitat for habitat generalists (Wilcove et al. 1986; Lee and Gosslink 1988; Laurance and Yensen 1991; With and Crist 1995).	▪ Theoretical threshold based on modelling.
	▪ 15% to 30% of forest in habitat conservation areas (Lamberson et al. 1994).	▪ Design for spotted owl reserves.
	▪ Landscape with 40% forest cover and 60% forage areas optimum for elk and deer (Thomas et al. 1979).	▪ Based on studies in Oregon.
	▪ >50% to 75% forest cover in areas greater than 9 to 10 km^2 (Hargis et al. 1999; Potvin et al. 2000).	▪ Recommendation to protect marten in eastern forests.
	▪ <10% of each Forest Operating Unit should be affected by logging (Horejsi 1996).	▪ Recommendation to protect grizzly bear in Yukon Territory.
	▪ <20% of watershed cleared within 10 years (McGurk and Fong 1995).	▪ Detectable changes in streamflow occur above this threshold.
	▪ 5% reduction in total forest area did not cause a detectable effect on relative abundance of forest interior songbirds, considered as a group; >10% reduction did (Rich et al. 1994).	▪ Based on studies in eastern forest.

4.1.2 Habitat Effectiveness

Some animals respond to human activities by modifying their behaviour and avoiding the affected area (Trombulak and Frissell 2000; Hamilton and Wilson 2001). **Habitat effectiveness** is an indicator of the value and amount of habitat available to an animal after accounting for the disturbance created by human infrastructure and activities. This indicator is calculated by species.

Calculation of habitat effectiveness is a three-step process (Axys 2001a; Salmo et al. 2001):

1. Habitat suitability models rate the quality of mapped ecological units for each species assuming *no human use*,

2. Human infrastructure and associated activities are stratified into groups with similar disturbance potentials. Each disturbance source is assigned a *zone of influence* (ZOI) that identifies the distance to which the species is affected by the activity. *Disturbance coefficients* (DCs) are assigned to each activity type, based on the degree to which the ZOI remains effectively useable by the species. DCs are determined from available literature or expert opinion; they range in value from 0 (i.e., no disturbance and no effect on the species) to 1 (i.e., high disturbance and probable exclusion of the species), and

3. An overlay of human use features and associated ZOI is integrated with the habitat suitability map, then the amount of effective habitat remaining is quantified for each analysis area (e.g., bear Wildlife Habitat Unit).

Table 3 presents some examples of established habitat effectiveness thresholds; most of these are outcome-based.

Table 3. Habitat effectiveness guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Habitat Effectiveness	▪ No net loss of habitat effectiveness for grizzly bear (BCF and MELP 1999a).	▪ Recommended British Columbia threshold.
	▪ >80% of all Bear Management Units with > 80% or greater habitat effectiveness for grizzly bear (Parks Canada 1997).	▪ Threshold adopted by Banff National Park.
	▪ >80% habitat effectiveness for elk (Servheen 1993).	▪ Grizzly bear management standard for Montana National Forest; based on Lyon (1979) elk-road model.
	▪ >80% habitat effectiveness for grizzly bear (Horejsi 1996).	▪ Recommendation calculated based on road density, assuming that 1 km/km^2 equates to 80% habitat effectiveness.

4.1.3 Edge

Natural and human-created edges of habitat patches differ ecologically from interior habitat, since edges have altered microclimate, changes in habitat type and quality, altered predation, parasitism and competition patterns, and changes in species composition and abundance (Taylor 1977; Wilcove et al. 1986; Laurance and Yensen 1991; Reed et al. 1996a; ESGBP 1998). Edge indices generally quantify the area where habitat quality is, or may be, affected by natural and man-made fragmentation.

Edge indicators take two forms:

1. Mathematical metrics that relate the total length of edge to the area of interior (km per ha) or a predefined analysis area (km per km^2), and
2. Estimates of the area affected by ‘edge’ effects (i.e., physical or chemical changes or reduced habitat effectiveness) in relation to a predefined area such as a mapped ecological unit (km^2 per km^2).

The diverse vegetation of the edge ecotone may contain a high density of birds, but a greater proportion of the individuals are juveniles, remain unpaired, experience high nest failure, fledge fewer offspring in a breeding season, and have lower survival rates (reviewed in Culling and Anderson 2001). Through these processes, edge habitat may act as an ‘ecological sink’ for forest interior bird species.

Edge effects are more pronounced in small forest fragments, since the ratio of edge to interior increases geometrically with decreasing forest size (Figure 4; Temple and Cary 1988; Collinge 1996; Culling and Anderson 2001). In very fragmented forests, virtually all remaining habitat may be so close to an edge that no functional interior habitat remains (Figure 4; Temple and Cary 1988). In general, larger forests contain more forest interior and neotropical migrant species, while smaller forests have a higher density and diversity of edge and interior-edge specialists (Askins et al. 1990, citing numerous authors). As patch size decreases, bird species richness in isolated fragments frequently decreases and the likelihood of local extinction increases (reviewed in Collinge 1996).

Forest fragmentation has been implicated as an important factor in the decline of ‘forest interior birds’, species such as warblers that are dependent on forest core (Askins et al. 1987, 1990; Paton 1994; Flather and Sauer 1996). Research in small, isolated habitat fragments surrounded by agricultural and suburban landscapes have detected local extinctions (Askins et al. 1987). However, few clear patterns are apparent in landscapes fragmented by forestry, where older forest fragments are imbedded in a matrix of younger forests and clearings (reviewed in Harris and Reed 2002).

Edge-related impacts on forest interior bird species include increased nest predation and reduced reproductive success (Askins et al. 1990; Virkkala 1991; Roth and Johnson 1993; Paton 1994; King et al. 1996; Rosenberg et al. 1999; Flaspohler et al. 2001). These factors could affect the continued presence of rare species with low local abundance (Hockin et al. 1992; Riffell et al. 1996).

Edge habitats are not all created equal. The nest success of disturbance-dependent bird species can be profoundly impacted by local land-use practices and the type of landscapes in which the edge is embedded (Harris and Reed 2002). Agricultural and abrupt, permanent edges (e.g., wildlife openings, campgrounds) had nest predation rates nearly twice as high as those of more gradual edges where plant succession had occurred (e.g., treefalls, streamsides, gaps created by selective logging) (Suarez et al. 1997). Abrupt edges may have higher prey visibility and increased predator activity.

A North America wide study found that the probability of finding cowbirds (nest parasites) increased with the degree of forest fragmentation and with lower elevation (Rosenberg et al. 1999). In general, nest parasitism rate was higher at more fragmented sites, but both of these results were somewhat variable across the wide area studied (Rosenberg et al. 1999). Other authors have also reported site-specific deviation from the general trend of elevated nest predation and parasitism rates at forest edges (Rudnicky and Hunter 1993; Yahner et al. 1993; Hahn and Hatfield 1995; Suarez et al. 1997; Tewksbury et al. 1998; Hannon 2000; Harris and Reed 2002).

A general theory emerging from the literature is that avian nest success is influenced by nest predator community composition and landscape structure on a relatively local scale (Virkkala 1991; Seitz and Zegers 1993; Yahner et al. 1993; Paton 1994; Darveau et al. 1997; Suarez et al. 1997; Tewksbury et al. 1998; Rosenberg et al. 1999; Bayne & Hobson 2000; Flaspohler et al. 2001). For instance, red squirrel (*Tamiasciurus hudsonicus*), a common nest predator, decreased with fragmentation of riparian forest in Montana (Tewksbury et al. 1998), but increased with fragmentation of boreal mixedwood forest in Saskatchewan (Bayne and Hobson 2000). Within an area, the net effect of increased fragmentation on predation rates will vary depending on prey availability, the regional abundance of forest-interior versus edge-adapted predators, the relative contribution of each group to overall predation rates, and the degree of fragmentation and types of edges found across the landscape (Culling and Anderson 2001).

4.1.3.1 Edge Widths

In coastal forest areas, microclimatic edge effects extend 100 m to 200 m into adjacent stands (BCF and BCE 1995a). Studies using artificial nests to investigate edge effects found predation and parasitism rates to be highest at the immediate edge, decreasing toward the forest interior (reviewed by Paton 1994). Biotic edge effects were most consistently found within 50 m of the forest edge (Paton 1994); research also suggests that abiotic and vegetation changes most consistently occur within 50 m of the forest edge (Paton 1994 citing several authors). Along clearcut edges in a heavily forested landscape, some species (Least Flycatcher [*Empidonax minimus*], Scarlet Tanager [*Piranga olivacea*] and Red-eyed vireo [*Vireo olivaceus*]) rarely nested within 50 m of the edge, while species commonly associated with edge and open canopy forest habitat (American Robin [*Turdus migratorius*], Rose-breasted Grosbeak [*Pheucticus ludovicianus*]) reached their highest nest abundance 50 m from the edge (Flaspohler et al. 2001).

The probability of nest survival was significantly decreased with distance to forest edge for ground-nesting birds (King et al. 1996; Flaspohler et al. 2001); canopy-nesting species were not impacted by distance to edge. However, all birds had higher nest density within 0 to 300 m than 301 to 950 m from the forest edge, suggesting that edges may be an ecological trap for sensitive species (Flaspohler et al. 2001). Flaspohler et al. (2001) suggest that the edge effect on nesting success for ground-nesting birds may extend 300 m into undisturbed forest; the effect of edge on nest density may extend further. King et al. (1996) concluded that ovenbird productivity may be reduced within a 200 m zone adjacent to small-scale clearcuts in forested landscapes. Temple and Cary (1988) report

82% nest failure in areas <100 m from the forest edge; 42% failure in areas 100 to 200 m from the edge; and 30% failure in areas >200 m from the edge respectively in 13 species studied in Wisconsin forests.

Edge ZOI detected for specific species are summarized in Table 4. The size and shape of the edge area depends on the sensitivity of the target species, the matrix surrounding the disturbance, and the intensity and duration of the activity. For example, predation rates on birds may be increased closer to the edge of clearings in agricultural areas (e.g., Sandstrom 1991), but edge effects may not occur in forested landscapes (Rudnicky and Hunter 1993). Effects of trails, pipelines, and cutlines are similar to roads, but the magnitude and ZOI of effects tend to be smaller for these narrow corridors because they are less physically disruptive and generally receive less human use (reviewed in Jalkotzy et al. 1998).

Table 4. Edge use guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Edge Use	▪ Deer used areas within 100 m of roads less than other areas, but no reduced use was evident at 600 m (Perry and Overly 1977)	▪ Based on studies in Oregon.
	▪ Elk and deer use highest within 180 m (Thomas et al. 1979).	▪ Based on studies in Oregon.
	▪ Forest bird productivity reduced within 100 to 200 m of high traffic roads (Reijnen and Foppen 1994; Reijnen et al. 1995).	▪ Based on studies in Netherlands.
	▪ Grizzly bear consistently under-use habitat within 500 m of high use roads; most grizzly bear mortality occurs within 500 m of roads and facilities and 200 m of backcountry facilities and trails (Mattson 1993; Gibeau et al. 1996; Mace et al. 1996; ESGBP, 1998).	▪ Based on work in Banff National Park, Montana, and Greater Yellowstone Ecosystem.
	▪ Boreal ecotype woodland caribou under-used areas <500 m of old wells during late winter and calving <250 m of these same features during summer; they also under-used areas <250 m of roads and <100 m of cutlines during late winter (Dyer 1999).	▪ Based on studies in north central Alberta.
	▪ Northern ecotype woodland caribou under-used areas adjacent to roads and streams but not cutlines (Oberg 2001).	▪ Based on studies in Alberta foothills adjacent to British Columbia border.
	▪ Predation rates for caribou may be higher <500 m of linear corridors (Stuart-Smith et al. 1997; James and Stuart-Smith 2000).	▪ Based on studies in north central Alberta.
	▪ Elk and moose may avoid linear features and ground-based disturbance by 100 m to 500 m (Penner and Duncan 1983; Ferguson and Keith 1982).	▪ Based on studies in Alberta.
	▪ Forest interior songbird relative abundance was reduced adjacent to paved secondary roads and powerline rights-of-way, but not unpaved trails (Rich et al. 1994).	▪ Based on study in eastern pine-oak forest.

4.1.3.2 Disturbance Buffers

While proposed or observed disturbance buffers are not thresholds in the sense used here, they are common management tools that help define the edge area (or ZOI). Some examples are summarized in Table 5.

Table 5. Edge area/disturbance buffer guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Disturbance Buffer	▪ **Identified wildlife buffers** Bull trout = 500 m on each side of stream Trumpeter swan (*Cygnus buccinator*) = 200 m (BCF and MELP 1999a; OGC 2002).	▪ Recommended British Columbia buffers.
	▪ No access within 300 m of sensitive wildlife sites (WLAP Backcountry Recreation Guidelines 2002).	▪ Recommended British Columbia buffers.
	▪ Nesting bald eagles (*Haliaeetus leucocephalus*) generally avoided habitat <400 m from a stationary boat (McGarigal et al. 1991 *in* Jalkotzy et al. 1998).	▪ Study in Pacific Northwest; range of response varied from 200 to 900 m.
	▪ **Raptor Buffers** Osprey (*Pandion haliaetus*) = 1,000 m (range 400-1,500 m) Cooper's hawk (*Accipiter cooperii*) = 525 m (range 400-600 m) Northern goshawk (*Accipiter gentiles*) = 450 m Sharp-shinned hawk (*Accipiter striatus*) = 450 m Golden Eagle (*Aquila chrysaetos*) = 800 m (range 200-1,600 m) Red-tailed Hawk (*Buteo jamaicensis*) = 800 m Ferruginous Hawk (*Buteo regalis*) = 500 m (range 200-800 m) Bald Eagle = 500 m (range 250-800 m) Prairie Falcon (*Falco mexicanus*) = 650 m (range 50-800 m) Peregrine Falcon (*Falco peregrinus*) = 800 m (range 800-1600 m) American Kestrel (*Falco sperverius*) = 50-200 m (Richardson and Miller 1997).	▪ Based on review of recommended North American disturbance buffers.
	▪ Goshawks nested 550 m from nearest house on average (range 250 to 1000 m; Tommeraas 1993 in Jalkotzy et al. 1998).	▪ Based on study in Norway.
	▪ 97% of Ferruginous hawk and 78% of Swainson's hawk (*Buteo swainsoni*) nests were >500 m from the nearest farmyard (Schmutz 1982 in Jalkotzy et al. 1998).	▪ Based on study in Alberta.
	▪ Edge effects for interior forest birds occur <300 m from clearcut edge or road (Harris and Reed 2002).	▪ Based on literature review.

4.1.4 Core Area

Remaining **core area** is a widely used habitat index that identifies the availability and location of areas with minimal human impacts. Core areas are relatively undisturbed, 'unroaded' areas that are often source areas for plant and animal populations or metapopulations.

In core areas, habitat effectiveness is enhanced for sensitive or 'interior' species such as grizzly bear, caribou, and warblers. Core area analysis is an accepted assessment technique for grizzly bear (Apps 1993; USFS 1993; Gibeau et al. 1996; Noss et al. 1996; CRC 1999) and has been adopted for other physical and ecological evaluations (Laurance and Yensen 1991; Reed et al. 1996b).

The abundance and diversity of forest-interior specialist and neotropical migrant bird populations is positively correlated with core forest area (Askins et al. 1990). Core area (defined as >100 m from the forest edge) was a better predictor of population abundance for forest interior bird species than total forest area (Temple 1986 *in* Askins et al. 1990). This was particularly true in forest sites with relatively little core area owing to their elongated or irregular shapes; predictions for these sites based on total forest area consistently overestimated abundances of interior species (Askins et al. 1990). Sites with long, narrow shapes, embedded open areas, and extensive linear corridors are less likely to be effective in preserving populations of forest-interior bird species, since they will have a higher edge to interior habitat ratio (Figure 5). The ideal configuration for forest preserves is solidly continuous and approximately circular (Askins et al. 1990; Collinge 1996).

Similar adaptations within forest-interior specialist and neotropical migrants may help explain their vulnerability to the high nest predation and parasitism rates associated with small forest patches. Compared to short-distance migrants and residents, they have a greater tendency to use open-cup versus cavity nests; to nest on or near the ground; and to have lower reproductive rates, with fewer broods/year and smaller clutch sizes (Askins et al. 1990 citing various authors). Hence, they are particularly vulnerable to predation and parasitism, and less likely to be able to compensate by renesting during the same breeding season (Askins et al. 1990).

In Canada, core areas have normally been defined to include those areas greater than a specified distance (often 500 m) from high use features (e.g., primary and secondary roads, truck trails, wellsites, petroleum and industrial facilities). A 500 m wide ZOI for all roads, wells, facilities, communities, and recreational sites is a conservative choice, since avoidance is generally related to activity levels rather than the features themselves (Mattson 1993; Dyer 1999; Gibeau 2000).

To be most effective, core areas should be larger than the minimum home range or territories of target species (Wilcove et al. 1986). Recommended and established core area thresholds are summarized in Table 6.

Table 6. Core area guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Core Area	▪ Minimize loss of core habitat (NCGBRT 2001).	▪ Recommendation for recovery of grizzly bear in the North Cascades of B.C.
	▪ No Net Loss of Core Area for grizzly bear (NCGBRT 2001).	▪ Grizzly bear management goal in the North Cascades of Washington.
	▪ >60% of available habitat as core area (Gibeau 2000).	▪ Grizzly bear management threshold for Banff National Park.
	▪ >58 to 68% of land area as core areas (NCGBRT 2001).	▪ Management goal for grizzly bear in Montana and Idaho National Forests.
	▪ >60% of Forest Planning Area as roadless core wildlife habitat (Horejsi 1996).	▪ Grizzly bear management recommendation for Yukon Territory.
	▪ Boreal-ecotype woodland caribou populations declined when core area <50%; threshold identified at <60% core area (Francis et al. 2002).	▪ Threshold based on review of Alberta population data; used 250 m buffer from all linear features.
Suitable Core Area	▪ Minimum viable core area of 450 to 1,000 ha (Gibeau et al. 1996).	▪ Grizzly bear core area used in western Canadian analyses based on 24 to 48 hr. feeding bout of an adult female grizzly.
	▪ Core area >10 ha in size, ideally >1,000 ha (NCGBRT 2001).	▪ Recommendation for recovery of grizzly bear in the North Cascades of B.C.
	▪ No vegetation change within established grizzly bear core areas for at least 11 years (USFS 1993).	▪ Management goal for Idaho National Forest.
	▪ Total weighted road density should be 0 km/km^2 in grizzly bear core areas (USFS 1993).	▪ Management goal for Idaho Nat'l Forest; Total Weighted Road density considers hiding cover, use intensity, and closure status to provide a common standard.

4.1.5 Patch and Corridor Indices

Clearing and other forms of natural and man-made disturbance introduce changes onto landscapes that affect the availability, distribution and juxtaposition of specific habitat types. These factors will influence population persistence, community composition, and ecosystem processes (Collinge 1996).

4.1.5.1 Patch Size

In forests of different sizes, the number of species and density of birds is similar, but the bird community composition is consistently different (Askins et al. 1987; Schmiegelow et al. 1997). Large tracts of forest contain greater diversity and density of forest-dwelling neotropical migrants than do small forest patches (Askins et al. 1987; Askins et al. 1990; Friesen et al. 1995; Schmiegelow et al. 1997). Dominant species in small forest patches tend to be widespread permanent residents and short-distance migrants (Askins et al. 1990). In general, the smaller the forest patch, the less effective it is as a preserve for forest interior specialists (Askins et al. 1990).

Reduced patch size may affect bird populations through some or all of the following mechanisms: (1) a decrease in the area of suitable habitat; (2) isolation of habitat patches, leading to reduced immigration/emigration and increased probability of local extinction; and (3) increased exposure to negative biotic factors affiliated with small patch size like brood parasitism and predation (Lord and Norton 1990; Boulinier et al. 1998 citing several authors; Rosenberg et al. 1999).

A short-term effect of forest fragmentation is a density increase in surrounding areas, as birds are displaced from the lost/altered habitat (Darveau et al. 1995; Hagan et al. 1996; Schmiegelow et al. 1997; Hannon 2000); this may create intense competition and attract predators, leading to high nest predation rates (Hagan et al. 1996). Bird densities will eventually decline and reductions will be faster and more extensive in smaller (Bierregaard & Lovejoy 1989 *in* Hagan et al. 1996) or narrower fragments (Darveau et al. 1995). Studies in the boreal forest found forest interior bird density declined below pre-fragmentation levels for 2 to 3 years following harvest, while the density of generalist species was unchanged (Darveau et al. 1995; Schmiegelow et al. 1997).

Reduced reproductive success may result from habitat fragmentation (Donovan et al. 1995a). Nest failure rates for three interior 'area-sensitive' species (Ovenbird [*Seirurus aurocapillus*], Wood Thrush [*Hylocichla mustelina*], and Red-eyed Vireo) were significantly higher in fragments than contiguous forests (Donovan et al. 1995a). In extensive forests, Ovenbird territorial male densities were higher (Villard et al. 1993); pairing success was higher (Porneluzi et al. 1993; Villard et al. 1993; Hagan et al. 1996); more nests were built (Hagan et al. 1996); more males successfully reproduced (Porneluzi et al. 1993); and more young were fledged (Porneluzi et al. 1993) than in fragments.

Habitat fragmentation may increase the temporal variability of communities through increased rates of local extinction and colonization (Boulinier et al. 1998). Over 22 years, larger forest patches in the eastern United States contained significantly more species and experienced less variation in the number of area-sensitive species present (Boulinier et al. 1998).

4.1.5.2 Patch Isolation and Habitat Heterogeneity

Patch isolation and habitat heterogeneity, including the presence of important microhabitats, also influence species richness in an area. Askins et al. (1987) found that large ($\geq$72 ha) eastern forests contained significantly more bird species as the amount of forest in the surrounding area increased. Clusters of wetland marshes (individually 20 to 30 ha) contained more species than large, isolated marshes (up to 180 ha), perhaps due to increased habitat heterogeneity (Brown and Dinsmore 1986). These mechanisms interact with one another; for instance, fragment size is more critical with increasing isolation (Lord and Norton 1990).

4.1.5.3 Patch and Corridor Guidelines

Patch size must be adequate to sustain enough territories for a viable breeding population (Askins et al. 1990). Many migrant songbirds are territorial so population size may be regulated by the number of nesting territories within an area (Donovan et al. 1995b citing several authors). Askins et al. (1987) reported that several bird species were restricted to forest patches significantly larger than would be expected if they were randomly distributed. These minimum patch sizes were much larger than the territory sizes of the species concerned (Whitcomb et al. 1981 *in* Askins et al. 1990; Askins et al. 1987; Collinge 1996). The Biodiversity Guidebook (BCF and BCE 1995a) specifies 600 m as the minimum width required for interior forest conditions to develop; however, this 113 ha area may substantially underestimate the needs of some sensitive forest interior species (Askins et al. 1987).

In remnant riparian vegetation corridors, species diversity and abundance was affected by corridor width. Interior forest species (e.g., golden-crowned kinglet [*Regulus satrapa*], Swainson's thrush [*Catharus ustulatus*], blackpoll warbler [*Dendroica striata*], and black-throated green warbler [*D. virens*]) were less abundant than widespread species in 20 m wide corridors; these species appeared to require riparian corridors >60 m wide (Darveau et al. 1995). In boreal mixedwood forests of Alberta, 100 m wide riparian buffers along clearcuts maintained adult movements and enhanced juvenile dispersal (Machtans et al. 1996).

Quantitative guidelines for size and geometry of habitat patches and wildlife corridors have been developed and implemented within the Bow River Valley west of Calgary, Alberta. The Bow Corridor Ecosystem Advisory Group (BCEAG 1999a) established four classes of patches and corridors derived from a number of information sources, including some long-term field studies:

1. *Primary Corridor* – multi-species corridor that will accommodate the needs of animals wary of human activities and settled areas, including large carnivores.
2. *Secondary Corridor* – multi-species corridor that will accommodate the needs of smaller wildlife species and those more tolerant of human activities and developments (e.g., elk, deer).
3. *Regional Habitat Patch* – area large enough (>1,000 ha) to contain adequate resources to support large carnivores for short periods of time; these areas are generally incorporated within a protected area designation.
4. *Local Habitat Patch* – contains adequate resources to meet the food, water, and rest needs of an animal travelling to a larger, regional habitat patch. Local habitat patches contain sufficient interior (core) habitat for an animal to feed or rest with security from human disturbance.

Table 7 and 8 summarize additional patch and corridor guidelines, respectively from field studies and literature reviews.

Table 7. Patch guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Patch	▪ Shrub or treed patches 0.8 to 2 ha optimum for deer thermal cover (Thomas et al. 1979).	▪ Based on literature review and studies in Oregon.
	▪ >4.5 ha patch area for small and tolerant species (BCEAG 1999a).	▪ Based on literature review and local studies.
	▪ 4.5 ha circular patch area optimum for elk and deer (Thomas et al. 1979).	▪ Based on literature review and studies in Oregon.
	▪ <10 ha patches have higher nest predation rate (Paton 1994).	▪ Based on literature review.
	▪ up to 24 ha undisturbed area for trumpeter swan (BCF and MELP 1999a).	▪ Recommended British Columbia core area.
	▪ >400 m patch radius for mountain-ecotype woodland caribou (Bloomfield 1979).	▪ Recommendation based on study in west central Alberta.
	▪ >40 ha of suitable habitat in patch for persistence of black-throated green warbler (Hannon 1992).	▪ Based on studies in central parkland of Alberta.
	▪ >40 ha woodlot for bald eagle roosts (Buehler et al. 1991 *in* Jalkotzy et al. 1998)	▪ Based on study in Washington State.
	▪ >100-200 ha of suitable habitat in a patch for marten (*Martes americana*; Buskirk and Ruggiero 1994).	▪ Based on literature review.
	▪ 200 ha undisturbed area for bull trout (BCF and MELP 1999a).	▪ Recommended British Columbia core area.
	▪ Optimum patch size based on protection of 20 to 25 breeding pairs (Lamberson et al. 1994).	▪ Spotted owl (*Strix occidentalis*) reserve design.
	▪ Minimum (Average) Patch Size Black-throated green warbler 187 ha (354 ha) Worm-eating warbler (*Helmitheros vermivorus*) 23 ha (477 ha) Brown creeper (*Certhia americana*) 50 ha (818 ha) Hermit thrush (*Catharus guttatus*) 323 ha (791 ha) Yellow-throated vireo (*Vireo flavifrons*) 347 ha (1366 ha) Cerulean warbler (*Dendroica cerulea*) 647 ha (1634 ha) (Askins et al. 1987)	▪ Based on studies in Connecticut mixedwood forests.
	▪ >183 ha in patch for functional breeding ovenbird population (Porneluzi et al. 1993).	▪ Based on studies in eastern Pennsylvania.
	▪ >1,000 ha patch area for carnivores and sensitive species (BCEAG 1999a).	▪ Based on literature review and local studies.

Table 8. Corridor width guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Habitat Corridor Width	▪ >200 m corridor width for moose (Nietfeld et al. 1985).	▪ Based on literature review.
	▪ >350 m corridor width for all species (BCEAG 1999a).	▪ Based on literature review; can be adjusted using prescribed formula to reflect security provided by vegetation and topography (BCEAG 1999a).
	▪ >400 m corridor width for mountain-ecotype woodland caribou (Bloomfield 1979).	▪ Recommendation based on study in west central Alberta.
	▪ Minimum Riparian Width Warbling vireo (*Vireo gilvus*) = 90 m Wood thrush = 145 m Blue-gray gnatcatcher (*Polioptila caerulea*) = 150 m Ovenbird = 175 m Scarlet Tanager = 200 m American Redstart (*Setophaga ruticilla*) = 200 m Rufous-sided Towhee (*Pipilo erythrophthalmus*) = 200 m (Stauffer and Best 1980).	▪ Based on study in Iowa.
	▪ >600 m width for interior forest habitat conditions to be present (BCF and BCE 1995a).	

4.1.5.4 Fragmentation Analyses

Fragmentation analyses are similar to land use analyses described later in Section 5, but consider the attributes of the matrix rather than the disturbance features themselves. Patch indices include: patch size, number, density, perimeter, frequency distribution, interpatch distance (McGarigal and Marks 1995), cohesion and contagion (a measure of clustering; O'Neill et al. 1988 and Schumaker 1996 *in* Gustafson 1998), structural contrast (magnitude of difference between adjacent habitats), and juxtaposition measures (i.e., percentage of area within a defined distance from patch of different habitat types).

Basic fragmentation analyses can be done within a number of GIS software packages. More sophisticated analysis can be undertaken using commercial software such as FRAGSTATS and UTOOLS, which were developed specifically to assess fragmentation (McGarigal and Marks 1995; Ager and McGaughey 1997). These programs generate a comprehensive suite of landscape and patch metrics, that can help assess specific analysis objectives (Reed et al. 1996a,b).

4.2 POPULATION INDICATORS AND THRESHOLDS

Population indicators are frequently used to manage populations of harvested species. Application of population indicators and thresholds is complicated by the inherent variability of fish and wildlife populations. Figure 11 provides a theoretical example of how population-based ecological thresholds can be developed and applied.

Human disturbances can change the population dynamics of species in the ZOI. Disturbances that change mortality, fecundity (recruitment), immigration, and/or emigration rates can affect the size, distribution, and/or viability of local fish and wildlife populations.

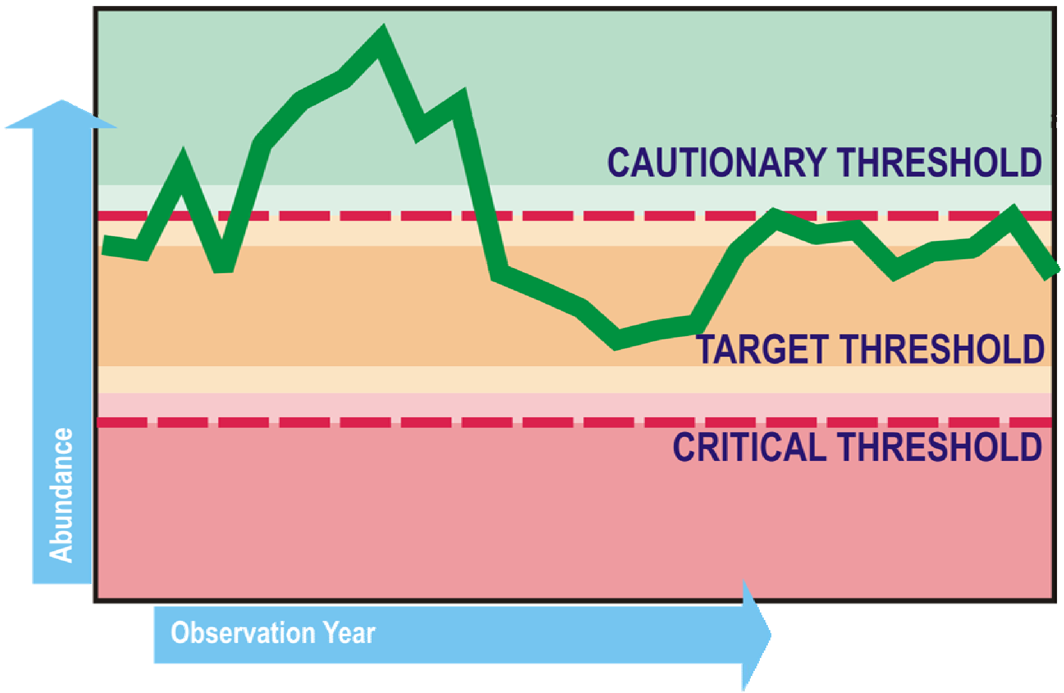

Figure 11. Tiered population thresholds.

Fish and wildlife managers conduct long-term monitoring of populations using a variety of standardized survey techniques. Field data on species and life history stages caught or observed are analyzed to determine distribution, relative abundance or density. These data provide useful information on historical and current trends and in theory, can also be used to monitor response to disturbance and habitat changes (Salmo et al. 2001). Numerical thresholds can be developed for most population parameters as summarized in Table 9.

Much early work to evaluate the ecological effects of human disturbance emphasized population indices such as abundance, population size, or density for indicator species or guilds. As noted in Section 2.2.2, population size is variable even under natural conditions, and this variability makes analyses of population parameters problematic for assessment and monitoring purposes (Karr and Chu 1997).

Table 9. Population guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Population Size	▪ Minimum viable population. ▪ Minimum population in defined area (e.g., 1,800 moose in a specified Management Unit). ▪ Average population calculated over defined interval. ▪ Maximum population in defined management unit. ▪ Minimum number of animals or redds detected during standardized field surveys (e.g., amphibian calls detected at more than 25% of survey sites). ▪ Minimum or average catch/harvest per unit effort. ▪ Species presence (e.g., designated invasive plants present on no more than 2% of area).	
Population Parameters	▪ Minimum or target cow:calf ratio. ▪ Defined age class composition. ▪ Minimum or defined sex ratio. ▪ Defined productivity (kg/ha).	

Population indicators are most useful for wildlife management at the regional scale or sport fish management in individual waterbodies. They have more limited utility for project-specific cumulative effects assessment since they require substantial supporting data and longer lead times, and are at best indirectly linked to proposed development activity. Project-specific studies generally emphasize habitat availability or quality because this can be readily quantified for both current and future scenarios.

4.3 BIODIVERSITY INDICATORS AND THRESHOLDS

Biodiversity is the diversity of plants, animals, and other living organisms in all their forms and levels of organization. This includes the diversity of genes, species, and ecosystems, as well as the evolutionary and functional processes that link them. As natural ecosystems become increasingly modified by human activities, natural patterns of biodiversity become increasingly altered and the risk of losing native species increases. The greatest degree of disruption occurs from land conversion for urban and agricultural purposes. Managed forest lands can support varying levels of biodiversity depending on management practices (BCE and BCF 1995a). The basic goal of biodiversity conservation is to maintain naturally occurring ecosystems, communities, and native species (CEQ 1993).

Suites of metrics have frequently been used as indicators of biodiversity and ecosystem integrity. Examples include the aquatic Index of Biotic Integrity (Karr 1991; Karr and Chu 1997), and Bureau of Land Management biodiversity indicators (seral stages, fragmentation, special habitats, special areas, riparian zones, species mix and hardwoods, snags, dead and down material, special status animals, special status plants; CEQ 1993).

Biodiversity analyses are most commonly used to identify and protect areas that are either representative, or rare/unique with respect to biological diversity (e.g., Kavanagh and Iacobelli nd). BCF and BCE (1995a) note that biodiversity management depends on a coordinated strategy that includes:

- a system of protected areas at the regional scale,
- maintenance of a variety of patch sizes, seral stages, and forest stand attributes at the regional and landscape scales,
- maintaining connectivity to ensure the continued dispersal and movement of animals and plants between landscapes, and
- protection of sensitive sites and features at the local scale.

In British Columbia, biodiversity objectives are related to Natural Disturbance Types (NDTs) to reflect the frequency of stand-initiating events (BCF and BCE 1995a). Examples of biodiversity management guidelines are summarized in Table 10.

Table 10. Biodiversity management guidelines.

Indicator	Guideline or Threshold	Comments
Landscape Diversity	▪ Landscape Unit Plans, including biodiversity objectives (BCF and BCE 1995a; BCF and MELP 1999b).	▪ Already applied by the forest sector in northeast British Columbia.
	▪ System of regional protected areas (BCF and BCE 1995a).	▪ Established in northeast British Columbia through LRMP process.
	▪ Old growth and wildlife tree retention BCF and BCE 1995a; BCF and MELP 1999a,b).	▪ Forest harvesting generally increases the amount of young forest and decreases the amount of older forest. The more that managed forests diverge from natural disturbance regimes, the greater the risk of loss of biodiversity (BCF and BCE 1995a).
	▪ Seral stage objectives (BCF and BCE 1995a; BCF and MELP 1999b).	
	▪ Maintain riparian areas in mid- to upper-seral successional states (CEQ 1993).	
	▪ Increase diversity of structure and age classes of specified vegetation communities (CEQ 1993).	
	▪ Define forest ecosystem networks (BCF and BCE 1995a; BCF and MELP 1999b).	▪ To maintain landscape connectivity.
Species Diversity	▪ Management of identified wildlife species (BCF and MELP 1999a).	
	▪ Defined species composition (BCF and BCE 1995a; BCF and MELP 1999a).	
	▪ Defined community structure (top-level predators maintained; CEQ 1993).	

4.4 RISK-BASED INDICATORS AND THRESHOLDS

Population viability analysis can be used to predict the probability of extirpation (loss) or decline of a population over a certain specified time period.

Viability analyses can explicitly consider the unpredictability of life history and environmental events, relative to a population's status. For instance, litter sizes for a species may range from 1 to 4 but individuals vary within that range. Stochastic population models describe events such as reproductive rate in terms of both their average value and their variance (standard deviation). Population models incorporate factors with uncertain outcomes by randomly deciding the outcome within the limits specified by the variance associated with the factor. Types of 'stochastic' variables in a population simulation model include: sex determination, survival rates, reproductive rates, dispersal rates, mate selection, effects of inbreeding, number of offspring, and catastrophes and their effects. Input parameters can also be varied to determine the effect of an activity on the probability of population decline or extirpation; land use thresholds can then be defined from the model results (Salmo et al. 2001).

This approach has been applied to forest reserve design for protected species in the western United States (e.g., Lamberson et al. 1994; Table 11). MELP (2000) has also developed a risk-based approach to environmental assessment in British Columbia that is able to incorporate cumulative effects indicators.

Table 11. Risk-based guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Population Viability	▪ Negligible risk of extinction of a viable salmonid population over a 100-year time frame (McElhany et al. 2000). ▪ High probability of survival after 250 years, considering dynamic landscape and random factors (Lamberson et al. 1994).	▪ US National Marine Fisheries Service draft guide for salmon recovery.

Risk-based indicators are generally most appropriate where species of concern are the management focus. They also require substantial supporting data and longer lead times to calculate (Dugas and Stenhouse 2000; Rohner and DeMarchi 2000; Axys 2001b; BCC 2001; Salmo et al. 2001; Harris and Reed 2002).

5. LAND USE INDICATORS AND THRESHOLDS

Over the last 30 years, and particularly during the last decade, a wide array of **land use indices** have been developed and applied to fish and wildlife management. These indices document existing and future human disturbance features such as linear corridors (e.g., roads, seismic lines, rail lines, pipelines), clearings (e.g., cutblocks, agricultural fields), facilities (e.g., mines, gas plants, borrow pits, wellsites), residential (e.g., houses, communities), and recreational sites (e.g., campgrounds, trails). A GIS summarizes information on the location and intensity of use of these features to evaluate potential effects on fish and wildlife populations and habitat.

Land-use indices provide meaningful information about existing disturbance levels, the incremental effect of proposed activities, and potential cumulative effects from existing, planned, and potential future activities (Hegmann et al. 1999; Antoniuk 2002). They are generally input-based; examples include:

- access density (km/km^2; the total length of roads or other linear corridors present in a defined area),
- total cleared or disturbed area (ha),
- core area (area greater than a specified distance from a land use feature),
- edge area (area within a specified distance of land use feature),
- stream crossing index (number of crossings/km of stream),
- valley roads (length of roads and utility corridors within 100 m of a stream), and
- riparian area cleared (ha; area cleared within 15 m of a stream or lake bank).

Land-use thresholds have been applied in the northwest United States for management of species at risk such as grizzly bear and spotted owl (e.g., Lamberson et al. 1992; Mattson 1993; Bart 1995). In Canada, similar thresholds have been applied within national parks; research is underway to establish disturbance-based thresholds for grizzly bear and boreal-ecotype caribou in Alberta (Dugas and Stenhouse 2000; BCC 2001).

5.1 HUMAN ACTIVITY

Repeated intrusions by recreational users and other groups within bird habitat can seriously alter avian behaviour, habitat use, reproduction, and survival (Riffell et al. 1996; Richardson and Miller 1997; Rodgers and Smith 1997; Gutzwiller et al. 1998). Human intrusion can uncouple foraging relations within guilds (Skagen et al. 1991), decrease song occurrence and singing consistency (Gutzwiller et al. 1994 *in* Riffell et al. 1996), alter nest height and location (Datta and Pal 1993 *in* Riffell et al. 1996), compromise nest defense (Keller 1989 *in* Riffell et al. 1996; Richardson and Miller 1997 citing several authors), and reduce hatching and fledging success (Safina and Burger *in* Riffell et al. 1996; Richardson and Miller 1997 citing several authors; Gutzwiller et al. 1998). A recreational trail may have a ZOI up to 100 m into adjacent habitat (Miller et al. 1998 *in* Hamilton and Wilson 2001). Additive and synergistic effects from multiple disturbance sources can cause reductions in fitness, even when individual disturbance types have no impact (Holmes et al. 1993).

Responses to human intrusion vary considerably between and within bird species (Tuite et al. 1984; Skagen et al. 1991; Holmes et al. 1993; Rodgers and Smith 1997; Gutzwiller et al. 1998; Jalkotzy et al. 1998). In general, birds with larger bodies, higher up the food chain, or that feed in flocks tend to be more sensitive to disturbance (Hill et al. 1997). Repeated human intrusion can potentially cause impacts that accumulate over time, eventually manifesting as progressive declines in avian richness and abundance (Riffell et al. 1996). Experimentally, repeated intrusion over 5 years in 1 ha patches within a large contiguous forest caused significant declines year-to-year in relative richness and abundance of common bird species, but no cumulative declines were detected (Riffell et al. 1996).

Most bird species are sensitive to human disturbance at their nesting sites, and frequently exhibit physiological or behavioural responses. These responses may or may not result in long-term population effects (Hill et al. 1997). Potential population-scale effects include nest desertion, reduced parental care of young, decreased feeding efficiencies, and increased offspring dispersal distances (Richardson and Miller 1997; Jalkotzy et al. 1998 citing numerous authors). A common short-term response to disturbance is temporarily leaving the nest or perch (‘flushing’) in response to: unfamiliar noises (Owens 1977; Tuite et al. 1984), pedestrian approach (Owens 1977; Tuite et al. 1984; Fraser et al. 1985; Grubb and King 1991; Skagen et al. 1991; Holmes et al. 1993; Riffell et al. 1996; Rodgers and Smith 1997; Gutzwiller et al. 1998); aircraft/ boat/ATV passage (Davis and Wiseley 1974; Owens 1977; Tuite et al. 1984; Grubb and King 1991; Rodgers and Smith 1997); and vehicular traffic (Grubb and King 1991; Holmes et al. 1993; Rodgers and Smith 1997).

Response distances vary considerably, and may be influenced by:

- body size (Holmes et al. 1993),
- degree of ‘conspicuousness’ (Gutzwiller et al. 1998),
- reproductive status (Bromley et al. 1995; Rodgers and Smith 1997),
- perch/nest height (Holmes et al. 1993; Gutzwiller et al. 1998),
- avian group size (Owens 1977; Belanger and Bedard 1989; Skagen et al. 1991; Gutzwiller et al. 1998), and
- dominant vegetation/landscape type (Rodgers and Smith 1997; Hill et al. 1997).

Human habitation appears to reduce habitat suitability for some raptor and songbird species. As the number of houses within 100 m of a forest edge increased, the diversity and abundance of neotropical migrant songbirds decreased, regardless of forest size (Friesen et al. 1995). They found that a 25-ha urban woodlot had a poorer, less abundant neotropical community than did a 4-ha woodlot without residences nearby. Both species diversity and individual abundance sharply declined with an increase from 8 to 15 residences to ≥25 residences adjacent to the forest studied (Friesen et al. 1995). Raptor nests are normally >250 m from houses (reviewed in Jalkotzy et al. 1998*).*

Human activity guidelines and thresholds and flushing distances are summarized in Table 12.

Table 12. Human activity guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Human Activity	▪ No trails within designated wildlife corridors (BCEAG 1999b). ▪ <30 km of linear corridors active in any township at any given point in time each year between December 1 and April 30 (Pedigree Caribou Standing Committee 1991). ▪ Restrictions on bus and private vehicle passage (Singer and Beattie 1986).	▪ Guideline for Bow Corridor near Canmore, Alberta. ▪ Threshold developed for woodland caribou in northwest Alberta. ▪ Traffic control in Denali National Park, Alaska.
Pedestrian Flushing Distance	▪ Mean flushing distance Golden eagle = 225 m (Holmes et al. 1993). Bald eagle = 338 m (Skagen et al. 1991); = 476 m (Fraser et al. 1985 *in* Jalkotzy et al. 1998). Rough-legged hawk (*Buteo lagopus*) = 177 m (Holmes et al. 1993). American kestrel = 44 m (Holmes et al. 1993). Merlin (*Falco columbarius*) = 76 m (Holmes et al. 1993). Great Blue heron (*Ardea herodias*) = 31 m (Rodgers and Smith 1997). American crow (*Corvus brachyrhynchos*) = 202 m (Skagen et al. 1991). Gray jay (*Perisoreus canadensis*) = 9 m (Gutzwiller et al. 1998). Mountain chickadee (*Parus gambeli*) = 4 m (Gutzwiller et al. 1998). American robin = 12 m (Gutzwiller et al. 1998). Yellow-rumped warbler (*Dendroica coronata*) = 9 m (Gutzwiller et al. 1998).	
Vehicle Flushing Distance	▪ Mean flushing distance Golden eagle = 82 m (Holmes et al. 1993). Bald eagle = 50 to 990 m (Fraser et al. 1985 *in* Jalkotzy et al. 1998). Rough-legged hawk = 71 m (Holmes et al. 1993). American kestrel = 40 m (Holmes et al. 1993). Merlin = 62 m (Holmes et al. 1993). Western sandpiper (*Calidris mauri*) = 19 m (Rodgers and Smith 1997).	
Boat Activity Distance	▪ Mean flushing distance Goldeneye (*Hiodon alosoides*) = >700 m (Hume 1976 *in* Tuite et al. 1984).	

5.2 HUMAN-CAUSED MORTALITY

Human-caused mortality results from legal and illegal harvest, trapping, management actions (e.g., problem wildlife control, defence of life and property), and vehicle collisions. Human activities and structures may also indirectly increase mortality; but detection and assessment of this mortality is difficult. For example, road construction causes limited or no direct mortality, but the road can increase mortality rates by increasing harvest effort and success. Increased mortality is a concern for species with low reproductive rates and limited ability to rebound from population declines (Ursus and Salmo 2002).

Vehicle collisions can be a significant mortality source for some species. In general, mortality increases with traffic volume (Trombulak and Frissell 2000). Often, high mortality road-kill locations are associated with substantially higher quality feeding opportunities available along the roadside (Gibeau and Heuer 1996; Lehnert et al. 1996). Annual road-kill and train-kill mortality rates for carnivores in the Banff Bow Valley were: coyotes (*Canis latrans*) - 25%, black bear (*Ursus americanus*) – 9 to 11%, cougar (*Felis concolor*) – 3 to 5%, and grizzly bear – 0% (Gibeau and Heuer 1996). Road-kill mortality is a serious concern for low-density populations (e.g., cougar) particularly because it is additive with natural mortality, management removal, and hunting mortality (Gibeau and Heuer 1996). Road-kill can have substantial demographic effects, particularly because it kills regardless of age, sex, or condition of the animal (Trombulak and Frissell 2000).

Examples of mortality guidelines are provided in Table 13.

Table 13. Mortality guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Mortality	▪ <4% grizzly bear harvest rate from all sources; females should be <33% of total kills, including estimated natural mortality, accidental kills, and illegal kills (MELP 1995).	▪ Sustainable harvest rate for British Columbia.
	▪ 40% hunter success rate with moose killed for every 15 days hunted (Harper 1988).	▪ Recommended management objective for northeast British Columbia.
	▪ <1% average annual mortality based on current population estimate (Gibeau et al. 1996).	▪ Recommendation for grizzly bear in Banff National Park.

5.3 ACCESS DENSITY

Roads and access corridors are of increasing concern for terrestrial and aquatic communities (Trombulak and Frissell 2000). Research indicates that some animals avoid, and are displaced by disturbances associated with roads. In addition, there is a tendency for trails and roads to be extended beyond their original destination, ultimately creating an access network. In many cases, it is difficult or impossible to manage these incremental impacts after the original access route is in place (Mychasiw and Hoefs 1988).

5.3.1 Observed Effects

Figure 7 illustrated the inverse relationship between access density and elk habitat effectiveness for a variety of road types. Actual terrestrial effects of linear corridors are complex and vary with species, sex, vegetative and topographic features of the landscape, prior exposure to disturbance, traffic volume and patterns, season, and hunting history (PRISM 1982; O'Neill 1993; Mace et al. 1996; Jalkotzy et al. 1998; Gibeau 2000). For instance, roads are commonly assumed to have a negative effect on moose at the local scale, at a regional scale, road density was positively associated with moose density in northern Alberta (Schneider and Wasel 2000). Some species can partly habituate to activities associated with roads, and reduce their behavioural reaction to them, but this does not occur in all cases (Mychasiw and Hoefs 1988).

Access effects on most wildlife species appear to be related to traffic volumes, out-of-vehicle activity, and predation rather than the linear feature itself. For example, roads through meadows have greater impact on elk and deer than roads through forests due to the visual screening provided by forest cover (Perry and Overly 1977). Vehicle traffic influenced caribou crossing success more than the presence of elevated pipelines and roads (Murphy 1984). Humans and stopped vehicles generally elicit a greater response than moving vehicles and other mechanical disturbances (Singer and Beattie 1986; Henson and Grant 1991; Andersen et al. 1996).

In Denali National Park, Alaska, a 50% increase in traffic volume on the sole access road reduced moose and grizzly bear sightings by 72% and 32% per trip, respectively. Caribou and Dall sheep (*Ovis dalli dalli*) sighting rates were unchanged, and relative abundance of all species was considered to be similar. The most severe responses occurred when visitors left their vehicles and approached animals (Singer and Beattie 1986).

In northern Montana, most grizzly bears exhibited a neutral or positive response to roads receiving less than 10 vehicles per day (300 per month), but avoided roads with higher traffic volumes. Grizzly bears can persist in areas with roads, but spatial avoidance increases and survival decreases as traffic levels, road densities, and human settlement increase. Average total road density was 0.6 km/km^2 in areas used by female bears as compared to 1.1 km/km^2 outside the composite home range. In this analysis, total road density included roads both open and closed to vehicle traffic, but excluded old roads reclaimed by natural vegetation or ‘in-block’ roads in timber harvest units (Mace et al. 1996). In southeast British Columbia, grizzly bears selected lower elevation riparian areas with average road density of 0.68 km/km^2 during spring and fall (McLellan and Hovey 2001). Mace et al. (1996) frequently found grizzly bears in regenerating cutblocks but these features were consistently selected less than other habitats in the Flathead valley (McLellan and Hovey 2001).

Caribou provide a good example of the variability in observed response. As noted above, caribou exposed to non-threatening behaviour in Denali Park did not appear to avoid a heavily used road (Singer and Beattie 1986). Barren ground caribou in other parts of Alaska appear to avoid roads where they are hunted (USBLM 1997). Habitat use of female barren ground caribou was altered by petroleum development during the calving and post-calving period; males and juveniles did not display the same response (Cameron et al. 1992; Nellemann and Cameron 1995; USBLM 1997). The local distribution of caribou on the calving grounds appears to be related to predators and insects (Cameron et al. 1992). In north central Alberta, boreal-ecotype caribou used areas adjacent to roads less than expected; this response was independent of the level of activity (Dyer 1999).

5.3.2 Road Density

Road density can be used as a numerical indicator of the habitat effectiveness and fragmentation associated with linear corridors. It is a useful summary index because it integrates so many ecological effects of roads and vehicles (Forman and Hersperger 1996). Relationships between access density and habitat effectiveness have been developed for some large mammals (e.g., Thomas et al. 1979, 1988; Lyon 1983,1984; Thiel 1985; Mace and Manley 1993; Reijnen and Foppen 1994; Mace et al. 1996; Jalkotzy et al. 1997; Rowland et al. 2000). Access density is commonly used for cumulative effects assessments in western Canada (e.g., CRC 1999; Alliance 1997; Hegmann et al. 1999; Kansas and Collister 1999), but the linear features considered to represent access may vary.

In aquatic evaluations, road density has been used as an indicator of land use and forestry activity and cumulative effects risk (BCF and BCE 1995b; Carver 2001). It has also has also been correlated with declines in salmonid species, including bull trout (USDA 1996; Rieman et al. 1997; Baxter et al. 1999). Roads may also increase the vulnerability of fish populations to other impacts like illegal harvest and non-native species introduction (Baxter et al. 1999; BCF 1999). The use of road density in watershed analyses is discussed in Section 5.5.

‘**Open road density**’ is an indicator that factors in the effect of road closures and revegetation that reduce actual use of roads and trails. An open road is defined as a linear feature that is passable by any type of four-wheel motorized vehicle and that is not closed to public use by gates or other methods. Formal definitions of closed roads vary; examples include roads that receive less than 5 round trips per week, and roads where use is restricted to one or two very short periods (<14 days total) during the year (Servheen 1993).

Established road and open road density effects, guidelines, and management thresholds are summarized in Table 14.

Table 14. Road density guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Road Density	▪ Road density <0.6 km/km^2 to protect high quality grizzly bear habitat (BCF and MELP 1999a).	▪ Recommended British Columbia threshold.
	▪ Density of calving barren-ground caribou highest at road density of 0 km/km^2 and declined by 86% at road densities >0.6 km/km^2; male and yearling density highest at 0.3-0.6 km/km^2 (Nellemann and Cameron 1998).	▪ Based on studies in Alaska petroleum development areas.
	▪ Road densities <0.6 km/km^2 in winter range used by northern-ecotype caribou (Salmo unpub. data).	▪ Based on studies in west central Alberta.
	▪ Road densities greater than 0.6 km/km^2 may affect habitat usage by wolves and elk (Lyon 1983, 1984; Thiel 1985; Edge and Marcum 1991; Mech 1989; Rowland et al. 2000).	▪ Effects can be mitigated by the presence of visual barriers and adjacent unroaded areas.
	▪ Areas selected by grizzly bears had average road densities of 0.6 to 0.68 km/km^2 (Mace et al. 1996; McLellan and Hovey 2001).	▪ Based on studies in northern Montana and southeast British Columbia.
	▪ Areas with road densities greater than 6 km/km^2 do not support grizzly bears (Mace et al. 1996).	▪ Based on study in northern Montana.
	▪ Forest Operating Area with road density >1.25 km/km^2 on <10% of area and additional 10% up to 0.6 km/km^2 (Horejsi 1996).	▪ Forest harvest management recommendation for grizzly bear in Yukon Territory.
	▪ <30% of Forest Operating Area with road density <0.3 km/km^2 (Horejsi 1996).	▪ Forest harvest management recommendation for grizzly bear in Yukon Territory.
	▪ Road density <1.5 km/km^2 to protect bull trout (BCF and MELP 1999a).	▪ Recommended British Columbia threshold; additional site-specific work required beyond this point.
	▪ Bull trout populations were seven times more likely to be strong in subwatersheds with road densities <1.55 km/km^2 (Rieman et al. 1997).	▪ Based on study in interior watersheds in Pacific Northwest United States.
	▪ Watershed road densities >2.5 km/km^2 increased sediment yield and affected downstream spawning habitat (Cederholm et al. 1981).	▪ Based on study in coastal watershed in Pacific Northwest United States.
	▪ 0 to 0.06 km/km^2 Very Low; 0.06 to 0.4 km/km^2 Low; 0.4 to 1.1 km/km^2 Moderate; 1.1 to 2.9 km/km^2 High; >2.9 km/km^2 Very High (Quigley et al. 1996).	▪ Rankings established for Interior Columbia basin in Pacific Northwest United States.
	▪ 0 to 0.9 km/km^2 Low; 0.9 to 1.72 km/km^2 Medium; >1.72 km/km^2 High Aquatic Hazard (BCFS and BCE 1995).	▪ Aquatic hazard rating developed for British Columbia.
Open Road Density	▪ Average open road density <0.45, <0.48, or <0.6 km/km^2 (Servheen 1993).	▪ Management objectives in 3 Yellowstone Grizzly Bear Recovery Zone National Forests.

Information provided in Table 14 demonstrates that cumulative effects risk increases along with road density. Females, especially those with young, are generally less tolerant of roads than males. Areas with few or no roads (i.e., less than 0.6 km/km^2) appear to be critical for long-term persistence of intolerant species such as woodland caribou, grizzly bear, and bull trout.

Models relating road density to terrestrial habitat effectiveness have been developed for a few species. Local circumstances have significant effects on actual species response, and models developed in other areas should be applied with caution:

- recent work to validate a commonly used elk-road model confirmed that elk avoid roads during spring and summer, but that the model overestimates the relative loss of habitat effectiveness (Rowland et al. 2000),
- differences in response of grizzly bears may reflect the amount of use that roads receive. In southeast British Columbia, high grizzly bear densities occurred in areas with higher open road densities than observed in American studies (McLellan 1990; Servheen 1993). However, these roads receive very little use except during hunting season, while use is much higher in most areas of Montana, Idaho, Wyoming, and Washington with comparable road densities. Proximity to human population centres, ease of access, and actual road use intensity are therefore important factors (Servheen 1993), and
- many different definitions of roads, trails and human use intensity have been used, and results may not be directly comparable.

5.3.3 Corridor Density

Access effects on most wildlife species appear to be related to traffic volumes, out-of-vehicle activity, and predation rather than the physical presence of the feature. There is currently scientific and public debate on whether cutlines and utility corridors (pipelines, powerlines, railway lines) have the same impact as roads. Jalkotzy et al. (1998) reviewed available literature and concluded that the effects of these features are similar to those of roads, but that the magnitude of effects is lower since their physical attributes are less disruptive (e.g., narrower and more curvilinear) and fewer people use them.

Studies in relatively unpopulated areas of the Alberta foothills indicate that human use of cutlines, utility corridors, and trails is generally lower than previously thought, and that most would be classified as ‘no-use’ features for grizzly bear habitat effectiveness purposes (Kansas and Collister 1999; Salmo unpub. data). In the Wapiti River drainage immediately east of the provincial border, 35% of cutlines were classified as unpassable because of vegetation regrowth (Salmo unpub. data).

Although no clear relationship has been demonstrated, all types of linear corridors may increase predation by providing packed travel routes for wolves, thus increasing their search efficiency and range (Bergerud et al. 1984). This may be especially significant for woodland caribou. In north central Alberta, wolf predation rates for caribou were higher in proximity to linear corridors (Stuart-Smith et al. 1997; James and Stuart-Smith 2000).

Boreal-ecotype caribou avoided cutlines in this area (Dyer 1999), but northern-ecotype caribou in west central Alberta did not (Oberg 2001).

Narrow linear corridors may act as filters or barriers to small mammals and songbirds (several authors cited by Machtans et al. 1996). Fleming and Schmiegelow (2002) observed that several bird species were unwilling to cross 15 to 16 m wide pipeline rights-of-way near Grande Prairie, although no effect on species number or abundance was detected. Desrochers and Hannon (1997) reported that gaps <30 m wide had little effect on bird movements, but gaps >70 m altered movement patterns significantly for some species. Dellasala (1986 *in* Jalkotzy et al. 1998) found that declines in density of Red-eyed vireo, a forest interior species, increased along with right-of-way width. The area affected by density declines also increased from 250 m adjacent to medium-width features, to 400 m adjacent to the widest rights-of-way.

'Corridor density' guidelines and thresholds were summarized in Table 15.

Table 15. Corridor density guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Access Corridor Width	▪ Forest interior songbird relative abundance was reduced adjacent to paved secondary roads (15 m) and powerline rights-of-way (23 m), but not unpaved trails (8 m); 10 m considered to represent the fragmentation threshold (Rich et al. 1994).	▪ Based on study in eastern pine-oak forest.
	▪ No effect on bird community structure was detected adjacent to 15 m pipeline rights-of-way (Fleming and Schmiegelow 2002).	▪ Based on study near Grande Prairie.
	▪ No effect on small mammal abundance and distribution was detected adjacent to 5 m wide trails in idle pasture or dense nesting cover; an edge effect was detected in delayed hay fields (Pasitschniak-Arts and Messier 1998).	▪ Based on study in south central Saskatchewan.
Corridor Density	▪ Boreal-ecotype woodland caribou populations declined when total corridors >1.8 km/km^2 (Francis et al. 2002).	▪ Threshold based on review of Alberta population data.
	▪ Boreal-ecotype woodland caribou populations do not persist when total corridors >3 km/km^2 (B. Stelfox pers. comm.).	▪ Threshold identified by caribou biologists in Delphi process.

5.4 CLEARED/DISTURBED AREA

Total cleared or **disturbed area** is used as a numerical index of forest habitat availability and fragmentation. This index is the inverse of the available habitat indicator discussed in Section 4.1.1.

Natural and anthropogenic disturbances introduce changes into forest landscape patterns that affect the availability, distribution and juxtaposition of specific habitat types. Studies in temperate and tropical areas have shown a positive relationship between the number of species and the area of contiguous suitable habitat, and habitat destruction is assumed to be the major cause of species extinctions (e.g., Forman and Godron 1986; Seagle 1986; Tilman et al. 1994; Flather and Sauer 1996). However, moderate levels of forest fragmentation may benefit habitat generalists with good mobility (Enns et al. 1993; Bayne and Hobson 2000).

Cleared or disturbed areas may also create barriers to movement, (Wilcove et al. 1986), create movement corridors, and affect stream flow and quality (e.g., Troendle and King 1985; Nip 1991). Movement rates of forest interior songbird species were significantly lower in clearcuts than in adjacent forests (Machtans et al. 1996). They also concluded that there appeared to be a maximum distance (unspecified) between undisturbed areas, below which birds may be more willing to cross openings such as clearcuts.

Clearing and disturbance guidelines and thresholds are summarized in Table 16.

Table 16. Clearing and disturbance guidelines and thresholds.

Indicator	Guideline or Threshold	Comments
Cleared/Disturbed Area	▪ Sediment accumulation increases when road area is >2.5% of basin area (Cederholm et al. 1981).	▪ Recommendation to prevent sediment accumulation in coastal streams.
	▪ <5% of any township cleared (Pedigree Caribou Standing Committee 1991).	▪ Disturbance threshold developed for woodland caribou in northwest Alberta.
	▪ <300 km new linear corridors in any township during a single winter (Pedigree Caribou Standing Committee 1991).	▪ Disturbance threshold developed for woodland caribou in northwest Alberta.
	▪ 5% reduction in total forest area did not cause a detectable effect on relative abundance of forest interior songbirds, considered as a group; >10% reduction did (Rich et al. 1994).	▪ Based on study in eastern pine-oak forest.
	▪ <20% of watershed cleared within 10 years (Bosch and Hewlett 1982).	▪ Literature-based threshold to prevent detectable changes in streamflow.
	▪ <30 to 33% of watershed harvested within 25 years (Chatwin 2001).	▪ Threshold established for a number of British Columbia watersheds.

Theoretical and field investigations have identified critical thresholds in the process of habitat fragmentation where rapid changes in the size and isolation of patches occur (Andren 1994; With and Crist 1995; Mönkkönen and Reunanen 1999). As habitat becomes increasingly fragmented, the number of local extinctions increases. In remnant patches, even moderate habitat loss increases the extinction risk of abundant species, although there is a 50 to 400 year lag before this is predicted to occur (Tilman et al. 1994).

5.5 WATERSHED ASSESSMENT

Cumulative effects on watersheds can result from the accumulation of the insignificant effects of small routine activities, or from changes in dominant watershed processes (Collins and Pess 1997). Studies in western North America have shown that clearings and road and trail networks created for timber harvest and resource extraction can create direct and indirect effects on flow rates, patterns, sediment yield, stream habitat, invertebrates, and fisheries (Furniss et al. 1991; McGurk and Fong 1995; Trombulak and Frissell 2000). Several models and indices have been developed to describe these effects.

Investigators in Alberta, British Columbia, and the northwest United States have developed watershed assessment techniques that use watershed indices to evaluate the potential for cumulative aquatic effects from combined land uses in a watershed. Most cumulative effects techniques consider disturbed area, potential for sediment yield, water quality, or changes in probable peak flow and channel characteristics (Klock 1985; Reid 1993; Lowrence and Vellidis 1995; Lull et al. 1995; McGurk and Fong 1995; Collins and Pess 1997; Carver 2001).

Chatwin (2001) discusses six types of watershed assessment techniques that have been applied in British Columbia:

- process models,
- empirical models,
- expert systems,
- critical thresholds,
- indicator models, and
- professional assessments.

Process and **empirical models** simulate or predict specific hydrological or aquatic habitat attributes. They are most applicable to strategic planning or research and are impractical for routine watershed assessment purposes (Chatwin 2001; Salmo et al. 2001).

Expert systems are standardized evaluations developed from the collective knowledge of a group of experts. Non-technical users input predefined parameters into a workbook or computer program, which then provides a rating of risk or sensitivity. This approach was abandoned because it did not help define why or when more detailed investigations were appropriate (Chatwin 2001).

The concept of **critical thresholds** has been discussed extensively in preceding sections. In British Columbia, thresholds have been established to define the maximum percentage of watershed that can be cut over a set period of time (Table 16). A variant, Equivalent Clearcut Area is discussed in more detail in Section 5.5.1.

The British Columbia Level 1 Interior Watershed Assessment Procedure (IWAP) is an example of an **indicator model**. This procedure used thirteen indices calculated as part of reconnaissance level analysis to examine the potential for cumulative effects due to past or planned forest harvesting (BCF and BCE 1995b). This procedure was designed to be completed by non-specialists with basic map and airphoto skills. Indices were ultimately used to generate 'hazard levels' for peak flow, erosion (sediment), riparian condition, and landslides. Rating procedures and scores were developed by experienced hydrologists working with detailed data from 20 watersheds. IWAP results from 1,400 subwatersheds were reviewed by Carver and Teti (1998, *in* Carver 2001 and Chatwin 2001). They found that the distribution of hazard scores was reasonable and conservatively segregated watersheds into no problem and possible problem groups. It was also inexpensive, consistent and repeatable (Chatwin 2001).

Unfortunately, the IWAP provisions for site-specific data collection were not consistently implemented, and a 'Revised IWAP' was developed (BCF 1999) that emphasized **professional assessments** based on field work. This procedure is considered to be more reliable because it is unique to each watershed. However, it is significantly more expensive and less consistent and repeatable (Carver 2001; Chatwin 2001).

A Landscape Unit plan that integrates watershed, riparian, and biodiversity objectives, is currently being piloted in the Okanagan watershed; this is considered to be the next logical step in watershed assessment and management (Chatwin 2001).

Individual watershed indicators have poor predictive power, especially when applied to large geographic areas with variable geological, climatic, and hydrological conditions. However, indicators appear to provide useful information about the risk of cumulative effects of watershed disturbance, and this information can be delivered consistently and quickly (Collins and Pess 1997; Carver 2001; Chatwin 2001).

IWAP indices used in the original Level 1 procedure (BCF and BCE 1995b) are summarized in Table 17; reference to their status in the 'Revised IWAP' is also included. They are described in more detail below.

Table 17. British Columbia Interior Watershed Assessment Procedure indicators and thresholds.

Indicator	Guideline or Threshold	Comments
IWAP Peak Flow Hazard	▪ Peak flow index greater than 0.18 km/km^2 indicates risk of cumulative effect. ▪ Road density greater than 0.3 km/km^2 above H_{60} line indicates risk of cumulative effect. ▪ Road density greater than 0.9 km/km^2 for entire sub-basin indicates risk of cumulative effect. ▪ % of watershed cleared.	▪ Index represents Equivalent Clearcut Area calculated by sub-basin. H_{60} weighted in Revised IWAP. ▪ H_{60} represents the upper 60% of a watershed based on area-elevation curve. Dropped from Revised IWAP. ▪ Road and trail network based on TRIM base map or Forest Cover Block maps. ▪ Added in Revised IWAP.
IWAP Erosion Hazard	▪ Riparian roads: riparian road density greater than 0.12 km/km^2 indicates risk of cumulative effect. ▪ Riparian road density greater than 0.06 km/km^2 on erodible soils indicates risk of cumulative effect. ▪ Stream crossings: stream crossings by roads greater than 0.24/km^2 indicates risk of cumulative effect. ▪ Road density greater than 0.9 km/km^2 for entire sub-basin indicates risk of cumulative effect. ▪ Road density greater than 0.06 km/km^2 on erodible soils indicates risk of cumulative effect.	▪ Riparian roads classified as those <100 m from a stream calculated by sub-basin. Dropped from Revised IWAP. ▪ Dropped from Revised IWAP. ▪ Stream crossing number calculated by sub-basin. Dropped from Revised IWAP, but used elsewhere. ▪ Erodible soils defined using available soils maps and reports or based on slope and soil attributes.
IWAP Riparian Hazard	▪ Riparian area cleared: greater than 0.09 km/km of streambank cleared indicates risk of cumulative effect. ▪ Riparian area cleared: greater than 0.15 km/km of fish-bearing streambank cleared indicates risk of cumulative effect.	▪ Riparian zone assumed to be within 100 m of streams. Dropped from Revised IWAP. ▪ Fish-bearing streams as defined in Forest Practices Code guidebooks. Replaced with S1/S2/S3/S4 streams in Revised IWAP.
IWAP Landslide Hazard	▪ Greater than 0.06 landslides/km^2 indicates risk of cumulative effect. ▪ Road density greater than 0.09 km/km^2 on unstable slopes indicates risk of cumulative effect. ▪ Riparian area cleared: greater than 0.09 km/km of streambank cleared on slopes >60% indicates risk of cumulative effect.	▪ Number of landslides calculated by sub-basin. ▪ Unstable slopes as defined in Forest Practices Code guidebooks. ▪ Riparian clearing on steep slopes calculated by sub-basin. Dropped from Revised IWAP.

5.5.1 Equivalent Clearcut Area

The Equivalent Clearcut Area index (ECA) was one of the earliest cumulative watershed effects indicators and was developed by the US Forest Service. This index was developed to evaluate the effect of forest harvest on stream channel conditions. It assumes that peak stream flows (and basin water yield) increase as a result of increased snow accumulation and reduced evapotranspiration due to forest clearing. Because most hydrologic impacts occur during periods of peak stream flow in a watershed, ECA provides an indicator of potential watershed impact. This index does not consider other sources of cumulative effect such as sediment input or persistent physical features such as landslides (e.g., Troendle and King 1985; Reid 1993; BCF 1999).

ECA calculations include all areas that have been harvested, cleared, or burned with factors applied to account for hydrological recovery due to forest regrowth, regeneration, or harvest system. In British Columbia, second growth is considered to be recovered when snowpack conditions approximate those prior to clearing (BCF and BCE 1995b).

Application of ECA models requires calibration to relate increases in water yield to vegetation type, elevation, and age of activity. Water yield values for each land type and disturbance feature are then compared to water yield values for a clearcut. The area of clearcut that would produce the same change (the 'equivalent clearcut area') is then calculated. This is used to calculate the ECA coefficients for each disturbance feature. The amount of monitoring required for full coefficient calibration is usually prohibitive, so professional judgement is often used to define ECA coefficients (Reid 1993).

In the United States, where most forestry-related watershed studies have been done, harvesting has been found to increase annual water yield by 100 to 500 mm per year; increases are inversely related to mean annual precipitation (Bosch and Hewlett 1982; Troendle and King 1985; BCF and BCE 1995b; Jones and Grant 1996; Burton 1997). Research has also shown post-clearing increases in water yield in the British Columbia interior and Alberta Rockies, but increases are less than those reported in United States studies (Nip 1991; BCF and BCE 1995b) and are highly variable (Scherer 2001). This is at least partly due to the snowfall-dominated hydrological regime in western Canada; most runoff occurs during a comparatively brief period of spring snowmelt (BCF and BCE 1995b; Scherer 2001).

Calculation of ECA requires information on the date, area, and type of clearings or disturbed areas and the length of roads, trails, and other linear corridors. The predictive power of the ECA index appears to be weak (Reid 1993; Carver 2001; Scherer 2001). However, when its limitations are acknowledged, it can help identify watersheds with an increased risk of cumulative effects. ECA hazard levels for ECA assigned in the original IWAP rating scheme (BCF and BCE 1995b) were:

- **Low**: 0 to 18% ECA,
- **Medium**: 18 to 36% ECA, and
- **High**: >36% ECA.

5.5.2 Riparian Area Cleared/Disturbed

Riparian areas include the banks and slopes next to streams, lakes and wetlands that are affected by elevated soil moisture levels for at least part of the year. These riparian areas protect water quality, stabilize stream banks, regulate stream temperature, and provide a continuous source of woody debris, nutrients, and food organisms (BCF 1999). Clearing of riparian areas can lead to increased bank erosion, altered stream channel dimensions, lowered groundwater table and summer flows, increased summer temperatures, and winter icing (Armour et al. 1994; BCF and BCE 1995b). Riparian habitat can also be altered by deliberate or inadvertent introduction of non-native or exotic vegetation that alters substrate, banks, or trophic relationships.

Riparian area conditions have been used as indicators of both cumulative land use and waterbody integrity. Riparian habitat conditions appear to influence aquatic (and terrestrial) species presence, distribution, and abundance at both local and watershed scales (e.g., Platts 1991; Waters 1995; Roth et al. 1996; Jones et al. 1999). In agricultural areas, riparian habitat conditions were found to be the best predictors of sediment-related habitat variables (Richards et al. 1996) and stream ecological integrity (Roth et al. 1996).

A frequently overlooked source of riparian habitat loss is stream crossings (Garant et al. 1997; Brown 1999). A recent study in the Prince George Forest District found that an average of 0.06 ha of riparian habitat was lost at each road crossing (Harper and Quigley 2000). In the Alberta foothills, linear corridors generally affect 2% to 5% of riparian areas, and are normally the largest source of riparian clearings (Salmo unpub. data). Forest harvest is no longer allowed in the riparian management zone in northeast British Columbia, so the incremental effects of linear corridors will continue to increase.

Hazard levels for riparian clearing assigned in the original IWAP rating scheme (BCF and BCE 1995b) were:

- **Low**: 0 to 9%,
- **Medium**: >9 to 18%, and
- **High**: >18%.

5.5.3 Road Density

Roads can directly and indirectly create all the cumulative effects described in Section 2.3 (Reid 1993; Trombulak and Frissell 2000). Road density appears to be one of the most useful watershed cumulative effects indicators for aquatic systems as well as terrestrial systems (BCF 1999; Bauer and Ralph 2001; Carver 2001). Aquatic road density guidelines and thresholds were summarized in Table 14.

On average, roads in southeast British Columbia watersheds contributed 24% of the total annual sediment yield (Henderson and Toews 2001; Jordan 2001); roads in the Alberta foothills were the largest source of sediments from human activities (Anderson and Anderson 1987). As with terrestrial habitat, the response of any particular stream is contingent to some degree on local conditions (MBTSG 1998). As a result, road density does not appear to provide reliable predictions of sediment yield (Henderson and Toews 2001).

Hazard levels for road density (calculated for entire subwatersheds) assigned in the original IWAP rating scheme (BCF and BCE 1995b) were:

- **Low**: 0 to 0.9 km/km^2,
- **Medium**: 0.9 to 1.72 km/km^2, and
- **High**: >1.72 km/km^2.

In general, observed or derived road density thresholds for fish and aquatic environments are higher than those for terrestrial species or habitat. This likely reflects the indirect link between upland roads and watercourses.

5.5.4 Equivalent Roaded Area

Equivalent roaded area (ERA) is another watershed disturbance index developed by the US Forest Service. This index assumes that peak stream flows (and basin water yield) increase as a result of compaction and reduced infiltration from road networks. By converting all land use activities to an ERA index, disturbances throughout a watershed can be considered. ERA calculations are similar to those described for ECA, but the method is customized to address issues relevant to each management area (Reid 1993; McGurk and Fong 1995).

ERA provides a screening tool to identify watershed with increased risk of cumulative effects, rather than to predict effects. ERAs are likely to be grossly correlated with many types of impacts (Reid 1993).

McGurk and Fong (1995) detected no effect on benthic communities when ERA was <5% of a subwatershed, but ERAs above this threshold were associated with a decline in macroinvertebrate density and an increase in dominance of the top five taxa.

5.5.5 Riparian Roads

The concept of roads within 100 m of a watercourse (riparian roads) was introduced in the IWAP as an index of surface erosion potential. The two most important factors in determining how much fine sediment will be delivered to watercourses from roads are the proximity of the road to the stream and the parent material the road was built on. The riparian road indicator attempts to quantify this hazard (BCF and BCE 1995b).

Many roads and trails follow valley plains and floodplains of watercourses, since the soil conditions and level terrain in these areas make construction relatively easy and economical. Roads in proximity to watercourses or those within the riparian zone can cause streambank destabilization, increased surface erosion and sedimentation, dewatered channels, obstruction to fish passage, and altered channel locations (BCF and BCE 1995b; Harper and Quigley 2000).

Hazard levels for riparian roads assigned in the original IWAP rating scheme (BCF and BCE 1995b) were:

- **Low**: 0 to 0.12 km/km^2,
- **Medium**: >0.12 to 0.25 km/km^2, and
- **High**: >0.25 km/km^2.

5.5.6 Stream Crossings

The number of road crossings of streams has been used as an indicator of land use activity for aquatic evaluations (Baxter et al. 1999; BCF 1999). This index is an easily calculated measure of sediment and mortality sources and stream habitat fragmentation in a watershed. It is expressed as the number of access corridor (road, trail, utility corridor, or cutline) crossings per kilometre of stream or watershed area. A watercourse that is repeatedly crossed is more likely to suffer increased erosion and water temperature, have higher angling pressure, and have temporary or permanent barriers to fish passage. Stream crossing indices can be calculated independently for each linear feature or similar features can be combined; an example was provided in Table 15.

Active stream crossings are often a chronic source of sediments and in-stream and riparian habitat changes. This can be either directly from the crossing construction, or indirectly from delivery of sediments along the right-of-way (Reid and Dunne 1984; BCF and BCE 1995b; Anderson 1996; Haskins and Mayhood 1997; Anderson et al.1996, 1998; Brown 1999; Reid and Anderson 1999). Road stream crossing density was positively correlated with fine substrate and embeddedness and negatively correlated with trout standing stocks in a foothills area of Wyoming (Eaglin and Hubert 1993).

Stream crossings also represent points of access for subsistence users and anglers as well as potential barriers to movement. An inventory of road crossings in the Prince George Forest District of British Columbia found that 36% of surveyed road culverts were barriers to movement (Harper and Quigley 2000). A similar study in the Alberta foothills near Edson found that 29% of surveyed road culverts were probable barriers to movement, and 40% were possible barriers (Marshall 1996).

Bull trout redd densities and counts were inversely correlated with the number of road crossings per catchment in the Swan River drainage of northern Montana (Baxter et al. 1999). As with terrestrial habitat fragmentation, actual effects depend on the extent and nature of the disturbance, watershed geology and topography, and species present, among others.

The stream crossing index provides the most direct indicator of cumulative effects erosion and mortality risk because it only includes features that intersect watercourses.

6. SPECIES-SPECIFIC INDICATORS AND THRESHOLDS

Appropriate species or guilds used for cumulative effects assessment and management are generally selected by considering the following criteria (adapted from Noss 1990):

- economic importance (e.g., **featured** species important for hunting, fishing, traditional land use, or recreation),
- sensitivity to potential development activities or early indicator of environmental stress or incremental demand on facilities and services (e.g., **ecological** and **sensitive** indicators: species that signal the effects of perturbations on a number of species with similar habitat requirements),
- importance in the food chain or ecosystem function (e.g., **keystones**: pivotal species upon which the diversity of a large part of a community depends),
- social importance (e.g., **flagships**: popular, charismatic species that serve as symbols and rallying points for conservation initiatives),
- special conservation status (e.g., **vulnerables**: species that are rare, genetically impoverished, of low fecundity, dependent on patchy or unpredictable resources, extremely variable in population density, persecuted, or otherwise vulnerable to extinction in human-dominated landscapes),
- ecological or economic significance for more than one discipline and disturbance type (e.g., **umbrellas**: species with large area requirements that incorporate many other species),
- ability to be quickly and cost-effectively calculated, estimated or assessed from existing data sources, and
- ability to be efficiently and cost-effectively monitored.

In British Columbia, the Identified Wildlife Management Strategy (BCF and MELP 1999a) was established under the Forest Practices Code to help conserve biodiversity. ‘Identified Wildlife’ are legally-designated species or plant communities that are considered to be at risk and require special management of critical habitats in order to maintain or restore populations or distributions. These critical habitats include breeding, denning, or feeding sites.

Summaries of indicators and thresholds are provided for the following focus species found in northeast British Columbia:

- bull trout (*Salvelinus confluentus*),
- grizzly bear (*Ursus arctos*), and
- woodland caribou (*Rangifer tarandus*).

6.1.1 Bull Trout

Bull trout are a char native to foothills streams of British Columbia, including the Peace and Liard River drainages. Although the species is widely distributed in the region, and is not in danger of extinction, bull trout are Blue-listed because populations are declining throughout its global range. In British Columbia, the declines are mainly due to habitat degradation, disruption of migration patterns, and overfishing (MELP 1997). Bull trout are also classified as an Identified Wildlife Species (BCF and MELP 1999a).

Spawning generally occurs in late August to late September in small streams with groundwater inflow. Three ecotypes are recognized based on movement and residence patterns. Stream **resident** populations spend their entire lives in headwater streams and overwinter in deep pools. Juveniles of **fluvial** populations remain in their natal stream or other small streams for one to five years after emergence. Fluvial adults spend most of their lives in larger rivers and migrate into smaller tributaries during summer to feed and spawn; they return downstream immediately after spawning to overwinter in deep pools and mainstem rivers such as the Peace and Fort Nelson. **Adfluvial** populations reside in lakes but use tributaries for spawning and juvenile rearing.

Bull trout are considered to be a suitable cumulative effects indicator species because of their importance for recreational fishing (flagship), they are fall-spawning predators associated with low productivity waterbodies (ecological), fluvial forms utilize a variety of habitats in different environments (umbrella), they are sensitive to overharvest, competition, and hybridization (vulnerable), and are reliant on localized spawning habitat (sensitive; Fraley et al. 1989; Weaver and Fraley 1991; Rieman and McIntyre 1993; MELP 1997; Salmo 2000).

Bull trout have lower ecological resilience than many other fish species in the region. They appear to have a narrower range of habitat preferences than other salmonids, and are uncommon where temperatures exceed 15^{o}C. Fluvial and adfluvial populations are often late maturing, and must survive for at least five years before spawning for the first time; this reduces reproductive productivity. Bull trout spawning areas are very restricted and localized; this makes them very sensitive to effects of altered groundwater flow, siltation, erosion, and removal of instream cover. Fluvial adults generally migrate long distances (up to 200 km) to spawning and poorly installed culverts or other barriers can restrict access to spawning areas. After migrating, virtually all adults in a run congregate in a single staging area (usually a pool) prior to spawning; at this time, mature adults can be easily harvested. Incubating eggs require stable, high quality water flow for a prolonged winter period. Finally, young-of-the-year and yearling bull trout may successfully overwinter in relatively small tributary streams where they often survive despite marginal discharge and apparent lack of flow. These small watercourses have high potential to be affected by human disturbance (Fraley et al. 1989; Ford et al. 1995; MELP 1997).

Cumulative effects guidelines and thresholds applicable to bull trout are summarized in Table 18. Most quantitative evaluations on this species have been conducted in the

United States near the southern limit of their range where bull trout are considered to be at risk (e.g., Rieman et al. 1997; MBTSG 1998).

Table 18. Indicators and guidelines relevant to bull trout.

Indicator	Guideline or Threshold	Comments
Disturbance Buffer	▪ Identified wildlife buffers Bull trout = 500 m on each side of stream (BCF and MELP 1999a).	▪ Recommended British Columbia buffers.
Road Density	▪ Watersheds with >1 km/km^2 road density had degraded bull trout habitat (Craig 2001).	▪ Based on studies in Yakima River basin, Washington.
	▪ Road density <1.5 km/km^2 to protect bull trout (BCF and MELP 1999a).	▪ Recommended British Columbia threshold; additional site-specific work required beyond this point.
	▪ Bull trout populations were seven times more likely to be strong in subwatersheds with road densities <1.55 km/km^2 (Rieman et al. 1997).	▪ Based on study in interior watersheds in Pacific Northwest United States.
	▪ Watershed road densities >2.5 km/km^2 increased sediment yield and affected downstream salmonid spawning habitat (Cederholm et al. 1981).	▪ Based on study in coastal watershed in Pacific Northwest United States.
	▪ 0 to 0.9 km/km^2 Low; 0.6 to 1.72 km/km^2 Medium; >1.72 km/km^2 High Aquatic Hazard (BCF and BCE 1995b).	▪ Aquatic hazard rating developed for British Columbia.
Stream Crossing Index	▪ >0.4 stream crossings by roads per km^2 indicates risk of cumulative effects (BCF and BCE 1995b).	▪ Part of IWAP procedure; calculated by subwatershed.
	▪ <0.6 stream crossings by roads per km^2 indicates risk of cumulative effects (BCF and MELP 1999a).	▪ Recommended British Columbia thresholds; additional site-specific work required beyond this point.
Substrate Index	▪ <40% median fine sediment (<6.4 mm) in substrate of bull trout spawning areas (Enk 1992; Weaver and Fraley 1991).	▪ Management threshold for Flathead River basin, Montana.

No studies exist that document in detail the linkages between land use and the biological responses of a specific bull trout population. The natural variability and complexity of abiotic and biotic factors means that a cause-effect model that relates human disturbance to bull trout populations throughout their range is unlikely to be developed. However, several comparative watershed or landscape studies have demonstrated some predictability or pattern in the status of bull trout populations relative to gross land use or watershed indicators. Although local conditions are known to be important, this likely reflects general patterns in watershed biophysical processes (Roth et al. 1996; Watson and Hillman 1997; MBTSG 1998).

A long-term study conducted in the Flathead River basin of northern Montana provides information on the effects of road construction and forest harvest on bull trout habitat and abundance. Investigators found that juvenile bull trout densities and redd counts were inversely correlated to a streambed condition index (% of sediments <6.4 mm; Shepard et al. 1984; Weaver and White 1985). These substrate indices were correlated with estimated sediment loads from road development (Enk 1984; Shepard et al. 1984; Leathe and Enk 1985) and were inversely related to embryo survival (Shepard et al. 1984; Weaver and White 1985; Weaver and Fraley 1991). Between 1982 and 1995, bull trout redd counts were inversely correlated with road density and the number of road crossings in spawning tributary watersheds (Baxter et al. 1999). Redd density appeared to be most strongly linked to road densities for the previous 7 to 12 years, suggesting a lag time of a decade or more between road construction and the full expression of its effects on bull trout spawning populations (MBTSG 1998; Baxter et al. 1999). A sediment condition index was developed for watershed management in this area (Enk 1992).

In the Columbia River basin, road density was also inversely correlated with aquatic habitat conditions, aquatic habitat integrity, known bull trout spawning areas, and bull trout presence (Henjum et al. 1994; Lee et al. 1997 *in* MBTSG 1998; Rieman et al. 1997).

6.1.2 Grizzly Bear

The grizzly bear is widely used as an umbrella species for the assessment and management of cumulative effects at regional scales (Noss et al. 1996; Kansas 2002). Grizzly bear are Blue-listed in British Columbia, classed as Special Concern by COSEWIC (2002), and are classified as an Identified Wildlife Species (BCF and MELP 1999a). Estimated grizzly bear numbers are below historic and current potential in northeast British Columbia and have been extirpated from the Peace River lowlands, local populations elsewhere in the region are not considered to be at risk (Cannings et al. 1999; Culling and Culling 2001).

Grizzly bears are considered to have low ecological resilience (Weaver et al. 1996) and display variable life history and home range sizes. Distinct seasonal habitat use periods occur during spring, summer and fall. Female home ranges are usually smaller than males and females with young appear to select isolated habitats; this is thought to minimize disturbance and encounters with mature males that kill cubs. Reproductive rate is the lowest recorded among North American land mammals. Subadult female dispersal outside maternal home ranges is rare in much of British Columbia, so areas where all resident adult females have been killed are usually not recolonized (reviewed in MELP 1995, 1997; Cannings et al. 1999; Culling and Culling 2001).

Development of cumulative effects assessment methods for grizzly bear began in the early 1980's and are now well established and generally accepted (Gibeau et al. 1996; Kansas 2002). The standardized Cumulative Effects Model (CEM; Weaver et al. 1986; USDA 1990) includes three components: habitat effectiveness (integrating habitat availability, quality, and disturbance); mortality; and connectivity (linkage zone prediction). Conventional Canadian grizzly bear cumulative effects assessment methods are described in Apps (1993), Gibeau et al. (1996), ESGBP (1998), and Kansas (2002) and will not be discussed further. Each component is applicable at different scales and for different cumulative effects pathways; they are most frequently used as a suite (Axys 2001a).

Recommended and established habitat guidelines for grizzly bear are summarized in Table 19.

6.1.2.1 Habitat Effectiveness

There is considerable evidence that grizzly bears avoid occupied and active human facilities (reviewed in Mattson 1993). The reported Zone of Influence (ZOI) varies by geographic setting, season and time of day, type of use (motorized or non-motorized; dispersed or point source), and intensity and frequency of use, among others (e.g., Archibald et al. 1987; McLellan and Shackleton 1988, 1989; Kasworm and Manley 1990; Manley and Mace 1992; Mace and Manley 1993; Mace et al. 1996; Gibeau 2000).

Habitat effectiveness models were discussed in Section 4.1.2. Grizzly bear cumulative effects assessments in western Canada have adopted a standardized 500 m ZOI (Axys 2001a), or the ZOI originally developed for the Yellowstone Ecosystem: 800 m ZOI for motorized access roads, and 400 m for non-motorized trails and corridors (e.g., Gibeau et al. 1996; ESGBP 1998).

Habitat effectiveness evaluations appear to be useful for comparing the relative amounts of effective habitat loss between subregional planning units, but the relationship between these numerical values and actual grizzly bear use or density is more tenuous. In studies west of Calgary, the lowest levels of habitat effectiveness (49% to 65%) were reported in areas where most resident grizzly bears occurred. Conversely, grizzly bears were much less common in areas with high habitat effectiveness values (76% to 82%; Kansas 2002).

Core area evaluation was discussed in Section 4.1.4; this technique was originally developed for grizzly bears

Table 19. Habitat indicators and guidelines for grizzly bear.

Indicator	Guideline or Threshold	Comments
Habitat Availability	▪ <10% of each Forest Operating Unit should be affected by logging (Horejsi 1996).	▪ Recommendation to protect grizzly bear in Yukon Territory.

Table 19. Habitat indicators and guidelines for grizzly bear (cont.).

Indicator	Guideline or Threshold	Comments
Habitat Effectiveness	▪ No net loss of habitat effectiveness for grizzly bear (BCF and MELP 1999a).	▪ Recommended British Columbia threshold.
	▪ Resident female range use appears to be severely restricted in areas with <50% habitat effectiveness (Kansas et al. 1997; Gibeau et al. 1996; ESGBP 1998; Kansas 2002).	▪ Based on studies in Banff Park and Kananaskis Country west of Calgary.
	▪ Most areas with resident female grizzly bears have habitat effectiveness >70% (Parks Canada 1997).	▪ Based on study in Jasper National Park.
	▪ >80% of all Bear Management Units with 80% or greater habitat effectiveness for grizzly bear (Parks Canada 1997).	▪ Threshold adopted by Banff National Park.
	▪ >80% habitat effectiveness for grizzly bear (Horejsi 1996).	▪ Recommendation calculated based on road density, assuming that 1 km/km^2 equates to 80% habitat effectiveness.
Edge Use	▪ Grizzly bear consistently under-use habitat within 500 m of high use roads; most grizzly bear mortality occurs within 500 m of roads and facilities and 200 m of backcountry facilities and trails (Mattson 1993; Gibeau et al. 1996; Mace et al. 1996; ESGBP 1998).	▪ Based on work in Banff National Park, Montana, and Greater Yellowstone Ecosystem.
Core Area	▪ Minimize loss of core habitat (NCGBRT 2001).	▪ Recommendation for recovery of grizzly bear in the North Cascades of B.C.
	▪ No Net Loss of Core Area for grizzly bear (NCGBRT 2001).	▪ Grizzly bear management goal in the North Cascades of Washington.
	▪ >60% of available habitat as core area (Gibeau 2000).	▪ Grizzly bear management threshold for Banff National Park.
	▪ >58 to 68% of land area as core areas (NCGBRT 2001).	▪ Management goal for grizzly bear in Montana and Idaho National Forests.
	▪ >60% of Forest Planning Area as roadless core wildlife habitat (Horejsi 1996).	▪ Grizzly bear management recommendation for Yukon Territory.
Suitable Core Area	▪ Minimum viable core area of 450 to 1,000 ha (Gibeau et al. 1996).	▪ Grizzly bear core area used in western Canadian analyses based on 24 to 48 hr. feeding bout of an adult female grizzly.
	▪ Core area >10 ha in size, ideally >1,000 ha (NCGBRT 2001).	▪ Recommendation for recovery of grizzly bear in the North Cascades of B.C.
	▪ No vegetation change within established grizzly bear core areas for at least 11 years (USFS 1993).	▪ Management goal for Idaho National Forest.
	▪ Total weighted road density should be 0 km/km^2 in grizzly bear core areas (USFS 1993).	▪ Management goal for Idaho Nat'l Forest; Total Weighted Road density considers hiding cover, use intensity, and closure status to provide a common standard.

Habitat effectiveness and core area evaluation must also be put into regional context. Effects on grizzly bears are different on forested public lands of northeastern British Columbia than in highly fragmented habitats in more densely populated areas. Human use and pressure in this region is much lower than many areas of southern British Columbia, Alberta, and the northwest United States where grizzly bears must coexist with large numbers of people and where most quantitative studies have been conducted (Servheen 1993; Culling and Culling 2001).

6.1.2.2 Road Density

Recommended and established land use guidelines for grizzly bear are summarized in Table 20. Road density standards have become an important component of grizzly bear management (Mattson 1993; Dugas and Stenhouse 2000). Road density thresholds have also been proposed for British Columbia (BCF and MELP 1999a).

In northern Montana, grizzly bear research found that most grizzlies exhibited a neutral or positive response to roads receiving less than 10 vehicles per day (300 per month), but avoided roads with higher traffic volumes. Grizzly bears can persist in areas with roads, but spatial avoidance increases and survival decreases as traffic levels, road densities, and human settlement increase. Average total road density was 0.6 km/km^2 in areas used by female bears as compared to 1.1 km/km^2 outside the composite home range. In this analysis, total road density included roads open and closed to vehicle traffic, but excluded old roads reclaimed by natural vegetation or roads in timber harvest units (Mace et al. 1996). In southeast British Columbia, grizzly bears selected lower elevation riparian areas with average road density of 0.68 km/km^2 during spring and fall (McLellan and Hovey 2001). Mace et al. (1996) frequently found grizzly bears in regenerating cutblocks but these features were consistently selected less than other habitats in the Flathead valley (McLellan and Hovey 2001).

Table 20. Land use and mortality indicators and guidelines for grizzly bear.

Indicator	Guideline or Threshold	Comments
Road Density	▪ Road density <0.6 km/km^2 to protect high quality grizzly bear habitat (BCF and MELP 1999a). ▪ Areas selected by grizzly bears had average road densities of 0.6 to 0.68 km/km^2 (Mace et al. 1996; McLellan and Hovey 2001). ▪ Areas with road densities greater than 6 km/km^2 do not support grizzly bears (Mace et al. 1996). ▪ Forest Operating Area with road density >1.25 km/km^2 on <10% of area and additional 10% up to 0.6 km/km^2 (Horejsi 1996). ▪ <30% of Forest Operating Area with road density <0.3 km/km^2 (Horejsi 1996).	▪ Recommended British Columbia threshold. ▪ Based on studies in northern Montana and southeast British Columbia. ▪ Based on study in northern Montana. ▪ Management recommendation for grizzly bear in Yukon Territory. ▪ Management recommendation for grizzly bear in Yukon Territory.

Table 20. Land use and mortality indicators and guidelines for grizzly bear (cont'd).

Open Road Density	▪ Average open road density <0.45, <0.48, or <0.6 km/km^2 (Servheen 1993).	▪ Management objectives in Yellowstone Grizzly Bear Recovery Zone National Forests.
Mortality	▪ <4% grizzly bear harvest rate from all sources; females should be <33% of total kills, including estimated natural mortality, accidental kills, and illegal kills (MELP 1995).	▪ Sustainable harvest rate for British Columbia.

Differences in response of grizzly bears may reflect the amount of use that roads receive. In southeast British Columbia, high grizzly bear densities occurred in areas with higher open road densities than observed in American studies (McLellan 1990; Servheen 1993). However, these roads receive very little use except during hunting season, while use is much higher in most areas of Montana, Idaho, Wyoming, and Washington with comparable road densities. Proximity to human population centres, ease of access, and actual road use intensity are therefore important factors (Servheen 1993). Studies in relatively unpopulated areas of the Alberta foothills indicate that human use of cutlines, utility corridors, and trails is generally lower than previously thought, and that most would be classified as 'no-use' features for grizzly bear habitat effectiveness purposes (Kansas and Collister 1999; Salmo unpub. data).

Ideally, grizzly bear road density standards should vary as a function of bear home range and human dimensions (e.g., population density and attitudes towards bears). They should reflect mortality risk more than habitat alienation, and allow the effects of vegetation cover, topography, and road closures to be considered. Studies to confirm cause-effect relationships between road density and grizzly bear populations should include an area equivalent to approximately 10 female home ranges, including areas not impacted by roads (Mattson 1993).

6.1.2.3 Mortality

Direct and indirect human-cause mortality appears to be the most significant factor affecting grizzly bear populations in western North America (Mattson 1993: ESGBP, 1998; McLellan et al. 1999).

Human-caused grizzly bear mortality is determined by the rate of encounter between humans and bears and by the probability that such an encounter will result in a bear's death. Encounter and mortality rates are affected by human population density and access patterns, human and bear behaviour during and following an encounter, grizzly bear population density and social structure, and the distribution of bear foods (reviewed in Mattson et al. 1996). Most grizzly bear mortality occurs within 500 m of roads and facilities and 200 m of backcountry facilities and trails (Mattson 1993; Gibeau et al. 1996; Mace et al. 1996; ESGBP 1998).

Because of their low productivity, grizzly bear populations respond slowly to impacts that produce a change in status. In British Columbia, maximum sustainable harvest has been set at 4% of the estimated population, including kills from all sources. The unreported kill from natural mortality and accidental and illegal kills is standardized at 50% of the total kill unless documentation indicates otherwise. Total harvest should include no greater than 33% females, and hunting seasons are not permitted in management units with fewer than 26 bears. Between 1984 and 1993, sustainable harvest rates were frequently exceeded in the province (MELP 1995; Table 20).

Human dimensions must be factored with changes in access to evaluate the effect of access management on grizzly bear mortality. Access reduction promises the greatest gains where people are unarmed visitors. In areas where people are armed and hostile towards bears, there may be no gains (or even increased mortality) by restricting access unless road closures are totally effective (Mattson et al. 1996).

6.1.2.4 Connectivity

Linkage zone prediction considers the degree of landscape fragmentation caused by human disturbance and identifies areas where grizzly bear movements are not adversely impacted. This is used to identify and protect critical movement corridors between important seasonal habitats, or to identify barriers to movement (Servheen and Sandstrom 1993; Gibeau et al. 1996). In Banff National Park, the busy Trans-Canada Highway appears to create a barrier to female grizzly bear movements. Male bear highway crossings occur more frequently, but this high use linear corridor could have profound effects on bear habitat use and movements in the Bow River valley (Gibeau and Heuer 1996).

Connectivity models are most applicable in highly fragmented landscapes or narrow valleys where movements are restricted by topographic features. They have not been tested for validity (Salmo et al. 2001).

6.1.3 Woodland Caribou

Two woodland caribou ecotypes are present in northeast British Columbia, 'northern' and 'boreal' (Heard and Vagt 1998). Most individuals of the 'northern-ecotype' spend the spring, summer and fall in subalpine spruce/balsam (*Picea*/*Populus* spp.) and lodgepole pine forests and alpine tundra. During winter, they use a combination of windswept alpine and low- to high-elevation mature to old coniferous forests where lichens are most abundant and available. Considerable individual and between-year variability in habitat use is observed. Some individuals remain in low elevation coniferous BWBS forests year-round. Deciduous and mixedwood forests appear to be avoided (Edmonds and Bloomfield 1984; Sopuck 1985; Murray 1992; Backmeyer 1994; Brown and Hobson 1998; Apps et al. 2001, Culling and Culling 2001).

'Boreal-ecotype' caribou reside in low elevation coniferous forests year-round. Treed fens and bogs appear to be important Black spruce forest and their distribution may overlap with northern-ecotype caribou during winter (Bradshaw et al. 1995; Stuart-Smith et al. 1997; Brown and Hobson 1998; Anderson et al. 1999, Rettie and Messier 2000).

Caribou have a relatively low productive rate. Females are generally 2.5 years old before first breeding, and rarely produce more than one calf per year. The most important source of adult caribou mortality is predation by wolves, followed by predation by bears, and legal and illegal hunting (Bergerud and Elliot 1998; Dyer 1999; reviewed in Culling and Culling 2001).

Relative safety from predators is assumed to be a key feature of habitat used by caribou. Woodland caribou are assumed to reduce the risk of predation by using habitat that separates them from other ungulates and existing at low densities over large ranges so that encounters with predators are minimized (Bergerud and Page 1987). A recent study in north-central Alberta found that boreal-ecotype caribou used areas adjacent to wellsites and linear corridors less than expected, presumably to reduce the risk of predation (Dyer 1999; James and Stuart-Smith 2000).

Access creation can increase hunting and predation rates by providing travel corridors that allow humans and wolves to increase their encounter rate and hunting efficiency (Bergerud et al. 1984; Cumming and Beange 1993; James 1999). In north-central Alberta, predation rates for caribou were higher in proximity to linear corridors (Stuart-Smith et al. 1997; James and Stuart-Smith 2000). In the west central Alberta foothills, however, non-forested clearings and linear corridors were least preferred by wolves (Kuzyk 2002). Clearing and access creation may alter caribou movements and distribution and also lead to increased predation and hunting pressure (Whitten et al. 1992; James and Stuart-Smith 2000; Dzus 2001).

Between 1960 and 1980, a precipitous decline of woodland caribou populations occurred in western Alberta (Bloomfield 1979; Dzus 2001), southern British Columbia (Harding and McCullum 1994), and northeast British Columbia (Harper 1988). In all areas, this decline occurred concurrently with an increase in road access and logging that is believed to have significantly increased mortality from hunters and wolves (Bergerud et al. 1984; Seip 1992; Harding and McCullum 1994). This is considered to be a good example of the cumulative effects of human development and habitat fragmentation (Harding and McCullum 1994).

Northern-ecotype caribou populations in northeast British Columbia are believed to have declined significantly in the past decade (Culling and Culling 2001). Fewer data are available for boreal-ecotype caribou, but monitored populations in Alberta have exhibited substantial declines over the last 4 to 9 years (Dzus 2001; BCC 2002).

No specific models relationships have been developed to relate access density to woodland caribou habitat effectiveness. Nellemann and Cameron (1998) found that the density of calving barren-ground caribou was inversely related to road density although non-maternal individuals did not display the same relationship (Dau and Cameron 1986). Vehicle traffic influenced barren-ground caribou crossing success more than the presence of elevated pipelines and roads (Murphy 1984).

Boreal-ecotype caribou in boreal habitat in north-central Alberta used areas near wells, roads and cutlines less frequently than expected (Dyer 1999; Table 21). Caribou in the nearby Redrock/Prairie Creek range of the west central Alberta foothills used areas within 500 m of active roads and 250 m of inactive roads and streams less frequently than expected during winter, but use of areas within 100 m of cutlines did not differ from that expected (Oberg 2001). Caribou in this range also used areas within 540 m of old cutblocks and 1.2 km of newly harvested cutblocks less frequently than expected (Smith et al. 2000). Research will be required to determine actual response of boreal- and northern-ecotype caribou in northeast British Columbia.

Table 21: Caribou reduced use buffers in north-central Alberta (Dyer 1999).

SEASON	FEATURE				
	ROADS		WELLSITES		Seismic Lines
	Open Coniferous Wetland	Closed Coniferous Wetland	New - <15.5 months old (drilling completion date)	Old - ≥15.5 months old (drilling completion date)	
Early Winter Nov 16 - Feb 21	*	*	250 m	0 m	100 m
Late Winter Feb 22 - Apr 30	250 m	250 m	250 m	500 m	250 m
Calving May 1 - Jun 30	100-250 m	0 m	1000 m	500 m	100 m
Summer July 1 - Sep 15	250 m	100 m	0 m	250 m	100 m
Rut Sep 16 - Nov 15	250 m	0 m	250 m	0 m	100 m

* Insufficient caribou had roads within their home ranges to perform analysis to examine avoidance of roads during this time period.

Current average corridor densities in the winter range of the Little Smoky boreal-ecotype herd are 0.23 km/km^2 for roads, 0.77 for truck trails, 0.15 km/km^2 for utility corridors (pipelines), and 2.90 km/km^2 for cutlines. Existing road and truck trail corridor densities are calculated to be Very Low or Low (<0.4 km/km^2) on about 35% of the Little Smoky caribou winter range and Moderate on approximately 28% of the area. High to Extremely High access densities are present on approximately 37% of the caribou winter range (Salmo unpub. data). This population is declining (Brown and Hobson 1998).

Based on information provided in Brown and Hobson (1998) and Smith et al. (2000), the main wintering areas of the nearby Redrock/Prairie Creek northern-ecotype caribou herd are located in subwatersheds with access densities between 0.35 and 0.6 km/km^2, and this herd appears to have at least temporarily moved away from areas with road and trail densities that exceed 0.6 km/km^2 (Salmo unpub. data). On this basis, an average road and trail density threshold of 0.6 km/km^2 was derived for northern- and boreal-ecotype caribou in west central Alberta (Salmo unpub. data).

Caribou in the Little Smoky herd have relatively large winter home ranges that average 14,700 ha (Edmonds and Bloomfield 1984). The most critical factor appears to be availability of suitable habitat during late winter when movements become restricted (Edmonds and Bloomfield 1984).

In the Graham River area, all but one radio-collared animal were considered to be migratory and moved more than 15 km between summer and winter ranges. Average annual home range size was 543 km^2 (range 161 to 963) and average migration distance between seasonal home ranges was 41.4 km. All caribou that wintered in low elevation BWBS forest returned to the alpine and subalpine habitats by early June (Backmeyer 1994).

No specific thresholds for core area were located for woodland caribou, however this concept was originally applied to grizzly bears and work on that species is applicable. Like caribou, female grizzly bears appear to require a portion of their home range that is secure from disturbance and mortality associated with high use human features. These core security areas are considered to be that portion of home range that corresponds to a 24 to 48 feeding bout of an adult female grizzly bear (Mattson 1993; Mace et al. 1996). Daily winter movement rates of woodland caribou in Alberta average 0.64 km/day (Stuart-Smith et al. 1997; Smith et al. 2000), which translates to a 24 to 48 hour range area of 130 to 515 ha.

Dzus (2001) concluded that the challenge for caribou conservation is to maintain sufficient quantities of suitable habitat through time in each caribou range and not unduly increase predation pressure. He recommended that cumulative effects thresholds be developed and incorporated into range management plans. The Alberta Boreal Caribou Committee guidelines also identify the need for habitat effectiveness and activity targets (BCC 2001). Preliminary analyses suggest that a habitat effectiveness threshold occurs below which caribou populations go into decline (BCC 2001); the threshold value was not reported.

Recommended and established land use guidelines for woodland caribou are summarized in Table 22.

Table 22. Indicators and guidelines for woodland caribou.

Indicator	Guideline or Threshold	Comments
Edge Use	▪ Boreal ecotype woodland caribou under-used areas <500 m of old wells during late winter and calving <250 m of these same features during summer; they also under-used areas <250 m of roads and <100 m of cutlines during late winter (Dyer 1999).	▪ Based on studies in north central Alberta.
	▪ Northern ecotype woodland caribou under-used areas adjacent to roads and streams but not cutlines (Oberg 2001).	▪ Based on studies in Alberta foothills adjacent to British Columbia border.
	▪ Predation rates for caribou may be higher <500 m of linear corridors (Stuart-Smith et al. 1997; James and Stuart-Smith 2000).	▪ Based on studies in north central Alberta.

Table 22. Indicators and guidelines for woodland caribou (cont'd).

Core Area	▪ Boreal-ecotype woodland caribou populations declined when core area <50%; threshold identified at <60% core area (Francis et al. 2002).	▪ Threshold based on review of Alberta population data; used 250 m buffer from all linear features.
Road Density	▪ Density of calving barren-ground caribou highest at road density of 0 km/km^2 and declined by 86% at road densities >0.6 km/km^2; male and yearling density highest at 0.3-0.6 km/km^2 (Nellemann and Cameron 1998). ▪ Road densities <0.6 km/km^2 in winter range used by northern-ecotype caribou (Salmo unpub. data).	▪ Based on studies in Alaska petroleum development areas. ▪ Based on studies in west central Alberta.
Corridor Density	▪ Boreal-ecotype woodland caribou populations declined when total corridors >1.8 km/km^2 (Francis et al. 2002). ▪ Boreal-ecotype woodland caribou populations do not persist when total corridors >3 km/km^2 (B. Stelfox pers. comm.).	▪ Threshold based on review of Alberta population data. ▪ Threshold identified by caribou biologists in Delphi process.

7. INDICATORS FOR NORTHEAST BRITISH COLUMBIA

In broad terms, resource development in northeast British Columbia is be conducted such that natural characteristics and wildlife habitat are maintained over time. In the Muskwa-Kechika Management Area, the management intent is to "…maintain in perpetuity its wilderness quality, and the diversity and abundance of wildlife and the ecosystems on which it depends…" (Muskwa-Kechika Management Act).

Reid (1993) reached the following conclusion on cumulative effects evaluation methods:

> *"When methods originate from management agencies, they tend to be simple, incomplete, theoretically unsound, unvalidated, implementable by field personnel, and heavily used. Methods developed by researchers are more likely to be complex, incomplete, theoretically sound, validated, require expert operators, and not used" (p. 35)*

In other areas, resource managers have concluded that a complementary suite of habitat and land-use indicators is the most practical and effective choice for cumulative effects assessment and management (Axys 2001b; BCC 2001). Land-use thresholds have been applied in Canadian national parks; research is underway to establish disturbance-based thresholds for grizzly bear and boreal-ecotype caribou in Alberta (Dugas and Stenhouse 2000; BCC 2001).

All indicators and thresholds presented here have some value for resource management. However, some commonly used indicators are not necessarily practical for cumulative effects assessment and management. Population-based indicators require substantial supporting data and longer lead times, and are at best indirectly linked to proposed development activity. Biodiversity and risk-based indicators are most appropriate in specific assessments where specific features or communities/species of concern are the management focus. Derivation of risk-based indicators may also require substantial supporting data and longer lead times (Dugas and Stenhouse 2000; Axys 2001b; BCC 2001; Salmo et al. 2001).

Indicators adopted for northeast British Columbia will be most effective when they are:

- based on management objectives identified in the Dawson Creek, Fort St. John, and Fort Nelson Land and Resource Management Plans (LRMPs),
- readily calculated, understood, and monitored,
- theoretically sound and science-based, ideally using regional data,
- protective of fish and wildlife species of management and public concern,
- compatible with existing development review and assessment processes, and
- applicable to a wide range of ecological settings and development activities.

7.1 RECOMMENDED CUMULATIVE EFFECTS INDICATORS

Based on the literature review, the complementary suite of cumulative effects indicators presented in Table 23 is most appropriate for cumulative effects assessment and management in northeast British Columbia. These generalized indicators are applicable to a wide variety of terrestrial and aquatic species, and do not require detailed site- or species-specific information to be applied.

Table 23. Generalized cumulative effects indicators for northeast British Columbia.

Habitat Indicators	Land Use Indicators
Core Area	**Access Density** (km of corridors per unit area)
Patch and Corridor Size	**Stream Crossing Index** (number of crossings per km of stream)
No Net Habitat Loss	**Activity Setbacks**

Access density is the best known and most widely applied land-use and access density indicator. This index represents the total length of roads or other linear corridors present in a defined land area or watershed. It is usually expressed as km/km^2. It is a useful summary index because it integrates so many ecological effects of roads and vehicles (Forman and Hersperger 1996).

Stream crossing index is an easily calculated measure of sediment and mortality sources and stream habitat fragmentation in a watershed. It is expressed as the number of road, utility corridor, or cutline crossings per kilometre of stream. A watercourse that is repeatedly crossed is more likely to suffer increased erosion and water temperature, have higher fishing pressure, and have temporary or permanent barriers to fish passage.

Remaining **core area** is a widely used habitat index that identifies the availability and location of areas with minimal human impacts. Core areas are relatively undisturbed, 'wilderness' areas that are often source areas for plant and animal populations or metapopulations.

Patches and **corridors** are reasonably uniform areas and linear features that differ from their surroundings. Patch and corridor criteria can help define suitable habitat in fragmented landscapes. In the habitat fragmentation process, there appear to be 'critical thresholds' where rapid changes in patch size and isolation occur; criteria should reflect these critical thresholds.

No net habitat loss is an indicator that is routinely applied to protect fisheries habitat. Although unavoidable habitat loss may be legally authorized at the regulator's discretion, it requires implementation of Enhanced Protection Measures, namely negotiation of a formal agreement to compensate for lost habitat. This requires habitat loss to be quantified and offset (DFO 1999).

While **activity setbacks (buffers)** are not indicators or thresholds in the sense described in this report, they are effective in protecting sensitive ecological sites (e.g., spawning and nesting areas), and thereby minimizing both project-specific and cumulative effects. Ideally, buffers used to calculate habitat and land use indicators should be based on established setbacks to help integrate assessment and management at all scales.

These indicators are tested in the Case Studies using detailed regional data (Appendices 2 and 3).

7.2 CUMULATIVE EFFECTS THRESHOLDS

Indicators present information about the likelihood of adverse cumulative effects, but do not directly measure the acceptability of these effects. Thresholds are objective, science-based standards used to define the point at which the indicator changes from an acceptable to unacceptable condition.

Established chemical thresholds are available to help proponents and regulators identify the point or range at which cumulative effects on air and water quality changes from an acceptable to an unacceptable condition. This allows development activities to proceed without detailed review until a defined threshold is reached. Once the threshold range is reached, however, additional review or regulation is implemented.

The British Columbia Approved Water Quality Guidelines (MELP 2001) are an example of science-based thresholds. Water quality problems are considered non-existent if the substance concentration is below the guideline value. In cases where the substance concentration exceeds its guideline, an enhanced evaluation of water quality is desirable. In some instances, local “Water Quality Objectives” may be developed to protect the most sensitive water use at a specific location, accounting for local circumstances (MELP 2001).

There is inevitably some uncertainty with science-based thresholds, and economic, social, and technical factors are normally considered when thresholds are established. Regulators may build in a safety margin by establishing a threshold below the point of irreversible effects or below the lowest point at which a behavioural, physiological, or population-level effect has been detected. In other cases, regulators may adopt a less stringent threshold that provides a lower, but still adequate level of protection at less cost to the proponent or society. Regardless, the rationale for threshold derivation should be clear and the process used to derive the threshold should be transparent.

Ecological thresholds have not been as widely applied as chemical thresholds, but thresholds based on meaningful ecological indicators can also be used to evaluate the acceptability of project-specific and cumulative effects.

7.2.1 Tiered Thresholds

The tiered threshold approach has been recommended since it provides a clear and integrated framework for derivation and implementation of ecological thresholds. With this approach, science-based and politically defined targets can be integrated with defined management actions so that operating rules are clear for all parties. As well, tiered thresholds provide the flexibility necessary for different land management regimes and ecological settings, and for a full spectrum of development proposals.

The primary strength of tiered thresholds is the formal link between predefined thresholds and management actions. Cautionary, Target, and Critical thresholds are defined to reflect increasing degrees of concern. A secondary asset is that tiered thresholds implicitly recognize the uncertainty inherent in our understanding of complex ecological relationships. In doing so, they provide a framework to gather data on actual responses and modify management actions where appropriate.

In northeast British Columbia, tiered thresholds can be directly related to objectives established in approved LRMPs, Landscape Unit Plans, or defined management areas. Risk-based thresholds would be most conservative in Protected Area and Special Management Zones (RMZs), intermediate in General Management RMZs, and most liberal in Enhanced Resource Development and Agriculture/Settlement RMZs.

To be most effective, cumulative effects thresholds must be able to deal with the full spectrum of development proposals, ecological settings, and administrative boundaries. In northeast British Columbia, project proposals are likely to undergo one of three types of regulatory review:

1. **Routine Review**: Simple or normal proposals with limited potential for significant adverse cumulative effects.
2. **Enhanced Review**: Simple or normal proposals in sensitive areas or those with some potential for significant adverse cumulative effects.
3. **Complex Review**: Complex proposals with high potential for significant cumulative effects

Table 24 summarizes the desirable indicator attributes for each review class, and provides examples of generalized, terrestrial, and aquatic indicators and thresholds.

Section 4 of the main report (Conclusions) presents candidate thresholds for the core area, patch and corridor size, access density, and stream crossing indicators.

Table 24. Desirable threshold attributes for cumulative effects assessment in northeast British Columbia.

	Routine Review *(Most Practical)*	Enhanced Review *(Intermediate)*	Detailed Review *(Most Comprehensive)*
Attributes	▪ Quickly and cheaply calculated ▪ Least data required ▪ Limited need for specialized expertise ▪ Standardized and generally applicable ▪ Easily measured and enforced ▪ Indirect measure of effects	▪ Quickly and cheaply calculated ▪ Moderate data requirements ▪ Specialized expertise generally needed ▪ Regional or RMZ-specific ▪ Measurable and enforceable	▪ Expensive and protracted to develop ▪ Comprehensive, site-specific data required ▪ Specialized expertise required ▪ Site-specific ▪ Most difficult to measure and enforce ▪ Direct measure of effects
Generalized Examples	▪ Maximum Access Density ▪ Maximum Disturbance ▪ Maximum Activity Level	▪ Maximum Access Density by Use Intensity	▪ Access Management Plan ▪ No Net Change in Carrying Capacity
Terrestrial Examples	▪ Maximum Road Density ▪ Minimum Corridor Width ▪ Minimum Patch Size	▪ Maximum Species-specific Road Density ▪ Minimum Habitat Effectiveness ▪ Maximum mortality rate	▪ Linkage Zone Modelling ▪ Minimum Viable Population Size ▪ Minimum Cow/Calf Ratio
Aquatic Examples	▪ Minimum Roadless Area ▪ Road Density Hazard Level ▪ Stream Crossing Density ▪ Riparian Road Density ▪ Riparian Clearing ▪ Maximum Surface Water Drawdown	▪ No Net Loss of Fish Habitat ▪ Road Density on Unstable Slopes or Erodible Soils ▪ Riparian Clearing on Unstable Slopes or Fish-bearing Streams	▪ Linkage Zone Modelling ▪ No Net Loss of Productive Capacity ▪ Minimum Viable Population Size

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