Coast Information Team

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The Coast Information Team is pleased to deliver the final version of the *CIT Scientific Basis of Ecosystem-Based Management*.

The Coast Information Team (CIT) was established to provide independent information for the central and north coasts of British Columbia and Haida Gwaii/Queen Charlotte Islands using the best available scientific, technical, traditional and local knowledge. The CIT was established by the Provincial Government of British Columbia, First Nations, environmental groups, the forest industry, and communities. It is led by a management committee consisting of representatives of these bodies; and is funded by the Provincial Government, the environmental groups and forest products companies, and the Federal Government of Canada. The technical team comprises nine project teams consisting of scientists, practitioners, and traditional and local experts. CIT information and analyses, which include this *CIT Scientific Basis of Ecosystem-Based Management*, are intended to assist First Nations and the three ongoing sub-regional planning processes to make decisions that will achieve ecosystem-based management (as per the April 4th 2001 Coastal First Nations—Government Protocol and the CCLRMP Interim Agreement).

In keeping with the CIT's commitment to transparency and highly credible independent analysis, the *CIT Scientific Basis of Ecosystem-Based Management* underwent an internal peer review and the CIT's independent peer review process chaired by University of Victoria Professor Rod Dobell. Peer reviews of the draft document and the authors' response are found at http://www.citbc.org/abostru-comm.html. The final document reflects changes made by the authors to address peer review comments.

We encourage all stakeholders involved in land and resource management decision-making in the CIT area to use the information and recommendations/conclusions of the *CIT Scientific Basis of Ecosystem-Based Management* in conjunction with other CIT products as they seek to implement EBM and develop EBM Land Use Plans. We are confident that the suite of CIT products provides valuable information and guidance on the key tenets of EBM: maintaining ecosystem integrity and improving human wellbeing.

Sincerely,

Robert Prescott-Allen, Executive Director
on behalf of the CIT Management Committee:
Ken Baker, Art Sterritt, Dallas Smith, Jody Holmes, Corby Lamb
Graem Wells, Gary Reay, Hans Granander, Tom Green, Bill Beldessi

Coast Information Team

The Scientific Basis of Ecosystem-Based Management

FINAL

Prepared by the Coast Information Team (CIT)
Compendium Team

March 2004

Acknowledgements

The CIT compendium team initially consisted of Rachel F. Holt, Andy MacKinnon, Jim Pojar and Karen Price (listed alphabetically), and this team was joined by Laurie Kremsater during the drafting process. Each member of the team authored some sections of the report, and reviewed others. Rachel Holt, Laurie Kremsater and Karen Price revised the first draft based on the many helpful comments provided by external reviewers.

The compendium team thanks all the authors of original reports prepared for CIT and other coastal projects whose work is summarised within. In addition to the original team, Brigitte Dorner, Pamela Dykstra, and Faisal Moola provided draft text for sections in their areas of expertise. We worked closely with the EBM Handbook team, particularly Dan Cardinall and Bill Beese and with the Hydroriparian Planning Guide team, particularly Mike Church and Nick Winfield. We thank Dave Daust, Robert Prescott-Allen, Gordon Butt, representatives of the Ministry of Forests Research Branch, Hans Granander (assisted by several forest industry representatives), Audrey Roburn, and Glenn Farenholtz for providing helpful reviews of earlier drafts.

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Introduction

This document provides the rationale and scientific background to the Coast Information Team's (CIT) approach to ecosystem-based management (EBM) as presented in the Ecosystem-based Management Handbook (EBM Handbook). It consists of two parts: a comprehensive summary of the ecological theory and data that informed development of the EBM Handbook (the "compendium"), and; a "compilation" of original sources of information prepared for the CIT or for coastal planning processes. The compilation is available on the Internet (www.citbc.org). The compendium is available on the Internet at this same website.

Together, the compendium and compilation provide

1. general theoretical background, and
2. rationales for specific ecological thresholds and guidelines given in the EBM Handbook.

The compendium has four sections:

- Section 1 defines EBM, identifying the maintenance of ecological integrity as a central focus.
- Section 2 defines ecological integrity and describes the components of ecological integrity in coastal British Columbia.
- Section 3 provides an overview to the general concepts and approach used in the EBM Handbook to maintain ecological integrity.
- Section 4 provides rationales for specific thresholds and guidelines identified in the EBM Handbook.

There is an understanding that ecosystem-based management is as much about managing people as it is about managing ecosystems (e.g., Szaro et al. 1998). However, the compendium was intended to focus primarily on identifying technical information relating to maintaining ecological values. It does not directly discuss the social science and process aspects of successful implementation of ecosystem-based management. These aspects however are being addressed by the CIT in other projects (e.g., Prescott-Allen 2004; CIT Institutional Analysis Project, in progress) and they will also be a key task for the three land-use planning tables. In this document we provide the background science relating to ecological recommendations made in the EBM Handbook, to assist tables with these decisions.

The compendium parallels, as much as possible, the EBM Handbook: Sections 1, 2, and 3 of the compendium elaborate upon the concepts discussed in Section 2 of the EBM Handbook; Section 4 of the compendium justifies the specific thresholds and guidelines provided in Sections 4, 5, and 6 of The EBM Handbook.

Each sub-section refers to relevant portions of the EBM Handbook. These pieces of text are set in boxes for quick reference. Throughout the document, concepts that are discussed in Section 3 are printed in **bold** text.

Much of the rationale provided in this document and the application of concepts to the CIT regions is based on the "best available information." A strong scientific foundation requires large-scale experiments repeated in different ecosystems: this foundation does not exist for the CIT region, or in fact for many ecosystems world-wide. In many cases, a limited number of relevant scientific studies have been published for other geographic regions. These studies, combined with expert opinion and consideration about how to apply results to the British Columbia coast, form the basis for the EBM Handbook.

Much of the information in this Compendium is summarised from a series of in-depth background reports. The intent is that questions about the background to the CIT's Ecosystem-based Management Handbook can be answered briefly, within a manageable summary document. Specific questions can then be directed to pertinent background reports that are available online (www.citbc.org). Throughout this document, each sub-section includes a list of the relevant background reports that provide more detailed information on the topic.

Annotated List of Background Reports

Below we list the background reports, describe the original audience for each report (e.g., LRMP table), list the type of report (e.g., literature review, analysis), document the described approach to studies from outside the CIT region, and briefly describe the contents of the report.

Beasley, B. and P. Wright. 2002. Criteria and indicators briefing paper.

Original audience: Background report for the North Coast LRMP table

Type of report: Framework; 30 p.

Regional focus: General

Summary: This report describes criteria and indicator frameworks used at international to local scales for planning, monitoring and demonstrating sustainable forest management and addresses the challenges in developing and applying indicators

Bunnell, F.L., G.D. Sutherland, and T.R. Wahbe. 2001. Vertebrates associated with riparian habitats on British Columbia's mainland coast.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Overview; based on author's previous literature review and analysis, but methodology not presented; 17 p.

Regional focus: CIT region

Summary: This report identifies terrestrial vertebrates in the CIT region with strong affinities to riparian habitat, summarises the habitat elements used by these species and recommends specific management activities for listed species.

Church, M. and B. Eaton. 2001. Hydrological effects of forest harvest in the Pacific Northwest.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Literature review; analysis of flood return period; 59 p.

Regional focus: Pacific Coastal Ecoregion

Summary: This report reviews literature examining the hydrological and sediment-related effects of forestry practices, interprets these effects and presents recommendations for management based on the documented experience.

Dorner, B. and C. Wong. 2003. Natural disturbance dynamics in the CIT area.

Original audience: Background report for the CIT Scientific Basis to Ecosystem-based Management

Type of report: Literature review augmented by information provided by specialists; framework for CIT region; 69 p.

Regional focus: Priority to regionally relevant papers; includes published literature from Pacific Coastal Ecoregion.

Summary: This report documents the occurrence, effects and interactions with forest management of natural disturbance agents, presents a conceptual framework for understanding natural disturbance dynamics in the CIT region, and lists strategic planning issues arising out of current scientific understanding of natural disturbance dynamics in coastal ecosystems.

Dykstra, P. (editor). 2003. Habitat supply thresholds: a literature synthesis. Draft.

Original audience: Draft report to Ministry of Sustainable Resource Management, Ecosystem Conservation Branch

Type of report: Literature review; draft

Regional focus: The paucity of regional information means that the review considers information from modeling studies and from a variety North American and European temperate and boreal forest and forest/farmland ecosystems.

Summary: This report examines the evidence for the presence of ecological thresholds, focusing on species responses to amount of habitat. It considers theoretical and empirical studies and concludes that the evidence supports the existence of thresholds that vary with features of particular organisms and landscapes.

Holt, R.F. 2001a. An ecosystem-based management planning framework for the North Coast LRMP.

Original audience: Background report for the North Coast LRMP table

Type of report: Framework; 20 p.

Regional focus: General

Summary: This report discusses definitions of ecosystem management and provides examples of inventory, analyses and questions to include in an ecosystem-based plan.

Holt, R.F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity.

Original audience: Background report for the North Coast LRMP table

Type of report: Analysis of risk to biodiversity; 50 p.

Regional focus: Specific to North Coast ecosystems

Summary: This report describes models used to assess the risk to North Coast ecosystems (timber analysis units within biogeoclimatic variants) by comparing predicted amount of old forest in each ecosystem with benchmarks based on predicted natural amounts of old forest. Risk levels are modified by representation in protected areas, conservation value and trends over time. A Bayesian Belief Network model incorporates expert opinion and uncertainty. The report highlights risk levels currently, and in future resulting from current management. It concludes that all high productivity ecosystems and most medium productivity ecosystems do, or will in the midterm show a high divergence from natural levels of old forest.

Hydroriparian Planning Guide. 2003.

Original audience: CIT

Type of report: Planning guide forming basis for hydroriparian recommendations in the Ecosystem-based Management Handbook.

Regional focus: Specific to CIT region; 87 p.

Summary: This guide provides procedures for planning forest management activities that maintain hydroriparian function. It lists low-risk precautionary guidelines and provides a risk-assessment procedure that can be used, accompanied by monitoring and adaptive management, when low-risk guidelines are exceeded. The planning guide is based on seven background reports summarising coastal ecosystems, riparian management policies and scientific literature on several topics and on specially convened expert workshops.

Pearson, A. 2003. Natural and logging disturbances in the temperate rain forests of the Central Coast, British Columbia.

Original audience: Report to the Ministry of Sustainable Resource Management

Type of report: Analysis of disturbances; 63 p.

Regional focus: Specific to part of the Central Coast

Summary: This report describes an analysis of natural and logging disturbances, based on air photo interpretation and existing databases, in 1.5 million hectares of the Central Coast. Only the abstract is available online.

Pojar, J., C. Rowan, A. MacKinnon, D. Coates, and P. LePage. 1999. Silvicultural options in the Central Coast.

Original audience: Background report for the Central Coast LRMP table

Type of report: Objective technical review of alternative harvesting methods and silvicultural systems; 102 p.

Regional focus: The report reviews alternative practices and guidelines that have been developed or applied elsewhere in British Columbia, and assesses their ecological suitability, sustainability, practicality, utility and applicability to the Central Coast.

Summary: The report distinguishes and describes three ecological subunits relevant for planning that have subsequently been used in several CIT projects. It describes disturbance regimes, silvicultural systems, harvesting systems and economics of timber production. Based on these descriptions, the report provides recommendations on how to apply alternative models and practices on the Central Coast.

Pojar, J. 2002. Rare ecosystems of the North Coast.

Original audience: Background report for the North Coast LRMP table

Type of report: List and rationale based on CDC lists and expert opinion

Regional focus: Specific to the North Coast

Summary: This report lists and describes rare ecosystems present in the North Coast. It includes those ecosystems listed by the Conservation Data Centre and other unlisted ecosystems.

Pojar, J. 2003. Biodiversity in the CIT region.

Original audience: Background report for the CIT Scientific Basis for Ecosystem-based Management

Type of report: List and rationale based on CDC lists and expert opinion; draft

Regional focus: Specific to the CIT region.

Summary: This report lists and describes rare ecosystems present in the CIT region. It includes those ecosystems listed by the Conservation Data Centre and other unlisted ecosystems.

Prescott-Allen, R. 2004. The well-being of North and Central Coastal British Columbia: a well-being assessment of British Columbia's north and central coastal zone including Haida Gwaii/Queen Charlotte Islands.

Original audience: CIT

Type of report: Description of approach and analysis; draft

Regional focus: Specific to CIT region

Summary: The well-being assessment measures environmental and socioeconomic conditions in eight coastal sub-regions of North and Central British Columbia. It provides a baseline and means of monitoring sub-regional progress toward ecosystem-based management and sustainability.

Price, K. 2003. Testing the Hydroriparian Planning Guide.

Original audience: Report for the CIT and North Coast LRMP table

Type of report: Map-based application of the Hydroriparian Planning Guide; 30 p. + maps

Regional focus: Specific to two North Coast watersheds

Summary: This report describes a map-based test of the watershed-scale planning steps described in the Hydroriparian Planning Guide. It clarifies the procedures listed in the guide and estimates the impact of applying the precautionary guidelines to timber harvest.

Price, K. and D. Daust. 2003. The frequency of stand-replacing natural disturbance in the CIT area.

Original audience: Report for the CIT

Type of report: Analysis of disturbance frequency based on existing databases; 30 p.

Regional focus: Specific to CIT region

Summary: This report provides an estimate of the frequency of stand-replacing natural disturbances in the CIT area. The analysis first divides the area into units with relatively homogeneous disturbance regimes, and then uses three methods to estimate the disturbance frequency for each unit.

Price, K. and M. Church. 2002. Risk to ecosystem functions. Summary of expert workshops.

Original audience: Report for the CIT Hydroriparian Planning Guide

Type of report: Summary of expert workshops; 21 p.

Regional focus: Some, but not all, participants were experts in coastal ecosystems. The workshops aimed to modify curves based on general published literature to be more specific to coastal ecosystems.

Summary: This report summarises workshops of experts held to consider risk for the hydroriparian functions. The results of these workshops form the basis of the hypothetical risk curves described in the Hydroriparian Planning Guide.

Price, K. and D. McLennan. 2001. Hydroriparian ecosystems of the North Coast.

Original audience: Background report for the North Coast LRMP table

Type of report: Literature review augmented by interviews with coastal experts; 80 p.

Regional focus: Priority given to regionally relevant papers (order of priority: 1. North Coast, 2. Pacific Coastal Ecoregion, 3. inland British Columbia, Washington or Oregon, 4. elsewhere). When information was not available for the North Coast, the report notes how relevant conditions in the North Coast may differ.

Summary: The report describes the variety of riparian ecosystems found in the North Coast, reviews the functions of riparian ecosystems and notes which functions are most important to which ecosystems. It briefly compares riparian management policies for British Columbia and neighbouring jurisdictions and lists potential strategic planning issues for consideration by the LRMP table.

Price, K. and D. McLennan. 2002. Impacts of forest harvesting on terrestrial riparian ecosystems of the Pacific Northwest.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Literature review and meta-analysis of impacts of harvesting based on data derived from literature; 41 p.

Regional focus: Pacific Coastal Ecoregion

Summary: This report reviews the literature on terrestrial riparian associates and their response to riparian harvesting. It describes a meta-analysis of 15 studies examining the impacts of harvesting and buffer effectiveness for amphibians, birds and small mammals using riparian areas next to small streams in the Pacific Coastal Ecoregion, and provides recommendations for management.

Rumsey, C. and many others. 2003. An ecosystem spatial analysis for Haida Gwaii, Central Coast and North Coast British Columbia.

Original audience: Report for the CIT

Type of report: Spatial analysis of conservation values and selection of sites; 180 p. + figs.

Regional focus: Specific to CIT region

Summary: This report integrates analysis of the biological values of terrestrial, freshwater and marine ecosystems across the region to identify priority areas for biodiversity conservation. It uses an automated site selection algorithm (SITES) to select portfolios of priority areas that maximise attainment of conservation goals in a compact set of sites.

Taylor, B. 2000a. An introduction to adaptive management.

Original audience: Background report for the North Coast LRMP table

Type of report: Definition and description; 12 p.

Regional focus: General

Summary: This report introduces "learning by doing" as a good option for managing in the face of uncertainty.

Taylor, B. 2000b. Implementing adaptive management through the North Coast LRMP.

Original audience: Background report for the North Coast LRMP technical team

Type of report: Framework; 25 p.

Regional focus: General

Summary: This report provides suggestions for incorporating adaptive management into the North Coast LRMP from the outset. It provides a decision tree to help decide when to invest in active adaptive management.

Trainor, K. 2001a. Geomorphological/hydrological assessment of the Central Coast plan area.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Analysis of differences in geomorphological and hydrological features of sub-regions; 21 p.

Regional focus: Specific to Central Coast

Summary: This report compares stream density and landslide frequency in different sub-regions of the Central Coast.

Trainor, K. 2001b. Ecosystem sub-units: Central Coast, North Coast and Haida Gwaii plan areas.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Synthesis of classification systems to provide consistent units for planning; 5 p.

Regional focus: Specific to CIT region

Summary: This brief report divides the CIT region into sub-regions based on existing ecological and physiographic classification systems and on previous reports.

Young, K. 2001. A review and meta-analysis of the effects of riparian zone logging on stream ecosystems in the Pacific Northwest.

Original audience: Technical report for the CIT Hydroriparian Planning Guide

Type of report: Literature review and meta-analysis of impacts of harvesting based on data derived from literature; 31 p.

Regional focus: Pacific Coastal Ecoregion

Summary: This report reviews the literature on aquatic ecosystem response and recovery from riparian logging, on ecosystem function and riparian buffer width. It then compares the general relationships derived from the literature with results of a meta-analysis of 34 studies in the Pacific Coastal Ecoregion. Sufficient data existed to allow meta-analysis of aquatic ecosystem response to logging over several decades, but not to examine the effectiveness of various buffer widths.

Section 1. Ecosystem-based Management

The following reports within the compilation provide more detailed information:

Holt, R.F. 2001a. An ecosystem-based management framework for the North Coast LRMP. Background report for the North Coast LRMP.

Prescott-Allen, R. 2004. The well-being of North and Central Coastal British Columbia: a well-being assessment of British Columbia's north and central coastal zone including Haida Gwaii/Queen Charlotte Islands**.**

1.1 Introduction

Ecosystem or ecosystem-based management, has emerged from concern that conventional resource management threatens biodiversity. The concept has existed for over 60 years, and has thoroughly permeated land management discussions within the last decade (Holt 2001a). Concerns driving ecosystem-based management have expanded from loss of biodiversity and of the intrinsic value of ecosystems, (Ludwig et al. 1993; Hilborn et al. 1995; Holling and Meffe 1995), to concerns that conventional practices also limit social and economic options for future generations (Grumbine 1994; Brussard et al. 1998).

1.2 Definitions of Ecosystem-based Management

A literature review quickly identifies the lack of a single, generally accepted definition for ecosystem-based management (Grumbine 1997; Holt 2001a). Some variations offer minor refinements; others reveal significant differences in theme. Almost all ecosystem-based management definitions include maintenance of ecological integrity (or healthy ecosystems) and continued human presence and use. Almost all recognise that humans are part of ecosystems. Definitions then diverge into those that give precedence to maintaining ecological integrity and those that do not (Table 1). This apparent dichotomy forms the crux of the conflict among many of the proposed definitions (Grumbine 1994; Stanley 1994; Wilcove and Blair 1995).

The first four definitions in Table 1, crafted during planning processes, identify the need to maintain ecological integrity and to provide desired products. They do not, however, mention ecological limits nor recognise that maintaining healthy ecosystems and providing desired products may be mutually exclusive (see reviews of the links between human activities and endangered species in Stein et al. 2000 and Holt et al. 2003). The vague wording does not provide guidance about decision-making when the two goals are incompatible in either the short-term or the long term. These definitions therefore fail to address the principal concerns that the initial development of EBM was intended to address (Noss 1999).

The remaining definitions, primarily academic, give precedence to ecosystem integrity and provide clearer language about the interaction between humans and ecosystems. Most relevant to the B.C. context is that of the Clayoquot Scientific Panel (1995). Their definition includes a list of principles including explicit recognition of ecosystem limits and the precedence of ecosystem integrity over the output of products or services.

Table 1. Selected EBM definitions developed through land use processes and by academics

EBM definition*	**Source**	**Ecosystem precedence**
*…the collaborative process of **sustaining the integrity of ecosystems** through partnerships and interdisciplinary teamwork. The long-term goal is **sustainability of** [Minnesota's] **ecosystems**, the **people** who live in them, and the **economies** founded on them.*	Minnesota Dept. Natural Resources	Unclear
*a land management approach that considers the biological needs of a large area of land; management for the **health of the whole ecosystem** by providing for the preservation and restoration of plants, animals, streams, forests, soil, and wetlands; management for the preservation and restoration of the biological elements while also **providing for things important to people**, such as food and fibre, crops and recreation. Ecosystem management is the skillful, integrated use of ecological knowledge at various scales to **produce desired resource values, products, services**, and conditions in ways that also sustain the diversity and productivity of ecosystems...*	Illinois Landowners Association	Unclear
*Ecosystem management is a goal-driven approach to restoring and **sustaining healthy ecosystems** and their functions and values using the best science available. It entails working collaboratively with state, tribal and local governments, community groups, private landowners, and other interested parties to develop a vision of **desired future ecosystem conditions**. This vision integrates ecological, economic, and social factors affecting the management unit defined by ecological, not political boundaries. The goal is to restore and **maintain the health of ecosystems while supporting economies and communities**.*	The Interagency Ecosystem Management Task Force (1994 – Clinton Administration)	Unclear
*Ecosystem management is a concept of natural resources management wherein national **forest activities are considered within the context of economic, ecological, and social interactions** within a defined area or region over both short and long term*	Thomas and Huke (1996): USDA FS	No
*...is management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to **sustain ecosystem composition, structure, and function**. Ecosystem management includes sustainability, goals, sound ecological models and understanding, complexity and connectedness, the dynamic character of ecosystems, context and scale, **humans as ecosystem components**, adaptability, and accountability.*	Ecological Society of America: Christenson et al. (1996)	Yes
*...is integration of **ecological, economic, and social principles** to manage biological and physical systems in a manner that **safeguards the ecological sustainability**, natural diversity, and productivity of the landscape*	Wood (1994)	Yes
*...managing areas at various scales in such a way that ecosystem services and biological resources are preserved while appropriate human uses and options for livelihood are sustained... appropriateness and **sustainability of human uses are dictated by the constraints** imposed by the biological and physical environment.*	Brussard et al. (1998)	Yes
*…forest practices that **maintain the integrity of watersheds and the diversity of natural ecosystems**, while providing for the needs of current and future generations.*	Clayoquot Sound Scientific Panel (1995)	Yes

*Our emphasis in **bold**.

A decade ago, in a review of existing literature, Grumbine summarised the key elements commonly included in discussions of ecosystem-based management and proposed a working definition:

> *"Ecosystem management integrates scientific knowledge of ecological relationships within a complex socio-political and values framework toward the general goal of protecting native ecosystem integrity over the long term".* (Grumbine 1994)

This definition, largely agreed to by other authors since (Jensen and Bourgeron 1994; Christensen et al. 1996; Haynes et al. 1996; Noss 1999), clearly focuses on protecting ecosystem integrity.

Grumbine (1997) reflected on his 1994 review some years later, in light of an exponential increase in literature, and following feedback from managers and practitioners. Although he concluded that the key elements, particularly the need to maintain ecological integrity, remained unchanged, he acknowledged that confusion over the essence of ecosystem-based management persisted. On reflection, he concluded that the dichotomy in definitions (some acknowledging ecosystem precedence, others not) was an "artefact of narrow problem definition, lack of contextual thinking, and our propensity to separate ourselves from nature. Over time there is simply **no way to sustain humans without sustaining ecosystems.**"[1]

The Ecological Society of America Committee of Scientists (a multidisciplinary committee including representatives from landscape ecology, silviculture, fire ecology, wildlife biology, economics, sociology, planning, environmental law, range management, and fisheries) also concluded that ecological sustainability is the fundamental basis for economic and social sustainability and must set bounds for economic and social goals (Dale et al. 1999).

Grumbine and other biophysical scientists identified the need to include social science and values into ecosystem-based management decision-making, however they have been criticised for not including social data more explicitly (e.g., Endter-Wada et al. 1998). Oddly, while decrying the biophysical scientists for apparently separating humans and ecosystems, Endter-Wada et al. (1998) appear to fall into a similar trap by failing to acknowledge that human sustainability is ultimately linked to ecological integrity.

Other social scientists have suggested that rigidly prioritising ecological integrity over human systems does not acknowledge the real-world way that decisions are made (e.g., Francis 1993). In a review of lessons to be learned from practical examples of EBM implementation, the importance of managing people is highlighted in similar fashion to that of managing ecosystems (e.g., Slocombe 1993; Slocombe 1998a). The essence of this argument appears not to fundamentally disagree with that from biophysical scientists, but stresses that without putting equal effort into the social aspects of ecosystem-based management, the ecological aspects are doomed to failure. Understanding how humans interact with their environment in this way has been termed taking a "sociobiophysical systems" view (Slocombe 1993). We note that the CIT Handbook and Compendium are primarily focused on ecological science; however the broader CIT process and land use planning tables have focused extensive effort on understanding and incorporating people and how they interact with ecosystems.

One key element that we have found poorly discussed in the literature is the aspect of timeframe. Ecosystem-based management should theoretically result in improved environmental and social conditions *over the long term*, and it is often in the short-term that direct conflicts arise. One key

[1] Our emphasis.

implementation problem to overcome is that economics and political decision-making both traditionally focus on the short-term, while true sustainability requires a long-term vision.

THE CIT DEFINITION OF ECOSYSTEM-BASED MANAGEMENT

A group of scientists (various disciplines) and stakeholders (industry, government, ENGOs) agreed on the definition included in the EBM Handbook:

An adaptive approach to managing human activities that seeks to ensure the coexistence of healthy, fully functioning ecosystems and human communities.

The intent is to maintain those spatial and temporal characteristics of ecosystems such that component species and ecological processes can be sustained, and human well-being supported and improved. (Handbook Section 2.1)

The group also specified the following principles for EBM:

- Maintain ecological integrity
- Recognize and accommodate Aboriginal rights, title and interests
- Promote human well-being
- Sustain cultures, communities and economies within the context of healthy ecosystems
- Apply the precautionary principle
- Ensure planning and management is collaborative
- Distribute benefits fairly

The EBM Handbook definition has elements of many of the definitions listed in Table 1. It acknowledges the importance of ecological and social values, but does not require provision of all desired products. Rather, it calls for promotion of human well-being within the context of healthy ecosystems. The definition itself is unclear about the relationship between ecosystems and humans, although the intent statement implies a recognition of ecosystem precedence and some of the listed principles further imply ecosystem precedence. The compendium focuses on describing the approach to sustaining ecosystems assuming some sort of management will occur in the CIT region. It provides a framework within which social and economic concerns and benefits are pursued at low risk to the overall ecosystem. The CIT approach supports some level of social planning by including and analysing real social data (Prescott-Allen 2004) and in so doing the CIT approach tackles a key criticism of other work (Endter-Wada et al. 1998). Additionally, specific barriers to implementation are being separately identified in additional product (CIT Institutional Analysis Project ongoing). Although the issue of timeframe is not explicitly addressed in the EBM Framework, some of the issues associated with implementation and transition are highlighted in the EBM Handbook.

The question that continues to be key is what constitutes "sustaining ecosystems"? Maintaining what levels of species and processes, and by what means, will allow persistence of ecosystem integrity that in turn will support social and economic systems?

This question will be addressed throughout the remainder of the Compendium.

Section 2. The Ecological Integrity of the CIT Region

2.1 Introduction

Maintaining ecological integrity is a central focus of ecosystem-based management (Compendium Section 1).

In the EBM Handbook, maintaining ecological integrity is both a guiding principle (Handbook Section 2.2) and goal (Handbook Section 2.3).

This section briefly defines ecological integrity and then provides an overview of the natural elements and processes of the CIT region. Section 3 describes the conservation approaches and concepts that can be used to maintain this ecological integrity and elaborates on natural elements and processes of the CIT region where appropriate.

2.2 Definition

Definitions of ecological integrity typically compare an area's organisms, communities, ecological functions, and processes with those found naturally in the region. Franklin (2000) uses a simple, yet comprehensive expression of these ideas, defining ecological integrity as

> a system's wholeness, including presence of all appropriate elements and occurrences of all processes at appropriate rates

Other definitions include

> The capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition and functional organization comparable to that of the natural habitat of the region. (Karr and Dudley 1981; Angermeier and Karr 1994)
>
> An ecosystem has integrity when it is deemed characteristic for its natural region, including the composition and abundance of native species and biological communities, rates of change and supporting processes. (Parks Canada Panel 2000)
>
> Functioning, self-sustaining systems undergoing no systematic changes as the result of unnatural (i.e., human-induced) manipulations. (Clayoquot Sound Scientific Panel 1995)

The EBM Handbook defines ecological integrity as

the abundance and diversity of organisms at all levels, and the ecological patterns, processes, and structural attributes responsible for that biological diversity and for ecosystem resilience

This definition does not explicitly compare the diversity and processes with those found naturally, but otherwise is consistent with other definitions. However, implicit in the Handbook is the understanding that the reference point against which ecological integrity is measured is the ecosystem as defined by natural disturbance processes.

Ecological integrity, ecosystem health and biodiversity are closely related, fuzzy terms with sometimes overlapping definitions. While "biodiversity" generally focuses on the variety of life and their interactions (e.g., UNEP 1992), "ecological integrity" or "ecosystem health" also explicitly considers the ecological and evolutionary processes that are responsible for that biodiversity (Noss 1999). It has been pointed out that "ecosystem health" is a useful but not comprehensive construct, and that in fact ecological integrity is a more useful term (Slocombe 1998b) in the management context. Key aspects of ecological integrity include maintaining normal functions under varied conditions, resilience to stress and continued self-organisation (Kay and Schneider from Slocombe 1998b). All three terms require operational definitions to make them useful to managers. For example, "normal function," "resilience," and similar components of ecosystem integrity are often captured operationally by looking at surrogates that can actually be measured such as: amount of unmanaged forest, amount of structural retention, amount of disturbed soil, amount of buffered stream length and other such indicators.

2.3 Ecological Integrity in the CIT Region

The following reports within the compilation provide more detailed information:

Dorner, B. and C. Wong. 2003. Natural disturbance dynamics in the CIT area. Report to the CIT.

Holt, R.F. 2001a. An ecosystem-based management framework for the North Coast LRMP. Background report for the North Coast LRMP.

Hydroriparian Planning Guide 2003. Report to the CIT.

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Trainor, K. 2001a Geomorphological/hydrological assessment of the Central Coast plan area. Hydroriparian Planning Guide Technical Report #1.

Trainor, K. 2001b Ecosystem sub-units: Central Coast, North Coast and Haida Gwaii plan areas. Hydroriparian Planning Guide Technical Report #2.

Under the goal of maintaining ecological integrity, the EBM Handbook lists objectives of maintaining the natural diversity of species, genes, and habitat elements across scales and through maintaining ecosystem functions and processes (e.g., stream flow, water quality, soil productivity, natural disturbance rates, and patterns) across scales and through the long term, and restoring damaged, degraded or under-represented ecosystems. (Handbook Section 2.3).

Ecosystems in the CIT region, as elsewhere, are products of large-scale elements and processes such as climate, physiography and natural disturbance, overlaid with elements and processes at finer scales.

2.3.1 Climate and Physiography in the CIT Region

The landscape in coastal British Columbia is varied, consisting of lowlands, intricate coastlines, extensive floodplains, and coastal mountain ranges with deeply incised valleys and steep-sided fjords. The climate is cool (mean annual temperatures range from 5 to 10°C) and wet (mean annual precipitation ranges from about 1,000 and 4,500 mm). The CIT area divides into four broad regions—Lowlands, Haida Gwaii Insular Mountains, Outer Coast Mountains and Inner Coast Mountains—differing in topography, climate, hydrology, natural disturbance regimes and ecosystems (Holland 1976; Pojar et al. 1999; Trainor 2001a, 2001b).[2] Trainor (2001b) provides a map and describes the existing biogeoclimatic and physiographic boundaries that delineate the four regions.

The Lowlands form narrow, low-lying, boggy strips along the coast and islands. Precipitation is high, but annual snowfall is relatively low (Table 2). The Insular, Outer Coast, and Inner Coast Mountain regions all feature steep, rugged mountains, ocean fjords, and watersheds of variable size. These mountainous regions are distinguished primarily by climate: the Insular Mountains are very wet and cool with little snowfall, the Outer Coast Mountains are very wet and mostly cool with heavy snowfall and the Inner Coast Mountains are drier with warmer summers and heavy snowfall.

[2] Because Northern Vancouver Island was added recently to the CIT area, most of the reports did not include Northern Vancouver Island.

Table 2. Some climate statistics for selected stations in the CIT region[1]

Region	Station	Mean annual precipitation (mm)	Mean annual snowfall (cm)	Degree days over 18°C
Lowlands	Dryad Point	2,526	45	8
	Egg Island	2,564	48	0
	McInnes Island	2,595	72	1
	Prince Rupert A	2,594	126	0
Insular Mountains	Langara	1,957	103	0
	Pallant Creek	3,342	78	--
Outer Coast Mountains	Falls River	3,633	291	--
	Kildala	2,158	299	26
	Kemano	1,897	260	35
Inner Coast Mountains	Bella Coola	1,652	160	36
	Bella Coola A	1,185	110	42
	Stuie	800	156	83

[1] From Environment Canada climate normal tables.

The four major regions can be further subdivided according to glacial runoff and hydrology, resulting in eleven sub-regions (Figure 1).[3]

Few quantitative summaries of climate and physiography exist for the CIT region. Trainor (2001a) summarises existing data on geomorphology and hydrology for the part of the region within the Central Coast planning district. She concludes that the drier watersheds of the Inner Coast Mountains have a lower density of small streams than either the Outer Coast Mountains or the Lowlands (Table 3) and that the wet Outer Coast Mountains have a higher landslide density than either the drier Inner Coast Mountains or less steep Lowlands (Table 3).

Table 3. Selected elements of hydrology and geomorphology for part of the CIT region[1]

Region	Density of small streams (km/km^{2}; mean ± standard error)	Landslide density (km^2/km^2; mean ± standard error)
Hecate Lowland	0.62 ± 0.12	1.41 ± 0.36
Outer Coast Mountains	0.61 ± 0.06	3.22 ± 0.67
Inner Coast Mountains	0.38 ± 0.22	1.50 ± 0.05

[1] From Trainor (2001a) Tables 3 and 10, except standard error for drainage density calculated from table 8.

[3] The planning area also contains a small area of an “interior” ecosystem—the Nechako Plateau. This area is not considered explicitly in the compendium. However, its unique natural disturbance processes should be considered during management.

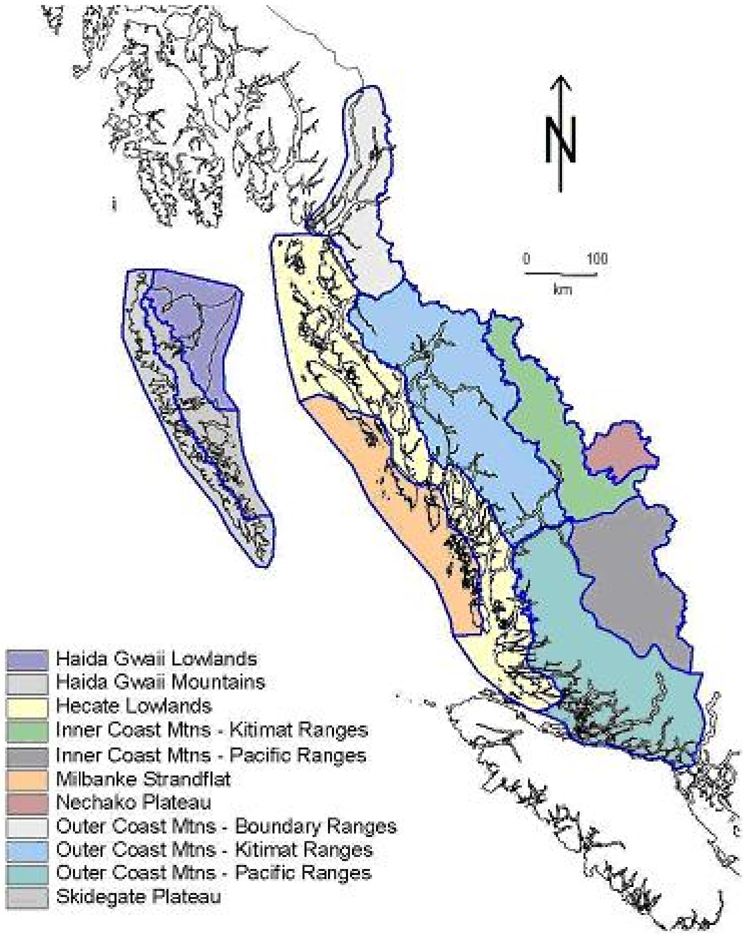


Figure 1. Eleven sub-regions of the CIT area, created by dividing four physiographic regions according to glacial presence and hydrology.

2.3.2 Natural Disturbance in the CIT Region

Dorner and Wong (2003) discuss natural disturbance in the CIT region. The following section is a partial summary of their report.[4]

Natural disturbances, along with the activities of First Nations peoples, have historically shaped the composition and structure of forests on the coastal landscape. A good understanding of natural disturbance dynamics is an important first step towards maintaining ecosystem function and habitat characteristics to which native species have adapted.

Over the majority of the CIT region, stand-replacing disturbances are rare and stands are very old (Figure 2). Death and re-establishment of trees primarily occurs in small canopy gaps of ten trees or fewer. These gaps are typically created by a combination of wind and pathogens and make up around 10–30% of the area in old-growth stands at any time. Geomorphic disturbances are the most important naturally occurring high-severity, stand-replacing events in the area. Wind (generally

[4] Please see complete document for references.

restricted to the exposed portions of the Insular Mountains and northern Vancouver Island) and fire (generally restricted to the Inner Coast Mountains) may also occasionally create large canopy openings. Return intervals for these stand-replacing events are on the scale of millennia throughout the majority of the area. However, wind disturbance on Northern Vancouver Island can be quite frequent (with a return interval in the order of hundreds of years), as it can on localised face units on particular areas of the coast (such as the windward side of Haida Gwaii and other localised exposed areas). Flooding is a key process shaping floodplains and estuaries. Other agents that may occasionally injure or kill trees include snow, ice, frost, and drought, as well as insects and mammals. The different types of disturbances do not occur homogeneously across the forest landscape. Rather, most of the agents are confined to, or occur predominantly in, specific stand types, site types or landscape positions.

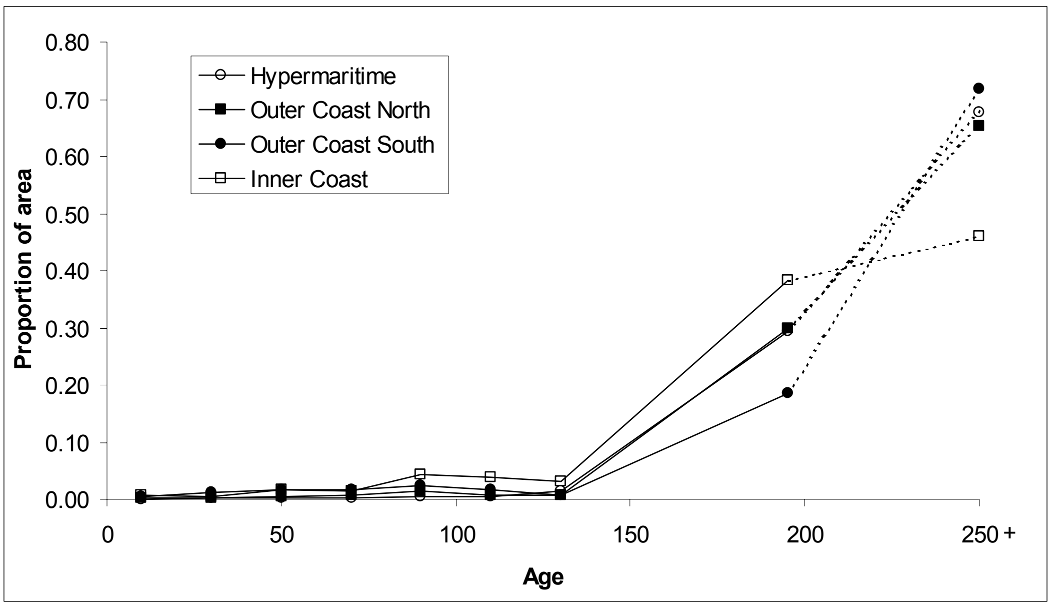


Figure 2. Proportion of CIT region in each age class (midpoint except for last class) in physiographic regions. For these estimates, young forest stands (greater than about 1 hectare) were classed as "natural," "probably natural," and "probably logged" (see Price and Daust 2003). "Probably logged" stands were then aged, allowing calculation of the proportion of young forest disturbed naturally. The amount of forest in the last two age classes is misleading; most of these stands are likely much older than 250 years.

Geomorphic disturbances, such as landslides and snow avalanches are the primary natural agents of stand-replacing disturbance in wet, steep coastal forests. Geomorphic disturbances dislodge trees, soil, and rocks and can deliver sediments and woody debris considerable distances to valley floors, often modifying stream channels and floodplains in the process. Jams of large woody debris alter flow patterns, create a complex channel morphology, and protect downstream habitat and back channels from flood scouring. Geomorphic disturbances also create and maintain early seral communities that provide forage opportunities in close proximity to cover for many species.

Blowdown, largely attributable to the frequent cyclonic storms originating over the Pacific, is also a key disturbance agent in the CIT region. The severity of wind effects is determined by topographic exposure to prevailing winds, tree species and growth form, stand structure, site characteristics, and precipitation. The vast majority of openings created by wind are small canopy gaps. Stand-replacing blowdown is very rare throughout most of the CIT area, except for the exposed western portions of the Insular Mountains (Pearson 2003) and northern Vancouver Island where the 1906 wind event blew down thousands of hectares. Snapped and uprooted trees serve an important ecological role as downed wood in both terrestrial and hydroriparian ecosystems.

Floods, triggered by rainstorms, rain on snow, or rapid snowmelt, cause varying degrees of tree mortality. Flooding, however, rarely kills a substantial number of mature trees unless debris flows or torrents are triggered. Those areas subject to flooding (floodplains, fans, and estuaries) are highly productive biodiversity hotspots. Depending on disturbance frequency and severity, they support forests of various composition and ages, including deciduous forests, stands of massive spruce and hemlock, as well as shrub, berry and herb communities.

With the exception of the drier Inner Coast Mountains, lightning caused fires are rare on the coast, due to the cool, wet climate. The closest published study of pre-colonial fire regimes in similar forests (southern Vancouver Island) found that the average time since the most recent fire ranged between 750 and 4,500 years ago (Gavin et al. 2003). Intentional low-severity burning by First Nations peoples to enhance berry production has been documented for the area, but the extent is unknown. Fire affects post-disturbance succession by reducing thick organic layers, improving nutrient availability, providing a good seed bed for western redcedar and removing advanced regeneration of mistletoe infected western hemlock.

Decay fungi (including heart, butt, and root rots) are the most important biotic disturbance agents in coastal British Columbia. Although pathogens rarely cause direct mortality in older trees, they can predispose infected trees to blowdown and other modes of disturbance. Thus, pathogens enhance structural complexity and accelerate succession to all-aged stands with old-growth characteristics.

Mammals may affect forest dynamics where conditions are favourable. Beavers affect hydrology and vegetation dynamics in hydroriparian ecosystems. Porcupines kill or injure trees and thus contribute to gap dynamics, especially in second-growth stands. Heavy browsing by deer can denude the understory and impede tree regeneration. This impact is especially dramatic on Haida Gwaii, where introduced deer have spread throughout the islands, favoured by the mild climate and virtually unchecked by natural predators.

Dorner and Wong (2003) summarise the disturbance characteristics estimated by studies conducted in coastal British Columbia and Alaska (their Summary Tables 2–5). Few studies have been completed within the CIT regions and fewer tease apart disturbances on unlogged and logged land. The most detailed study, as part of the Central Coast Planning District, found that wind disturbed 0.3%, geomorphic disturbances (including landslides, avalanches, and flooding) disturbed 1.4% and fire disturbed 1.3% of forested land in the study region over the past 140 years (Pearson 2003). Fire was limited to the Inner Coast Mountains; wind was essentially limited to the Lowlands and geomorphic disturbances occurred throughout the study area.

Price and Daust (2003) estimated the total frequency of stand-replacing natural disturbance in the CIT area (removing the effects of logging, but not distinguishing among disturbance types). They found that disturbance increases moving inland, and that fluvial ecosystems have higher disturbance frequencies than upland ecosystems (Figure 3).

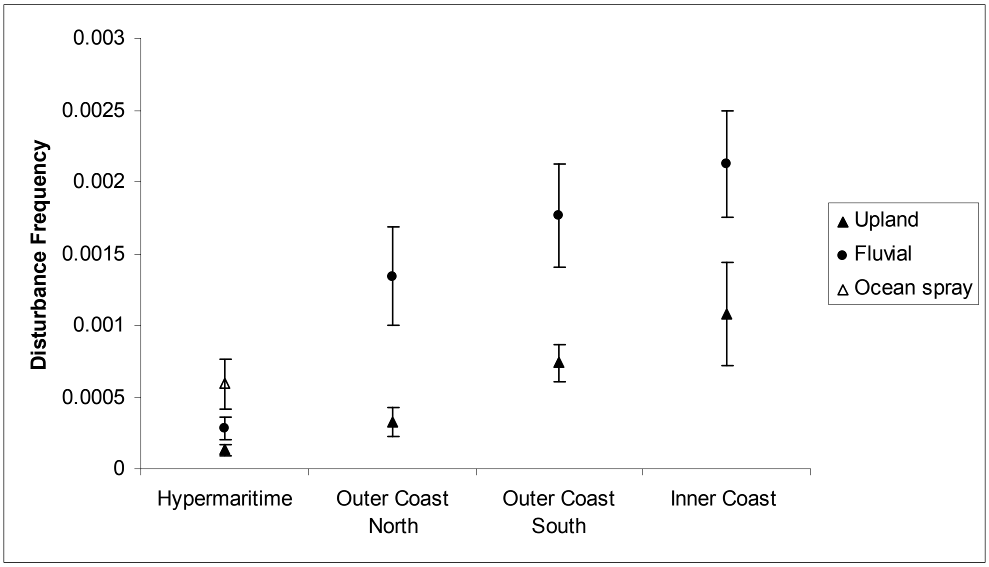


Figure 3. Mean disturbance frequency (proportion of area disturbed per year) for upland fluvial and ocean spray ecosystems in four physiographic regions[5] of the CIT area. Bars are standard errors.

2.3.3 Ecosystems of the CIT Region

The moist, cool climate, mountain and lowland physiography, and non-random natural disturbances create patterns of ecosystems on the landscape. In the Lowlands, patches of productive forest often occur within a matrix of non-forested bog lands and stunted bog woodland. In the watersheds of the Insular and Coast Mountain ranges, the forest mosaic typically consists of large, continuous tracts of all-aged, structurally diverse old-growth coastal western hemlock forest, separated by cliffs, gullies, wetlands, and shrub-covered avalanche and landslide tracks.

Within a climatically and physiographically homogeneous sub-region, slope position, and substrate influence soil moisture and nutrient regime, and consequently, productivity and species composition. Where water accumulates, the wet, cool climate has led to areas of extensive low productivity bog ecosystems. Where water drains freely, the history of infrequent, stand-replacing disturbances and wet, mild climate has led to highly productive forests characterised by large, old trees and tremendous accumulations of biomass, including downed wood and snags (Pojar and MacKinnon 1994). Large trees, snags and downed wood play significant ecological roles in the CIT region, providing habitat and influencing hydroriparian processes.

The biogeoclimatic ecosystem classification system identifies and classifies forested ecosystems in the CIT region (Banner et al. 1993). The Coastal Western Hemlock and Mountain Hemlock biogeoclimatic zones make up about 80% of the CIT region (Table 4). Within a biogeoclimatic subzone or variant (e.g., CWHvh2 or very wet hypermaritime coastal western hemlock subzone, central variant) different site series (e.g., CWHvh2 01) support different late seral plant communities (in this case, a scrubby

[5] Note that the physiographic regions delineated for the purposes of estimating natural disturbance differ slightly from the four units discussed above. The “Hypermaritime” unit includes both Lowlands and Insular Mountains; the Outer Coast Mountains divide into two regions with different disturbance regimes (Pacific Ranges in Outer Coast South; Kitimat Ranges in Outer Coast North).

forest of western redcedar, yellow-cedar, and western hemlock with salal dominating the shrub layer). Because late seral plant community integrates differences in climate, topography and soil, and represents habitat for other organisms, site series indicate general ecological types. Most plants and animals are not restricted to a particular site series (e.g., Kremsater 2001), so grouping similar site series within biogeoclimatic variants is appropriate when considering rates of disturbance and appropriate targets for acceptable risk (see Sections 3 and 4).

Table 4. Area of each major biogeoclimatic subzone (those covering more than 100,000 ha) within the CIT area (excluding Gwaii Haanis Park). Data are from ssPEM (small-scale predictive ecosystem mapping) produced for the entire coastal region.

Subzone	Area (x 1,000 ha)	Percentage (%)
AT	1,276	17
CWH[1]v[2]h[3]	1,729	23
CWHwh	600	8
CWHvm	1,633	22
CWHwm	114	1
CWHws	208	3
CWHms	133	2
MHwh	152	2
MHmm	885	12
MHmmp	163	2
Other[4]	472	6
Total	7,365	

[1] CWH = Coastal Western Hemlock; MH = Mountain Hemlock
[2] v = very wet, w = wet, m = moist, d = dry, x = very dry
[3] h = hypermaritime, m = maritime, s = submaritime, w = warm, c = cold, p = parkland
[4] Other subzones include CWHmm, dm, xm, ds, ESSFmw, mc, mk, xv, IDFww, dw, MSdc, SBSmc, SBPSmc

On this wet, cool coastal environment, hydroriparian ecosystems are particularly prevalent. In the Lowlands, wetlands (bogs, ponds and small lakes) cover 51–75% of the landscape (Banner et al. 1986, 1988). Small low-gradient streams are very common, draining the extensive slope/blanket bogs. There are many small, but few large estuaries and floodplains, because Lowland watersheds are small and primarily rain fed (McKenzie et al. 2000). Exposed marine shores are common. On some of these marine shores, sensitive ecosystems can be found. The Insular and Coast Mountains contain a variety of hydroriparian ecosystems, including small steep headwater streams and gullies, running down into fans and wide floodplains. Moderately sized linear lakes head some valleys and a variety of small wetlands dot floodplains. In these mountainous regions, large estuaries fed by rivers, rain, glaciers, and permanent snow are common (MacKenzie et al. 2000). In the entire CIT area, over a fifth of the ecosystems are hydroriparian (based on site series groups; Table 5).

Table 5. Area of each ecosystem type (group of biogeoclimatic site series) within the CIT area (forested area only). Data are from ssPEM (small-scale predictive ecosystem mapping) produced for the entire coastal region.

Site series group	Area (x 1,000 ha)	Percentage
Estuary	<1	<1%
Fluvial	62	2%
Fan	452	11%
Ocean spray	27	<1%
Wetland	345	9%
Avalanche	87	2%
Upland	2,955	75%
Other	31	<1%
Total	3,960	

As the interface between water and land, hydroriparian ecosystems are ecologically important in four ways (Price and McLennan 2001; Hydroriparian Planning Guide). First, the land influences adjacent waters. Vegetation moderates temperature and water input, provides structure and nutrients and stabilises banks. Bedrock and soil determine water chemistry and channel form. Second, water influences adjacent land. Flow erodes banks and deposits sediments. Flooding, in well-drained soils, creates mosaics of diverse and productive communities. Third, hydroriparian ecosystems link landscapes by transporting water and solutes, sediment, food and organisms. In essence, they form the "*circulatory system of the ecological landscape*" (Clayoquot Scientific Panel 1995). Fourth, because of their diverse forms, relatively frequent disturbance and high productivity, hydroriparian ecosystems are home to diverse plants and animals, including rare communities and species—they are hotspots of biodiversity.

The diversity of species within the CIT region is far greater than previously thought even 10 years ago as studies have expanded to consider more than vertebrates and vascular plants. While bears, salmon, and large trees are integral ecosystem components, inconspicuous species such as mosses, lichen, fungi and soil fauna comprise the majority of species in the region and play crucial ecological roles (Marcot 1997). There are an estimated 50 terrestrial mammal, 20 marine mammal, 200–250 bird (terrestrial and marine), 600 fungi, 700 lichen, and 500–600 bryophyte, 800 vascular plant, and as many as 10,000 terrestrial invertebrates and about 5,000 marine invertebrates (over 2,500 marine invertebrates around Haida Gwaii alone; Sloan et al 2001).

About half of the terrestrial vertebrate species (89 species) are associated with riparian ecosystems (Bunnell et al. 2001). No summaries exist for less familiar organisms due to lack of information (Price and McLennan 2001).

In addition to productive, structurally diverse old-growth ecosystems and unique bog complexes, important ecological elements in the CIT region include unregulated rivers supporting large populations of spawning salmon and grizzly bears, estuaries, kelp beds, seabird colonies, archipelago/fjord terrain, deep fjord and cryptodepression lakes and inter-tidal flats loaded with invertebrates and resident/migrating waterbirds. Haida Gwaii is an especially significant part of the region, internationally renowned among ecologists because of its isolation, its complicated geological history and its insular biota with distinctive, disjunct, and endemic taxa, all threatened by introduced species (Scudder and Gessler 1989; Martin and Gaston 2003). Brooks Peninsula, on the northwest coast of Vancouver Island, is also significant as a glacial refugium.

Section 3. Maintaining the Ecological Integrity of the CIT Region: the Concepts

3.1 Introduction

The following report within the compilation provides more detailed information:

> Holt, R.F. 2001a. An ecosystem-based management framework for the North Coast LRMP. Background report for the North Coast LRMP.

In addition, the following textbook provides thorough overviews of many of topic areas discussed below:

> Lindenmayer, D.B. and J.F. Franklin. 2002. Conserving forest biodiversity: a comprehensive multiscaled approach. Island Press, Washington, D.C.

Maintaining ecological integrity entails sustaining biodiversity and the ecological functions and processes responsible for that biodiversity. The following, more specific goals increase the probability of maintaining ecological integrity (Noss 1992; Noss and Cooperrider 1994; Schwartz 1999; Soulé and Terborgh 1999):

- represent, in a system of reserves, all native ecosystem types across their natural range of variability;
- maintain viable populations of all native species in natural patterns of distribution and abundance;
- maintain ecological and evolutionary processes, such as energy flows, nutrient cycles, disturbance regimes, hydrological processes, and biotic interactions;
- design and manage the system for resilience to environmental change and for maintenance of the evolutionary potential of lineages.

A number of different planning approaches (Noss and Cooperrider 1994; Mangel et al. 1996; Lindenmayer et al. 2000; FSC 2002) can be used to attempt to maintain ecological integrity, in particular:

a) a system of reserves (some of which will be legally defined Protected Areas) that is defined for all scales from region down to site level,

b) precautionary management regimes in the managed landbase,[6] and

c) restoration where appropriate.

Each of these tools likely provides a different benefit, for example Protected Areas often provide the highest probability that certain values will be maintained but are often limited in their application over the landbase by political will. Management regimes within the managed landbase are often very

[6] The managed landbase is sometimes termed the 'matrix'. We don't use this term here because the matrix can also refer to the larger landbase outside Protected Areas, not all of which is included in the 'managed landbase' in this rugged region.

important because the managed portion of the landbase often represents a high percentage of the total landbase (which is less true in this region than in other areas, but there remains a south/ north gradient of percent operability). However, the managed landbase also tends to predominantly consist of a non-random sample of ecosystems, and is often particularly lacking in those with high productivity supporting large structured, diverse forest types. Restoration activities tend to occur due to necessity at smaller scales, and are often extremely costly (in terms of money and/ or time) and with sometimes unpredictable outcomes.

In practice, a combination of the strategies is usually used, and can be used in multiple combinations to attempt to reach the same goal. For example, highly precautionary management over a larger percentage of the landbase may reduce the need for reserves, though will not reduce the need to zero. Alternatively, less cautious management may lead to an increased requirement for both protected areas in the short-term and restoration over multiple time scales. The appropriate combination of strategies will also depend on the condition of the area being considered, and should be partially guided by the pragmatic constraints associated with the development of a particular landscape[7].

Application of the three primary land use planning tools (protected areas/reserves; precautionary management and restoration) requires the use of a number of additional concepts and approaches from several fields:

Reserve design and population viability have been central topics in conservation biology circles for several decades. Landscape ecology has provided theory in the area of ecological hierarchies, landscape spatial patterns, and the ecological processes that create or result from these patterns (Urban et al. 1987). **Risk assessment** and **adaptive management** are relatively new approaches within environmental management. Ecosystem-based management has also championed new concepts, including using the **range of natural variability** as a benchmark. Often, concepts are shared among fields, though jargon and definitions may differ. While discussion of subtleties is academically interesting, the ensuing confusion can hamper planning processes. For example, some authors have suggested that **reserve** design, **natural disturbance** and **coarse/fine filter** approaches constitute fundamentally different approaches to land use planning (e.g., Quigley and Arbelbide 1997) because the concept of **reserves** seems fundamentally opposed to an approach that acknowledges the dynamism of ecosystems. In some ecosystem types, natural disturbances are frequent enough to warrant dynamic reserves that shift locations over time. However, the rate of stand-replacing natural disturbance events is sufficiently low in coastal ecosystems that dynamic reserves would not be relevant here. In addition, reserves are also crucial dynamic benchmarks for assessing the effectiveness of forest practices in managed areas. Each of the approaches outlined above has positive features that, taken in concert, can increase the robustness of any planning process. Comprehensive plans reduce the chance that important elements are missed, and therefore improve the odds of maintaining ecological integrity.

The EBM Handbook uses a variety of approaches in its attempt to maintain ecological integrity on the B.C. coast. (Handbook Section 2.4.1)

[7] This is not to suggest that development constraints should ultimately drive this planning process, however some options may simply be unfeasible. For example harvesting with 70% retention over the entire Coast is likely not economically viable therefore in this case a zoning approach was considered a more feasible option.

Different aspects of the science guiding planning are summarised in Section 3. They are organised into four groups: A: Ecological Elements of Interest, B: How Much is Enough?, C: Management Options and D: Addressing Uncertainty raised in A, B, and C. A schematic of this is shown in Figure 4.

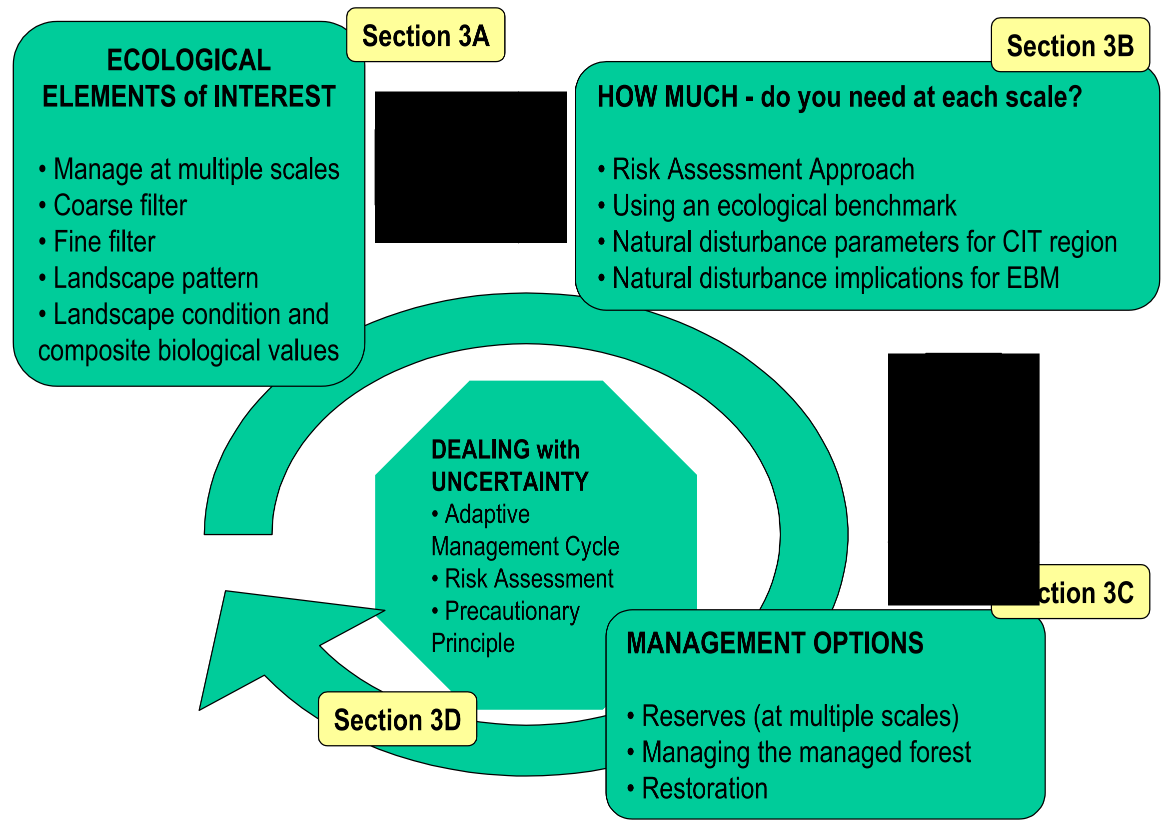


Figure 4. Overview of Section 3.

3A: Ecological Elements of Interest

This section provides an overview of different concepts relevant to determining what components of the landscape will be required to maintain ecological integrity.

3A.1 Managing at Different Spatial Scales

The elements, function, and processes of ecosystems are scale-dependent (Turner 1989; Levin 1992). Grizzly bears, trees, ground beetles and humans experience ecosystems very differently. The scale at which humans perceive ecosystem boundaries, habitat patches and structural elements could have little relevance to the energetics of the system or to the population dynamics of many of its organisms (Pojar et al. 1994).

No single scale is sufficient to assess or maintain ecological integrity. Because ecosystems exist at multiple spatial scales, conservation efforts at multiple scales increase chances of success (Poiani et al. 2000), and multi-scaled systems (such as forested landscapes) are divided into an ordered progression of distinct but inter-related elements and processes functioning at different scales (Urban et al. 1987; Bourgeron et al. 1994). Each level is embedded in the one above, and each level has a characteristic and distinct rate and frequency of processes (Eng 1998). Typically, processes at higher levels (either spatially or temporally) are less frequent and slower and occur over larger areas than processes at lower levels (Urban et al. 1987). Larger scale processes constrain the processes and patterns occurring at lower levels. Hence, conservation efforts that focus on small scales will be ineffective if the larger scale has been functionally altered (Jennings 2000).

Forest management usually occurs at sub-regional, landscape, watershed, and stand levels (Eng 1998). Stands (also referred to as sites) are the scale of forest operations and field assessments. Watersheds are prominent natural features in the CIT region, and provide a logical scale for planning (Naiman 1992; Lertzman et al. 1997; Gomi et al. 2002). Landscapes are mosaics of ecosystems and landforms (Urban et al. 1987), usually containing several watersheds and providing context for watershed planning. Sub-regions include sufficient area for wide-ranging species and are appropriate units to consider as a primary layer of protected areas.

Table 6 shows the range of sizes considered at each scale, based primarily on ecological considerations. Categories intentionally overlap because physiographical and ecological units are characterised by a lack of uniformity in size. For example, in the CIT region, sub-regions with relatively homogeneous hydrology, physiography, climate, natural disturbance types and ecosystems cover from about 100,000–1.7 million hectares (Figure 1). Most territorial or sub-regional planning processes consider even larger areas up to 3 million hectares. The numbers within the EBM Handbook fall within the ranges included in Table 6 and were determined by considering additional social and economic constraints.

Table 6. Size range of area considered within each scale

Level	Size (ha)	Map scale
Region	> 3 million	1:1 million
Sub-region/territory	100,000 – 3 million	1:100,000 – 1:250,000
Landscape	10,000 – 250,000	1:50,000
Watershed	1,000 – 50,000	1:20,000
Site/stand	<250	1:5,000

The distinction between watersheds and landscapes is particularly fuzzy. Watersheds can be defined with more rigor than landscapes, as watershed size depends on the size of its principal stream or river. In the CIT region, watersheds vary from a few hectares to more than 100,000 hectares. Choosing a lower limit for watershed management is a crucial task. The 1,000 hectare bound in the EBM Handbook is set for hydrological and practical reasons; an expert group determined that risk to hydrology should be considered in all watersheds or sub-basins down to 1,000 hectares (Price and Church 2002).[8] Some experts considered that 500 hectares would be more appropriate, but inventories are currently inadequate to support resolution at this scale. The upper limit for watersheds is practical: it is difficult to complete detailed planning processes on areas larger than 50,000 hectares. However, particularly when considering hydroriparian ecosystems, it is important to use as

[8] Some watersheds smaller than 1,000 hectares will need special consideration (see Section 4.5 Fish Habitat)

large a watershed as possible because management in any part of a watershed can potentially influence areas downstream. Once watersheds of interest have been defined, landscapes must be chosen to be sufficiently large to provide a context for planning within these watersheds. Where watersheds of interest are relatively small, small landscapes may provide sufficient context; large watersheds of interest likely require consideration of large landscapes.

Reserves at the sub-regional scale will include entire landscapes or watersheds and address the needs of wide-ranging species (e.g., grizzly bears). Reserves at the watershed and landscape scale will seek to maintain a network of representative ecosystems, hydroriparian ecosystems and linkages among sub-watersheds. Reserves at the site level will concentrate on important structural elements, small streams and ecologically sensitive areas.

3A.2 Coarse and Fine Filter Approaches to Planning

Ecosystems and species can be used in both coarse filter and fine filter approaches. In the CIT region, representative and hydroriparian ecosystems and some wide-ranging focal species are used in a coarse filter approach, and rare ecosystems, rare species and focal species of concern are used in a fine filter approach.

Coarse and fine filter approaches are key concepts in the EBM Handbook (Handbook Section 2.4.1).

3A.2.1 Coarse Filter

Coarse filter approaches to maintaining ecological integrity acknowledge that there are myriad species about which we know little or nothing, and that our ability to predict ecosystem processes is, at best, very limited (Hunter 1991). Maintaining **representative ecosystems** in suitable abundance and distribution across watersheds, landscapes and regions may be the only way to maintain these species and processes (Franklin 1993). Maintaining ecosystems also provides a pragmatic approach to planning, because ecosystems are already classified and, to some extent, mapped. Assessing whether the abundance and distribution of ecosystems is adequate can be aided by using the **range of natural variability** as a benchmark (Quigley and Arbelbide 1997; Landres et al. 1999), combined with **risk assessment** (e.g., MoELP 2000).

The EBM Handbook identifies **biogeoclimatic site series** (or surrogate) **within biogeoclimatic variant** and **hydroriparian ecosystems** as **representative ecosystems** in a coarse filter approach (Handbook Sections 4.1, 4.2, 5.1, 5.2).

Coarse filter approaches can also use individual species as surrogates for ecosystems. Three classes of species capture elements of ecosystems beyond their own requirements. Umbrella species have broad habitat requirements that encompass the needs of others (Noss 1999). Retention of sufficient habitat to maintain these species may also sustain smaller and/or lesser-known species. On the coast, grizzly bears are umbrella species, needing large areas, a diversity of habitats and remoteness. Keystone species play a disproportionately large role in ecosystem functioning (Mills et al. 1993, Simberloff

1998); salmon are a potential keystone species in the CIT region. Indicator species are highly sensitive to particular changes in an ecosystem and Northern goshawks, for example, seem dependent on particular structural elements of certain old forests.

Grizzly bears, salmon and northern goshawks have been identified as suitable coarse filter focal species for the CIT region (Jeo et al. 1999, 2002; Holt and Reid 2002). The EBM Handbook does not attempt to provide detailed guidance for each species, but identifies the need for planning tables to consider the CIT ecosystem spatial analysis (ESA) and locally relevant species (Handbook Sections 5.3, 5.4).

Coarse filter strategies include

1. a system of fully protected reserves outside the managed forest, considering representation analysis
2. reserves within the managed forest matrix (e.g., seral stage targets) considering representation analysis
3. management zones with specific objectives, and
4. baseline stand-level retention targets within the managed forest.

3A.2.2 Ecosystem Representation – But Which Ecosystems?

The following reports within the compilation provide more detailed information:

Holt, R.F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity. Final report to the North Coast LRMP.

Hydroriparian Planning Guide. 2003.

Price, K. 2003. Testing the Hydroriparian Planning Guide. Report to the CIT and North Coast LRMP.

Price, K. and D. McLennan. 2001. Hydroriparian ecosystems of the North Coast. Background report to the North Coast LRMP.

Reserving a portion of every ecosystem occurring in a region is a coarse-filter approach to maintaining ecological integrity. The first task in reserving representative ecosystems is to identify units (ecosystem- or species-based) that capture ecological elements and processes most appropriately and efficiently. Research in other regions suggests that no single surrogate will be sufficient. In the tropics, managing for invertebrates protected vertebrates and plants better than vice versa (Moritz et al. 2001). In the US, surrogate species, including large carnivores, performed no better than random in protecting other species (Andelman and Fagan 2000). In Australia, ecosystems defined by vegetation classes represented trees, birds, mammals and habitat structure reasonably well, but did not capture the requirements of reptiles and invertebrates (MacNally et al. 2002). Effective representation should incorporate a variety of ecosystem- and species-based units (Franklin 1993; Poiani et al. 2000; Groves et al. 2002; Kintsch and Urban 2002). Coastal British Columbia has a well-developed terrestrial ecosystem classification system based on soil and vegetation (Banner et al. 1993). It has a less well-developed hydroriparian ecosystem classification (Price and McLennan 2001).

Terrestrial Ecosystems

British Columbia's biogeoclimatic ecosystem classification system (BEC; Pojar et al. 1987) provides a practical way of distinguishing many terrestrial ecosystems. Because late seral plant community integrates differences in climate, topography and soil, and represents habitat for other organisms, site series indicate general ecological types. Within site series, different seral stages (ages) add further complexity to ecosystem types, creating differing structure and species composition. Even further complexity and variation in structure and species is created by a site's disturbance history, thus the characteristics of any particular ecosystem incorporate many influences.

> The EBM Handbook provides guidelines to reserve representative ecosystems as part of its **coarse filter** approach to maintaining ecological integrity. The EBM Handbook uses site series or groups of site series within biogeoclimatic variants to represent ecosystems. It recommends assessing representative ecosystems at the sub-regional, landscape and watershed scales and provides guidelines for the amount of old forest to be maintained at each scale. In addition, the EBM Handbook recommends developing objectives for locally relevant focal species as appropriate (Handbook Sections 4.3, 4.4, 5.3, 5.4).

While analysis using site series is precautionary and ideal at watershed and landscape scales, accurate data are not available for the entire CIT region, sometimes necessitating use of a surrogate. Indeed, over large areas, site series may be too fine a division for practicality: the large number of site series becomes unwieldy; and many site series are relatively rare, complicating analysis. Division into BEC variants or subzones, conversely, are too broad, encompassing a wide range of ecosystems. In the CIT region, it is possible to retain a high percentage of the area of a particular BEC variant, yet to harvest all of the productive ecosystems within that variant (Holt and Sutherland 2003, Price 2003).

Several projects have used groups of site series or other surrogate to classify ecosystems. Most organisms are not restricted to particular site series, but are often associated with groups of sites series (Kremsater 2001). While it is not surprising that mobile vertebrates are associated with several site series (and broader ecosystem typing), even less mobile organisms, such as invertebrates, vascular plants, fungi, lichens and mosses, seem to respond more to microclimate, substrate and particular site characteristics than to individual site series per se. Small-scale predictive ecosystem mapping, based on terrain modelling and satellite imagery, lists aggregates of site series (inseparable by remote means) for the region. These data are a good starting point, but require verification. Other processes have analysed clusters of site series (grouped after analysis because of similarities in vegetation; Huggard 2000, 2001), "Broad Ecosystem Units" (http://srmwww.gov.bc.ca/ecology/bei/) or "Analysis Units" (Holt and Sutherland 2003), which are defined by dominant tree species and productivity (based on widely used forest cover information) and correspond broadly to overlapping groups of site series (A. Banner, pers. comm.).

No surrogate is ideal. The costs and benefits of each available system must be assessed prior to use (see Holt and Sutherland 2003 for discussion of potential pitfalls associated with Analysis Units). For example, all remotely sensed surrogates require special consideration of rare site series because ground-truthing has shown that remote techniques under-represent these site series on maps (e.g., Ketcheson et al. 2002). The ultimate goal is to have site series mapping available for all areas (e.g., TEM), which can then be grouped as appropriate. In the interim other surrogates may have to be used. It is also worth note that an appropriate unit for strategic planning may be inappropriate at other scales. For example broadly assessing ecosystem representation using a surrogate such as analysis units may be appropriate regionally, but operationally the best ecosystem description available (usually site series) must be used to capture the rare elements.

Hydroriparian Ecosystems

Unfortunately, site series mapping does not classify all ecosystems along streams, rivers, wetlands and shorelines. Neither does it account for the influence of upstream ecosystems on those downstream. Because the BEC system was designed for classifying forested ecosystems, it does not directly consider hydrological features, provide landscape context or combine sites into ecosystem complexes—all important aspects of hydroriparian ecosystems. A new classification framework, based on the Canadian Wetland Classification System, BEC site units, and other sources, is being designed specifically for hydroriparian ecosystems (MacKenzie and Banner, draft 2001), but is not yet available. Principle hydroriparian ecosystem types include estuaries, ocean spray forests, floodplains, fans, small steep streams, other small streams, lakes, and wetlands. Appendix 3 of the Hydroriparian Planning Guide lists important hydroriparian ecosystems for the CIT region in more detail.

> To ensure representation of hydroriparian elements and processes, the EBM Handbook includes representing hydroriparian ecosystems in its **coarse filter** approach. It recommends assessing hydroriparian ecosystems at the watershed scale and provides guidelines for the amount of natural riparian forest to be maintained next to each type of hydroriparian ecosystem. (Handbook Section 5)

Until hydroriparian ecosystems are included in a complete classification system, there will be some duplication of effort. For example, while ecosystems around small streams cannot be identified by biogeoclimatic site series, site series can sometimes delineate fans and can generally define floodplains. However, hydroriparian ecosystems are relatively easy to identify based on water features and terrain information. Given their disproportionate importance and the current uncertainty associated with site series surrogate measures, it is reasonable to use two methods (site series representation and representation based on water and terrain features) to ensure that hydroriparian ecosystems are evaluated appropriately.

3A.2.3 Fine Filter

Some elements (populations, species, ecosystems, special features) may not be maintained adequately using a **coarse filter**. **Rare ecosystems** are more closely considered through a fine filter approach. Fine filter species include **rare species** and those with specific habitat requirements or limited ranges. Some individual species of interest act as ecosystem surrogates under the coarse filter approach (e.g., grizzly bears). Others have more specific requirements, or are less well known (e.g., tailed frog, marbled murrelet, species of fish limited to a particular watershed, plants, or fungi limited to a particular site) and might be under-represented under a coarse filter.

Fine filter strategies include

1. identifying rare or sensitive species, or those with requirements not likely to be covered in the coarse filter,
2. identifying **rare ecosystems** or ecosystem elements not included in the coarse filter,
3. using reserves and management zones to protect fine-filter ecosystems and habitat for fine-filter species.

The EBM Handbook considers **rare species**, **rare ecosystems,** and certain focal species as part of its fine filter approach. (Handbook Sections 4.4, 5.4, 5.5, 6.3, 6.4).

Note that the coarse filter should be applied first and the fine filter is intended to pick up elements not covered by the coarse filter.

Rare Ecosystems

The following reports within the compilation provide more detailed information:

Pojar, J. 2002. Rare ecosystems of the CWHvh2. Report for the North Coast LRMP.

Pojar, J. 2003. Biodiversity in the CIT region. Report for the CIT.

The EBM Handbook suggests special consideration of rare ecosystems as part of a **fine filter** approach to maintaining ecological integrity in the CIT region (Handbook Sections 4.4, 5.4, 5.5, 6.3, 6.4).

In British Columbia, the Conservation Data Centre (CDC) tracks rare plant associations defined by biogeoclimatic site series and structural stage, and classifies them as "red-listed" or "blue-listed" associations. These plant associations, with their accompanying habitat features, can be considered ecosystems. A red-listed plant association is "imperilled provincially because of extreme rarity or because of some factor(s) making it especially vulnerable to extirpation or extinction", and typically has fewer than 20 high-quality occurrences within the province. A blue-listed plant association may have from 21 to 100 occurrences and is considered vulnerable to either large-scale disturbance or small-scale but chronic, human-caused disturbance. Both red- and blue-listed associations can be either naturally rare, or depleted and rare due to human activities.

CDC lists a handful of red-listed plant associations for the CIT region. Typically they are represented by productive hydroriparian ecosystems. In many cases, these red-listed floodplain associations adjoin blue-listed associations, often on fans. A second major group of blue-listed associations occur in salt-spray zones along windward shores or in brackish shoreline habitats.

There are several unlisted rare, sensitive, or threatened ecosystems in the region (Pojar 2003):

- productive stands of yellow-cedar
- forests on recent, post-glacial volcanic landforms

- karst forest ecosystems on the northern mainland coast
- seabird islands
- marine mammal rookeries and haul-outs
- estuaries and other tidal wetlands
- sand beaches
- rich fens and marshes
- hot springs.

The EBM Handbook includes rare ecosystems from official lists and those classified as rare based on local expert opinion.

Rare Species

The following report within the compilation provides more detailed information:

Pojar, J. 2003. Biodiversity in the CIT region. Report to the CIT.

Because legal mandates (e.g., Federal Species at Risk Act) focus on individual species rather than ecosystems, rare species have traditionally received more attention than any other element of ecological integrity. This history has led to inappropriate use of lists of rare species to set conservation priorities and determine management actions (Pojar 2000; Possingham et al. 2002; Pojar 2003). Identifying, listing, and assessing species and ecosystems at risk are means to achieving conservation goals not the goal itself. Listing is a worthy activity that highlights elements of conservation concern, affects the allocation of resources to conservation, and can influence management. The official federal and provincial lists are logical places to start, but not the sole considerations. In British Columbia, COSEWIC currently considers 20 taxa of vascular plants endangered or threatened. All 20 of them occur in extreme southern British Columbia, have the bulk of their distribution in the U.S. Pacific Northwest or elsewhere, and are peripheral species in our province. Additionally, only five of them are ranked globally rare or globally "important" (G1, G2, G3). CDC's looser filter yields 600+ extant red- and blue-listed vascular plant taxa. But most are **not** globally rare/important (about 78% and 92%, respectively) and most occur peripherally in B.C. More than half of the globally rare/significant taxa are also peripheral in British Columbia. A relative handful are endemic or have the bulk or a significant portion of their populations in British Columbia. Certainly, there are legitimate conservation values of peripheral, regionally rare populations. They also warrant conservation concern, but perhaps more so in the aggregate, as components of endangered ecosystems.

COSEWIC's constrained lists have limited relevance to the conservation of vascular plant resources in British Columbia. CDC appears to emphasize local and regional rarity, which can have unfortunate consequences, most notably a lopsided conservation effort. Such listing tends to focus conservation attention and allocation of scarce resources on what can be unrewarding or ineffective efforts, and can divert attention and funding from other, perhaps worthier elements of conservation concern. Maintaining and conserving biological diversity, ecological integrity, and ecosystem services involve other kinds of species (and biodiversity elements) in addition to rare species—mostly peripheral ones at that. Our conservation goals require a more expansive and balanced approach.

Consideration of rare species alone cannot maintain ecological integrity. Even within the realm of protecting individual species, a focus on peripheral species (rare because of jurisdictional boundaries) diverts limited resources away from "*species for which the province has global responsibility*" (Bunnell and Campbell 2002). Several authors have suggested ways to improve the correspondence between listed species and conservation value (Dunn et al. 1999; Bunnell and Campbell 2002).

Rare species are not the only species that should be listed and monitored. Bunnell and Campbell (2002) recommend that species conservation efforts (listing, tracking and recovery) address several criteria in order of priority, based on Dunn et al. (1999):

- endemics (species and subspecies/varieties)
- significant world populations and ranges in British Columbia
- population trends
- vulnerability
- geographically or ecologically marginal populations.

Endemic taxa are native to, and restricted to, a particular place or geographic region. Island endemics seem to be most vulnerable. Several endemic subspecies of birds and mammals, some found only on the islands of Haida Gwaii, occur in the CIT region (see Pojar 2003 for list). Seven species or subspecies of vascular plants are endemic to the CIT region (Douglas 1996). Many of these insular endemic taxa probably result from isolation and survival in Pleistocene glacial refugia (Warner et al. 1982; Hebda and Haggarty 1997). Biogeographic evidence supporting a Haida Gwaii refugium comes from mammals (Foster 1965), birds (Cowan 1989), insects (Kavanaugh 1989), bryophytes (Schofield 1989), vascular plants (Taylor 1989), and lichens (Brodo 1995).

Common, but potentially vulnerable endemic taxa should also be listed and considered for monitoring. For example, *Sinosenecio newcombei,* a very distinctive plant, is known only from Haida Gwaii, where it suffers from grazing by introduced deer (Pojar 2003).

In addition to the endemics, coastal British Columbia supports significant world populations, or contributes a large part of the breeding or wintering range, of more than a dozen species of birds (Bunnell and Campbell 2002), including the ancient murrelet, marbled murrelet, Cassin's auklet, rhinoceros auklet, black oystercatcher, Barrow's goldeneye, bald eagle, great blue heron, trumpeter swan, varied thrush, other vertebrates, including the tailed frog, and plants, including western redcedar, Sitka spruce, copperbush, and devil's-club. Although many of these species are not rare, the importance of the CIT region to global populations can justify their inclusion on lists of organisms to be considered for fine-filter monitoring. Choices for monitoring, however, also depend on criteria other than responsibility and vulnerability. Species at the northern extreme of their range may become more important to monitor if climate change or practices in the Pacific Northwest cause declines in southern populations.

Alternative lists for vascular plants of the Queen Charlotte Islands, using criteria of stewardship responsibility, population trends, vulnerability, rarity, significant disjunctions and marginal populations, and virulent weediness have been initiated (Jim Pojar, pers. comm., and included in CIT's Ecosystem

Spatial Analysis). Such an approach could lead to more effective biological conservation in British Columbia.

> The EBM Handbook suggests special consideration of rare species and species of concern as part of a **fine filter** approach to maintaining ecological integrity in the CIT region (Handbook Sections 4.4, 5.4, 5.5, 6.3, 6.4).

Introduced Species

Introduced species pose one of the greatest threats to ecological integrity over the globe (Mack et al. 2000) and in British Columbia (Holt 2001c, 2001d). Introduced animals can cause extinction of native species; introduced plants can alter ecological processes; many introduced species can hybridise with native species (Mack et al. 2000). Control of established invaders is rare (Mack et al. 2000).

In the CIT region, introduced species have had substantial impacts on the ecological integrity of Haida Gwaii (Holt 2001c; RGIS 2002; Martin and Gaston 2003). The impact of introduced deer is especially dramatic, denuding understory vegetation in both upland and floodplain forests, threatening the islands' endemic and rare flora and changing succession (e.g., Pojar 1999; Pojar 2002). Introduced beavers have altered hydroriparian processes. More than ten other mammals and about a quarter of the plants in Haida Gwaii are introduced. Japanese knotweed and broom have already spread widely and dominated sites; a long list of other introduced plants are successful in disturbed habitats (Pojar 2002). Introduced species also affect other areas of the coast.

> The EBM Handbook does not directly address introduced species. The planning process and risk methodology outlined in The EBM Handbook, however, provide guidance. A set of introduced species should be included among locally relevant focal species.

Many of the species that turn into "problem species" are strong competitors in disturbed areas. Forestry, through silvicultural trials, harvesting operations and road-building, has contributed to the introduction and spread of introduced species. Roads are corridors of invasion. Seeding of roadcuts and disturbed areas with non-native mixes contributes to the spread of introduced vegetation. Young production forests help support the high population of deer on Haida Gwaii (Pojar 2002).

Because it is difficult to identify a problem species until it is well established, and because control of established invaders is rare, precaution dictates that all introduced species are considered harmful. Some species remain at low levels for decades before their population explodes (Sakai et al. 2001). Knowledge of local exotics and how they are spread is important. Tracking of key exotics is prudent.

3A.3 Landscape Pattern

The following reports within the compilation provide more detailed information:

Price, K. and,D. McLennan. 2002. Impacts of forest harvesting on terrestrial riparian ecosystems of the Pacific Northwest. Hydroriparian Planning Guide Technical Report #7.

Young, K. 2001. A review and meta-analysis of the effects of riparian zone logging on stream ecosystems in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #4.

Habitat Pattern

A null hypothesis in landscape ecology is that the overall suitability of a landscape, for a given species, declines linearly with the amount of habitat available. The degree to which overall suitability declines non-linearly is an indication of the role that pattern and non-habitat areas play in maintaining or impeding population persistence. A non-linear decrease in suitability at approximately 30% habitat remaining (Andrén 1994) suggests that pattern effects may be more important when habitat abundance drops below this threshold.

Habitat abundance exerts considerable control over landscape pattern. On modeled landscapes, as habitat proportion increases, patch size increases and the distance between patches decreases (Andrén 1994) in a predictable manner until a percolation threshold (where one patch connects the entire area) is reached (Stauffer and Aharony 1992). When patches are uniformly spaced on a grid, the ratio of patch size to patch spacing varies with habitat abundance in a predictable manner (Daust 1994). Although modelled landscapes do not match real landscapes, they show that for a given habitat abundance, landscapes can have a few widely spaced patches or many small patches that are closer together (or some combination).

The effects of ecological patterns are well documented in some ecosystems. The classic work on island biogeography (McArthur and Wilson 1967) identifies two ecologically relevant aspects of pattern (summarised in Bunnell 1999):

> *Area*: larger islands contain more species than do small islands. This occurs because small islands experience more extinctions (small populations are more vulnerable to chance events) and receive fewer immigrants (species wandering from the mainland to nearby islands are not as likely to encounter them—a kind of "target size" effect).
>
> *Distance*: equivalent-sized islands more remote from the mainland or source population will have fewer species because the extinction rate is the same but the immigration rate is lower (fewer immigrants reach the island).

Other lines of investigation corroborate the importance of area and distance effects. Meta-population theory supports the importance of patch size and isolation on extinction and colonization processes (Levins 1969, 1970; Hanski 1994). Theory on minimum viable populations suggest that species persistence is greatly compromised when habitats support less than 50 individuals (Shaffer 1981). Some species, particularly habitat specialists, may have trouble finding habitat in sufficient density to support a home range in heavily fragmented forests (e.g., Chapin et al. 1998).

Area (of a patch) and distance (between patches) effects, however, are poorly documented in forested settings (Bunnell 1999). Some of the studies that have found effects have not controlled for habitat abundance. Many studies that have controlled for abundance have not been able to demonstrate a strong affinity of species to habitat pattern (Bunnell 1999). Evidence from vertebrates suggests that habitat abundance has a greater impact on species persistence in temperate forested landscapes than habitat pattern (McGarigal and McComb 1995; Fahrig 2002; Schmiegelow and Monkkonen 2002). Managed forests may differ from island biogeography theory because forest fragments are not true islands and because the non-habitat matrix is not a true sea (Hunter 1990). While many vertebrate species prefer natural old forest, they can often inhabit young forest, particularly young forest with natural structure. Other taxa, particularly poor dispersers and habitat specialists, such as some epiphytic lichens, canopy invertebrates and fungi, may be more sensitive

(Price and Hochachka 2001; Sverdrup-Thygeson and Lindenmayer 2003) though further studies are necessary.

Despite the differences between forest and island settings, some studies (e.g., Laurance and Bierregaard 1996; Jansson and Angelstam 1999) have found effects consistent with Island Biogeography theory in forested landscapes. In addition, species-specific simulation models also predict pattern effects (Fahrig 2002) in any landscape, depending on the suitability of the matrix for a given species.

Maintaining pattern is one of the objectives listed in the EBM Handbook (Handbook Section 2.3). The EBM Handbook recommends that natural disturbance patterns guide harvest pattern (Handbook Section 5.5), but does not describe targets for landscape pattern, because the generally precautionary nature of representation targets will tend to prevent landscapes from reaching pattern thresholds.

Connectivity and Corridors

When intervening habitat is hostile to both survival and movement, the importance of connectivity is well established (Bunnell 1999). In managed forests, some species move more within than outside corridors, suggesting that clearcuts may be somewhat hostile to some species (Bunnell 1999). Dispersal of individuals links different populations, and it is critical to manage managed stands in a way that facilitates movement for species with limited dispersal abilities (Wiens et al. 1993)

Historically, strategies to enhance connectivity meant providing linear corridors. The benefits of corridors remain equivocal, however (Noss 1987; Simberloff and Cox 1987; Hobbs 1992; Rosenberg et al. 1997; Beier and Noss 1998), and there has been very little evidence that corridors enhance movement between patches (Rosenberg et al. 1997). Corridors seem beneficial for some species, and detrimental for others, sometimes increasing predation and promoting introduced species. Riparian corridors are an exception and are inherently valuable as habitat regardless of their ability to facilitate movement (Rosenberg et al. 1997).

Approximately 74% of the Central and North Coast forested landscape is constrained from harvesting, especially at higher elevations (Holt and Sutherland 2003; Wells et al. in review), suggesting that connectivity among watersheds may not be an issue. However, in this topographically rugged landscape, is it possible that a small area of disturbance within an area which traditionally has facilitated movement (e.g., a low pass or relatively benign headwaters) may in fact create unpredicted but significant barriers to movement. Within-watershed connectivity is also a concern, particularly in productive valley bottoms that have been heavily harvested and impacted by roads. A recent comprehensive study of mammal communities in the fragmented forests of northwestern Washington suggests that riparian forest corridors are an effective supplement to continuous old forest for maintaining biodiversity (Lomolino and Perault 2000; Perault and Lomolino 2000). In the riparian corridors, forest mammal communities were related to the proportion of adjacent old growth, corridor width and isolation. There was evidence for reproduction in corridors, but at lower levels than in continuous old-growth (see discussion of the value of riparian corridors in Price and McLennan 2002). Partial retention in logged stands and the maintenance of riparian buffers can facilitate movement along valleys.

The EBM Handbook recommends identification and reserve of wildlife migration corridors, riparian corridors, connectivity for rare species, and landscape connections, in conjunction with reserving representative ecosystems, in watershed-level planning (Handbook Section 5.5).

An important means of maintaining connectivity is by managing harvested areas so that they are favourable for movement. Providing patches of retained trees and other structures creates "stepping stones" of habitat and generally may increase the permeability of early seral stages to those organisms preferring certain structures to facilitate movement.

Edge Effects

Inherent edges result from long-term features of the landscape such as the underlying soil type, topography, geomorphology, or microclimate (Swanson et al. 1988). These inherent edges are distinguishable by a shift in the plant communities within the forest landscape (Yahner 1988; Hunter 1990). Inherent edges are very common in the CIT region, along hydroriparian ecosystems, rocky outcrops and cliffs. In contrast, induced edges are relatively short-term entities resulting from disturbances (harvesting or natural) creating adjacent seral stages.

Induced edges change physical and biological elements of the ecosystems to either side. There is tremendous variability in responses around edges (see review by Kremsater and Bunnell 1999). Consistent changes include microclimatic effects and shifts in community composition.

Reported edge effects range from 0 to several hundred metres into the forest. Microclimate effects are well documented (Young 2001) and typically extend 1 to 3 tree heights into the forest, depending on aspect and variable measured. Microclimate effects appear to increase structural diversity of vegetation near the edge. Although microclimate responses are detectable sometimes as far as 150 to 200 m from the edge (on extremely exposed edges, but as little as 15 m on sheltered edges), responses of organisms are concentrated within the first 50 m, most plants responses are seen only within first 25 m. Effects on animals have been reported as extending much farther, but some of these studies are flawed and unconvincing (see Paton 1994 for review). From studies in the Midwest and east, there is clear evidence for increased predation and parasitism near edges (Paton 1994, Cotterill and Hannon 1999), again usually within the first 50m. Evidence in forested ecosystems in the west is less clear, although corridors can become population sinks, attracting breeding pairs of birds, but providing low reproductive success (Bunnell et al. 1998). For example, increases in Steller's jay abundance in buffers in the Pacific Coastal region (Price and McLennan 2002) suggest that nest predation risk may increase in western riparian buffers.

Most of the current literature examines edge effects from forests surrounded by non-forested ecosystems; studies of edge effects between forest stands of different ages are few. While there are no vertebrates that, consistently in all studies, seem to require interior forest conditions (i.e., more than 150 m from edge), forests away from the edges contain different communities than forests close to edges. This may be particularly true for species more sensitive to microclimate such as lichens and mosses, which may need forest interior, and likely edge effects may become more pronounced as old forest habitat decreases.

The inherent edge present in riparian ecosystems presents a special case. In unmanaged forests, riparian ecosystems include a natural edge created by water. Many organisms use these productive riparian edge ecosystems. Microclimatic gradients around small streams (2–4 m) can extend over 50 m (Brosofke et al. 1997), and vegetation communities can be influenced by moderately sized streams (3–30 m) for 100 m (Chan-MacLeod 1996). Forest harvesting adjacent to riparian ecosystems can change the conditions and communities found within these ecosystems. In undisturbed areas, the inherent water-land edge merges into continuous, interior forest away from the water. Conversely, when forest adjacent to riparian ecosystems is harvested, leaving a riparian buffer strip, the riparian ecosystems provide only edge (natural on one side, managed on the other). Young (2001) and Price and McLennan (2002) performed meta-analyses of studies in coastal ecosystems examining the impacts of harvesting on riparian edges. While both studies found that riparian buffer strips mitigated the impacts of harvesting on riparian ecosystems, neither was able to examine quantitatively the effectiveness of different buffer widths on biological (rather than physical) aspects due to lack of data. A recent comprehensive study of mammal communities in the fragmented forests of northwestern Washington suggests that riparian buffers are an effective *supplement to*, rather than *replacement for*, continuous forest (Lomolino and Perault 2000; Perault and Lomolino 2000).

The most recent comprehensive study in British Columbia of edge effects (examining 21 variables including microclimate, soil chemistry, snow depth, windthrow, regeneration, vegetation, and animals) was at Sicamous Creek in the high elevation Engelmann Spruce–Subalpine Fir ecosystem. Huggard and Vyse (2003) summarized the multiple studies. Most variables showed very short edge effects, interior forest levels (or clearcut levels) were reached within 10 to 20 m of the cutblock edge. Some physical variables such as temperature and snowmelt showed effects from up to 30 to 50 m from the edge, but no biological effects were seen at that distance. There appeared to be an effect of edge creep due to windthrow that extended effects beyond the initial edge. There are studies underway to examine edge effects in coastal forests (see Weyerhaeuser's Adaptive Management program investigating lichen and moss and invertebrates at different distances from edges).

Edge effects around roads include those listed above and additional effects listed under **access**.

Managing patch sizes, aggregating harvesting and reducing roads are management tools to avoid negative edge effects. To accommodate uncertainty, some forest interior should be maintained. An estimate of 100 to 150 m edge effect would be conservative.

> Because of the questionable benefits of protecting the functions of small streams by using narrow buffers (effectively all edge), the EBM Handbook recommends small stream protection areas that surround a group of streams (Handbook Section 5.5).

Although buffers mitigate impacts of harvesting, Price and McLennan (2002) conclude that riparian buffer strips may not be the best way to maintain riparian biodiversity around small streams because riparian associates also use upland habitat, because buffers include only edge with changed microclimate, structure and community, and because buffers often blow down. Church and Eaton (2001) also discuss the impracticability of buffering all small, particularly intermittently flowing, streams. Because these channels include more than half of the entire length of a drainage system, systematically applying reserves along them would fragment the landscape.

3A.4 Composite Biological Values and Landscape Condition

All the information summarised above suggests that multiple single values will help guide the process of maintaining ecological integrity. While it is obviously important to understand which individual values are located in individual geographic areas, it will also be key to summarise single indicators into indices of biological value. Integration of suites of values should allow a general ranking of watersheds and landscapes in terms of composite biological values. This kind of summary will clearly be important in interpreting and applying all the types of information summarised above. For example, ranking watersheds/ landscapes for their biological values would aid in applying reserve design.

Approaches for combining single values into a composite of values has been used in many other processes, and key approaches used are summarised under Section 3.9 (Reserve Selection) below.

The EBM Handbook recommends applying low to high risk management strategies at the different scales according to the biological values of geographic areas. For example, high risk strategies and high ecological values would likely be incompatible (Handbook Section 3.2, Section 7).

In addition, understanding the current condition of a particular area will also influence how low to high risk strategies, including restoration, should be applied.

The EBM Handbook outlines how current condition of the landscape (in relation to targets at each scale) should be used to determine appropriate future action (management/ restoration etc.). (Sections 4, 5, and 6 for regional, landscape and site level planning).

3B: How Much Should Be Reserved to Maintain Ecological Integrity?

This section provides an overview of how science can help the process of determining what levels of protection at each scale may have a high probability of maintaining ecological integrity. The general approach is summarised, and then specific implications for ecosystem-based management in the CIT region are summarised (see Figure 4 for overview).

3B.1 The Approach

Maintaining ecological integrity involves retaining sufficient area to maintain viable populations of species across their natural ranges, and to maintain natural processes. The ecological effects of habitat abundance are well documented. Both population size (Fahrig 2002) and species richness (Preston 1962) decrease as habitat abundance decreases, and habitat loss is generally regarded as the biggest single threat to biodiversity (Erhlich 1988). This relationship has been most clearly demonstrated when one habitat type is being replaced by a significantly different one (e.g., forest being replaced by agriculture or urban development; Bunnell 1999). In coastal forested systems where old growth is replaced by earlier seral conditions rather than rural or suburban conditions, the effects of losing old forest habitat are not as clear. Young seral forest around natural or old forests can be hospitable to some organisms that typically occur in old growth; it is those organisms with tighter associations to older forest that reflect the impact of its reduction. Retaining structural attributes of older forests increases the habitat value of young managed forests for the majority of species associated with old growth (see Section 4.4 on in-block/variable retention). Structural retention in managed forests can complement and may reduce the requirement for reserves in some ecosystems.

A crucial question is, "at each scale, how much retention of each ecosystem or habitat is enough to maintain natural elements and processes?" Science cannot provide a generic answer (Holt 2001a). Estimates range from 4 to 99% depending on the ecosystem, on assumptions about the contribution of the managed landbase and other variables (Noss and Cooperrider 1994). Within a region, however, science can provide guidance on the range of amounts that are more likely to maintain ecological integrity. Sufficient area will vary with **natural disturbance** regimes, and more specifically, with the **range of natural variability** of ecosystem elements and processes. For example, assuming that species have adapted to their environment, the amount of old seral forest required to maintain ecological integrity will increase as natural levels of old forest increase. Because ecosystems differ, it is essential that the amount reserved is considered separately for each ecosystem. For example, within a BEC variant, each site series or group of site series should be represented by the amount of old forest calculated as appropriate based on the natural disturbance regimes for that variant. Even high levels of overall retention in a region may fail to maintain ecological integrity if they do not include critical habitats in **reserves**. In the CIT region, for example, the small extent of productive floodplain vastly underestimates its ecological significance, and ensuring overrepresentation of these types would likely be key.

Ecosystems with high variability in natural disturbance patterns and consequently high variability in the natural amounts of a particular habitat type such as old forest, may be more resilient to divergence from natural conditions. Consequently, ecological integrity may be maintained using lower percentages of old forest. Conversely, more stable ecosystems, with lower variability in natural disturbance rates and patterns are potentially less resilient to a high divergence away from the natural amount due to species' adaptation to a stable environment.

Risk assessment, based on the range of natural variability in natural disturbance regime, and including the results of theoretical and empirical studies, can be used to develop curves relating the amount reserved and risk to ecological elements and processes.

The EBM Handbook provides guidelines for how much to reserve of each **representative ecosystem** (site series or hydroriparian ecosystem) based on the **range of natural variability** and **risk assessment**.

3B.2 Risk Assessment

The following reports within the compilation provide more detailed information:

Beasley and Wright. 2001. Criteria and indicators briefing paper. Background report to the North Coast LRMP.

Holt, R.F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity. Final report to the North Coast LRMP.

Taylor, B. 2000a. An introduction to adaptive management. Background report to the North Coast LRMP.

Risk assessment, along with adaptive management and the precautionary principle, is a method used to address uncertainty. While Section 3D describes adaptive management and the precautionary principle, risk assessment is particularly germane to deciding how much to reserve, and hence is included here.

Risk assessment is a key concept in EBM (Handbook Section 2.5). The EBM Handbook uses risk assessment to provide guidance about which activities pose low or high risk to various aspects of ecological integrity (Handbook Sections 4.4, 5.5, 6.4).

Risk assessment evaluates the probability that a particular management activity will have an adverse impact on some component of ecological integrity. It provides an opportunity to make informed decisions about complex questions using best available information while acknowledging uncertainties and knowledge gaps explicitly (Everest et al. 1997). Risk assessment procedures can be highly complex, or simple 'back of the envelope' processes (Swanson et al. 1996; Ministry of Environment, Land and Parks 2000).

The EBM Handbook uses an explicit risk assessment because of the need to present the presumed implications of any decisions explicitly.

Risk is the probability of an adverse outcome and can be considered as the probability that an ecosystem or ecosystem function will be changed or lost following a particular management activity[9]. In ecosystem-based management, ecological risk can be represented as [1 – the probability of maintaining a particular element, process or function]. Risk assessment is a way of estimating risk based on curves relating indicator values to risk level. Risk curves relate the level of risk to planning indicators developed for each ecosystem function, and are most useful as tools when thresholds or points which likely indicate significant ecological change are identified (see below for guidance around what constitutes low and high risk points). These curves can be used to assess the risk associated with a particular management activity. The level of risk can then be compared with an acceptable level of risk. Determining ***acceptable*** levels of risk is a task for planning tables but the approach taken by the EBM Handbook is to attempt to inform these and other decision-makers of the potential implications of alternate decisions.

Risk assessment consists of five steps: a) determine acceptable level of risk, b) identify indicators for the elements, processes or functions that are the focus of the assessment, c) define the relationship between indicator values and risk, including appropriate benchmarks against which the indicator values will be compared, risk classes, thresholds and uncertainty, d) calculate indicator values for different management options, e) determine risk associated with indicator values using risk curves and benchmarks, and compare with acceptable levels.

[9] This definition follows that used by the BC Ministry of Water, Land and Air Protection rather than an industrial definition, which entails the probability of loss multiplied by the value of the object.

The EBM Handbook used risk assessment to set precautionary "low risk targets" at appropriate scales (e.g., sub-regional scale for ecosystem representation, all scales for rare ecosystems and species, watershed scale for hydroriparian processes) and to determine "high risk thresholds" for each indicator. Provided that low risk targets are met at appropriate scales, the EBM Handbook allows deviation at subordinate scales up to the high risk threshold. Identifying both low and high risk points allows for a range of activities and provides some flexibility for operations. In order that there remains a high probability of maintaining ecological integrity, however, low risk targets must be reached at the appropriate scale.

A decision to plan a particular activity in a particular area will be based on risk assessment. Risk assessment will show that risks of further development are either currently within the targets set for ecosystem-based management or outside the targets for certain ecosystems.

The CIT scientists and professionals used the following rationale to provide risk-based guidelines to bound management decisions under ecosystem-based management:

- The low risk targets represent the point where detrimental ecological changes can be observed.
- The high risk thresholds represent conditions where ecologically significant loss of function occurs (i.e., an ecological limit has been crossed).
- The thresholds identified are scale dependent, and were determined from the regional down to the site level. The lower levels (e.g., site level) are defined with knowledge of the higher levels (e.g., landscape/regional), and may no longer be appropriate if the higher level targets were changed.
- Ecosystem-based management requires a low risk approach overall (i.e., at the sub-regional scale). However, this requirement does not demand a single approach to management. The low risk target and the high risk threshold provide flexibility for multiple management strategies at subordinate scales. However, for ecosystem-based management to be successful, the low risk targets must be met at an appropriate scale, and high risk approaches can only be applied on a few appropriate geographic locations. Additional work will be required to identify those geographic locations where application of the high risk thresholds would not result in loss of overall ecological integrity.
- When activities tend towards the high risk threshold in a particular geographic location, active adaptive management must be used to test the implications and revise planning as necessary. However, the scientific limits of adaptive management must be taken into account. For example, the scientific ability to monitor and predict species loss is often low due to lack of robust sampling ability and/ or statistical power problems. Where an irreversible change may be possible (e.g., local extirpation of a species), adaptive management should not be used as an excuse for taking a high risk approach.

3B.2.1 Indicators

Indicators are measurements that index the state of complex, difficult to assess, ecological functions. Good indicators respond to management actions, are related clearly to the function considered, can be measured or described simply, are relatively insensitive to factors beyond the management actions considered, and are appropriate for the purpose and scale considered (Beasley and Wright 2001).

Indicators can be developed for planning, for monitoring management actions (e.g., seral stage distribution along streams) and for monitoring the effects of actions (e.g., populations of riparian-associated organisms). Planning indicators usually describe landscape state and include spatial summaries (e.g., equivalent clearcut area) that can be read from maps and projected through time. These indicators can be used as the basis for risk assessment.

Several indicators may be used for each element or function. The EBM Handbook selects a subset of indicators at three main scales (territory/ subregion, landscape/ watershed, and site) to minimise complexity and redundancy (EBM Handbook Sections 4, 5, 6).

3B.2.2 Determining Appropriate Risk Classification

The EBM Handbook has attempted to provide a precautionary approach to management over a large geographic area. It provides a general framework that can be used for setting management approaches and targets across sub-regions. Where flexibility is provided in the targets, the application of high risk levels should be tempered by consideration of an ecosystem's importance, influence and abundance. Some ecosystems are more important for maintaining a particular function than others (e.g., valley bottom ecosystems are usually more ecologically diverse and associated with more rare plant communities than upslope ecosystems). Other ecosystems exert a strong influence elsewhere, particularly downstream (e.g., small steep streams provide more organic material to a system than do ponds). Finally, some ecosystems are especially abundant, defining the character of a region (e.g., bogs in the Hecate Lowland), or are especially rare. In addition, more sensitive landscapes, watersheds and ecosystems requiring a higher degree of protection should be assigned lower risk targets.

Risk targets are also influenced by the larger landscape context. Where high risk conditions occur in surrounding areas (at any scale) because of past modification, lower risk targets should be used.

Generally, acceptable levels of risk will be determined at the strategic level through public processes (e.g., Land and Resources Management Plans and First Nations Land Use Planning). Planning teams will set risk targets within sub-regions and landscapes to guide decisions at lower scales. Planning teams may be more specific and also set targets for specific watersheds or hydroriparian ecosystems. While this is a planning table decision, it should be informed by the Ecosystem-based Management Planning Guide. In this way, ecological objectives can be balanced with other resource management objectives to best fit with different ecological characteristics and conditions at various scales. Basic information provided in the guide could inform tables about where risk increases rapidly and about the types of risks encountered in different sub-regions. Tables could then examine the current level of risk in particular areas and model changes to risk resulting from different management options. Modelling exercises could demonstrate how decisions affect activities at the watershed and site level. Interactive watershed simulation games, with animations of future conditions, could be useful to facilitate understanding of trade-offs and to identify experimental options for adaptive management.

Acceptable levels of risk should be linked to the adaptive management strategy. For example, a planning team may accept a moderate level of risk in a certain area provided it is linked to a well-designed adaptive management plan that will document the consequences of the actions taken. Hence, levels of risk should be tied to the level of commitment to a well-designed adaptive management program with secure long-term funding. The limits of adaptive management should always be considered when making such trade-offs.

3B.2.3 Determining Actual Risk

Risk curves are explicit hypotheses about how management activities influence ecosystems. Explicitly drawn risk curves are useful to summarise current knowledge and to force consideration of uncertainty. Without explicit risk curves, management decisions are based on hidden, implicit risk curves, often confounding values with knowledge. Explicitly drawn curves can also help separate knowledge from values in multi-stakeholder discussions. In planning processes, different stakeholders have different implicit risk curves, complicating debate. A set of common risk curves illustrating current knowledge can facilitate discussion. Alternatively, management could consider that current knowledge is insufficient to even draw hypothetical curves and begin with a hypothesis of no information about the relationship in question (such an assumption could still be drawn explicitly, with uncertainty bands covering all possibilities). This latter approach seems to avoid making use of the considerable research and management experience to date.

Risk curves within this guide are based on published literature, as summarised in the technical reports, and on expert opinion gathered at specially convened risk workshops. In some cases, expert opinion was used to modify more general curves to apply better to the ecosystems of the CIT region. In other cases, where published literature was sparse or ambiguous, expert opinion was used to draw preliminary curves based on the little information available. Price and Church (2002) list workshop participants and describe workshop methodology for Hydroriparian curves. Precautionary guidelines correspond either with indicator values at which risk curves become steeper, uncertainty about risk increases, or, for linear curves, values between the very low and low risk categories on a 5-point scale.

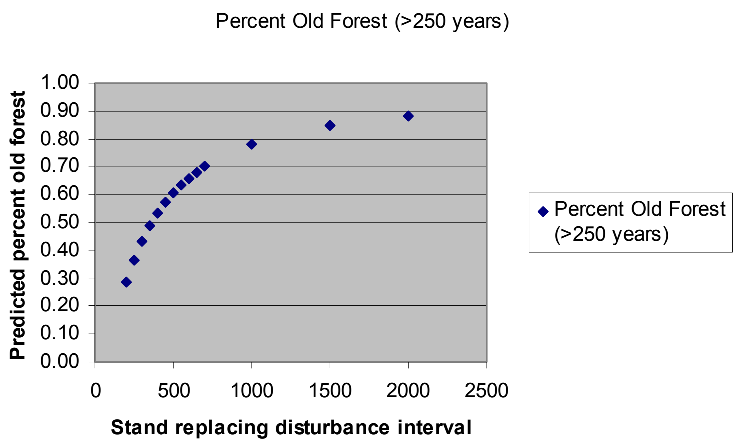
In early stages of development of the EBM Handbook, the specification of quantitative risk curves for application to forest management planning decisions was still novel. Very few data were presented in a manner appropriate for developing the curves; hence the reliance on expert opinion. In most cases, data were insufficient to draw different curves for different landscapes, watersheds or ecosystems.

All the curves provided remain hypotheses and empirical testing and refinement of these curves is still necessary. Carefully designed adaptive management experiments and monitoring will be critical to improving the general curves and to designing suites of curves that can be applied to different types of landscapes, watersheds and ecosystems. Over time, it should be possible to differentiate between risk curves in landscapes or watersheds or different character, based on appropriate indicators. In some cases, local knowledge may already be sufficient to redraw general risk curves to be more specific for landscapes or watersheds of a particular character. Providing that the rationale is documented and the newly drawn curves are used as a basis for adaptive management, such refinements are within the intent of EBM.

There is no easy or transparent manner to combine risks to different functions. Planners will need to decide which function (e.g., hydrology or biodiversity) are of most concern within particular geographic areas or relevant to particular decisions.

Risk is expressed as a function of an indicator value, derived from risk curves. For indicators of ecological function, the planning guide uses natural disturbance regimes as the benchmark for comparison. The approach assumes that indicator values lying closer to values associated with natural systems (i.e., the mean of the natural variability) pose the lowest risk, and that risk increases as values diverge further from natural. In reality defining the full range of natural variability can be difficult. However, in coastal ecosystems with infrequent disturbance, sensitivity analyses show little difference between using means and a range as benchmarks (Figure 5; Holt and Sutherland 2003). The figure shows that the predicted percent of old forest varies little whether disturbance is, for example, 800 or 1,000 years, and so describing the mean or the range has relatively little influence on the predicted amount of old forest. This would not be the case if the study area were defined by a much higher frequency of disturbance (e.g., between 200 and 600 years where predicted old forest varies between 29–66%) or by high variability in natural disturbance patterns. In this case a more complex approach that fully described the range would be required.

Figure 5. Change in predicted amount of old forest with different disturbance intervals.



Ecologically based risk curves are often assumed to be sigmoidal (S-shaped, with areas of relative insensitivity at both extremes on the X-axis joined by a steep portion; for example, see curve under representative old forest in Section 4). Levels of uncertainty, however, mean that evidence may be insufficient to distinguish a linear from a sigmoidal curve. For management purposes, thresholds (where the curve becomes steeper) are most crucial to identify as these represent areas where a small change in management can change risk substantially. In some cases, change may be sudden and obvious (a step function); more often, change will be more subtle, gradual—and hence more difficult to detect precisely.

3B.3 Ecological Benchmarks: Range of Natural Variability

The following report within the compilation provides more detailed information:

Holt, R.F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity. Final report to the North Coast LRMP.

Determining the probability of success in maintaining ecological integrity requires an appropriate benchmark for comparison. Given ignorance about the specific needs of most species and processes, many authors have identified the 'range of natural variability' as an appropriate option (Morgan et al. 1994; Province of BC 1995; ,Haynes et al. 1996; Cissel et al. 1999; Landres et al. 1999; Swetnam et al. 1999). Use of the range of natural variability assumes that species have adapted to the range of habitat patterns resulting from historical disturbance events and that the probability of survival declines as habitat elements and patterns move beyond this range (Jensen and Bourgeron 1994; Bunnell 1995; Cissel et al. 1998). Using recent, ecologically relevant conditions to provide context and guidance for current and future land management assumes that the closer current landscapes are to the range of natural variability, the higher the probability of maintaining ecological integrity (Swanson et al. 1994; Cissel et al. 1998; Landres et al. 1999).

The range of natural variability is a key concept of EBM (Handbook Section 2.4.2). The EBM Handbook uses the range of natural variability and **risk assessment** to determine the amount of each **representative ecosystem** that should be old and to guide harvest pattern. Estimation of the range of natural variability is part of sub-regional assessments (Handbook Section 4.2, 4.3).

The range of natural variability can identify boundaries, beyond which the implications of pushing the ecosystem become unknown and potentially unpredictable. It can provide guidance about the amount of **representative ecosystems** to maintain, about **landscape pattern** and about structural retention within cutblocks. When historic data are lacking, 'pristine' landscapes can possibly be used as reference ecosystems to set benchmarks.

Natural range of variability, current landscape condition, and desired future condition (as set by land managers under social policy) can be compared to clarify management direction (Landres et al. 1999; Holt and Sutherland 2003). Where desired future condition is outside the natural range of variability, the risks associated with attempting to reach this goal should be explicitly acknowledged in the decision-making process (Morgan et al. 1994), and social goals should perhaps be re-evaluated because of the likely ecological and economic impacts of trying to maintain the landscape outside the natural range (Landres et al. 1999). Where current conditions are outside the natural range of variability, **restoration** may be the appropriate management tool. Where current conditions, and desired future conditions are within the range of natural variability, management can likely be maintained (Landres et al. 1999).

It is crucial to choose the spatial and temporal scale used to analyse the range of natural variability carefully. Assessment should be performed over relatively consistent climatic, edaphic, topographic and biogeographic conditions (Morgan et al. 1994; Landres et al. 1999). In the CIT region, sub-regions or BEC variants are the most appropriate large-scale units, and site series are the most appropriate fine-scale unit.

Some authors have criticised the term "range" (e.g., Landres et al. 1999) because range includes rare events. Depending upon the element, other parameters (e.g., mean or median with a measure of variability) may be more useful. Application of the range of natural variability concept provides a suite of options to the manager while increasing the probability of maintaining ecological integrity. In some of the CIT region, particularly the hypermaritime BEC subzones, the range of natural variability is extremely narrow and provides little in the way of options.

Limitations of the concept include lack of historical data, difficulties in interpreting historical data, possibility that future conditions may be without precedent and the difficulties of synthesising over appropriate multiple spatial and temporal scales (Swanson et al. 1994; Swetnam et al. 1999). There are also practical reasons limiting close emulation of natural disturbance patterns. In particular, if there is a large discrepancy between the rate of natural disturbance and the rate of harvesting disturbance, altering the pattern of harvest to emulate gap dynamics (i.e., partial harvest) but not the rate would result in management activities being distributed over a much greater area in a given period. Many more stands would be impacted, and the extra roads required to reach these dispersed patches may also impose high ecological costs.

3B.4 Estimates of Natural Stand-replacing Disturbance Intervals

Price and Daust (2003) estimate the frequency of stand-replacing natural disturbances for the CIT area using existing databases (Table 7). They divide the region into units with relatively homogeneous disturbance frequency and use three methods to estimate the mean disturbance frequency in each, excluding non-forested areas such as alpine parkland and avalanche tracks. Return intervals are calculated as the inverse of disturbance frequency. The return intervals presented in Table 7 represent the uncertainty around the estimated mean return interval for stand-replacing disturbances (resolution down to about 1 hectare). Because natural disturbance frequency is so low in some areas, estimated upper values for return intervals are tens of thousands of years. In these areas, many stands are not replaced by fire, wind or geomorphic disturbances for millennia. Within stands, however, gap dynamics function continuously during these intervals to create the multi-aged character of coastal stands

Table 7. Estimated return intervals for stand-replacing disturbances for three groups of site series within four physiographic regions of the CIT area (not including Vancouver Island)

Region	Site series group	Return interval*
Hypermaritime	Upland	4,500 – 10,000
	Fluvial	2,200 – 10,000
	Ocean spray	1,000 – 5,600
Outer Coast North	Upland	1,800 – 10,000
	Fluvial	500 – 2,100
Outer Coast South	Upland	900 – 2,500
	Fluvial	400 – 1,400
Inner Coast	Upland	500 – 5,600
	Fluvial	300 – 900

* Rounded to the nearest 100 years; note that numbers above 10,000 years have been truncated.

The variation in disturbance intervals across the CIT region provides a convincing argument for stratifying the region into relatively homogeneous areas. Even within the units summarised in Table 7, there will be variation, some of which is known locally. For example, within the CWHvh1, the frequency of larger scale blowdown is much greater for hemlock-amabilis fir stands than for cedar-hemlock stands (Scott 2001).

3B.5 Ecological benchmarks, natural disturbance processes, and implications for EBM

The following reports within the compilation provides more detailed information:

Dorner, B., and C. Wong. 2003. Natural disturbance dynamics in the CIT area. Report for the CIT.

Price, K. and Daust, D. 2003. The frequency of stand-replacing natural disturbance in the CIT area. Report for the CIT.

The **range of natural variability** in disturbance regimes are a central concept in ecosystem-based management.

> The EBM Handbook uses estimated range of variation in natural disturbance to determine the amount of each **representative ecosystem** that should be old and to guide harvest patterns (Handbook Sections 4.2, 4.3).

Section 2 of the Compendium described the prevalent types of natural disturbance within the CIT region. This section lists some of the implications of these disturbances to maintaining ecological integrity in the CIT region.

1. Planning and management should stratify the CIT region into areas dominated by different disturbance types and regimes.

Landscape dynamics on the coast are strongly influenced by the underlying mosaic of physiography and topography.[10] The paradigm of a shifting mosaic of seral stages[11] that underlies the management approach proposed in the Biodiversity Guidebook (Province of BC 1995) does not fit this type of landscape very well, because much of the early seral vegetation is confined to a few areas subject to repeated disturbance, whereas most of the operable, productive forests of interest to harvesting rarely experience severe, large-scale disturbance. A planning approach that fails to stratify analyses appropriately will make it difficult to incorporate knowledge about natural dynamics in a meaningful way.

> In the EBM Handbook, analysis of natural disturbance will be performed by sub-regions and finer units

2. Planning and management should consider very long time frames.

The coastal forest is a system of long time scales. Tree species are long-lived, intervals between stand- replacing disturbances are measured on the scale of centuries, and processes of decay and

[10] For references, see Dorner and Wong (2003).

[11] This paradigm applies to a landscape where forest dynamics are similar across the landscape and occurrence of natural disturbances is not restricted to particular topographic or other site characteristics. In such a landscape, the spatial distribution of young and old forest throughout the landscape is similarly independent of topography or other site characteristics, and the forest mosaic therefore 'shifts' at random across the landscape over time as disturbances create new openings and older openings gradually merge back into the matrix of old forest.

decomposition progress equally slowly. Thus, undesired effects of forest management, such as potential negative impacts on terrestrial and aquatic habitat because of reduced inputs of large woody debris, may take a long time to manifest and even longer to reverse.

The Biodiversity Guidebook, used to guide ecological planning since 1995, overestimates disturbance frequency over much of the CIT region. The available evidence indicates that, in productive and operable forests, disturbance return intervals for natural, large, stand-replacing events are much longer than commercial rotation periods.

> The EBM Handbook considers long time periods in using the **range of natural variability** as a benchmark for the amount of **reserves**.

Climate also drives coastal forest dynamics. Predicted changes in future climate will likely increase the incidence and/or severity of most disturbance agents and affect how forests respond to these agents and other processes.

3. Variable retention with high levels of retention is an appropriate silvicultural system.

Because coastal forests are primarily driven by gap dynamics, natural disturbance regimes provide a limited template for clearcut silviculture systems, or traditional approaches to even-aged management in general. If the objective of forest management is to emulate natural disturbance patterns, a silvicultural system that emulates a gap-phase replacement regime, such as variable retention with high retention levels, is the closest match for extensive use (Pojar et al. 1999). In cases where high retention levels are rejected because of economic considerations, the potential risk of negative ecological impacts from silvicultural practices that do not maintain natural structure and dynamics may be reduced through extended rotations, and by constraining the proportion of the landbase over which harvesting occurs. However, if a reduction in harvesting landbase is to be offset by more intensive management zones, it will be important to carefully consider the allocation of such areas, because in many cases the most productive areas from a timber growth perspective also tend to be of primary importance for biodiversity.

> The EBM Handbook recommends variable retention, with a variety of levels of retention, over the entire landbase (Handbook Section 6.4).

4. Management should retain structural elements in harvested stands.

Standing dead and downed wood are important habitat elements in both terrestrial and aquatic ecosystems. Natural disturbances generally ensure an ample, sustained supply of snags, root wads, and downed wood. Reduced recruitment of snags and large woody debris in managed stands may have long-term effects on nutrient cycling, hydroriparian processes, tree regeneration, and habitat quality (see variable retention in Section 4).

Productive valley-bottom areas and estuaries often have high timber value, as well as very high habitat value, and are thus especially vulnerable. Narrow riparian buffers are susceptible to blowdown and may not provide effective protection of hydroriparian ecosystems (Price and McLennan 2002). To sustain input of large woody debris into stream channels at historical levels and allow for the meandering of floodplain channels throughout the floodplain area, most, if not all, of the valley bottom will require protection in some cases.

The EBM Handbook recommends protection of all hydroriparian ecosystems within the transportation process zone with windfirm buffers.

5. Uncertainty about future disturbance rates means that monitoring will be necessary.

Most of the interactions between harvesting and natural disturbance on the coast are synergistic, i.e., harvesting increases the frequency and/or severity of natural disturbances (Dorner and Wong 2003). Harvesting can promote the occurrence of landslides and erosion, and increase the severity of flooding. By opening up the canopy, harvesting also makes stands more susceptible to blowdown, and wounding of trees during harvesting operations encourages fungal infections. Furthermore, several disturbance agents, including root rots, porcupines, and some defoliating insects, are more prevalent in harvested landscapes because they primarily affect younger, second-growth stands. On the other hand, harvesting is likely to reduce the incidence of disturbance agents that are slow to develop or affect primarily older stands, such as heart and butt rots and mistletoe.

Climate change will likely also increase the frequency or severity of many natural disturbance types (Dorner and Wong 2003). Whereas predictions of climate change once varied widely, most predictions now suggest increased temperatures, increased winter storm severity, and increased flooding. Because of climate change, overall disturbance rates are likely to go up in the future even without harvesting, and harvesting will almost certainly lead to disturbance rates substantially higher than the area has historically experienced. Appropriate harvesting and management will have to change to accommodate new disturbance regimes. Retention of connected old growth may need to increase to allow species to move as conditions change. Uncertainty emphasizes the need to monitor conditions to recognize and avert potential negative impacts on ecological integrity.

Monitoring is a central concept in the EBM Handbook (Handbook Section 2.5.7)

3C: Management Options:

This section provides an overview of how science can aid in reserve design, in management of the 'managed' forest landbase, and in determining restoration needs and priorities (see Figure 4 for overview).

3C.1 Reserves

The following reports within the compilation provide more detailed information:

Holt, R.F. 2001a. An ecosystem-based management framework for the North Coast LRMP. Background report for the North Coast LRMP.

Rumsey, C. and many others. 2003. An ecosystem spatial analysis for Haida Gwaii, Central Coast and North Coast British Columbia. Report to the CIT.

Reserves are a key concept of ecosystem-based management. The EBM Handbook includes guidelines to create reserves based on analysis of representative ecosystems, gap analysis, rare ecosystems, and rare and focal species at watershed, landscape and sub-regional scales (Handbook Sections 4.4, 5.5, 6.4).

The importance of reserves, or protected areas, as a central feature of ecosystem-based management is well documented (Margules et al. 1994; Dellasalla et al. 1996; Noss 1996a). The purposes of watershed, landscape and sub-regional reserves include i) providing refuges for natural processes, ii) retaining representative samples of ecosystems (for use as benchmarks) and iii) providing core habitat for sensitive species (though maintaining species solely within a reserve system is considered impractical and likely does not meet the requirements of maintaining ecological integrity). It has been well demonstrated that *ad hoc* reserve selection is inefficient (Pressey 1994; Pressey et al. 1996). Three main approaches for reserve selection have been suggested: i) gap analysis for representation (e.g., Burley 1988; Scott et al. 1993), ii) reserve selection algorithms (e.g., Pressey and Nicholls 1989; Margules et al. 1994) and iii) protection of special elements (Myers 1990; Noss 1999b). Protected areas that have other management foci (scenic, recreation, etc.) may fail to meet biological objectives (Noss and Cooperrider 1994).

Reserves can have fixed locations (e.g., rare ecosystems, hydroriparian ecosystems, unstable terrain) or can be flexible (e.g., representative ecosystem targets, old seral targets, corridors), moving over the landscape through time. Fixed-location reserves form anchors for flexible reserves.

The EBM Handbook considers fixed reserves and two types of flexible reserves (variable and target). Handbook Section 5.4.

3C.1.1 Gap Analysis

The EBM Handbook recommends gap analysis as part of sub-regional assessment (Handbook Section 4.4).

Gap analysis identifies the ecosystems, or other elements, missing from existing systems of reserves (Margules and Pressey 2000). It is a relatively straightforward evaluation that uses available data. The scale of analysis impacts the results, and must be appropriate to the size of planning area and the variation in ecosystems throughout the area.

Gap analysis can be used within a **coarse filter** approach (e.g., Burley 1988; Iacobelli 1995) to identify under-represented ecosystems. Effective gap analysis considers i) the potential need for over-representation of rare, ii) increasing the representation of widely impacted ecosystem components, iii) assessing structural/seral stage. In much of the CIT region, where the timber harvesting landbase is limited to low elevation areas, gap analysis and **ecosystem representation** in combination with **risk assessment** helps determine suitable areas for **reserves** at the sub-regional scale.

Gap analysis can also be used as part of a **fine filter** approach, evaluating ecosystem elements according to rarity, uniqueness, viability, diversity, vulnerability and significance (Noss et al. 2002).

Coarse-filter, and some aspects of fine-filter, gap analyses have been completed for the CIT region (Ronalds 1995, Prince Rupert Regional Protected Areas Team 1996, Lewis et al. 1997, Jeo et al. 1999, 2002). The results of these analyses provide a basis for making regional scale reserve decisions, and will aid in application retention targets at lower planning scales.

3C.1.2 Reserve Selection Algorithms

Reserve selection algorithms have been designed for use mostly where there are no existing reserves. They select, from a predetermined pool of choices, the combination of reserves that matches selected ecological criteria most efficiently. The approaches applied to real situations (mostly in Australia and South Africa) have used the concept of 'complementarity'—where the first reserve chosen has highest ecological value (often number of species), and further reserves are added chosen based on the maximum value. Refinements of this process have included using species abundance rather then presence/absence data to deal with maximising population viability (Turpie 1995), and considering locating reserves to consider meta-population dynamics (Nicholls and Margules 1993). Limitations of the process include the necessity for species distribution data or other information on ecosystem distributions. However, a combination of these concepts, even when applied informally, can aid in small- and large-scale reserve selection (e.g., Holt 1998, 2000).

The CIT process includes potential reserve selection using the SITES algorithm as part of the Ecosystem Spatial Analysis.

Rumsey et al. (2003) describe the selection algorithms used in the CIT Ecosystem Spatial Analysis in detail. Briefly, the tool that was used "SITES" is an automated site selection algorithm that attempts to maximise attaining conservation goals using the fewest planning units and smallest overall area possible. SITES first examines the conservation value of each planning unit, then selects groups of units to meet stated conservation goals. The algorithm adds and removes units iteratively to find the most efficient option, incorporating a random element so that initially unpromising options can be reconsidered later.

3C.1.3 Special Elements

Special elements can include many different variables: species richness (hotspots of diversity), rare species or ecosystems, critical habitat, etc. Difficulties in reserve design using this method are often based on lack of data about the specific elements in question. Examining special elements is a fine filter approach, and should be included as part of an analysis for reserve selection, and within a fine filter approach.

3C.2 The Managed Forest

The following reports within the compilation provide more detailed information:

Church, M. and B. Eaton. 2001. Hydrological effects of forest harvest in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #3.

Holt, R. F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity. Final report to the North Coast LRMP.

In forested regions used extensively for timber extraction, large reserves are necessary but not sufficient to ensure maintenance of ecological integrity; management of ecosystems outside reserves is also important (Franklin 1993; Lindenmayer and Franklin 1997; Norton 1999; Fahrig 2002). The managed landbase can perform three vital roles in the maintenance of biodiversity (Franklin 1993; Parminter 1998): a) providing habitat at smaller scales and throughout the landscape, b) buffering reserves and therefore increasing the effective reserve size, and c) providing connectivity within the landscape and between reserves. The connectivity of protected areas to other unmanaged areas is often dependent upon activities in the managed forest (Margules and Pressey 2000; Boutin and Hebert 2002; Fahrig 2002).

> The EBM Handbook recommends applying reserves at multiple scales within the managed landbase, and using variable retention within all stands (Handbook Section 4, 5, and 6.2).
>
> Within these multiple scales, areas with higher ecological values should be managed with higher levels of precaution, or lower risk.

Typically, in British Columbia, the landbase outside reserves is divided into management zones with differing emphases. The EBM Handbook is consistent with a zoning approach, but does not describe explicit categories. Managing areas for different levels of risk below the high risk threshold allows for operational flexibility; achieving low-risk targets at higher scales ensures that cumulative risk remains low. The EBM Handbook recommends that areas with higher ecological value be managed with higher levels of precaution. Following the principles of ecosystem-based management, however, and taking guidance from the coastal patterns of natural disturbance, the EBM Handbook requires some retention at all scales at all locations. Because all harvesting in the CIT area will use variable retention, the managed forest will contain some elements (particularly structural elements) of old forest.

There is ample evidence that many species from a wide range of taxa are closely associated with habitat elements more common in older forests. These elements typically include large live trees, large and small dead trees, down wood, and well-developed forest floor. Other elements such as deciduous trees and shrubs, although not necessarily associated with old forest, are elements of natural forests that are also critical for some species. Bunnell et al. (1998) review the associations of habitat elements for vertebrates, and Bunnell et al. (2003) expand that review to include lichens, invertebrates, fungi and plants. We do not repeat their findings here, but repeat their summary, that retaining late-successional attributes in amounts that are not limiting to sustain native populations is an important goal for ecologically sound management. There are two scales at which this goal should be addressed: the landscape scale and the stand scale. The larger scale is addressed by reserves of various types and limits on the amount of harvesting. At the stand scale, these attributes are maintained by retaining portions of stands during harvest. While retaining structures provides habitat immediately for some species, the a primary goal is to help the regenerating forest sustain a larger suite of species earlier in its successional path than it otherwise would. Hagar et al. (1995) noted that while 25 vertebrates seem to reach maximum densities in older seral conditions, most of those species require only a single habitat element to inhabit earlier seral conditions. The same phenomenon is likely true for groups other than vertebrates, though data are lacking. The amount of suitable habitat for many species can be increased and become available more quickly simply by retaining structures during harvesting. Additionally, because the managed landbase will be put on a rotation length considerably shorter than that required to develop "old growth" (e.g., approximately 120–140 years compared to 250 plus years), retained stand structure should be sufficient to maintain at least some old forest elements through time. This aspect of structural retention requires that elements are to be retained for a minimum of one rotation and preferentially indefinitely.

On the coast, the appropriate configuration of retained trees varies (Bunnell et al. 2003). While some sort of group retention is beneficial for many species, others benefit from dispersed retention, and others would benefit most from aggregating cutting to reduce roads and limit the extent of disturbance at any time. For example, ectomycorrhizal fungi and other organisms that dwell below-ground may benefit most from the coverage of root zones allowed by retaining dispersed trees, but dispersed retention may not benefit some passerine birds because of predation from aerial predators that perch on the dispersed trees. Small patches may allow lichens and mosses to persist by ameliorating effects of desiccation. Larger patches may house more species of birds than smaller patches or dispersed retention (beyond the effects of simple species-area curves). Minimizing within-stand retention so that harvest can be clumped to allow fewer roads, larger openings, and less disturbance by humans may be the most appropriate pattern for ungulates and larger carnivores sensitive to human disturbance. One of the keys to retaining structure in an ecologically appropriate way is to ensure a variety of pattern of retained structure over the landscape. In general, patch retention seems to favour the most species, but given the variety of species requirements it is appropriate to use a range of opening sizes, and a range of sizes of retention patches.

Appropriate amounts of retention cannot be assessed at one scale. Effects of stand level practices combine with landscape level and sub-regional level practices. The entire range, from zero within-stand retention to high proportions of retention, will likely be appropriate across the entire landscape. Where the landscape context retains higher proportions of unmanaged or relatively natural stands, then within-stand retention is less important than in landscapes that are extensively managed. The less unmanaged forest retained at the landscape scale, the higher the recommended retention at the stand level. Longer rotations can be used as a means of recovery in cases when low retention levels are applied (Perry 1998).

3C.2.1 Access

A potential negative consequence of high levels of retention applied over wide areas is the increased length of road required to access the same volume of timber. Nelson and Finn, and Dunsworth and Northway (in Bunnell et al. 1999) modeled road construction under a variety of forest practices rules. The amount of road needed increased as cutblocks became smaller or as green-up increased, because more roads are built to access the same amount of timber. Generally as cutting becomes more dispersed more roads need to be built and maintained, fewer roads can be shut down or "put to bed". While improved placement, construction practices and culvert use have reduced negative impacts of roads, they still often alter hydrogeomorphic processes (Church and Eaton 2001) and have several affects on organisms (e.g., Gucinski et al. 2000; Trombulak and Frissel 2000; Hamilton and Wilson 2001). This trade-off is important to consider when basing management on natural gap dynamics.

Most studies evaluating the effects of roads on organisms have focused on direct mortality by vehicles and roads as access for predators, including humans. Fewer studies have examined roads as barriers to movement (Clancy and Reichmuth 1990; Furniss et al. 1991) or as agents creating edges effects on physical and ecological processes. Bunnell et al. (1999) summarized the literature on roads to that date, noting the importance of roads as avenues for vehicular mortality and predators. They also note several studies documenting effects of disturbance of traffic on large animals (studies of disturbance on small mammals are largely undocumented) that can have far-reaching effects particularly where roadside cover is low (400 m disturbance for elk and deer). Roads can either facilitate or hinder movement and have been recognized as barriers to large and small animals, depending on the size of the road (Mader 1984; Swihart and Slade 1984; Merriam et al. 1988). Anecdotal information suggests roads often become travel corridors for predators, as well they are documented avenues for the spread of exotic plant (and animal) species. Generally roads create high contrast edges that do not decrease with time, although smaller roads and roads that are deactivated create smaller edge effects than do larger roads. However, edge effects around roads of various widths have not been compared in the literature.

Because of the several well-documented negative effects of roads (increased mortality, disturbance barriers to movement, and facilitation of spread of exotic plants) are associated with roads of any sizes, the major management effort should be to reduce overall road density, and particularly the density of active roads. This argues for concentrating harvesting rather than widely dispersed partial cutting for the same amount of timber volume. Secondarily, roads should be constructed the minimum safe width to minimize barriers to movements and negative edge effects. Proper location of roads and design of culverts to avoid negative downstream impacts of road failures and sedimentation are critical to reducing road impacts on fish survival (Scrivener and Brownlee 1989)

Some biologists consider impacts of roads to be greater for some species than the impact of habitat alteration, so attention to road density is a key concern. This is particularly true for species that are hunted or poached. Grizzly bears are particularly sensitive because of their low reproductive rates, road-associated mortality (hunting mostly), and displacement from areas near roads (McLellan and Shackleton 1988).

3C.2.2 Constrained Areas

In the CIT region, not all area outside reserves will be harvested. Approximately 74% of the area in the Central and North Coast planning areas are currently estimated to be constrained from harvesting for various reasons (Holt and Sutherland 2003; Wells et al. in review). While representing only a subset of ecosystems, these areas will also contribute towards maintaining ecological integrity (e.g., Huggard 2001). However, analyses including these areas as "inoperable" must also remain cognisant that their status is very different from that of a protected area. In this particular case the vast majority of the "inoperable" area is currently considered not economically viable but this may change overnight as markets change, as wood supply decreases, or as the use of wood becomes more efficient (e.g., through value-added industries).

3C.2.3 Management Practices

Over the past decade, basic forest management and road-building practices have improved considerably. A number of guidebooks document best management practices for various activities.

The EBM Handbook does not prescribe specific practices, but assumes that best management practices will be met everywhere.

During the CIT process the arena of normal forest practices has changed considerably in British Columbia. In particular there has been a move from a regulatory system to the results-based system. It is not yet clear the extent to which Best Management Practices will be maintained.

3C.3 Restoration

Ecological restoration is "…the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed." (SER 2002).

Active restoration includes a specific activity aimed at restoring an ecosystem or ecosystem component. Passive restoration allows the natural restoration of an ecosystem or ecosystem component. In each case, the intention is to return an ecosystem to its natural trajectory (SER 2002), and therefore the use of natural conditions as benchmarks is a key facet of ecological restoration.

A successful restoration program includes identification of a) an appropriate restoration goal, b) barriers that may prevent the restoration goal been reached (e.g., ongoing human activities), and c) an endpoint where restoration will be said to have been achieved (detailed overview in Clewell et al. 2000).

In an assessment of restoration needs in British Columbia, ecological experts were canvassed to identify those ecosystems most in need of restoration for each region of the province (Holt 2001b,c). Relevant to the CIT region, both landscape pattern and stand level management were identified as issues within the coastal temperate rainforest where restoration activities would be required to increase ecological integrity, due to historic management activities.

The EBM Handbook requires restoration activities when current landscape conditions are below precautionary targets. (Handbook Sections 5.5, 7.3). In landscapes where historic activities have resulted in deviation from natural at all scales, specific activities to restore elements such as stand level legacies or connectivity may be necessary.

For many areas where current conditions are below a management targets the restoration is most likely to take it passive form. In particular site series or site series groups that are currently underrepresented should be removed from the timber harvesting landbase or put on to very long rotations so that old forest elements can recover. Initial regional, landscape and watershed analysis should highlight those ecosystems where some form of representation will be required. Note that this is one example where an initial analysis using a site series surrogate (e.g., analysis units) may be necessary because of the lack of good site series information regionally, however as the scale of analysis becomes smaller more ecologically relevant information should be used.

Restoration activities are costly and often somewhat uncertain. Once an ecosystem has been removed from its natural trajectory, there is little evidence that the appropriate conditions and trajectory can be recreated. For example, invasive species are often associated with ecosystem degradation and can be responsible for significantly altering biological processes. Haida Gwaii within the CIT area is already heavily impacted by invasive species making restoration of natural forest succession extremely difficult if not impossible. Preventing the need for restoration is the most cost-effective approach to maintaining ecological integrity.

3D: Dealing with Uncertainty

Uncertainty is inevitable in resource management. Ecosystems are complex, dynamic and unpredictable (Taylor 2000a). Several approaches exist to address uncertainty while making decisions. Risk assessment was described in detail under Section 3B; the following section discusses adaptive management and the precautionary principle, and introduces some important analysis techniques.

Risk assessment, adaptive management and the precautionary principle are intimately related. To account for uncertainty, the precautionary principle calls for a low-risk approach. Often, information is insufficient to assess risk accurately. In these cases, ecosystem-based management decisions will be based on the precautionary principle, and will choose an activity level hopefully below the point at which risk becomes high (i.e., likely to result in loss of ecological functioning). Adaptive management attempts to reduce uncertainty about risk by systematically improving knowledge—in essence, it tries to refine risk assessments. To decrease uncertainty, adaptive management should be applied whenever feasible to planned low- or higher-risk activities. Consciously chosen higher-risk activities should, however, always be accompanied by carefully designed adaptive management. Where rigorous adaptive management is not possible, ecosystem-based management activities should follow the precautionary principle. The targets and thresholds discussed in Section 4 include low-risk precautionary guidelines and targets, specific risk relationships (where known) and suggestions for adaptive management.

3D.1 Adaptive Management and Monitoring

The following reports within the compilation provide more detailed information:

Hydroriparian Planning Guide. 2003. Appendix 9 lists questions that could profit from adaptive management

Taylor, B. 2000a. An introduction to adaptive management. Background Report to the North Coast LRMP.

Taylor, B. 2000b. Implementing adaptive management through the North Coast LRMP. Background Report to the North Coast LRMP.

Adaptive management is a systematic approach to improving management and accommodating change by learning from the outcomes of management interventions (Taylor et al. 1997). With adaptive management, not only are objectives and policies adjusted in response to new information, but also, management policies are deliberately designed and implemented as experiments to enhance the rate of improvement (Holling 1978, Walters 1986; Taylor et al. 1997). Note that the EBM

Handbook discusses adaptive co-management. This terminology has been suggested to improve the changes of adaptive management coming to fruition, however, it is not commonly used in the ecological literature, and so is not included directly here.

Adaptive management and monitoring are key concepts in ecosystem-based management (Handbook Section 2.5, 3.4).

Adaptive management is useful when there is significant uncertainty about the best management option to achieve a desired future condition and when resolving the uncertainty will improve management (Taylor 2000a, 2000b). Taylor (2000b) provides a decision tree to guide decisions about when to apply adaptive management to reduce uncertainty. When adaptive management cannot be designed to resolve uncertainties, a precautionary approach may be more appropriate (Taylor 2000a).

Successful adaptive management requires a large commitment. Designing experiments with adequate statistical power over large spatial scales and in systems with high natural variability is challenging. Continuing effective monitoring over sufficiently long time periods is also difficult. Although many processes plan to use adaptive management, few are successful (Taylor et al. 1997).

The North Coast LRMP decision tree notes that if uncertainty can only be resolved by adaptive management and if a response will not be detectable within a reasonable time, a precautionary approach should be considered appropriate (Taylor 2000b). Similarly there the Hydroriparian Planning Guide requires that a precautionary approach be used in areas of uncertainty unless adaptive management can be applied.

There are active and passive adaptive management studies occurring in coastal forests that can help inform management options on the CIT region as results begin to accumulate (e.g., Bunnell et al 2003).

Successful implementation of adaptive management will require teams of experts to conceptualise powerful experiments. Rigorous application of classical statistics, based on the altar of the 0.05 P-value, has lead to ecological disasters world-wide. This issue arises because many studies have mistaken the message of statistics. Germano (1999) reviews misconceptions about statistical significance, lists difficulties associated with determining ecological integrity and offers suggestions for alternative methods. The sections below introduce two of the most important considerations for studies examining ecological integrity. Adaptive management studies will need to thoroughly integrate these concepts into their design.

3D.1.1 Power Analysis

Authors have generally concentrated on ensuring that they do not wrongly reject a null hypothesis (type I error; e.g., inferring a population decline when there is none), while ignoring the reverse (type II error; e.g., accepting the null hypothesis of no decline when there is one; Toft and Shea 1983, Peterman 1990). In maintaining ecological integrity, however, the second type of error is often of greater concern, with potentially higher costs. Power analysis determines the ability of a sample to detect a real effect. Only if power is sufficiently high can a null hypothesis (e.g., of no population decline) be accepted. High sample variability and/or low sample size leads to low power (i.e., poor ability to detect a real effect). This phenomenon is particularly pervasive in ecology because of the difficulty of obtaining a sufficiently large sample size, and because of inherent ecological variability, which results in the need for even larger sample sizes.

Assessments of population trends rarely have sufficient power to detect anything but the largest changes. Peterman (1990) cites the example of de la Mare's (1984) study of the data used to determine whale population trends. The data used to assess whale abundance had only a 29–70% chance of detecting a true population decrease of 50% over 20 years—surely a biologically significant level of decrease. In the decade since Peterman's (1990) review, power analysis has been used more regularly in fisheries research and design of monitoring programs (e.g., USGS 1999). Simple calculations reveal that monitoring programs often cannot detect biologically significant population trends in less than 10–20 years.

Peterman (1990) provides recommendations for scientists and managers, including the following: design for high power, consider the detectability of ecologically relevant effect sizes and consider the ecological ramifications of type II errors.

The EBM Handbook recommends that all studies include analyses of statistical power.

Understanding and calculating power will prevent incorrect inferences, but will not solve all problems. There will remain many situations where data are inconclusive (i.e., fail to reject null hypothesis, but have insufficient power to accept null hypothesis). In these cases, Peterman (1990) suggests caution and a reversed burden of proof (i.e., follow the precautionary principle). Even the establishing and testing of null hypothesis can be misleading in ecological fields (Huggard 2003 has several startling examples).

3D.1.2 Bayesian Approaches

The application of Bayesian analytical techniques, which specifically allow an assessment of both expert opinion and uncertainty, can provide information where classical statistics fail (Marcot et al. 2001). Biologically significant population declines that are undetectable using classical statistics, can be detected more readily using Bayesian methods (Ludwig 1996, Wade 2000).

Instead of the strict acceptance or rejection of a null hypothesis, Bayesian analysis can assign intermediate probabilities to hypotheses. In essence, Bayesian analysis is an explicit technique for updating belief in a hypothesis whereas classical analysis can only be used to determine that the data are unlikely given that a null hypothesis is true (when we really know the data are true and want to know how likely it is the hypothesis is true). Bayesian techniques have been used to achieve consensus about the effects of management given divergent initial beliefs (Crome et al. 1996). Bayesian techniques were used for risk analysis in the North Coast LRMP process (e.g., Holt and Sutherland 2003; Steventon 2003) where they were specifically used to incorporate expert opinion and to report out on "the most likely outcome" for a particular scenario.

Bayesian techniques can also be used in formal decision-making analyses, which assess the probability of making various types of error and compare the relative costs of each (e.g., Berger 1985).

3D.2 Precautionary Principle

The precautionary principle is a response to scientific uncertainty (Holling 1986). Because the elements and processes making up ecological integrity vary, are difficult to measure, and may be inherently unpredictable, science, even with unlimited funding, cannot provide certainty (Ludwig et al. 1993). In an attempt to prevent damage to ecological integrity, the precautionary principle has been included in a range of national and international environmental and resource management documents over the past 15 years[12].

The precautionary principle is a guiding principle in the EBM Handbook (Handbook Section 2.2). The EBM Handbook recommends increased precaution in landscapes and watersheds with high ecological or cultural values (Handbook Section 7.0).

Ludwig et al. (1993) summarise a common-sense method of decision making under uncertainty that captures many of the elements of the precautionary principle:

> consider a variety of plausible hypotheses about the world; consider a variety of possible strategies; favour actions that are robust to uncertainties; hedge; favour actions that are informative; probe and experiment; monitor results; update assessments and modify policy accordingly; and favour actions that are reversible. (Ludwig et al. 1993)

A precautionary approach gives the benefit of the doubt to the resource, and reverses the burden of proof (i.e., proponents prove no impact). No longer does a management action need to be shown unequivocally to be damaging before it is judged inappropriate, rather there needs to be some certainty that it will not be damaging to proceed. Often the precautionary principle is implemented by considering **statistical power**, encouraging **adaptive management,** and providing guidance where adaptive management will not occur so that decisions can be made before scientific consensus is achieved (Peterman and M'Gonigle 1992; Ludwig et al. 1993; Kaufmann et al. 1994).

The low risk targets provided by The EBM Handbook were designed after considering "*a variety of possible strategies*" and to "*favor actions that are robust to uncertainties*". Many of Ludwig et al.'s (1993) remaining directions point towards performing adaptive management experiments to improve knowledge.

[12] For example, Government of Canada 2001. A Canadian perspective on the precautionary approach/principle. Discussion document; Principle 15 of the Rio Declaration on Environment and Development, UNCED 1992; the Convention on Biological Diversity; Decision II/10 on conservation and sustainable use of marine and coastal biological diversity, adopted by the Conference of the Parties in Jakarta in November 1995.

Section 4. Maintaining the Ecological Integrity of the CIT Region: Targets and Thresholds

The following section provides specific input on individual targets and thresholds identified within the EBM Handbook. They include

- representative old forest ecosystems
- representative hydroriparian ecosystems and processes (including hydrological regime, stream morphology, unstable terrain, downed wood and hydroriparian biodiversity)
- rare ecosystems
- in-block retention
- hydroriparian process zones
- riparian corridors
- fish habitat

4.1 Representative Old Forest Ecosystems

The following reports within the compilation provide more detailed information:

Dorner, B., and C. Wong. 2003. Natural disturbance dynamics in the CIT area. Report for the CIT.

Dykstra, P. 2003. Habitat supply thresholds: a literature synthesis. Report for MSRM. Draft 2.

Holt, R.F. and G. Sutherland. 2003. Environmental risk assessment: base case. Coarse filter biodiversity. Final report to the North Coast LRMP.

Pearson, A. 2003. Natural and Logging Disturbances in the Temperate Rain Forests of the Central Coast, British Columbia. Report to Ministry of Sustainable Resource Management, Victoria, B.C.

4.1.1 Definition

"Old forest ecosystems" in CIT region are maintained primarily by gap-phase processes. As a result, stands are often considerably older than individual trees. Structural attributes vary by ecosystem. In the Insular and Coast Mountains, most old stands have high vertical and horizontal heterogeneity, and contain large structures (live, standing dead and downed trees). In the Lowlands, old forests are similarly heterogeneous, but most often without large structures. Due to a lack of inventory of structural attributes and of accurate estimates of stand age (particularly in older age classes), old forests are considered to be those mapped as older than 250 years.

Old forest ecosystems vary among sites and across the region. "Representative" old forest will vary with the classification system chosen. This work focuses on site series within BEC variant as an appropriate ecological unit. A number of different units have been suggested as appropriate surrogates where site series are unavailable, which include analysis units (leading species and productivity) , and site series grouped on the basis of plant associations irrespective of their associated biogeoclimatic variant. Each of these approaches may provide some interim information, however their limitations in terms of adequately reflecting ecological variation may also be severe and should be very carefully considered prior to their use. Analyses (e.g., of representation) using these units should carefully consider and state the limitations (e.g., Holt and Sutherland 2003).

4.1.2 Importance of Element

Maintaining representative ecosystems in suitable abundance and distribution is a coarse filter approach to maintaining the biodiversity and ecological processes that make up ecological integrity (Compendium Section 3; Franklin 1993; Noss 1996; Nally et al. 2002). Over most of the CIT region (with the exception of North Vancouver Island), natural stand-replacing disturbances are rare and constrained to susceptible locations and therefore the vast majority or forest stands tend to be very old (Compendium Section 2; Dorner and Wong 2003; Pearson 2003). In these very old, heterogeneous stands, structural legacies of centuries of small-scale disturbances provide a variety of habitats and influence many ecosystem processes including nutrient cycling, tree regeneration, water transport and stream morphology. Based on the assumption that landscapes more closely resembling natural landscapes present lower risk to ecological integrity (Compendium Sections 2 and 3), maintaining adequate amounts of representative old forest is necessary to maintain ecological integrity on the coast.

4.1.3 Impacts of Forest Harvesting

At the landscape level, harvesting removes old forest habitat, often from specific ecosystems (Holt and Sutherland 2003). In addition to habitat loss, fragmentation of remaining habitat also occurs (Debinski and Holt 2000) in a pattern which is very different from that created by natural disturbances (in these landscapes, typically small openings, or vertical avalanches or landslides; Dorner and Wong 2003). In addition to loss of old forest, the amount of early and mid seral forest can be increased by considerably more than 100% over natural amounts. Mid seral forest is naturally uncommon in coastal rainforest ecosystems, but predominates in managed landscapes. This change in seral stage structure poses two problems. First, each stage is preferred by a different suite of species, and domination by a naturally uncommon stage reduces the availability of other stages (Hunter 1990). Second, while mid seral forests are a natural part of a seral trajectory, they tend to sustain fewer species of plants and animals than do earlier or later seral stages (Bunnell et al. 1999).

At the stand level, harvesting of old forests eliminates much of the stand structure that developed over centuries and naturally was maintained by small-scale natural disturbances (Franklin 1989; Price et al. 1998; Franklin et al. 2000).

4.1.4 Selected Indicators

- Minimum amount (specified as a ***percent of predicted natural***) of each representative ecosystem (site series , site series group, or surrogate within BEC variant) in old seral condition

- Maximum amount (specified as a percent of predicted natural) of each ecosystem in mid seral[13] condition

Habitat loss is the biggest single threat to biodiversity (Compendium Section 3). In the CIT region, old forest ecosystems predominate throughout the natural landscape (Compendium Section 2). The amount of old forest in each ecosystem (defined by site series or surrogate within variant; Compendium Section 3) is therefore the primary indicator for representation. Rather than using the absolute amount of old forest, the indicator expresses this amount as a percentage of predicted natural old forest (estimated from natural disturbance return intervals; Province of British Columbia 1995; Holt and Sutherland 2003) to mirror natural patterns across ecosystems with different disturbance regimes.

Identifying a maximum value for mid seral forests is also appropriate since harvesting can result in multi-fold increases in mid seral.

The EBM Handbook uses site series within biogeoclimatic variants as the basis for ecosystem representation (Compendium Section 3; Handbook Sections 4.4, 5.5, 6.4). Grouping site series may be appropriate, but groupings should never contain rare site series.

4.1.5 Risk

The EBM Handbook gives different guidelines for different scales:

- By sub-region, maintain minimum 70% of the natural old seral distribution in each site series[14] by sub-region
- By landscape, maintain minimum 50% of the natural old seral distribution in each site series; maintain >70% average distribution across landscapes
- By watershed, maintain minimum 30% of the natural old seral distribution in each site series; the average across watersheds = landscape target
- By site, maintain 15–70% retention, depending on watershed risk targets (see Compendium Section 4, In-block retention)
- Maintain less than 50% of each site series in mid seral condition (watershed and landscape scale)

The EBM Handbook also gives guidance about landscape pattern:

- protect critical habitat and maintain connectivity for red/blue listed and focal wildlife species

When considering the risk imposed by loss of old forest on ecosystems and species, it is important to consider the level of loss of habitat that results in changes to community structure or causes species to be unable to persist over the long term. Loss of habitat will reduce the abundance of species and

[13] Definitions for mid seral differ by biogeoclimatic variant as per the Biodiversity Guidebook (1995), but usually include stands aged 40–100 or 40–120 years.

[14] These percents may be applied to appropriate site series groups rather than to each site series

the composition of ecosystems; the question is at what level of habitat loss species and ecosystem functions are likely to be significantly impacted?

To guide management, it is helpful to identify "thresholds"—i.e., levels of habitat loss that result in a *change in rate* of loss of biodiversity, or where changes become ecologically significant. Thresholds are points at which linear or unobserved responses become rapid change. Threshold changes are long-lived and, in many cases, irreversible (Wissel 1984). The scientific search for the existence of habitat thresholds is relatively new, and as such there are a number of theoretical, but fewer empirical studies; however, in those studies that have searched for thresholds, most have found them (Dykstra 2003).

Ecological thresholds can occur at multiple scales. Thresholds present at the ecosystem level (e.g., community composition) have implications for biodiversity and ecosystem function (Chapin et al. 2000). At the species level, there may be "habitat amount" thresholds, beyond which a species tends to extinction (Lande 1987). Stand-level thresholds are also reported in forested systems (Edenius and Sjöberg 1997; Hargis 1999; Janssen and Angelstam 1999; Rodriguez and Andrén 1999; Penteriani and Faivre 2001). The overall effect of losing an individual species from an ecosystem varies, depending on it interactions with other species and its ecosystem role (Chapin et al. 2000).

As available habitat decreases, patches become smaller and/or more isolated, leading to a fragmentation threshold (Andrén 1996). For example, loss of up a certain percent of habitat often results in an approximately linear decline in population size; after this point, populations decline at a faster rate than habitat loss alone would predict. In essence, at low levels of habitat, the effects of pattern add to the effect of habitat loss (Fahrig 1998, 1999, 2002; King and With 2002). Several studies comparing the relative effects of fragmentation and habitat abundance find that habitat abundance is more important than habitat arrangement to species abundance and presence (McGarigal and McComb 1995; Trzcinski et al. 1999; Schmeigelow and Mönkkönen 2002), at least a higher levels of habitat abundance.

Theoretically, and empirically, thresholds appear to vary with respect to the species and the ecosystem of concern (Summerville and Crist 2001; Holt and Utzig 2002). In small number of empirical landscape-level studies, habitat thresholds in temperate and boreal forest range between 10 and 80%, but occur most frequently between about 30% and about 70% (see review in Dykstra 2003; this background report includes an extensive table of modeling and empirical studies). In general, threshold changes occur at much lower levels of habitat loss for rare species and habitat specialists than for habitat generalists (With and Christ 1995; Gibbs 1998; Schmeigelow and Mönkkönen 2002).

Simulation modeling shows a similar range of habitat thresholds, across a range of assumptions about species traits (e.g., Dytham 1995; With and King 2001; King and With 2002; Fahrig 2002). In a theoretical landscape, the loss of about 30% of habitat resulted in patches starting to become isolated (Franklin and Forman 1987; Stauffer and Aharony 1992), and at 70% loss of habitat, all remaining patches were isolated (Franklin and Forman 1987).

Understanding the implications of these studies for management also needs to acknowledge the increasing evidence for time lags in response to habitat change, and that most research to date has occurred over short time periods (Kattan et al. 1994; Lamberson et al. 1994; Hanski et al. 1996; Petit and Burel 1998; Brooks et al. 1999; Renjifo 1999; Hanski and Ovaskainen 2002). As well, reproductive parameters tend to show threshold changes at an earlier stage of habitat loss than presence or abundance parameters (e.g., Swift and Hannon 2002). Studies of population abundance may not reflect population viability (van Horne 1983). Hence studies based on presence or abundance values may underestimate habitat needs.

Generalizing management thresholds from the literature should be cautiously approached. Although thresholds are well documented by models, the empirical data include mostly short-term research, few in forested and particularly coastal ecosystems and few addressing potential delays in response. A generalised sigmoidal risk curve matches current theoretical and limited empirical knowledge (Figure 6; see Uncertainty, below). Note that this general model is not novel – a similar approach was used in development of the Biodiversity Guidebook (Province of BC 1995), and has been well advocated elsewhere (e.g., Landres et al. 1999). However, in this case attempts have been made to specifically hypothesise the shape of the curve and any thresholds, rather than make arbitrary/uninformed assumptions about how risk can be distributed.

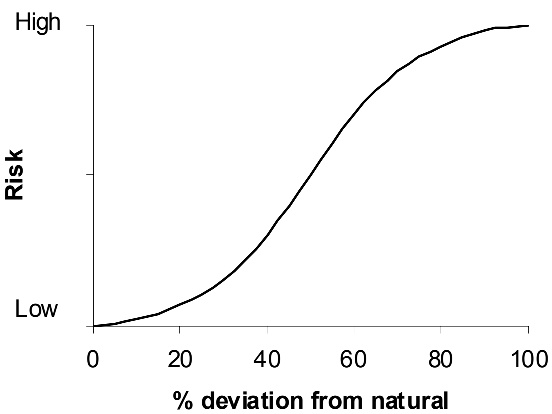


Figure 6. Generalised sigmoidal risk curve relating the deviation from natural amounts of habitat /or representation to risk to species and/or ecological integrity.

The EBM Handbook uses the inflexion points of a sigmoidal curve to locate the "low risk" target where detrimental ecological changes can be observed (30% deviation from natural) and the "high risk" threshold (70% deviation from natural) where ecologically significant loss of function occurs (i.e., an ecological limit has been crossed).

The sub-region is the appropriate scale to consider cumulative impacts of maintaining representative ecosystems. To ensure that cumulative risks are precautionary, the guideline for sub-regions uses the low risk point as a target. Because variability in natural amounts of old forest vary more in smaller areas (Wimberley et al. 2000), the guidelines for reserves in landscapes and watersheds are more flexible, allowing for habitat loss up to the high risk threshold in some watersheds. As the high risk habitat amount is approached, the spatial arrangement of habitat is increasingly important. Because the effects of fragmentation on the extinction threshold increase as habitat is altered, more intensive planning is appropriate to mitigate spatial effects in the more developed watersheds. Long range planning will also help guide risk distribution at the landscape level.

The EBM Handbook relates the amount reserved to the range of natural variability for each ecosystem under consideration. Table 8 provides an example of the predicted amount of old forest by physiographic region and site series group for the CIT area based on estimated disturbance frequencies for these units (Price and Daust 2003). For these estimates, young forest was divided into "natural", "probably natural," and "probably logged" stands, based on three existing databases. "Probably logged" stands were then aged, allowing calculation of the proportion of young forest disturbed naturally

A combination of physiographic unit and site series group better described homogeneous units than biogeoclimatic variant; predominant biogeoclimatic variants are listed in the table because of their familiarity. The fourth column shows how the 70% would be applied to each unit. For example, in upland hypermaritime ecosystems, 70% of 95% is 67%; hence the target calls for at least 67% of each site series group within this unit to be old. Similarly, in the Inner Coast fluvial systems, the target would be a minimum of 30% (70% of 43%). Price and Daust (2003) note that the estimated mean disturbance frequency is more reliable than the range and that areas should be managed to vary around the mean. They suggest that sampling areas within each region would better describe the actual historical range of old forest and that simulation modeling experiments would better describe the potential range of old forest.

Table 8. Estimated return intervals and natural proportion of old forest in different disturbance units as defined by physiographic region and site series group

Region	Site series group	Return interval*	Proportion of forest > 250 years (%)	70% target	Biogeoclimatic subzones**
Hypermaritime	Upland	4,500 – 10,000	95 – 98	67	**CWHvh**, **CWHwh**, MHwh, CWHvm
	Fluvial	2,200 – 10,000	89 – 98	62	
	Ocean spray	1,000 – 5,600	78 – 96	55	
Outer Coast North	Upland	1,800 – 10,000	87 – 98	61	**CWHvm**, CWHvh, MHmm, CWHms
	Fluvial	500 – 2,100	61 – 89	43	
Outer Coast South	Upland	900 – 2,500	76 – 90	53	**CWHvm**, **MHmm**, CWHms, CWHxm, CWHdm, CWHws
	Fluvial	400 – 1,400	54 – 84	38	
Inner Coast	Upland	500 – 5,600	61 – 96	43	**CWHws**, **MHmm**, CWHds, ESSFmw, CHWms
	Fluvial	300 – 900	43 – 76	30	

* Rounded up to nearest 100 years; return interval truncated to 10,000 years

** In order of predominance within physiographic region: **bold** subzones make up more than 100,000 ha; other listed subzones make up at least 10,000 ha.

Based on 70% of natural, within a sub-region, at least 30–67% of each site series (or site series group) will be reserved in the units listed. Less area will be reserved in areas with more frequent or more variable disturbance regimes such as fluvial ecosystems within physiographic units, or more southerly or interior physiographic regions. Note that we would expect future local refinement of natural disturbance information to reduce the range of predicted natural estimates on which targets are based – for example, within the Inner Coast Fluvial, refining the range currently estimated at 43–76% will be an important area for further research.

The low risk targets for representative old forest are generally consistent with recommendations from other areas. Over a range of forest ecosystem types, estimates for appropriate amounts of unmanaged ecosystems required to maintain biodiversity range from 4 to 99% of the landbase (Noss and Cooperrider, 1994). This wide range reflects differing ecosystems, assumptions about the contribution of the managed landbase, and other variables. Noss and Cooperrider (1994) recommend that from 25 to 75% of the landbase be reserved depending on conservation objectives. In Oregon, Noss (1996) suggested that about 50% of the landbase should be reserved, or managed for non-human species, and that this target could include 100% of certain ecosystems, such as those with very rare natural occurrence.

More locally, based on knowledge of the range of natural variability in the area, the Scientific Panel for Sustainable Forest Practices in Clayoquot Sound (1995) recommended 40% of that region be reserved, defined as the minimum requirement to maintain ecological processes. Also relevant to the CIT area, the Biodiversity Guidebook (1995) recommends reserving a minimum of 50% of natural old forest (including 12% of the total area assumed in Protected Areas; Appendix 4 of Biodiversity Guidebook) as a low-risk strategy (i.e., "high" biodiversity emphasis), representing about 27–53% of the total landbase when applied to the North Coast variants in the Table above.

4.1.6 Uncertainty

- The existence of thresholds is generally demonstrated in the literature. However their specific application in the CIT region is uncertain; there are no empirical studies available in this region. However, the low frequency of stand-replacing disturbances in coastal ecosystems, and the high divergence between typical harvesting systems and natural disturbance types suggests that ecosystems in the CIT region would not be expected to be more resilient to habitat loss than others where empirical studies have been completed. The use of existing literature on thresholds as a basis for low and high risk was therefore considered reasonable.
- Science is convincing that a very high level of disturbance has a very high probability of causing loss of ecological integrity, and conversely that very little disturbance has a low probability of causing loss of integrity. Uncertainty is highest between the inflexion points (Figure 7). That said, the shape of the curve likely only has a limited set of possibilities even if evidence for specific thresholds is lacking.
- Application of threshold literature in a primarily forested landbase is uncertain. There are clearly ecological differences between old forest and managed forest, however the extent of differences when forms of selective harvesting are applied is as yet unknown, especially at the regional scale.
- Habitat loss and fragmentation usually occur simultaneously, compromising the ability to distinguish their effects.

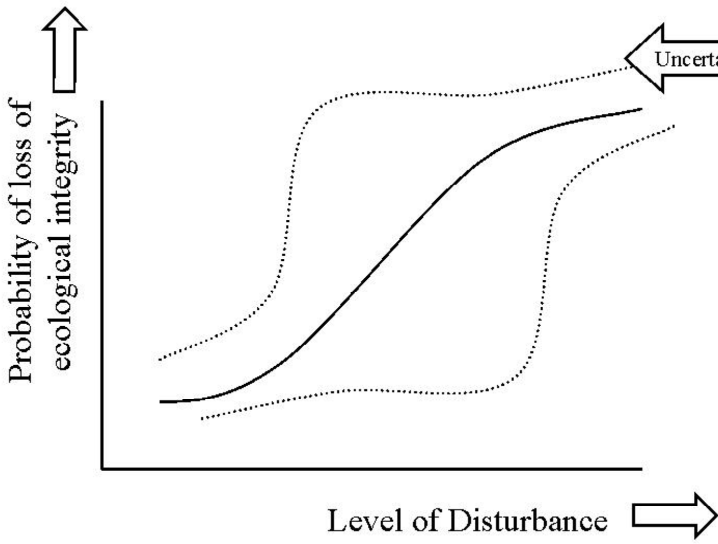


Figure 7. Uncertainty around the generalised sigmoidal curve.

4.1.7 Important Questions for Adaptive Management

There is a need for empirical studies of the implications of ecosystem/habitat loss on ecological/species parameters in this ecosystem. Important questions include investigating how retention of attributes at the stand level may or may not sum up and influence ecological integrity at the regional level. Unfortunately, this work requires application of relevant empirical data into multiscalar landscape models. Although some large-scale experiments have been successful in the past (Carpenter et al. 1995), it requires much effort and coordination to undertake a scientifically robust large landscape level experiment. Landscape level management recommendations is one area where adaptive management may not be very feasible, at least in the short-term, and therefore decisions at this level should remain on the precautionary side.

4.2 Representative Hydroriparian Ecosystems

The Hydroriparian Planning Guide and technical reports describe in detail a risk assessment procedure and rationale:

Bunnell, F.L., G.D. Sutherland. and T.R. Wahbe. 2001. Vertebrates associated with riparian habitats on British Columbia's mainland coast. Hydroriparian Planning Guide Technical Report #5.

Church, M. and B. Eaton. 2001. Hydrological effects of forest harvest in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #3.

Hydroriparian Planning Guide. 2003.

Price, K. and M. Church. 2002. Risk to ecosystem functions. Summary of expert workshops. Hydroriparian Planning Guide Background Information.

Price, K. and D. McLennan. 2001. Hydroriparian ecosystems of the North Coast. Background report to the North Coast LRMP.

Price, K. and D. McLennan.2002. Impacts of forest harvesting on terrestrial riparian ecosystems of the Pacific Northwest. Hydroriparian Planning Guide Technical Report #7.

Young, K. 2001. A review and meta-analysis of the effects of riparian zone logging on stream ecosystems in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #4.

4.2.1 Definition

Hydroriparian ecosystems consist of aquatic ecosystems plus those of the adjacent terrestrial environment that are influenced by and influence the aquatic ecosystem (Clayoquot Sound Scientific Panel 1995; Hydroriparian Planning Guide). Hydroriparian ecosystems include stream, wetland and marine ecosystems (e.g., small very-steep streams, small low-gradient streams, floodplains, fans, bogs, fens, pond, lakes, salt spray forests, estuaries; see Hydroriparian Planning Guide Appendix 3 for a description of important hydroriparian ecosystems of the CIT region).

4.2.2 Importance of Element

Water defines the domain of aquatic ecosystems; water flow transports materials to distant parts of watersheds and links surface and underground ecosystems (Edwards 1998). In essence, hydroriparian ecosystems are the "*circulatory system of the ecological landscape*" (Clayoquot Sound Science Panel 1995). Stream morphology is the net result of the mobilisation, transfer and deposition of sediments along channels. Sedimentation processes are critically important to maintaining hydroriparian ecosystems, creating rich, productive floodplains, fans and estuaries, and renewing the gravel substrates used by aquatic invertebrates and fish (CSSP 1995; Montgomery and Buffington 1998; Church and Eaton 2001). Downed wood influences stream morphology by creating pools and patterns of sediment and nutrient deposition (Bilby and Bisson 1998). Wood increases channel complexity and provides habitat, food, and shelter from the current for invertebrates and fish, and habitat for amphibians, birds and small mammals (Bilby and Bisson 1998). Large pieces of wood often form complex log jams that considerably alter stream morphology and habitat features for many decades (Hogan et al. 1998). Wood from riparian sources drifts to the ocean where it provides habitat and food for marine ecosystems, including estuaries, beaches and the deep-sea floor (Maser and Sedell 1994).

Around 90 terrestrial vertebrates are associated with riparian ecosystems (Bunnell et al. 2001). Deciduous ecosystems are an important component of natural riparian forest and provide habitat for almost half of the terrestrial vertebrates dependent upon riparian habitat (Bunnell et al. 2001). Many other organisms live in these complex, productive ecosystems (Price and McLennan 2002).

Because BEC was designed for classifying forested ecosystems, it does not consider hydrological features, provide landscape context or combine sites into ecosystem complexes—all important aspects of hydroriparian ecosystems. Particularly in the water-rich CIT region, it is important to consider representation of hydroriparian ecosystems in addition to representation of BEC site series. In coastal watersheds, hydroriparian ecosystems change fairly predictably from steep headwaters to valley bottoms. Headwater streams influence moisture regimes over a short distance from the water and site series near these streams may be those found in any upland ecosystem. Conversely, valley-bottom streams influence adjacent terrestrial ecosystems considerably.

The EBM Handbook provides guidelines that maintain hydrological regime, stream morphology, downed wood and hydroriparian biodiversity.

4.2.3 Extent of Riparian Ecosystem

Riparian ecosystems are those terrestrial ecosystems that are influenced by, or influence, water. Terrestrial influences on water usually extend at least one site-specific tree height from aquatic systems (Young 2001), depending on adjacent vegetation and gully characteristics. Aquatic influences extend a variable distance from open water depending upon landform and surficial materials. Plant communities indicate the extent of direct influence (Banner et al. 1993). Hyporheic flow and movement of hydroriparian-associated animals may extend further. Aquatic-breeding amphibians and riparian small mammals range at least 200m from water (Price and McLennan 2002); large mammals such as fishers and grizzly bears range much further (Bunnell et al. 2001). While it may be possible to define coastal hydroriparian ecosystems from the perspective of plants, it is not possible to define a distinct boundary for animals, and too little is known about fungi and microbes even to list hydroriparian associates. The strength of the association with hydroriparian ecosystems varies by species along a continuum from obligates to generalists and edge species—there is no all-species boundary between hydroriparian and upland ecosystems (Bunnell et al. 2001; Price and McLennan 2002).

The EBM Handbook defines riparian forest by plant community, including high-bench or dry floodplain communities, and/or landform (e.g., gullies) plus one and a half site-specific tree heights (horizontal distance) beyond. In the transportation and deposition zone, the guide includes the entire valley flat plus one and a half tree heights.

The first tree height captures the influence of the terrestrial system on the aquatic system; the additional half tree height serves to protect conditions within the buffer. The extent of influence of the terrestrial and aquatic ecosystems on each other differs depending whether adjacent land remains forested or has been harvested (Brosofke et al. 1997). Some studies recommend buffers of several tree heights to maintain conditions within the buffer (Brosofke et al. 1997; Reid and Hilton 1998).

Even small streams can influence adjacent conditions and vegetation. A study in western Washington found microclimatic gradients around small (2–4 m) streams extending between 31 and 62 m from the stream (Brosofke et al. 1997). Where channels are unconstrained, the highly variable flow periodically enriches adjacent land and leads to narrow bands of larger trees and diverse vegetation. In Clayoquot Sound, more species of plants grew within the first 10 m of small streams (0.5–5 m; Kim 1997), and vegetation communities were influenced by moderately sized streams (3–30 m) for 100 m (Chan-MacLeod 1996). Plant communities within gullies are obviously influenced by the continuously moist, cool microclimate, but impacts beyond the gully are not intuitively obvious. Interestingly, a preliminary study in Clayoquot Sound found that the number of species next to entrenched streams (3–30 m wide) continued to change over 100 m, even though the transects began on top of 20–50 m high cliffs (Chan-MacLeod 1996). Ephemeral channels may be expected to have a smaller influence than seasonal or perennial ones.

4.2.4 Impacts of Forest Harvesting

Impacts on hydrology and stream morphology include the following (Church and Eaton 2001):

- Harvesting increases total runoff and changes some patterns of low and peak water flows.
- In many watersheds, effects have been observed to become significant where more than about 20% of a watershed is cleared within a decade or two; effects on storm flows have been observed to disappear after about 10 years in the absence of roads.
- Roads have been found to increase storm runoff and advance runoff timing; road effects appear to be permanent; road building and use are disproportionately important sediment sources; roads perpetuate the presence of fine material in stream channels.
- Forest land use has accelerated sediment delivery to streams by 2–10 times.
- Streambank disturbance sharply increases erosion and sedimentation. Modest initial disturbances can propagate into major morphological change along a stream and reduce aquatic habitat quality
- Because wood jams modulate sediment transfer, changes to downed wood input influence sedimentation processes.

Other studies have described potential impacts of harvesting including increased bog formation in the Hecate Lowlands (Asada 2002), increased in fine sediments for at least 35 years (Young 2001), changed amount and pattern of downed wood input (Scrivener et al. 1998, Bilby and Bisson 1998, Hogan et al. 1998, Young 2001).

As well as changing aquatic conditions (Young 2001), removal of riparian trees changes the microclimate, the levels of light, water, nutrients and large and small organic materials reaching the ground and, consequently, the habitat features of terrestrial ecosystems (Price and McLennan 2002). Any change to downed wood deposition, resulting from upstream or local forest harvesting (May and Gresswell 2003; Reeves et al. 2003), impacts floodplain structure and vegetation (Naiman and Bilby 1998). Disturbed hydroriparian ecosystems are particularly sensitive to invasion by opportunistic weeds (Price and McLennan 2002).

A recent meta-analysis of the impacts of harvesting on terrestrial vertebrates (Price and McLennan 2002) found that

- amphibian, avian and mammalian communities all changed following harvesting and recovered over decades,
- buffers mitigated impacts on avian and mammalian communities and mitigated changes in amphibian abundance,
- some species were missing in 70 m buffers.

All reviews found high levels of variation among regions and studies, implying that site effects are highly important and that management planning must include adaptation for site conditions following on-site inspections.

4.2.5 Selected Indicators

- Hydrology: Equivalent clearcut area within watersheds or sub-basins over 1,000 ha.

Because trees intercept and use water, hydrologic responses are related primarily to the extent, rather than to the methods, of harvesting (Ziemer and Lisle 1998; Church and Eaton 2001). The annual rate of cut is a simple indicator. The indicator has been adjusted for watersheds with and without roads.

- Stream morphology: index of road length in terrain classes IV and V + area cut in terrain classes IV and V
- Stream morphology: % streambank cleared in transportation and deposition zones

Incremental sediment delivered to streams derives from road building and maintenance, from activities in unstable terrain, and from destabilisation of stream banks. The first indicator covers the increased probability of landslides following road building and harvesting on unstable terrain. The index uses guidelines established in the Coastal Watershed Assessment Guidebook (Province of BC 1995).

The second indicator is related to loss of root strength and covers sediment derived from streambanks.

- Downed wood: % forest younger than 30 years in the source zone
- Downed wood: % deviation from natural riparian forest in the transportation and deposition zones

In the source zone, most wood travels downslope during mass wasting events. In the transportation and deposition zones, most wood falls in from adjacent riparian forest. In the source zone, with smaller streams, smaller pieces of wood may be effective.

- Biodiversity: deviation from natural amount of riparian forest

This indicator is the same as that used for representing old forest, applied to riparian forest.

4.2.6 Risk

Hydrology

Significant changes occur to hydrology with rates of forest harvest higher than 1% of the forested area of a watershed per year averaged over 20 years (Church and Eaton 2001). Studies have been unable to detect effects below this threshold, but these findings are likely due to statistical ambiguity: “physical principles dictate that any level of cut must have some effect” (Church and Eaton 2001). As well as increasing physical, erosion damage, changes in hydrology affect organisms. Many stream-dwelling organisms are adapted to episodic extreme events, but may not respond well to elevated stresses lasting for longer periods.

An expert panel convened to develop risk curves for British Columbia's north and central coast drew two curves (one with, one without, roads) relating risk to hydrology to forest clearance (Figure 8; Price and Church 2002).

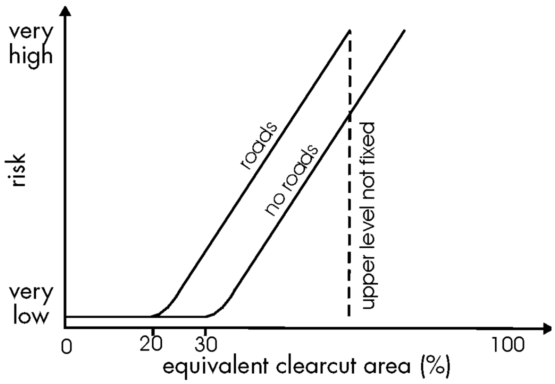


Figure 8. Risk to hydrology as a function of forest clearance.

The curve including roads defines a threshold at 20% clearance, where risk to hydrological regime starts to increase. The panel did not specify a high-risk threshold, beyond showing that the risk to hydrological function becomes a certainty at some point before 100% clearance. The 20% clearance threshold defined by the expert panel matches the 1%/year rate-of-cut threshold averaged over 20 years concluded from the literature review (Church and Eaton 2001) and the recommendations of the Clayoquot Sound Scientific Panel (CSSP 1995).

The EBM Handbook provides a precautionary guideline at the watershed scale for maintaining hydrological regime:

- Equivalent clearcut area of <20% (with roads) or <30% (without roads) applied to every watershed and sub-basin over 1,000 ha

Stream Morphology

A review of sedimentological effects of forest harvesting on the Pacific Northwest coast recommends that hillslopes are treated appropriately and that streambanks are undisturbed (Church and Eaton 2001). In addition, the review recommends that roads are eliminated from the entire valley floor (except where there is no practical alternative) to maintain natural processes, including normal sedimentation and channel shifting. Similarly, the Clayoquot Sound Scientific Panel recommended that the entire wet floodplain be reserved (CSSP 1995). To protect stream morphology, Church and Eaton (2001) recommend no activity in wetlands, active fluvial units (FAp, FAv, FAb, FAj, FAf) or floodplains of unknown activity (Fp) (see Table 9 for description of terms).

An expert panel convened to develop risk curves for British Columbia's north and central coast drew a single risk curve relating to the activities on unstable terrain (Figure 9; Price and Church 2002).

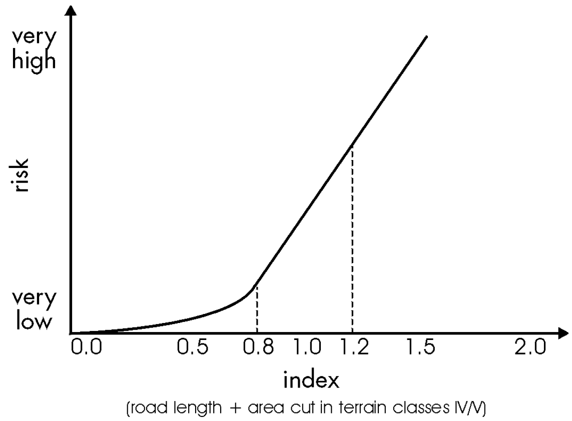
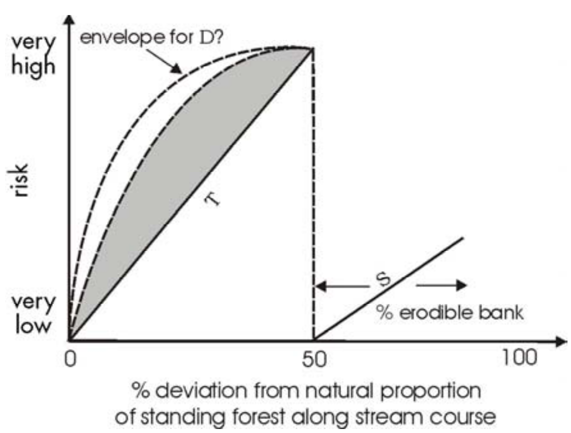


Figure 9. Risk to stream morphology as a function of activities on unstable terrain.

The panel noted that risks are low for index values of less than 0.8, and high for values above 1.2. Due to lack of confidence in the low-risk threshold, however, the expert panel felt that zero risk was most appropriate as a precautionary guideline and recommended no activities be planned on Class IV or V terrain. Site level inspections would be necessary to confirm or reject the potential instability of terrain mapped as Class IV at the watershed level.

The panel drew a suite of curves relating the risk to streambank stability to deviation from natural riparian forest (Figure 10; Price and Church 2002). Within the source zone (S), the panel felt that risks to streambank stability were minimal until more than 50% deviation from natural riparian forest along erodible banks. They suggested no precautionary guidelines for the source zone. The panel drew a linear to convex risk curve (shaded area) in the transportation zone (T) and, suggested that the deposition zone (D) might be even more sensitive to disturbance. The shape of these curves supports the requirement for a wind-firm buffer in the transportation and deposition zones suggested by



Church and Eaton (2001).

Figure 10. Risk to streambank stability as a function of deviation from natural riparian forest.

The EBM Handbook provides precautionary guidelines at the watershed scale for maintaining stream morphology:

- Reserve class IV and V terrain, failure of which might cause sediments to be delivered to any stream channel.
- Reserve all terrain units classified as active fluvial units or as floodplain of unknown activity.
- Reserve all wetlands.
- Reserve windfirm buffers around all streams in the transportation and deposition zones. This buffer will be adjusted at the site scale.

At the site scale, the EBM Handbook recommends

- Reserve windfirm buffers around small steep streams and gullies with high susceptibility for debris flow.

Downed Wood

Two expert panels convened to develop risk curves for British Columbia's north and central coast (one considering hydrological elements, the other considering terrestrial elements) drew curves relating risk to downed wood to riparian forest (Price and Church 2002).

The two panels independently drew almost identical curves for the transportation and deposition zone, with risk increasing after 20% deviation from natural riparian forest (Figure 11; Price and Church 2002). One panel drew a second curve relating the risk in the source zone to forest cleared within 30 years.

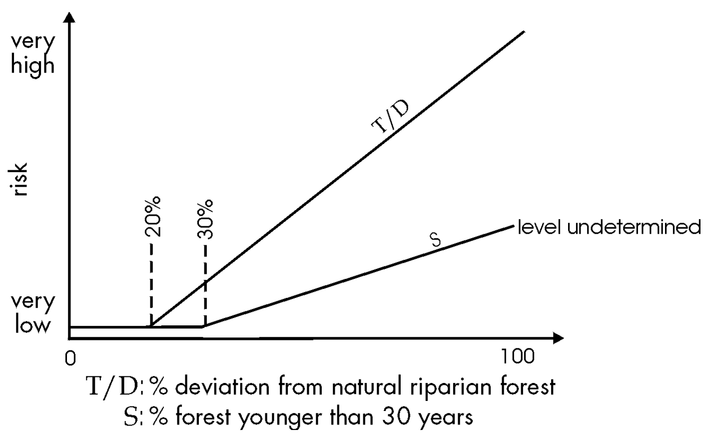


Figure 11. Risk to downed wood as a function of deviation from natural riparian forest.

The EBM Handbook provides precautionary guidelines at the watershed scale for maintaining downed wood:

- Less than 30% of forest in the source zone should be younger than 30 years.
- Less than 20% deviation from natural riparian forest in the transportation and deposition zones.

Biodiversity

An expert panel convened to develop risk curves for hydroriparian ecosystems of the CIT region drew three sets of curves relating risk to biodiversity to deviation from natural habitat for different hydroriparian ecosystems (Figure 12; Price and Church 2002).

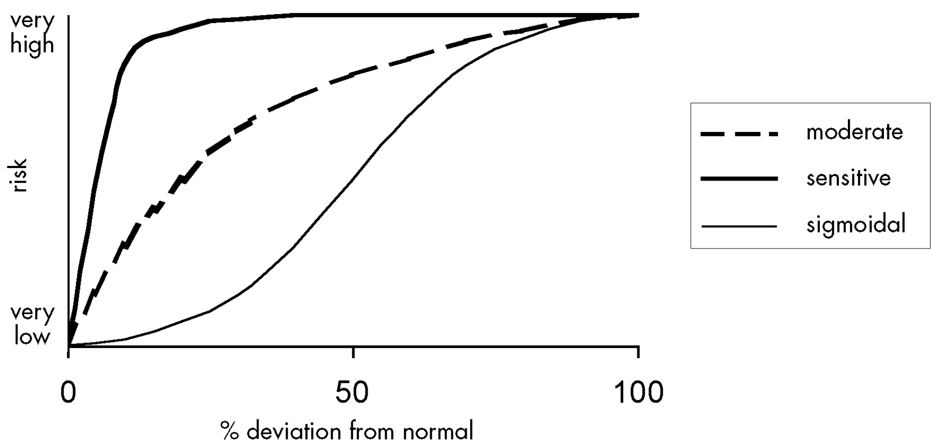


Figure 12. Risk to hydroriparian ecosystems as a function of natural riparian forest.

Estuaries, karst ecosystems and a special class of small very steep streams and gullies (those with high susceptibility to debris flow) follow the highly sensitive curve. Floodplains, fans, forested swamps, and a second class of small very steep streams and gullies (those with low susceptibility to debris flow but distinctive microclimate; e.g., creating spray zones with unique communities of plants) follow the moderately sensitive curve. Small low gradient streams, all other small steep streams and gullies, shoreline forests and wetlands (lakes, ponds, sedge fens, bogs) follow the standard curve (sigmoidal to reflect literature values).

At the watershed scale it will not be possible to discriminate among types of small streams. For planning purposes, small streams can all be included within the low susceptibility to debris flow category (streams on unstable terrain will already be reserved through stream morphology guidelines). Field checking will then identify streams with high susceptibility to debris flow and those with unique microclimates. These streams can then be treated as appropriate. Over time, information about the relative abundance of each stream type in different regions will be available for planning.

The EBM Handbook provides precautionary guidelines for biodiversity at sub-regional scales (SR) and watershed (W) scales:

- Maintain 97% (SR) and 90% (W) of natural riparian vegetation on estuaries and in karst ecosystems
- Maintain 90% (SR) and 70% (W) of natural riparian forest next to floodplains, fans, forested swamps, small very steep streams and gullies with unique microclimate
- Maintain 70% (SR) and 30% (W) of natural riparian forest around remaining hydroriparian ecosystems

In all cases, the average over watersheds must meet the sub-regional target.

4.2.7 Uncertainty

- Certain hydrological and sedimentological effects may last for a substantial portion of forest succession; yet no experimental studies have been continued for more than 40 years. The legacy of roads can last beyond 40 years. (Church and Eaton 2001; Young 2001)
- Flood size naturally varies tremendously, making detection of changes difficult. If the size of the largest annual flood varies by 3x from year to year (not at all unusual), then a change in average flood size of even 2x may remain difficult to detect for many years, yet it may have major ecological effects.
- Forest terrain varies considerably. There are too few studies to allow generalisation. This concern is particularly important in relation to British Columbia. Most hydrological studies have been conducted in unglaciated terrain with deeply weathered soils and relatively short slopes. The shallow, glaciated soils and long slopes in the CIT region probably create a different hydrological system—likely more sensitive to disturbance—than has been analysed (Church and Eaton 2001).
- Most experiments have not covered sufficient area to reveal the effect of episodic, major slope failures (Church and Eaton 2001).
- Studies of fine sediment have often confounded effects of roads and harvest (Church and Eaton 2001).
- The role of downed wood, particularly in moderately large to large streams, and the general impacts of forestry are now well recognised. The importance of downed wood of different sizes in small streams is less well known. In coastal British Columbia, the episodic input of wood (e.g., four episodes over 110 years on Haida Gwaii; Hogan and Schwab 1991) creates difficulty in interpreting short-term studies.
- The impacts of forestry on ocean ecosystems are unknown. Impacts could include changes to wood in deep-sea ecosystems and fecal contamination of near shore waters (Kingzett and Palzat 2002). In particular, little is known about species and ecosystem processes that may be affected by forestry in small basins that run directly into the sea.
- The extent of riparian forest remains a controversial topic. There are insufficient data to draw curves similar to the one presented for biological components of hydroriparian ecosystems (Young 2001). Effective buffer widths are unknown; recovery time for riparian organisms is unknown; impacts of partial cutting are unknown (Price and McLennan 2002).

4.2.8 Important Questions for Adaptive Management

What are appropriate rate-of-cut criteria in roaded and in non-roaded watersheds to assure maintenance of hydroecological functions?

Rate-of-cut remains a controversial topic. It is unlikely that the precautionary specifications that can be drawn from current research literature are appropriate everywhere on the coast. The risk curve presented in the hydroriparian planning guide represents only an arbitrary (although expert) extrapolation of those guidelines. Forest management deliberately designed to be sensitive to, and to monitor, rate-of-cut and hydrological effects, through an experimentally articulated program of comparative gauging, will eventually allow more secure and more flexible guidelines to be developed to govern rate of cut.

Applying adaptive management to resolve this question presents difficulties related both to time scale and to spatial variability in the landscape. To tackle the question of rate-of-cut most directly, a series of drainage basins should be identified and subjected to different harvest rates (including zero harvest controls), with monitoring of water outputs. Time constraints suggest that some form of community-industry-public sector partnership will have to plan and execute the program (it has proven difficult to maintain institutional focus on programs of this kind for the requisite time). However, the greatest difficulty will lie in selecting basins that reasonably can be compared, so unequivocal lessons can be learned. Recent developments in hydrological scaling theory may help to resolve the problems, but this circumstance suggests that this program may have to begin with a specialist (scientific) review.

What criteria reliably identify watersheds that are "high risk" in the sense that high sediment delivery is apt to compromise the function of hydroriparian ecosystems?

Because geology, topography, and landscape history all vary, certain parts of the landscape are more prone to yield excessive amounts of sediment to stream systems than others, especially under certain land use practices. It is important to identify these areas before development occurs. The current guidelines propose a method based on the occurrence of terrain of stability classes IV and V, which is available from terrain mapping, and is consistent with current CWAP criteria. The method requires to be critically tested, and the possibility for more refined means to identify "high risk" areas needs to be examined. It is probable that initial work on this problem would constitute research (rather than adaptive management) but, eventually, means need to be tested and adapted in the operational environment of forest land management.

The question of identifying high risk basins might be pursued by compiling operational experience for analysis in a statistical experiment. In this case, the focus would have to be placed on defining operational measurements that are sufficiently standard to permit construction of a database that can be properly evaluated statistically.

How do log jams change over time?; What are the ecological and physical roles of downed wood in small streams?

Current knowledge about log jams is based mainly on comparisons of dated jams in different places rather than on long-term studies of individual jams. The second question addresses one of the main uncertainties in relation to downed wood. Both questions are probably research projects to be conducted with industry cooperation.

What are the general relationships between biological ecosystem components and riparian forest width?

Some research projects are beginning to examine the influence of buffer width on biological components of natural riparian forest (e.g., Cockle and Richardson 2003; Kiffney et al. in press). It may be possible to extend these projects and to improve their amenity to adaptive management.

What are the impacts to biodiversity (at site and downstream) of various management practices around small streams? For example, is biodiversity best conserved through narrow buffers on most streams or by reserving selected patches including several streams?

The impacts of forest management around small streams remains controversial. Although harvesting around single streams likely have negligible downstream impacts, cumulative impacts can be high. Conversely, leaving buffers around all small streams can remove most harvest opportunity in some watersheds. In addition, buffers are susceptible to blowdown and may not achieve their aim.

This topic includes questions about impacts to the hydrology of very small, intermittently flowing streams. A relatively simple study looking at the number of weeks of flow of ephemeral, intermittent and perennial small streams throughout the summer before and after harvest would be very useful to explore how flow regimes and potentially stream communities change following harvest.

Scientists and managers have realised the inadequacy of practices to protect small streams over the past decade, and research and adaptive management has begun to address this question already (e.g., Weyerhaeuser's small stream retention program). Further studies in other geographic areas are necessary.

4.3 Rare Ecosystems

The following reports within the compilation provide more detailed information:

Pojar, J. 2002. Rare ecosystems of the CWHvh2. Report for the North Coast LRMP.

Pojar, J. 2003. Biodiversity of the CIT area. Report for the CIT.

4.3.1 Definition

Ecosystems can be rare locally, regionally, nationally and/or globally. Some ecosystems naturally occur over a small portion of the land (e.g., estuaries); others have been depleted due to human activities (e.g., floodplain forests).

4.3.2 Importance of Element

Maintaining biodiversity requires maintenance of rare ecosystems. Rare ecosystems often include specially adapted suites of organisms. Some rare ecosystems are among the most productive and complex on the coast. Pojar (2002) and (2003) lists rare ecosystems of the CIT region.

4.3.3 Impacts of Forest Harvesting

Some old-growth forests in the Central and North Coast contain live trees and decaying wood over 1,000 years old; many dominant trees are 300 – 600 years. Once harvested, the structural and functional equivalents of these old-growth ecosystems cannot be replaced within a meaningful management timeframe; some changes may be irreversible. Rare ecosystems are particularly sensitive because organisms depending on old seral stages may be unable to recolonise from distant similar ecosystems. Reserving rare ecosystems from harvest aims to maintain biodiversity by protecting all of the organisms within the ecosystem, including unclassified bacteria, fungi and invertebrates. Harvesting has the potential to alter ecological processes. For example, activities in the globally rare bog mosaic might change the productivity of these ecosystems (Banner et al. 1999).

4.3.4 Selected Indicators

- Minimum amount (specified as a percent of predicted natural) of each rare ecosystem in old seral condition

4.3.5 Risk

The EBM Handbook provides guidelines for listed and other regionally rare ecosystems:

- reserve 100% of known or potential red-listed and other regionally rare ecosystems at all scales
- reserve 70% of known or potential blue-listed ecosystems at all scales

In the CIT area, most CDC listed ecosystems are rare in the old seral stage. "Potential" red- or blue-listed ecosystems are listed site series at younger seral stage. Some of these site series may not be naturally rare (i.e., they are rare due to harvesting). It will not be necessary to reserve all of these areas once a plan has been developed that ensures that a sufficient amount has been reserved to remove the ecosystem from listed status.

No studies provide guidance on the amount of rare ecosystems to reserve. Levels of precaution for blue-listed associations are the same as for general representation of any site series at the sub-regional scale, but are not allowed to drop below the precautionary guideline at smaller scales. Because of their extreme rarity, all red-listed plant associations and other regionally rare ecosystems are reserved from harvesting.

4.3.6 Uncertainty

Many rare ecosystems are missed from maps at all scales above the site because they are often small and difficult to identify remotely (Ketcheson et al. 2002). Conversely, some listed ecosystems are not naturally rare (they have been depleted due to human activity) and may not require complete protection. Local knowledge is crucial for identifying unlisted rare ecosystems and for ensuring that an appropriate suite of ecosystems is included on the rare ecosystem list.

4.4 In-block Retention

The following reports within the compilation provide more detailed information:

Dorner, B. and C. Wong. 2003. Natural disturbance dynamics in the CIT area. Report for the CIT.

Pojar, J., C. Rowan, A. MacKinnon, D. Coates, and P. LePage. 1999. Silvicultural options in the Central Coast. Report for the Central Coast Land and Resource Management Planning Table.

4.4.1 Definition

Site-level, or in-block, retention refers to live or dead trees left within a harvested stand. Trees can be retained individually (dispersed retention) or in groups (grouped retention). Variable retention refers to different amounts and patterns of live and dead trees retained in different sites (see 3.12). Retained trees are not scheduled for later logging.

4.4.2 Importance of Element

Retaining old-forest elements (e.g., big trees, dead and decaying trees, downed wood, shrubs, vertical and horizontal heterogeneity) in the managed landscape can (Franklin et al. 1997, Lindenmayer and Franklin 2002)

- leave a legacy of older structural features in the managed stand,
- enhance connectivity through landscapes,
- buffer reserves,
- moderate microclimate changes,
- maintain refugia for species that might be extirpated from the harvested area due to their sensitivity ("life-boating").

The stand is the basic unit of forest management and for some species, the effects of disturbance are directly expressed at this level (Roberts and Gilliam 1995). This is especially true for species that are sensitive to the direct impacts of harvesting (e.g., saprophytic and mycorrhizal vascular plants; Schoonmaker and McKee 1988) and/or recover slowly during secondary succession (e.g., epiphytic lichens; Price and Hochachka 2001). Where an objective of management includes retaining the pre-harvest community of flora and fauna (manage for resistance) or ensuring their rapid long-term recovery (manage for resilience), a proportion of the pre-disturbance stand needs to be retained within the cutblock.

4.4.3 Impacts of Forest Harvesting

Traditional clearcut logging simplifies stands by removing the structural legacies of centuries, thus removing heterogeneity. Organisms dependent on these structures will disappear from stands lacking legacies and some groups of motile organisms will use the stand less (Franklin et al. 1997). Harvesting, but retaining some retention may alleviate some of these effects through space, and through time perhaps increasing the recovery rate of the stand (Holt and Sutherland 2004).

4.4.4 Risk

The EBM Handbook provides a range of options for the amount of retention within stands:

- retain 15 – 70%
- retain more in landscapes or watersheds with higher ecological values
- retain more in landscapes or watershed where past logging has removed more old forest.

The EBM Handbook also offers guidance about pattern:

- design retention using fixed elements (e.g., rare ecosystems, unstable terrain, tributary junction) to anchor remaining patches and corridors (e.g., downed wood, stream reserves, targets for old forest representation)

The targets in the handbook at the site level can also apply to meet higher level retention targets **IF** the retention is permanent and is larger than 1 ha in size.

High percent variable retention (>70%) can also count to meeting higher level retention targets, providing the species composition remains equivalent to the natural stand, and there are no road / trail impacts associated with harvest.

The amount of retention should be sufficient to "achieve specified management goals" (Lindenmeyer and Franklin 2002). Because goals will vary among areas, retention levels will also vary. Although the goal of ecosystem-based management is to maintain ecosystem integrity across all scales, ecosystem-based management does not necessarily require retention of all late successional species and attributes everywhere, all the time. Precautionary late-seral representation targets at the regional, landscape and watershed scales are meant to ensure that ecological integrity is maintained at these scales, including maintaining populations of species throughout the region that depend upon late-seral habitat. Retaining late-seral structure at the stand level, in addition to the higher level reserves helps support this goal. The amount of appropriate retention needs to be assessed in the context of the watershed and landscape reserves or other unmanaged areas.

Emulating natural disturbance patterns at the stand scale would require a significant amount of retention on most cutblocks (Pojar et al. 1999), because gap dynamics typically produce patches less than 0.05 ha and impact fewer than 10 trees per patch (Lertzman et al. 1996). Although harvesting that removes only a few trees per patch is ecologically appropriate on many areas of the coast, economic and safety considerations will limit where individual tree or small group selection is practised. In addition, ecological costs arise because, for the same volume of timber, there is a trade-off between removing small patches and minimising road length. On the rest of the harvested landbase, the objective of stand level retention is not aimed at mimicking the natural disturbance pattern, but rather is generally aimed at leaving a legacy of structures in regenerating stands so that those stands retain some attributes of old growth and act as older stands earlier in their successional pathway than they would in the absence of those retained structures. As well as increasing the habitat value of harvested stands for species associated with older forests, stand level retention is also used to buffer reserves and enhance connectivity. Note that this approach would not constitute a low risk strategy in isolation; if management was limited to the stand level then reasonable emulation of natural disturbances would be appropriate. However, the multi-scalar approach taken in the EBM Handbook is intended to provide sufficient regional and landscape level protection of attributes that full life-boating of species at the stand level is not required everywhere.

Several references (USDA Forest Service and USDI BLM 1994, Clayoquot Sound Science Panel 1995; Franklin et al. 1997) have suggested that to leave important legacies and increase the habitat values in the harvested matrix, levels of retention in variable retention should range between 15% and 70% of the stand. Although these recommendations were made by expert panels, there is little scientific justification for these percentages in the literature. As well as the amount of retention, the spatial arrangement of retention is also important (Franklin et al. 1997). These issues are discussed in detail in a review paper by Franklin et al. (1997) on the effectiveness of stand-level retention for maintaining late-seral structure and diversity.

One assumption to be made is that maintaining stand structure will provide an area-weighted benefit in terms of stand recovery (Holt and Sutherland 2004); effectively, the greater the retention the faster the recovery of the second-growth stand towards old forest. The minimum target is therefore particularly of interest: maintaining a minimum of 15% retention results in the majority of an opening being under "forest influence", defined as the biophysical effects of trees on the environment of the surrounding land. The degree, type and distance of influence can vary widely; however, within and adjacent to harvested areas, most forest edge and residual tree influences begin to diminish significantly at distances greater than one tree length from a standing tree, group of trees or forest edge (Keenan and Kimmins 1993). Specific influences vary by species, aspect and slope, and single trees do not have the same influence as groups or stand edges. Forest influence is one criterion for retention (Mitchell and Beese 2002). Although some level of forest influence can be maintained with 10% retention in 0.25 ha groups spaced no more than four tree lengths apart, the 15% level allows for greater flexibility of layout and potential losses to windthrow. The effectiveness of this minimal level of forest influence will vary by species, therefore, a range of retention levels is desirable to meet watershed and landscape-level objectives.

In certain cases, perhaps where a watershed is already heavily harvested, stand level retention may be required as a tool to lifeboat species or to provide habitat for species requiring forest interior conditions. (Ideally in most cases those functions would be fulfilled by older harvested stands and stands outside of the harvested landbase). A review of literature indicates that the life-boating of most late-seral species can be achieved at retention levels of about 30%, much lower than typically maintained after gap disturbance (Moola 2003, unpublished results), because species can remain on site under open conditions as buried propagules, rhizomes, etc. or are able to use whatever habitat remains after logging until forest cover is re-established (Halpern 1989; Bunnell et al. 2003). The assembled evidence indicates that at retention levels between 30–70% of the pre-disturbance stand, the abundance of late-seral species remains much more similar to pre-disturbance levels (Beese and Bryant 1999). These numbers for lifeboating are preliminary, little data has been collected from gap-driven coastal temperate forests to support specific thresholds and providing general recommendations is complicated by the great variability in life-history and population structure among potential target species (Vanha-majamaa and Jalonen 2001).

If stand retention is intended to accommodate species associated with forest interior (e.g., possibly Rattlesnake plantain, *Goodyera oblongifolia*; Chen et al. 1996, and other species Kremsater and Bunnell 1999), then patches must be large enough and intact enough to contain some interior conditions. Due to the edge influences associated with smaller isolated forest remnants, such conditions can only be met at the stand-level if retention levels are considerable (50–70%) and available in unfragmented patches (>1 ha; Franklin et al. 1997). Although large intact forest patches are probably more efficient at life-boating late-seral species than smaller ones (Vanha-majamaa and Jalonen 2001), the pre-disturbance richness of some groups (e.g., vascular plants, invertebrates) can be maintained in even small fragments due to the limited spatial extent to which edge effects negatively impact most species within these groups (Bunnell et al. 2003). This may not be the case with many mosses, liverworts and lichens which are more sensitive to the microclimatic changes associated with patch fragmentation at the stand-scale, particularly those species that are prone to desiccation (Sillet 1995). In Clayoquot Sound, lichen communities in small (<1 ha) young-mature stands with 25% retention differed from paired old-growth communities (Price and Hochachka 2001). Some findings, however, suggest even some of these species will persist in small patches (see review in Bunnell et al. 2003).

Decisions about how much to retain at the site level will be based on landscape- or watershed-level analyses and site-level features. How the retention is distributed within a block will usually be based on stand-level analyses. The ecological features of a site, along with operational requirements, will largely determine the spatial pattern of retention. Generally, group retention is favoured over dispersed retention; from a biological perspective, group retention retains remnants of intact forest structure and understory habitat, including snags that are often lost in dispersed retention. However, dispersed retention may be important for maintaining old trees distributed across a block, to serve as sources of inoculum (e.g., lichens aboveground, mycorrhizae below ground) for a new stand.

Retention may be for social or cultural values as well (e.g., visuals, culturally modified trees). Retention for non-biological values is incremental to the percentages retained for biological reasons.

4.4.5 Uncertainty

The limits of 15% and 70%, defined by expert panels, have not been tested for effects on a variety of organisms, but studies are ongoing. A 30% minimum for life-boating some species seems better established. The risks to various groups of organisms of different amounts and patterns of retention are uncertain. The incremental value of different levels of retention to watershed-level reserves is unknown.

4.4.6 Important Questions for Adaptive Management

Important questions (Bunnell et al. 2003) for adaptive management include

- How do different groups of organisms, elements, or processes respond to different retention amounts?
- How do different groups of organisms, elements or processes respond to different retention patterns (dispersed vs. aggregate vs. mixed)?
- How are these responses affected by landscape-level effects (e.g., distance to nearest patch of mature forest)?

In addition, the temporal aspects of retention, i.e., how does retention influence the recovery rate towards true old forest characteristics may be particularly important in understanding longer term effects.

4.5 Fish Habitat

The following reports within the compilation provide more detailed information:

Church, M. and B. Eaton. 2001. Hydrological effects of forest harvest in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #3.

Hydroriparian Planning Guide. 2003.

Price, K. and M. Church. 2002. Risk to ecosystem functions. Summary of expert workshops. Hydroriparian Planning Guide Background Information.

Price, K. and D. McLennan. 2001. Hydroriparian ecosystems of the North Coast. Background report to the North Coast LRMP.

Young, K. 2001. Review and meta-analysis of the effects of riparian zone logging on stream ecosystems in the Pacific Northwest. Hydroriparian Planning Guide Technical Report #4.

4.5.1 Definition

High-valued fish habitat includes physical components of aquatic ecosystems that are important for the completion of fish life cycles. Alterations of these physical components have the potential to affect fish at the individual and at population levels. In this context, however, the specific management approach relates to the conservation of fish populations. Examples of high-valued fish habitat areas on the Central Coast of British Columbia include estuaries, eel grass beds, salmonid and eulachon spawning areas, off-channel rearing habitats.

4.5.2 Importance of Element

High-valued fish habitats provide an important function in allowing the completion of fish life cycles, specifically spawning and rearing. They also provide the function of nutrient exchange areas between aquatic and terrestrial ecosystems.

After spawning, anadromous salmonids provide marine-derived nutrients to freshwater ecosystems through excretion and decomposition of fish carcasses (Bilby et al. 1996; Gende et al. 2002). These nutrients can be important to the productivity of the aquatic ecosystems in which they spawn. When salmon die after spawning, their carcasses become food for many species of birds and mammals thereby affecting terrestrial biodiversity. Nutrients from salmon carcasses can also benefit riparian vegetation (Bilby et al. 1996).

4.5.3 Impacts of Forest Harvesting

Changes in salmonid populations integrate influences of habitat, fishing pressure and marine conditions, making interpretation difficult. Changes in habitat lead to increases in abundance in some species and declines in others (Reeves et al. 1998). There has been considerable research on the habitat requirements of fish on southern Vancouver Island and the Queen Charlotte Islands/Haida Gwaii (Hogan et al 1998). Principal requirements are for stable, diverse channels with downed wood and for clean gravel for spawning. Hence, activities that reduce channel complexity, reduce downed wood or change patterns of sedimentation can alter the quality of fish habitat and thereby influence fish populations.

Young (2001) provides a meta-analysis of the impacts of forest harvesting in coastal forests and recommends that protection of aquatic ecosystems requires a buffer width of at least one standard tree height for hillslope constrained channels capable of transporting water, sediment and organic debris over ecologically relevant timescales.

4.5.4 Selected Indicators

Ensuring healthy fish habitat requires maintenance of all hydroriparian ecosystem functions throughout watersheds. The additional indicators selected are designed to capture high-valued fish habitat that may not be covered by other indicators.

- location of forest development activities (roads, landings, log dumps, log sorts, cutblocks etc.) in or adjacent to high-valued fish habitat
- clearance (% of forest area) in watersheds smaller than 1,000 ha

4.5.5 Risk

The EBM Handbook gives a precautionary guideline:

- Reserve all high-valued fish habitat and adjacent areas from development; "adjacent" refers to any land from which there may be direct effects on the habitat as a result of development; "direct effects" refers to changes in temperature, water quality, sedimentation and streambank stability.

Precautionary guidelines listed under this section are intended to prevent direct effects to high-valued fish habitat. Indirect effects are covered by guidelines listed under Representative Hydroriparian Ecosystems (Section 4.2).

Precautionary guidelines listed under Representative Hydroriparian Ecosystems should be followed for watersheds over 1,000 ha. Because process characteristics differ between these systems and smaller watershed systems, a different set of precautionary guidelines apply to watersheds smaller than 1,000 ha (Gomi et al. 2002). For primary watersheds (i.e., those that drain directly into the ocean and are not part of a larger drainage basin) smaller than 1,000 ha (10 km^2), that contain high-valued fish habitat zones, the following guidelines apply:

- Identify all hydrologically active areas, for example, hydroriparian zones, headwater seepage zones
- Limit disturbance to 10% of the forest area averaged over 3 years.
- Deactivate road networks after harvesting to restore natural hydrological regime.

For high-valued fish habitat, there is no option to follow a risk-assessment procedure; any development is considered to pose an unacceptable risk.

The Hydroriparian Planning Guide recommends that refugia be set aside in watersheds where harvesting has impacted fish habitat until habitat is judged to have recovered (Sedell et al. 1990).

4.5.6 Uncertainty

- Influence of climate change on aquatic productivity
- Influence of marine survival on salmon populations

4.5.7 Important Questions for Adaptive Management

Can reserved or habitat refugia serve as control sites for adaptive management purposes?

At present DFO has a series of monitored streams on the Central Coast that are used for enumeration and to measure run returns. These systems have no reserved status under the current land management framework but could be built upon for adaptive management.

4.6 Hydroriparian Process Zones

The following reports within the compilation provide more detailed information:

Hydroriparian Planning Guide (2003)

Price, K. 2003. Testing the Hydroriparian Planning Guide. Report to the CIT and North Coast LRMP.

This section describes the rationale and methodology, based on terrain units, for division into process zones.

4.6.1 Definition

Several of the hydroriparian guidelines divide the watershed into three "process zones": source, transportation and deposition zones. Hydroriparian functions, processes and risks vary among the three zones.

Source zone Source zone channels are small upland/headwater streams where channelled drainage begins. Channels in source zones receive material directly from hillslopes via snow avalanches and landslides, deliver fine sediments and nutrients to larger channels continually, and large sediment and organic debris to larger channels episodically via debris torrents, avalanches, and landslides. In wetland source zones, streams may receive mainly organic materials, and may transport only fine organic material. Source zone channels comprise the majority of channel length within the watershed.

Transportation zone channels occur in valleys that include a valley flat and that receive material from source zone channels and directly from adjacent riparian zones and transport them to deposition zones. Transport zone channels may be partly confined by hillslopes, migrate across valley floors, or alternate between confined and unconfined. They are associated with a discontinuous or continuous floodplain.

Deposition zone channels are unconfined valley bottom rivers characterised by horizontal migration across flood plains and valley bottoms. Deposition zones include alluvial fans and deltas. Deposition zone channels receive material from origin and transport zone channels as well as from adjacent riparian zones. Deposition zones are small but disproportionately important. In many cases, management in the transportation and deposition process zones may be similar, but there will be cases where management should differ.

There will be some overlap among zones. For example material will be both transported and deposited on fans. The process zones are designed to capture the primary process occurring. At finer scales, pockets of transportation or deposition (e.g., wetlands) may be detected in the source zone. Process zones are a conceptual planning tool, rather than hard and fast lines.

4.6.2 Delineating Process Zones

The EBM Handbook includes targets for reserves within process zones: • reserve buffers around all streams in the transportation and deposition zones

Terrain mapping provides the information required to classify areas into the deposition, transport, and source process zones. Table 9 identifies the common terrain symbols found in the deposition and transport zones. The source process zone is simply defined as the area not included in the deposition and transport zones.

Variants (including composite and stratigraphic symbols) exist and some familiarity with the B.C. Terrain Classification System will aid in accurately identifying which terrain symbols are associated with a given process zone. The table is intended as a guide, to be used in conjunction with the detailed descriptions of the three process zones given above. Because of the complexity of terrain symbols, drawing process zones simply by using GIS queries can result in area missed from the transportation and deposition zones because of the high number of possible combinations of descriptors. A terrain map should be used in conjunction with GIS queries.

No reliable procedures exist to map process zones for watersheds with no, or questionable, terrain mapping. Designation of landform in the current version of ssPEM does an extremely poor job of capturing floodplains and fans. Slope criteria could identify major floodplain units, but would lose pocket units, particularly fans, which could represent important habitat islands.

Table 9. Terrain symbols used to define transportation and deposition process zones

Process zone	General terrain symbol (surficial material/surface expression); comments
Deposition	**Ff** (fluvial fan); Essentially a low gradient (here defined as 0–7%) alluvial fan/delta situated at the outlet of the watershed in question.
	Fp (floodplain) or **F^{A}p** (active floodplain); Extensive Fp or FAp terrain polygons found near the outlet in larger systems may also be included in this process zone.
Transport	**Fp** (floodplain), **F^{A}p** (active floodplain), other active fluvial units (e.g., **F^{A}b, F^{A}v**)
	Ft (fluvial terrace)
	Cf , **Cc** (colluvial fan, colluvial cone); These landforms typically flank valley sides in the outer and inner coast mountain sub-regions. Only include colluvial cones that are associated with water.
	Ff (fluvial fan); Only if it does not fit the criteria for fluvial fans in the deposition process zone (e.g., low gradient, situated at outlet)
	L^{A}j, L^{A}p (lacustrine gentle slope, lacustrine plain)
	Op (organic plain) , **Ov** (organic veneer); When found in composite or stratigraphic symbols in conjunction with Fp

Add a buffer around the transport polygons defined by either

1. terrain polygons adjacent to the transport zone with a slope class range of 2 – 3 or less or
2. one and a half tree heights.

This buffer includes most of the extent of terrestrial ecosystems (Price 2003). Price (2003) tests the process zone concept in two watersheds in the North Coast and compares the two methods of adding a buffer.

4.7 Riparian Corridors

The following reports within the compilation provide more detailed information:

Dykstra, P. 2003. Habitat supply thresholds: a literature synthesis. Draft report to MSRM.

Price, K. and M. Church. 2002. Risk to ecosystem functions. Summary of expert workshops. Hydroriparian Planning Guide Background Information.

Price, K. and D. McLennan. 2001. Hydroriparian ecosystems of the North Coast. Background report to the North Coast LRMP.

Price, K. and D. McLennan. 2002. Impacts of forest harvesting on terrestrial riparian ecosystems of the Pacific Northwest. Hydroriparian Planning Guide Technical Report #7.

4.7.1 Definition

Riparian corridors include windfirm natural riparian vegetation extending to the edge of the hydroriparian zone. Refer to Price 2003 for detailed examples.

4.7.2 Importance of Element

Although the amount of habitat is paramount, species presence and movement are affected by corridors and connectivity (Dykstra 2003). Riparian ecosystems, in particular, form linear connections in naturally forested landscapes and are used as corridors by plants, invertebrates and vertebrates (Price and McLennan 2001), although evidence for movement of terrestrial vertebrates along strips of riparian vegetation left after forest harvesting is sparse and limited to birds (Bunnell et al. 1998). Riparian corridors in managed landscapes pose risks: whereas the edge on one side of undisturbed riparian ecosystems merges into continuous forest away from the water, linear buffers provide only edge (Bunnell et al. 1998) and often blow down (Reid and Hilton 1998). Riparian forest corridors may be an effective supplement to continuous forest for maintaining biodiversity (Lomolino and Perault 2000; Perault and Lomolino 2000).

4.7.3 Impacts of Forest Harvesting

The study of impacts of partial harvesting within riparian corridors is in its infancy and there are insufficient data to draw even tentative conclusions (Price and McLennan 2002).

4.7.4 Selected Indicators

- percentage of streams in each process zone with natural levels of cover from origin to outflow.

The corridor function requires a different indicator to deal with connectivity. Overall abundance values would not discriminate between a watershed with half of the streams cut completely and half untouched and a watershed with all streams cut along half of their length. Modifying the indicator to look at the percentage of streams with natural riparian forest captures watershed patterns at a gross scale.

Apart from the valley-bottom, mainstem stream, the transportation zone generally contains short stream sections. Precautionary guidelines under bank stability require that these short portions have wind-firm buffers—hence a corridor guideline is redundant in the transportation zone if precautionary guidelines are followed. The indicator is of primary importance in the source zone.

4.7.5 Risk

The EBM Handbook provides a precautionary guideline:

- Create small stream protection areas to include riparian corridors. Maintain a minimum of 60% of streams with natural levels of cover along their length.

Due to high uncertainty about the use of riparian corridors, an expert group convened to develop risk curves for hydroriparian ecosystems of British Columbia’s north and central coast and they drew a linear curve (Figure 13). Although natural levels of connected cover vary across the coast (e.g., Skidegate Plateau had streams with naturally connected forest from head to mouth; whereas mainland streams usually cross avalanche tracks, bogs and other non-forested ecosystems), a single curve for all regions reflects the current uncertain knowledge. The curve does not apply at the level of individual streams, but to the population of streams within a watershed. Hence it looks at the proportion of all streams within a process zone with natural cover along their length. For example, if the predicted amount of old forest in the source zone of a particular watershed is 80–90%, the indicator counts the number of streams with more than 80% cover along the reaches in forested areas of the watershed, and expresses this as a percent of the total number of streams.

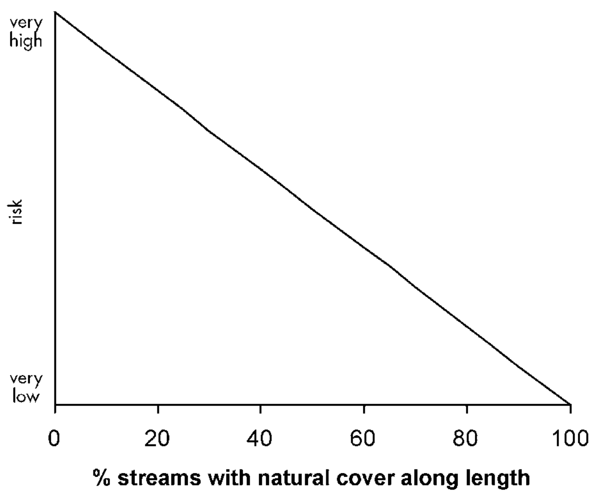


Figure 13. Risk to riparian corridors as a function of the number of streams with natural levels of riparian cover.

Calculating the corridor indicator involves counting streams within a process zone. In the source zone, it is relatively easy to count the number of channels crossing the boundary between the source and transportation zones. These channels include mainly first and second order streams with seasonal or perennial flow. In the transportation zone, it is possible to count the number of channels joining the main channel. With forest cover overlaid, it is possible to count the number of these streams in each zone with at least a certain proportion over a given age.

There are challenges associated with the corridor indicator. Most source streams will not be marked on 1:20,000 maps in watersheds without harvesting history. Streams are difficult to count and vary tremendously in their character. However, a simple rule, counting the number of streams entering the transportation zone (and ignoring stream order or persistence), still provides novel information (Price 2003).

4.7.6 Uncertainty

“Further work is necessary to understand what aspects of spatial arrangement of habitat patches affect the process of extinction” (Dykstra 2003). It is unclear how theoretical thresholds examined over uniform landscapes apply to linear riparian ecosystems.

Little is know about the effectiveness of riparian strips in managed landscapes as corridors

Little is known about the impacts to biodiversity of management around small streams

4.7.7 Important Questions for Adaptive Management

Are continuous strips of riparian vegetation valuable as corridors?

The value of remnant strips remains contentious. Unfortunately, this area of study is highly complex, and frequently troubled by ambiguous results. Initial work should document "natural" levels of connectivity along streams. This can be viewed as baseline data collection. Subsequent active studies and passive monitoring should examine how a variety of organisms (not just birds) use these corridors. This question will involve research (field and landscape-level modeling experiments) as well as adaptive management. Answers to this question are important to determining the best way to manage the pattern of harvest around small streams.

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