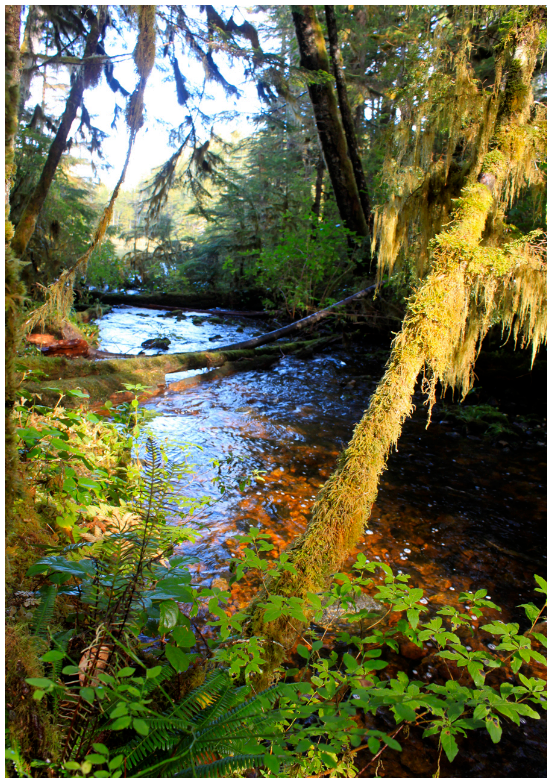
Ecosystem-based management in the Great Bear Rainforest:

A knowledge summary for priority ecological questions and experimental watersheds design



Prepared for the Coast Forest Region, BC Ministry of Forests, Lands and Natural Resource Operations

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April, 2013

Executive Summary

In response to concerns that conventional forest management results in biodiversity loss and limits social and economic opportunities for future generations, Ecosystem-based Management (EBM) has been adopted within the Central and North Coast area of British Columbia (Price et al. 2009). The Land Use Objectives (LUOs) were developed to support implementation of EBM (Central and North Coast Order 2009) and it is expected that they will be reviewed and amended if they do not perform well in terms of improving ecological and human well-being. Adaptive management will guide research and monitoring to examine how well these management goals are being met.

The Experimental Watershed Programme was established to conduct large-scale research and monitoring initiatives that examine the outcomes of forest management practices under EBM. To inform the design of this Programme, the Coast Area Forest Research Team of the Ministry of Forests, Lands and Natural Resource Operations (MFLNRO) compiled a list of priority research questions related to uncertainty around how forest management impacts physical and biological components of forest function, and around the efficacy of critical definitions within the LUOs.

The four priority research questions are as follows:

- Priority Question 1 - Hydrological Issues associated with definitions of function and relationship to management.
- Priority Question 2 - Information on buffers (management implications, characteristics of streams that interact with and affect role and application of buffer management).
- Priority Question 3: Issues of range of natural variation in amounts of forested systems, habitat supply thresholds, and landscape-level conservation.
- Priority Question 4: Interactions between stand-level retention and landscape-level representation/conservation.

Responses to these questions will update the existing body of knowledge available to MFLNRO on these issues (e.g. as discussed in Fenger et al. 2009), and are anticipated to help guide experimental work as well as inform revision of the LUOs. Therefore, there are three primary objectives of this report:

1) To synthesize primary and grey literature published from 2002 to present pertaining to the priority forest management research questions;
2) To review the efficacy of the BC EBM LUOs and make recommendations for their revision based on primary literature; and
3) To identify knowledge gaps in the literature that could be investigated using an experimental watershed approach and suggest methods to be incorporated in the design, implementation, and data analysis of experimental watershed projects.

Main Findings

Priority Question 1:

- The terms 'functional' in functional riparian forest and 'hydrologically effective greenup' as defined by the Orders should be clarified.
- Equivalent Clearcut Areas (ECA) of 20% in coastal watersheds may still pose significant risks to stream structure and function.
- Headwater streams should be afforded increased levels of protection to minimize forestry-related risks to hydrologic function.
- Forest harvesting impacts to hydrologic function are predicted to vary considerably across the diverse environmental gradients present in the Great Bear Rainforest. The ECA framework would thus benefit by including the diverse factors that affect stream flow such as climate, vegetation, elevation, topography, soil types, presence of lakes, glaciers, etc.

Priority Question 2:

- At the landscape scale, connectivity of channels to uplands, headwaters to ridgelines, and linkage areas should be maintained.
- 1.5 times tree height is not likely to be effective in maintaining the diversity of riparian reserve functions if the dominant trees are 20 m or less in height.
- TSFA stability ratings are unlikely to appropriately identify downslope areas that are at risk of upslope landslide activity.
- Requiring riparian reserve zones only for S1 to S3 streams is not sufficient to protect these streams from negative impacts of upstream harvesting.

Increasing riparian buffer width for S4 to S6 streams (fish-bearing or not), will reduce downstream impacts of forest harvesting.

- The foundation for determining riparian practices is largely fish-based, yet there is no scientifically-sound basis for managing riparian and aquatic values on the presence of game fish alone. Research continues to support that monitoring stream-associated amphibian populations provides a better index of riparian forest functioning and biodiversity.

Priority Question 3:

- Species loss and population declines will take place above threshold levels.
- Critical habitat for priority species has in most cases been identified and in some cases mapped.
- Fungi, lichens, bryophytes are among the most sensitive to logging. Monitoring these taxonomic groups is likely to provide sensitive tests of the impacts of the spatial arrangement of logging at both stand and landscape scales.
- Habitat quality is a key predictor of species distributions; therefore, not accounting for it in studies of species responses to habitat loss and configuration may lead to inaccurate conclusions.

Priority Question 4:

- The literature agrees that most species, especially late-seral species, decrease in abundance and richness with less than 15% retention.
- The literature appears to support definitions of high risk at the stand scale as 15% retention, and low risk as retention that exceeds 70%. Kremsater et al. (2008) would recommend a minimum of 30% retention.
- Landscape-level studies of habitat representation are lacking.

Implications for Experimental Watershed Design

Although there has been a substantial amount of research on topics relevant to the priority ecological questions in the past few years, there remain major gaps in our understanding. Most uncertainties stem from the fact that there are currently very few studies at the landscape scale. The Experimental Watersheds Programme has the potential to fill significant knowledge gaps in our understanding of ecosystems at the landscape level. For example, with BC's BEC classification system, a unique opportunity exists to link studies to BEC Site Series in order to measure landscape-scale representation.

Throughout this report, we list considerations for Experimental Watershed design (EWD) that have been informed by recent literature. We recommend the design of the Programme to include:

- A minimum of 50 S1 to S3 streams and 100 to 1000 S4 to S6 streams, with differences in the intensity and frequency of surveys depending on stream type.
- A partnership with the Central Coast First Nations in monitoring priority watersheds on the Central Coast should be implemented.
- A partnership with academia (e.g. the Hakai Network) as well as among other stakeholders (DFO, MoE, forest companies, Hakai Institute).
- Greater than 10 years of data collection to account for delayed responses.
- Investigation into the short- and long-term response of headwater ecosystems to disturbances.
- Linking changes in species responses to measures of landscape disturbance, rather than just amount of forest cover, to provide a more relevant measure of risk.
- Assessments of population density, distribution and habitat relationships for both priority and non-priority species (e.g., brown creeper, northern flying squirrel) that are sensitive to harvesting.
- An assessment to understand the state of knowledge and level of uncertainty associated with the range of natural variability in each BEC zone.
- Use of late seral-associated forest species as indicators of retention effectiveness
- Economic analyses that evaluate tradeoffs between different forest practices, including the possibilities surrounding carbon credits, more community-based processing of wood products, and tourism.

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Background

Forest harvesting and road construction can have substantial negative impacts on watershed hydrology, stream channel geomorphology, hillslope processes, and the biodiversity of terrestrial and aquatic ecosystems (Pike et al. 2010). These impacts are highly context-specific, but generally result in changes to forest systems and species population viability that can persist for decades.

The level of disturbance that forests may buffer across spatial and temporal scales and the efficacy of disturbance-mitigation practices (such as riparian buffers) are ongoing areas of study. In British Columbia, the 10-year results of forestry management have only recently been summarized for the province's first prescriptive forestry legislation: the 1995 *Forest Practice Code* (FPC; Forest Practices Board 2005). Many of the rules-based elements in the FPC continue to be regularly applied (e.g. fixed minimum riparian reserve widths, deactivation of roads), despite the fact that the FPC's successor, the current *Forest and Range Practices Act*, focuses on results-based approaches. Both primary and grey literature have increasingly concluded that more precautionary management regimes are needed.

In response to concerns that conventional forest management results in biodiversity loss and limits social and economic opportunities for future generations, Ecosystem-based Management (EBM) has been adopted within the Central and North Coast area of BC (Price et al. 2009). EBM is defined as *"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"* (Coast Information Team 2001). Figure 1 shows Land Use Zone designations agreed upon for this region, also known more popularly as the Great Bear Rainforest. Land use zones include Biodiversity/ Mining/ Tourism Areas, New Protected Areas, Existing Parks and Protected Areas, and Resource Development Areas to be managed under EBM.

The Land Use Objectives (LUOs) were developed to support implementation of EBM (Central and North Coast Order 2009) and it is expected that they will be reviewed and amended if they do not perform well in terms of improving ecological and human well-being. Thus, First Nation involvement, stakeholder participation, and adaptive management are critical to the success of EBM. Adaptive management will involve developing research and monitoring plans to examine the outcomes of management strategies.

To help enact this adaptive management strategy, the Ministry of Forests, Lands and Natural Resource Operations (MFLNRO) established an Experimental Watershed Programme with a goal to conduct large-scale research and monitoring that examines outcomes of forest management practices under EBM. As one of the first steps in the Experimental Watershed Programme, the Coast Area Forest Research Team of MFLNRO compiled a list of priority research questions related to uncertainty around how forest management impacts physical and biological components of forest function, and around the efficacy of critical definitions within the LUOs. Responses to these questions will update the existing body of knowledge available to MFLNRO on these issues (e.g. as discussed in Fenger et al. 2009), and are anticipated to help guide experimental work as well as inform revision of the LUOs.

There are three primary objectives of this report:

1) To synthesize primary and grey literature published from 2002 to present pertaining to the priority forest management research questions (Table 1);
2) To review the efficacy of the BC EBM LUOs and make recommendations for their revision based on primary literature; and
3) To identify knowledge gaps in the literature that could be investigated using an experimental watershed approach and suggest methods to be incorporated in the design, implementation, and data analysis of experimental watershed projects.

Table 1. Priority Questions compiled by the Coast Area Forest Research Team (MFLNRO)

Priority Question 1 - Hydrological Issues associated with definitions of function and relationship to management
a. How do we define the term 'functional' in functional riparian forest as defined by the Orders? What characteristics should functional riparian forest provide? b. How can concept of Equivalent Clearcut Area (ECA) be refined for coastal BC? (no targets exist for HVFH or general aquatic habitat)? c. How does the level of cut in small watersheds affect hydrological function and recovery?
Priority Question 2 - Information on buffers (management implications, characteristics of streams that interact with and affect role and application of buffer management).

a. What are the impacts of forest management activities within and adjacent to hydropriparian buffers on biodiversity and productivity of terrestrial and aquatic systems?
    i. Species level impacts
    ii. Impacts specific to small upland streams (>6% gradients such as S4, S5 and S7s)
b. How effective is 1.5X tree height for maintaining riparian function? And what are the impacts of wind damage in hydroriparian buffers on species diversity and site productivity and water quality?
c. Does forest management under EBM LUOs affect High Value Fish Habitat channel morphology (or general aquatic habitat channel morphology)? And does HVFH channel morphology change (beyond RONV) in managed watersheds?
d. How does forest management under EBM LUO's affect sediment supply to HVFH (or general aquatic habitat) and impact HVFH (or general aquatic habitat) function? Specifically, what is the impact of harvesting and road building activities on:
    i. Class IV terrain and around small steep streams with high potential for debris transport?
    ii. Streambank stability, channel movement and sediment input for active fluvial units under differing forested buffer widths? (riparian buffers)
e. Does 70% retention of the forest around small streams maintain function? Can there be some partial harvest and to what level?

Priority Question 3: Issues of range of natural variation in amounts of forested systems, habitat supply thresholds and landscape level conservation.

a. How effective at maintaining ecological integrity and biodiversity across scales are the current targets for ecosystem representation?
    i. What level of habitat actually does maintain ecological integrity and biodiversity across scales?
    ii. What are appropriate definitions of risk associated with loss of/removal of ecosystems (site series) at a landscape scale?
    iii. How do sensitive species respond to low levels of landscape-level reserves?

iv. For priority species/ecosystems what is critical habitat and what level of stewardship will result in low risk management? v. What extent and spatial arrangement of mature forest habitat is necessary for capturing biodiversity in taxa such as cryptogams, fungi, arthropods? b. How does configuration of Ecosystem Representation influence ecological integrity as measured using suites of response variables? c. What are current levels of natural disturbance by ecosystem type? How does natural disturbance change over time?
Priority Question 4: Interactions between stand level retention and landscape level representation/conservation
a. What is the effectiveness [for various processes and measures of function] of stand level retention in the context of different levels of landscape level representation? i. Are there combinations of stand level and landscape level retention levels that pose high risk to species populations within watersheds / landscapes? ii. What impact do stand-level retention targets have on western redcedar regeneration and growth on the landscape? iii. What habitat elements are critical for ecological integrity in long-term retention within harvested cutblocks? iv. Does 15% retention retain important structure or ecological diversity? v. With >=15 ha, what does distributing 50% of the retention do for structure and diversity? b. What constitutes "excellent retention" from an ecological perspective? c. Are current recommended accounting strategies for stand level retention contributions to landscape level ecosystem representation targets valid?

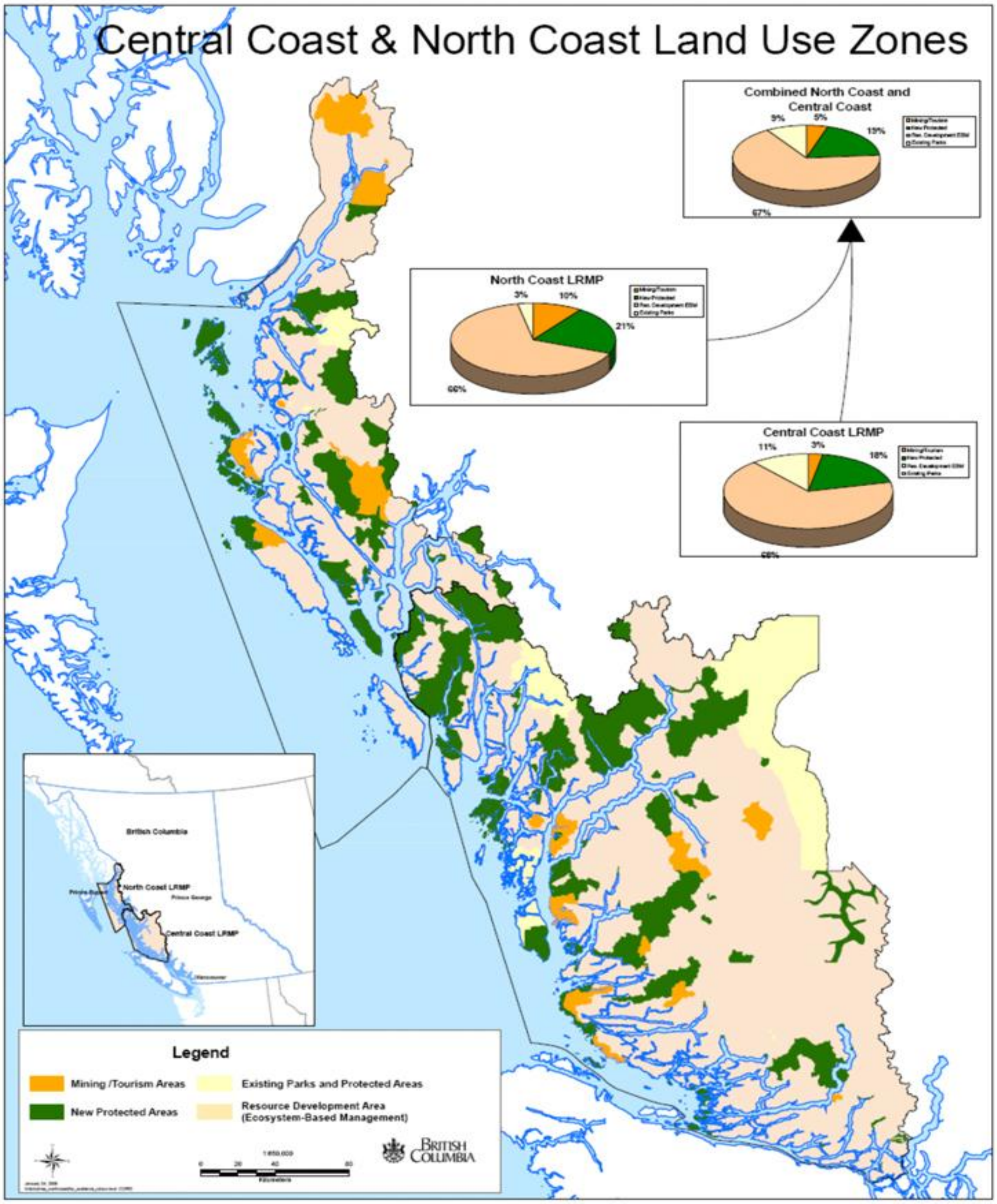

Figure 1. Central and North Coast land use zones including conservancies (green = new protected areas, yellow = existing parks), biodiversity, mining and tourism areas (orange), and resource development areas under EBM (beige).

Priority Question 1.

Hydrological Issues associated with definitions of function and relationship to management

a. How do we define the term 'functional' in functional riparian forest as defined by the Orders? What characteristics should functional riparian forest provide?

Summary of Knowledge

In the Central and North Coast Land Use Order 2009, “**functional riparian forest**” is defined as: “*forest that has reached hydrologically effective greenup and contains some large trees adjacent to streams to provide for large organic debris*”, while “**hydrologically effective greenup**” means “*the stage in the process of hydrologic recovery of a disturbed area at which a regenerating stand of trees has sufficient height, stocking density and canopy closure to prevent hydrologic response of the disturbed area from causing material, adverse changes in hillslope hydrology, stream channel condition, or stream flows.*”

Thus, in management terms on the Central and North Coast, the term ‘functional’ generally refers to the hydrologic functions provided by the loss and gain (through re-growth) of riparian and watershed forest cover and does not include all aspects of riparian functioning (e.g. biodiversity). Further, it is not clear from the orders what specific criteria of forest re-growth are used on the ground (e.g. time, canopy height, density etc.) to specify “hydrologically effective greenup”. Despite this lack of specificity, timber supply reviews on the Central and North Coast provide an indication of the measures used to assess this concept. For example, the Mid Coast Timber Supply Review 3 used a hydrologically effective greenup height of 9 m, and did not mention other characteristics. These timber supply reviews are intended to reflect the best available knowledge about on the ground forest practices.

The focus on hydrologic functioning in the LUOs for “functional riparian forest” is because reductions in forest cover strongly affect watershed hydrology, including the amount of precipitation that reaches the ground, evaporation, transpiration, and snow melt, and the depth of the water table (review by Winkler et al. 2010). There are three main indicators of hydrologic function and recovery of watersheds that have been monitored: a) water yield, b) peak flows, and c) low flows. There is strong evidence that forest harvesting affects all three of these streamflow attributes, with subsequent effects on channel morphology, fine sediments, water temperature and water quality, and biodiversity (reviews in Tschaplinski and Pike 2010; Winkler et al. 2010). Further details relating to hydrologic recovery and the concept of equivalent clearcut area (ECA) are discussed in response to question 1b below.

The literature highlights an expanded definition of "functional riparian forest". In addition to watershed hydrology, riparian vegetation affects a range of watershed functions including the amount of light that reaches streams, stream water temperature, the rate and kind of organic matter inputs to streams, bank stability, water quality, channel structure (e.g. Large Woody Debris (LWD) inputs), and biodiversity (summarized by Moore and Richardson 2010). The strength of coupling between riparian vegetation and stream function depends substantially on stream size and location within the watershed, with greater coupling adjacent to smaller streams, hillslopes and headwater streams compared to large downstream reaches.

Depending on stream width and canopy characteristics, forest canopies can intercept 95% or more of the light that reaches streams. Increased light levels occur after disturbances such as forest harvesting, which can enhance stream primary production (e.g. algal and periphyton growth) and increase water temperatures, which, in turn, can affect stream invertebrate communities and fish (Kiffney et al. 2003; Pike et al. 2010). In most small to medium-sized streams, terrestrial organic matter in the form of leaf litter, wood, and invertebrates provide the main form of energy and thus foundation for stream food webs. Within larger systems, the processing of these materials to finer particles provides an enormous flux of particles to downstream reaches (Moore and Richardson 2010). Forested riparian areas act as filters, intercepting sediments and nutrients that would otherwise enter streams (e.g. Nitrogen). Forested riparian areas also maintain stability and ground-water flow, limit erosion, and provide a crucial source for LWD, which maintains stream channel structure. By extension, this provides cover and regulates flows for aquatic organisms. Stream and riparian habitats also provide critical habitat for a range of terrestrial animals, plants and other organisms at some point in their life cycles – often referred to as obligate-riparian species (Moore and Richardson 2010).

Tschaplinski and Pike (2010) provide a broader definition for how we might define the word 'functional' in functional riparian forest in Chapter 15 titled "Riparian Management and Effects on Function" in the *Compendium of Forest Hydrology and Geomorphology in B.C*. The Province of B.C. has set up the Forest and Range Evaluation Program (FREP) to examine post-harvest management outcomes and effects on stream and riparian functioning. A series of riparian, stream, and aquatic habitat indicators are used to assess riparian management effectiveness (Table 2).

Table 2. Riparian, stream, and aquatic habitat indicators used for the routine-level assessment or riparian management effectiveness evaluations in BC (from Tschaplinski and Pike 2010).

Riparian, stream, and aquatic habitat indicators	
Channel bed disturbance	Aquatic invertebrate diversity
Channel bank disturbance	Windthrow frequency
LWD characteristics	Riparian soil disturbance/bare ground
Channel morphology	LWD supply/root network
Aquatic connectivity	Shade and microclimate
Fish cover diversity	Disturbance increaser plants/noxious weeds/invasive plan
Moss abundance and condition	Vegetation form, vigour, and structure
Fine sediments	

A total of 1441 streams have been assessed using the FREP system from 2005-2008 with "Properly functioning condition" defined as the ability of a stream and riparian area to:

- Withstand normal peak flood events without experiencing accelerated soil loss, channel movement, or bank movement;
- Filter runoff;
- Store and safely release water;
- Maintain connectivity of fish habitats;
- Maintain an adequate riparian root network and LWD supply; and
- Provide shade and reduce bank microclimate change.

Of the 1441 streams assessed using the FREP "Properly functioning condition" criteria, 87% of streams assessed maintained some degree of functioning condition. This means that some indicators may have failed but that most indicators passed, despite forest harvesting. Streams defined as “Not functioning” were mostly S4 and S6 streams. However, increased fine sediments affected 63% of all streams.

Sheldon et al. (2012) provide a similar framework to test “properly functioning condition” from the Freshwater Ecosystem Health Monitoring Program in Australia. Their aim was to identify the spatial scale of land use and disturbance that affects river ecosystem health scores from 116 streams in eastern Australia. They used linear mixed-effects and Bayesian model-averaging to generate models for an overall ecosystem health score and 5 component indicators (fish, macro-invertebrates, water quality, nutrients, ecosystem processes) (Figure 2). They competed models using metrics at different spatial scales (reach, riparian zone, and catchment) and found an overriding influence of forest cover close to the stream throughout the catchment in influencing ecosystem health across all component indicators. Their results suggest that good ecosystem health can be maintained in catchments where 80% of hydrologically active areas have mid-dense forest cover, while moderate health can be maintained with 60% cover.

Despite major investment in research such as the FREP system described above, and long-term fish-forestry programs in BC such as in Carnation Creek, significant gaps remain in our understanding of forest harvesting impacts on stream and riparian function. Tschaplinski and Pike (2010) make the following statements that provide a perspective on the current state of knowledge of forest harvest effects on stream function in BC:

“*Different physical and biological attributes of streams and other aquatic ecosystems respond differently to riparian forestry according to the influence of climate, geology, natural disturbance regimes, channel type, aquatic communities, and channel interconnections within basins.”*

“*… the exact thresholds of riparian, channel and aquatic ecosystem responses to streamside management activity, site disturbance and vegetation retention remain unknown*.”

In summary, the biophysical attributes of streams strongly mediate the effects of forest harvesting on stream functioning, although the exact nature of many of these responses is not well known. This means that risks to function will vary considerably with environmental attributes both within and across watersheds.

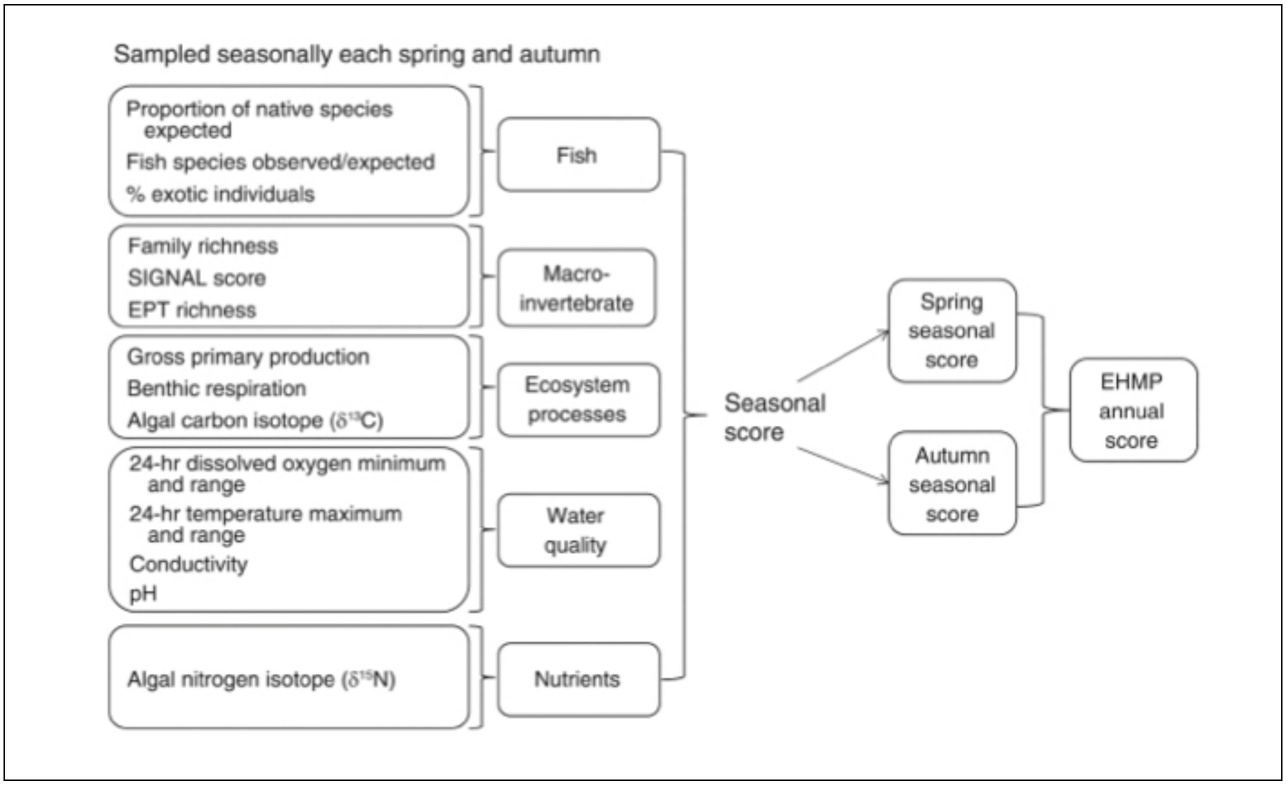

Figure 2. ***Diagram of types of data collected to generate an Ecosystem Health Monitoring Program (EHMP) annual score from 116 streams in Australia (from Sheldon et al. 2012).***

Relevant EBM LUOs

Section 8.(1) Maintain an equivalent clearcut area of less than 20% in important fisheries watersheds as set out in the areas shown in Schedule 3.

Section 10. Objectives for aquatic habitat that is not high value fish habitat

Section 11. Objectives for forested swamps

Section 12.(1) Maintain 70% or more of the forest, in the portion of the watershed where upland streams occur, as functional riparian forest.

Section 13.(1) Adjacent to active fluvial units, retain 90% of the functional riparian forest in a management zone with a width, on average, equal to 1.5 times the height of dominant trees.

Recommendations and Revisions

1. Based on research in BC using the FREP monitoring system (summarized in Tschaplinski and Pike 2010), impacts to stream and riparian function can be substantially reduced if riparian practices:

- Limit introduction of logging debris and riparian management area-related sediment into channels;
- Limit physical contact with streambanks and streambeds when falling and yarding around streams; fall and yard trees away from the channel wherever possible; and
- Retain more vegetation around S4 to S6 streams.

2. The definition of the term 'functional' in functional riparian forest as defined by the Orders should be expanded to include those outlined by Tschaplinski and Pike (2010) and utilized to evaluate properly functioning condition in the BC Government FREP program. Research needs to be conducted on the feasibility of collecting data based on this expanded definition of functional riparian forest. Further, the definition should be re-defined in more specific terms; specifically the section stating that functional riparian forest "*contains some large trees adjacent to streams to provide for large organic debris.*"

3. There is a clear relationship between stream size and/or location in the drainage network and the degree of coupling of riparian forest with stream functioning (e.g. Moore and Richardson 2010). In management terms, retaining more vegetation in S4 to S6 streams will decrease adverse risks to watershed function. One priority should be to retain more vegetation in S4 to S6 streams that are a part of the drainage network of important fisheries watersheds.

4. What does "hydrologically effective greenup" mean in practice on the ground? How long (in years) and what tree height or density are used to set "hydrologically effective greenup"? Hydrologic recovery will vary depending on many watershed factors, some of which are described below for question 1b and are relevant to how we might refine the concept of equivalent clearcut area. In general, we recommend that more specific guidelines for

“hydrologically effective greenup” be set in the Land Use Orders that reflect major environmental gradients such as climate, elevation, topography and watershed size.

5. The Government of BC should conduct a more thorough analysis of the FREP program data. Have more streams been analyzed for their function using FREP since 2008? With so many streams assessed (>1400), there is substantial power to build across-watershed predictive models of how forest harvesting treatments and environmental features of watersheds interact to affect stream and riparian function.

Considerations for Experimental Watershed Design

1. We recommend a large-scale across watershed study of the effects of forest harvesting on stream and riparian functioning on the Central Coast of BC where EBM is being implemented. Ideally this would include a minimum of 50 S1 to S3 streams and 100 to 1000 S4 to S6 streams, with differences in the intensity and frequency of surveys depending on stream type. This will allow for quantitative modeling of the effects of forest harvesting on stream functioning across the range of watershed attributes.

2. The BC Ministry of Forests, Coast Forest Region should consider a partnership with the Central Coast First Nations in monitoring priority watersheds on the Central Coast including the Heiltsuk, Kitasoo/Xai’Xais, Nuxalk and Wuikinuxv Nations. Additional possibilities for partnership include with Federal Departments such as Department of Fisheries and Oceans and Environment Canada, forest companies, and academia (e.g. the Hakai Network for Coastal People, Ecosystems and Management).

3. Results have shown that forestry-related impacts on streams and aquatic habitats can occur over two decades or more, particularly where impacts are related to mass wasting events in the headwaters and propagated over time down the stream channel network (Tschaplinski and Pike 2010). As such, a long-term monitoring and research framework is needed to assess the full impacts of forestry on stream function.

b. How can the concept of Equivalent Clearcut Area (ECA) be refined for coastal BC?

Summary of Knowledge

Forest harvesting increases the fraction of precipitation that is available to become stream flow (Moore and Wondzell 2005). This is because the amount of forest cover strongly affects watershed hydrology, including the interception of rain and snow, evaporation from soil surfaces, rates of snow melt and evaporation (together called snow ablation), plant transpiration, and soil water storage (review by Winkler et al. 2010). For example, the interception of annual precipitation by forests can range from 10-50% (Buttle 2011), and thus the amount of watershed area that is clearcut, or disturbed by fire, insects or disease directly increases the amount of precipitation that reaches the ground. Significantly greater snow water equivalents (SWE) are also observed in logged forests compared to unharvested forests. The combination of greater SWE and faster melt rates due to snowpack exposure leads to greater water inputs to flow. Net increases in precipitation and reduced evapotranspiration losses with vegetation removal results in increases in soil water storage. This leads to increases in groundwater recharge and discharge following harvesting, and higher water tables. The ability of water to infiltrate soils and recharge groundwater stores also decreases with soil compaction on roads and skid trails. The result is increased overland flow into channels (Buttle 2011). Increased rates of overland flow often generate high erosion rates and increased peak flows (USDA Forest Service 2010).

While many of these hydrologic impacts of forest harvesting at the site scale are reasonably well understood, it is more difficult to make watershed-scale predictions because of the natural variability in driving factors such as climate, geology, topography, and forest cover within and across watersheds (Winkler et al. 2010). Total water yield, and the intensity and frequency of peak flows and low flows are three watershed-scale variables of interest that integrate site-scale effects of forest harvesting. The subsequent changes to stream flow from harvesting are one of three interconnected factors (sediment supply, riparian vegetation, and streamflow) that affect channel morphology (Hogan and Luzi 2010). This is discussed in more detail in questions 2c and 2d.

In the context of forestry, hydrologic recovery (or as above 'hydrologically effective greenup") is defined as the process by which regeneration and regrowth of vegetation restores the hydrology of an area to pre-logging conditions. Equivalent clearcut area (ECA) is then defined as the area of a watershed that has been clearcut, with a reduction factor to account for the areas that have experienced hydrologic recovery through regeneration.

In the Central and North Coast Land Use Order 2009, "**equivalent clearcut area**" means: "*an indicator that quantifies the percentage of forested portion of a watershed where hydrologic response resulting from alteration of the forest by harvesting, fires, insects and disease is equivalent to the hydrologic response of a clearcut*". In other words, ECA describes a second-growth block in terms of its hydrological equivalent as a clearcut.

Indices of stand regeneration, such as canopy height, basal area or leaf area are generally used to determine hydrologic recovery and thus set ECA (Hudson and Horel 2007; Winker et al. 2010). While tree height is often used to set ECA, it may not be as good as leaf area, canopy density, or basal area at representing hydrologic processes, including interception, evaporation, soil moisture and water yield.

In snow-dominated watersheds, stand height is the principal stand descriptor for recovery rather than canopy density (Hudson and Horel 2007). In these systems, the hydrologic recovery threshold occurs at a level where the tallest trees in the stand are at a height roughly equal to the mean peak snow depth (Hudson 2000). Hudson (2000) found that 50% recovery occurred with trees at 4m height or a canopy density of 20%; 8m height or canopy density of 45% = 75% recovery; and 20m or more height or 95% canopy density = close to full recovery. In rainfall recovery watersheds, Adjusted Stand Height (ASH), which is the sum of stand height and a residual of the regression of stand height and canopy density, is now used to indicate hydrologic recovery (Hudson and Horel 2007).

Increasing ECA values are generally correlated with increasing hydrologic impacts and effects on water quality, quantity, and stream function (Hudson 2000, 2002; Schnorbus and Alila 2004; Klein et al. 2012). Recovery to pre-harvest conditions appear to occur within 10-20 years in some coastal catchments, but may take decades in snow dominated, mountainous catchments (Moore and Wondzell 2005). For example, in the Coast Range Mountains north of San Francisco, turbidity data was collected from 28 streams historically impacted from harvesting. ECA values for the 10-15 years preceding the collection of data were the best predictor of chronic turbidity, suggesting that harvesting could have much larger impacts and last for much longer time. A key reason for this lag time was hypothesized to relate to loss of root biomass and root decay. Though riparian buffers are commonly used to limit sediment from reaching the channel, hillslope-eroded sediment can pass through a buffer nonetheless, particularly when too much of the land is harvested on erosion-prone terrain over too short a time period (Klein et al. 2012).

The relative changes in peak flow can be large for relatively small changes in ECA (Hudson 2002). In a small coastal catchment on northern Vancouver Island, increases in ECA of 5% increased peak flow from 22-48%. Increases in ECA of 10% increase peak flow from 37-63%. These changes in flow are not based on roads, they are based on changes in forest cover and regeneration (Hudson 2002).

Small-watershed data support that watersheds located in rain-dominated regions (29% harvested area detection limit) are less sensitive to peak flow changes than those in the transient snow zone (15% harvested area detection limit) (Grant et al. 2008). Increases in peak flows observed in a BC interior Columbia Mountains watershed with harvesting are fundamentally a function of the frequency structure of snowmelt runoff (Schnorbus and Alila 2004). The characteristics of snow line retreat in this watershed produce the clear threshold effect at H60 elevation with respect to harvesting.

Harvesting in riparian areas can have a significant effect on hydrology. For example, in the Oregon Cascades, changes in riparian vegetation from conifers to deciduous species following harvesting increased transpiration by streamside vegetation and reduced dry weather stream flow (Winkler et al. 2010).

Hydrologic recovery is affected by (see Hudson and Horel 2007; Winker et al. 2010):

- Regional climate and hydrology;
- Canopy conditions (stand height and age, canopy density, canopy species, crown closure, and patchiness);
- Understory community;
- Elevation, which can determine dominance of rain, snowmelt or rain-on-snow;

- Site topography;
- Soil and bedrock types;
- Watershed features such as lakes and glaciers; and
- Watershed size.

We predict that hydrologic recovery will vary considerably across the gradient in hypermaritime to mountainous watersheds that are present on the Central and North Coast.

Relevant EBM LUOs

Section 8.(1) Maintain an equivalent clearcut area of less than 20% in important fisheries watersheds as set out in the areas shown in Schedule 3.

Recommendations and Revisions

1. It is generally assumed that ECAs at or below 20% pose low risk to stream and riparian function (CIT Hydroriparian Planning Guide 2004; Grant et al. 2008). Important fisheries watersheds on the Central Coast are mandated through the Land Use Orders to keep ECA at or below 20%. Hudson (2002), however, shows clear impacts of forest harvesting to peak flows in a small coastal watershed on Vancouver Island at ECA values of 17.5%. Thus, ECA's of 20% in coastal watersheds may still pose significant risks to stream structure and function. This may be particularly true if the land is harvested over too short a time period, or is on erosion-prone terrain (e.g. Klein et al. 2012). Risks may also be higher when significantly greater harvest (or ECA's) occurs at higher elevations in the watershed basin. Hydrologic recovery can differ between rain and rain-on-snow zones within watersheds (e.g. Hudson and Horel 2007), which should be factored into the calculation of ECA within watersheds.

2. How are ECA values calculated on the BC Central and North Coast? What environmental factors are included? Generally, the ECA framework would benefit from integrating this approach with other factors that affect stream flow such as climate, elevation, topography, soil types, presence of lakes, glaciers, etc. (e.g. Hudson and Horel 2007). On the Central and North Coast of BC there is major variation in watershed sizes and types that will affect hydrologic impacts and recoveries from harvesting (e.g. Banner et al. 2005). For example, we predict a strong west to east gradient in the elevational impacts of forest harvest on the flow

regime as systems move from rain, to rain-on-snow, to snow and glacial melt dominated runoff.

3. Is it only important fisheries watersheds that should keep ECA's less than 20%? What defines an important fisheries watershed? Almost all small streams on the Central Coast without impassable barriers support anadromous salmon (Harvey and MacDuffee 2002; Price et al. 2009), which provide a portfolio of salmon population diversity (e.g. Schindler et al. 2010) of relatively unknown significance for regional fisheries and ecosystem processes.

4. Within large watersheds, individual sub-basins also need to maintain ECA's at less than 20%.

Considerations for Experimental Watershed Design

Quote from Winkler et al. (2010): "Further research that combines field studies and modeling with long-term monitoring is needed to quantify the hazards associated with forestry related peak flow increases in BC watersheds of varying sizes, biophysical characteristics, and hydrologic regimes."

1. It is notoriously difficult to make inferences about hydrological processes from paired watershed experiments. Matching peak flows chronologically can be challenging because storms in control and treatment watersheds do not always coincide in time, duration, intensity, or spatial extent (Alila et al. 2009). We recommend an across watershed study where flow gauges are installed into as many watersheds as logistically feasible. Ideally, these experimental watersheds should be sited across the range of watershed types from the low hypermaritime to interior high-elevation systems on the central coast. Information-theoretic and Bayesian analyses across watersheds offer compelling advantages to single or paired watershed studies.

2. Hydrologic recovery refers to the forest's ability to maintain normal stream flows. It does not test how changes in flow or thresholds in recovery may affect stream structure and functioning. Thus, any across watershed experiment established to monitor how forest

harvest affects stream flows should be coupled with standardized monitoring of stream structure and functioning.

3. High priority questions identified by the National Research Council (NRC) in 2008 (USDA Forest Service 2010):
1) What are the magnitude and duration of hydrologic effects due to timber harvest?
2) What are the hydrologic effects of removing or retaining riparian forests over the long term and in large watersheds?
3) What are the cumulative watershed effects of forest cover loss in large watersheds?
4) How do past forest cutting patterns affect water quantity and quality?

c. How does the level of cut in small watersheds affect hydrological function and recovery?

Summary of Knowledge

Small streams occur in both the headwaters of larger systems, or may represent small coastal catchments that drain directly into the ocean. Either way, both are generally subject to low levels of riparian protection in forest management (Moore and Richardson 2003). This is because stream ecologists and managers have mainly focused on fish-bearing systems, and thus research on small streams has lagged our understanding of larger reaches lower in a drainage network. This is despite that headwater streams often comprise up to 70-80% of watersheds. Predictions for how the level of cut affects hydrologic function and recovery in these systems is described more broadly above in questions 1a and 1b. How forest harvest and buffering affects small steep streams and headwaters with respect to biodiversity, productivity, hillslope processes, sediment transport, and channel alteration is also discussed in more detail in questions 2a, 2d, and 2e.

Generally, there is much stronger coupling between riparian and stream functioning in small streams compared to large ones (Moore and Richardson 2010). For example, headwater streams primarily receive sediments from adjacent slopes rather than hydraulic transport, as in larger streams (Hassan et al. 2005). As a part of the Stuart-Takla Fish/Forestry Interaction

Project BC, Macdonald et al. (2003a) found increases in peak flow and mean freshet discharge in several small headwater streams following harvesting. No sign of hydrologic recovery was observed after 5 years, likely due to vegetation loss and road construction (Macdonald et al. 2003a). Beaudry (2003) found that harvesting 40% of the watershed causes a substantial increase in peak flows of small headwater streams in BC, accelerating channel erosion. These headwaters can be important for fish and amphibians and thus protection requires maintaining peak flows within natural range of variability.

Small streams high in a watershed network often experience higher rates of local extinction (Moore and Richardson 2010). This occurs because of the limited population sizes they can support, increased disturbance, lower re-colonization rates, and ultimately a decrease in both environmental and population stability. Thus fewer species, especially larger bodied ones, are supported higher up a drainage network. The function of these smaller meta-populations within a drainage network for watershed and landscape-scale population stability (e.g. salmon, amphibians) is an emerging topic in ecology (e.g. Olsen et al. 2007; Schindler et al. 2010).

Small watersheds may have more rapid hydrologic recovery than larger ones. This may be particularly true if there are differences in regional hydrology that parallel shifts in watershed size, as is seen across coastal gradients from small rain dominated streams to large interior watersheds with snow and glacial melt dominated runoff. For example, watersheds located in rain-dominated regions (29% harvested area detection limit) are less sensitive to peak flow changes than those in the transient snow zone (15% harvested area detection limit) (Grant et al. 2008). Recovery to pre-harvest conditions appear to occur within 10-20 years in some coastal catchments, but may take decades in snow dominated, mountainous catchments (Moore and Wondzell 2005). These results are based on very few studies, and do not include streams in the low productivity cedar-hemlock hypermaritime zone.

Natural disturbance regimes in stream and riparian zones vary with catchment scale and stream size (Moore and Richardson 2012). Headwater reaches are more strongly influenced by debris flows while large downstream reaches are more heavily influenced by floods. An emerging concept in forest management involves the potential application of emulation of natural disturbance (END) (Kreutzweizer et al. 2012; Moore and Richardson 2012; Naylor et al.

2012). How END could be applied to the Central Coast is an important question that requires further research.

Relevant EBM LUOs

Section 8.(1) Maintain an equivalent clearcut area of less than 20% in important fisheries watersheds as set out in the areas shown in Schedule 3.
Section 12.(1) Maintain 70% or more of the forest, in the portion of the watershed where upland streams occur, as functional riparian forest.

Recommendations and Revisions

1. Headwater streams should be afforded increased levels of protection to minimize forestry-related risks to hydrologic function. If small streams are managed more conservatively, can some forestry opportunities be increased by re-allocating riparian retention from reserve zones of larger streams to headwater streams?

2. Forest managers are assessing the feasibility of managing low productivity hypermaritime cedar-hemlock stands for timber production (Banner et al. 2005). On the Central Coast, low-productivity hypermaritime stands drained by small streams comprise a significant portion of the landscape. Banner et al. (2005) observe large hydrological responses in these unlogged hypermaritime forests from relatively small storm events because of the consistently high water tables. It is unclear what the impacts of forest harvest will be to watershed hydrology. Will ECA need to be reduced because these watersheds already have a high water table, have slower re-growth, and thus are more sensitive to harvesting? Alternatively, are these systems more resilient to hydrologic variability because of the high natural range of variability already present? Clearly, there will be large differences across the Central and North Coast region in the time to reach ‘hydrologically effective greenup’ that maintains watershed function.

Considerations for Experimental Watershed Design

1. It is essential that the full natural range of watershed sizes and hydrologies be included in any experimental watershed program that is established on the Central Coast of BC.

2. How could forest management emulate natural disturbance regimes in coastal forests of the Great Bear Rainforest? An experimental watersheds program should include a number of “control” watersheds that exist in conservancies and do not have a significant history of harvesting. This will allow comparisons between treatment watersheds impacted from harvesting and streams that are only experiencing natural range of variation in hydrologic functioning.

3. Although this research question discusses rate of cut in small watersheds, it is likely that harvest rates affect many variables across all sizes of watersheds. Accounting for this temporal component is important in the Experimental Watershed study. For example, the approach to EBM in Clayoquot Sound requires that all major watersheds have an annual allowable cut calculated independently, essentially spreading out harvesting effort. In contrast, EBM on the Central and North Coast has no such requirements for watershed scale harvest rates, meaning that harvesting can be concentrated to a greater extent (requirements for ECA and upland streams obviously provide some constraints to harvest rates). From a conservation and operational perspective, there is a debate about the relative merits of each of these approaches.

Priority Question 2.

Information on buffers (management implications, characteristics of streams that interact with and affect role and application of buffer management).

a. What are the impacts of forest management activities within and adjacent to hydroriparian buffers on biodiversity and productivity of terrestrial and aquatic systems?

i. Species-level impacts

Summary of Knowledge

The impacts of forest management activities, including the relative effectiveness of riparian buffers, vary widely depending on the species or community of concern, and across terrestrial and aquatic plants, invertebrates, and vertebrates.

Stream primary productivity

In-stream primary productivity typically increases with harvesting activities within riparian buffers due to increased light availability and water temperatures from increased canopy opening and nutrient loading (e.g., Kiffney et al. 2003). Conversely, forest harvesting may have negative effects on periphyton biomass through streambed destabilization, increased freshet discharge, and sedimentation (MoF, FFIP 2013). The magnitude of these contrasting effects varies with stream size, gradient, substrate composition, and upslope gradient.

Benthic macroinvertebrates

Benthic macroinvertebrates (BMIs) are affected by harvesting-linked increases in light, water discharge, sediment runoff, changes in basal resource supply (e.g., periphyton, plankton, and terrestrial detritus), in-stream large wood, water temperature, and shifts in food web dynamics (Richardson 2008 and references therein).

The cumulative effects of decreasing buffer width, or increasing harvesting within riparian forest are often negative. Compared to BMI assemblages in non-impacted streams, impacted streams are generally characterized by lower species diversity, richness, and in many cases, biomass (e.g. Martel et al. 2007). However, these responses vary markedly between species and functional feeding groups. Detritivores and sensitive groups such as the Tricoptera often decline with increasing harvest intensity, and assemblages become dominated by generalist grazers and disturbance-associated groups (e.g. Price et al. 2003). The magnitude of these effects typically increases with proximity of harvesting and access roads to streams banks, and with the proportion of watershed harvested. However, differing biotic and abiotic conditions, topography, climate, forest management practices, and the spatial scale of studies also influence the magnitude of impacts (Martel et al. 2007; Richardson 2008).

In sum, harvesting around streams initially increases overall BMI productivity due to increased aquatic and terrestrial basal resource inputs, but subsequently causes declines in diversity, richness, and abundance, which may persist for several decades after harvesting activities, particularly in small headwater streams (e.g. Cole et al. 2003; Zhang et al. 2009). Although research has largely focused on local, site-specific effects of harvesting on BMIs, there is increasing evidence that impacts at this scale are much weaker than those at basin or watershed-wide scales (Martel et al. 2007, but see Sakamaki and Richardson 2011).

Fish

Logging activities affect stream fish populations by altering environmental features that structure fish populations (Deschênes et al. 2007). At local habitat scales, harvest of riparian canopy increases light penetration and water temperatures, which increases primary productivity and invertebrate biomass (e.g. Kiffney et al. 2003) and potentially provides increased resources for fish populations (Wilzbach et al. 2005; Kiffney and Roni 2007). However, for cold-water species such as many salmonids, increased water temperature may have negative effects if it exceeds their thermal optima. This is particularly the case in smaller headwater streams, where canopy cover strongly buffers stream temperatures (McCullough et al. 2009; Groom et al. 2011a). At the drainage-basin scale, harvesting can increase suspended sediment loading to streams and increase freshet discharge. Increased stream flow can in turn have negative effects on fish populations by increasing sedimentation, scouring and destabilizing stream channels, and decreasing habitat heterogeneity and cover (MoF, FFIP 2013). Therefore, the effects of logging on fish populations may largely depend on the scale and environmental context. For example, Deschenes et al. (2007) found that logging activities had strong, negative effects on juvenile Atlantic salmon abundance across 120 streams in Quebec. However, these effects were largely scale-dependent with longer-term, negative effects on densities increasing with the spatial scale of logging.

A long-term study in the Carnation Creek watershed on Vancouver Island illustrates the detrimental effects of forest harvesting practices on Pacific salmon populations (MoF, FFIP 2013). Mean returns of coho and chum have declined by roughly 10–26%, respectively, from pre-harvest levels. This has primarily been attributed to decreased survival in early life-stages as a result of increased sedimentation, substrate deposition, streambed scour, and increased water temperatures. Juvenile coho densities have declined by 50–70% from pre-harvest levels; a trend that is associated with decreased habitat complexity and cover and ongoing habitat degradation over 20 years post-harvest. Increased water temperatures resulting from decreased shade can also shift age and size structure of fish populations. In Carnation Creek, higher water temperatures increased summer growth and overwinter survival, increased smolt production in juvenile coho, but decreased steelhead smolt production. These outcomes are in contrast to those documented by Leach et al. (2009), who found that shifts in temperature and food availability as a result of forest harvest resulted in decreased summer growth rates, increased growth in spring and fall, and smaller over-winter body size in cutthroat trout populations in coastal BC.

Harvest activities in riparian forests can also influence the ecological role of keystone species in freshwater habitats. Tiegs et al. (2008) found that in unharvested streams, spawning Pacific salmon had a positive effect by transferring nutrients from marine to freshwater ecosystems and increasing biofilm growth on sediments. However, streams impacted by harvesting had smaller sediment particle sizes than unharvested sites and biofilm declined in these streams despite increases in dissolved nutrients because the fine sediments were more easily bioturbated by the spawning salmon. Forest harvesting-related effects on sediment size can therefore shift the ecological role of salmon from one of ecological subsidy to one of disturbance (Tiegs et al. 2008).

Riparian fauna

Riparian buffers have frequently been applied as an umbrella approach for conserving terrestrial species, in lieu of selective interventions targeting sensitive species (Marczak et al. 2010). However, forest management practices have highly taxon-specific impacts and riparian buffer forests are not effective at conserving all terrestrial taxa. Marczak et al. (2010) conducted a meta-analysis to compare species abundance in harvested sites with riparian buffers versus abundance in un-harvested riparian sites. Of the 397 comparisons, they found arthropods, edge-associated birds, and other edge-associated species were more abundant in buffers compared to unharvested sites, while amphibians were less abundant in buffers (Figure 3 and Figure 4). The variability of responses by taxa to buffers as narrow as 5 meters and as wide as 200 meters illustrates that riparian buffers alone cannot reliably provide the same protection for all terrestrial taxa.

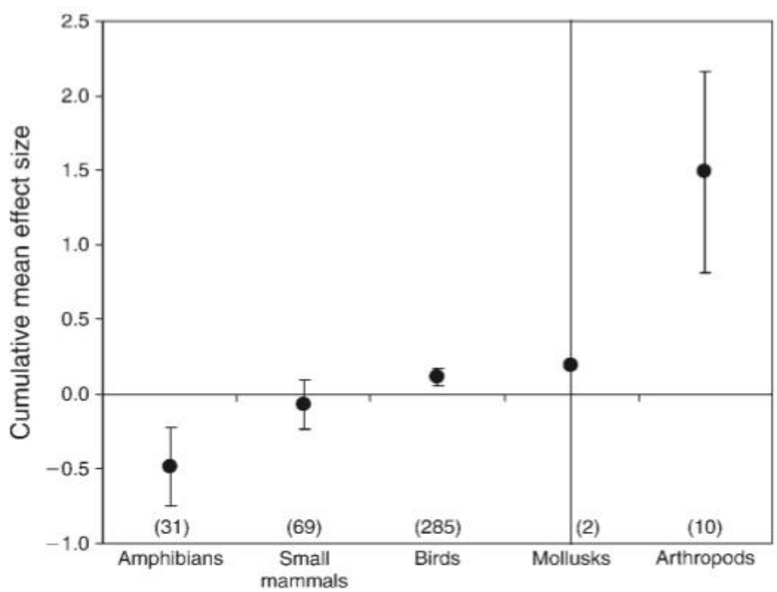

Figure 3. ***The cumulative mean effect size between riparian buffers and paired intact riparian forests, by major taxonomic groupings. Sample sizes are shown in parentheses for each taxon at the bottom of the figure. Error bars are 95% confidence intervals, CIs; CIs that intersect 0 indicate no significant effect (Fig. 1 in Marczak et al. 2010).***

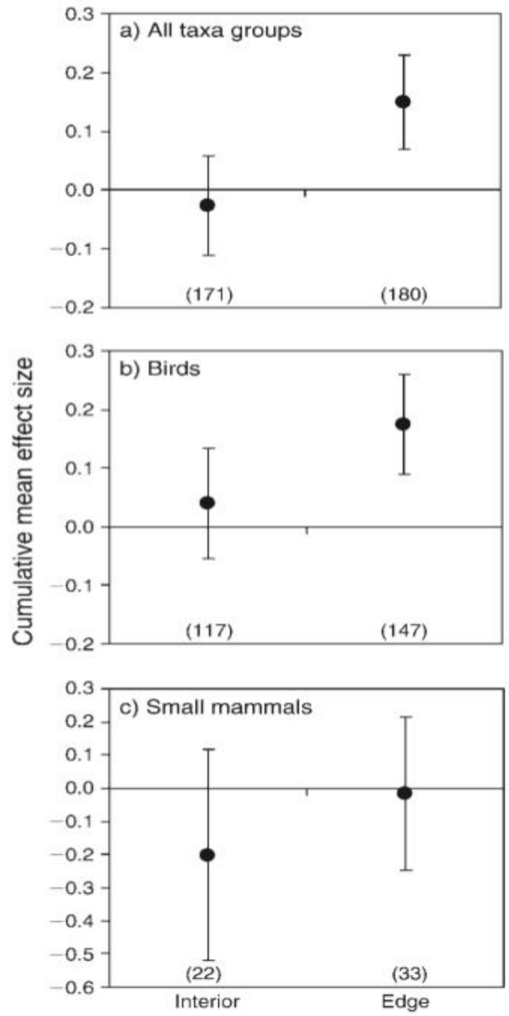

Figure 4. Difference between cumulative effect size for riparian species identified as preferring interior habitats and edge habitats for (a) all the taxa groups, (b) birds, and (c) small mammals. Error bars are 95% confidence intervals, CIs; CIs that intersect 0 indicate no significant effect (Fig. 2. in Marczak et al. 2010).

Small mammals

In a southwestern BC study, small mammal riparian species richness was considerably lower in clearcut sites than in sites with 10–30 m wide riparian buffers. Abundance in sites with buffers was still lower than in unlogged sites (Cockle and Richardson 2003). By contrast, in western Washington, Wilk et al. (2010) found that small mammal species abundance and community composition along riparian zones of headwater streams were lower in clearcut, continuous buffer, and patch buffer treatments relative to unharvested sites. However, total abundance, richness, evenness, and diversity were no different between all sites.

Birds

Birds are often more abundant in riparian buffer strips than in associated undisturbed riparian forests (Marczak et al. 2010). There are two general explanations for this: 1) birds displaced by harvest in upland areas move to remaining riparian-buffer habitat, resulting in temporarily increased bird populations in riparian forests or buffers; and 2) riparian buffers may provide edge-associated species with more of their preferred habitats and increase their populations. These explanations likely explain spatial patterns of abundance for species other than birds as well. Thus, overall increases in bird populations within riparian buffers are often driven by an increased abundance of edge species. Kardynal et al. (2009) offers additional evidence that harvesting effects on birds do not approximate those of natural disturbance regimes. They compared bird community responses to early post-wildfire and post-harvested riparian habitats with varying buffer widths and found higher rate of natural variation (RONV) in post-wildfire than post-harvest bird communities. However, a similar study in forests with disturbance regimes not dominated by fire would be more informative to management on the Central and North Coast of BC.

Amphibians

Amphibian life cycles are more complex, and more strongly dependent on the availability of both water and terrestrial habitat than those of other hydroriparian vertebrates. For this reason, amphibians are ideal representatives of the “reciprocal subsidies” perspective of stream-riparian zones (Olson et al. 2007). Traditional linear riparian buffers created to conserve aquatic biota do not allow for natural dispersal patterns of amphibians and many other obligate or facultative stream-riparian associates. Thus, studies of amphibians highlight the importance of landscape-level, rather than site-level forest management approaches. Crawford and Semlitsch (2006) found that 77–92.6 m buffers (27–42 m of core habitat + 50 m buffer for edge effects) were needed to fully maintain salamander assemblages in North Carolina Appalachian streams. Ficetola et al. (2008) found that 100–400 m buffers were necessary to maintain amphibian assemblages in Italy. In contrast, Hawkes and Gregory (2012) found that both 7.5 m buffers and wider, more variable buffers were adequate to mitigate the effects of upland logging for ensatina and red-backed salamanders but not for coastal tailed frogs. The latter were noticeably absent from upland habitats 10 years after logging.

The coastal tailed frog (Aschapus truei) is a Species of Concern under COSEWIC and designated of concern relative to logging in the EBM LUOs. Habitat connectivity over ridgelines could be maintained for such forest-dependent species by creating “linkage areas” between hydrologic units (in the EBM context, hydroriparian zones) (as uncut blocks, thinning only, and islands; Olson and Burnett 2009). This involves planning at three spatial scales: landscape, drainage basin, and forest stand, and incorporation of species’ life histories, habitat preferences, and dispersal capabilities. Such a planning approach is consistent with the recommended planning framework in the CIT Hydroriparian Planning Guide (2004).

Riparian vegetation

In the Pacific Northwest, forest harvesting may have lasting impacts on plant communities along streams. D’Souza et al. (2012) examined the legacy impacts of historical harvesting on riparian plant communities in Washington. Both stand age and distance from streams independently influenced the vegetative communities. Diversity and density were highest during early succession, lowest in mid-successional stands, and intermediate in late-successional stands. Herb species richness was greatest in young stands, whereas shrub and species richness were highest in old stands. Importantly, they identified that riparian plant communities extended from the stream edge out 9 m and that transitional plant communities existed 10-29 m from the stream edge. This suggests that current riparian buffer requirements of 15–29 m may protect riparian plant communities but not transitional plant communities.

Partial cutting in riparian management zones can also have considerable effects on riparian vegetation communities. Zenner et al. (2012) found that understory biomass and density increased, but the presence of conifer in understories decreased with overstory harvest. They also found the functional width of riparian buffers decreases with increased harvest of overstory trees.

Relevant EBM LUOs

Section 8. Objectives for important fisheries watersheds

Section 9. Objectives for high value fish habitat

Section 10. Objectives for aquatic habitat that is not high value fish habitat

Section 12. Objectives for upland streams

Section 14. Objectives for landscape level biodiversity

Section 16. Objectives for stand level retention

Recommendations and Revisions

1. Olson et al. (2007) offer key recommendations for forestry-compatible stream riparian management that preserves stream-riparian associates like amphibians:

- At landscape scale, consider connectivity of channels to uplands, headwaters to ridgelines, and linkage areas.
- Apply a mix of riparian buffers: 10m for bank stability, 15–30 m for water quality and habitat attributes, 40–100 for riparian-dependent species.
- Seasonal restrictions on management activities and disturbances that reflect the important life cycle activities of species of concern (e.g. Coastal tailed frog).

2. Buffer prescriptions of 5–50 m are unlikely to maintain all terrestrial organisms in buffer strips at levels comparable to undisturbed sites (Marczak et al. 2010).

3. More flexible/variable harvesting including selective harvest in riparian zones may better approximate disturbance patterns (Kardynal et al. 2009).

Considerations for Experimental Watershed Design

1. There is a lag time after buffer creation before interior-dependent and other species are lost due to habitat loss or degradation (Marczak et al. 2010). Short-term studies (<10 years following forest harvesting) should be viewed with caution.

2. Following consideration no. 1, longer-term, larger-scale research studies are needed to assess the extent to which current buffer requirements are sufficient for conserving species

biodiversity and abundance (Deschenes et al. 2007; Cockle and Richardson 2003). The length of the study should span several times the generation time of the species of concern (e.g. A 5 year study is not adequate to detect a population decline in an amphibian species with a 10 year lifespan).

3. Survey coastal ecosystem and species-specific requirements for buffer widths and within buffer harvesting levels on the Central Coast in order to inform adaptive, context-dependent buffer width and forestry management guidelines for specific ecosystems and species in this region.

4. Study ecosystem and community response to forest harvest over a temporal gradient by surveying streams in forest stands that have been historically harvested at different times and compare to uncut reference sites to elucidate recovery time of riparian and aquatic ecosystems to forest practices.

ii. Impacts specific to small upland streams (>6% gradients such as S4, S5 and S6s)

Summary of Knowledge

Generally in BC, riparian buffers are only mandated for S1 to S3 streams or streams with habitat suitable for salmonids. By contrast, small, fishless upland and headwater streams are often logged extensively without adequate protection against negative impacts of forestry activities (Richardson 2008). However, headwater streams drain 70–80% of watersheds and recent research increasingly suggests that these upper reaches play a much stronger role in ecosystem functioning than lower reaches, and may be more important for maintaining biodiversity and productivity of watersheds (Clapcott and Barmuta 2010; Olson et al. 2007; Sheridan and Olson 2003).

Upper reaches of watersheds are particularly important for stream-associated amphibian (SAA) assemblages, such as the Coastal tailed frog. Sheridan and Olson (2003) found SAA occur at higher densities and diversity in upper basins, relative to downstream reaches, which suggests smaller buffers in these upper basins may be more effective at maintaining biodiversity than wider buffers along lower stream reaches.

Headwater streams are more heavily dependent on allochthonous inputs from surrounding forest. Consequently, forest harvesting can have marked effects on terrestrial subsidies and communities dependent on these, such as benthic macroinvertebrates (BMIs). In 1st and 2nd order headwater streams of the Coast Range of the Pacific Coastal ecoregion, inputs of needles, twigs, and total particulate matter were significantly lower at clearcut sites than at sites with 10 and 30 m riparian reserve buffers (56 times lower in the fall than the other treatments) (Kiffney and Richardon 2010). This persisted for 2, 6, and 7 years for twigs, total, and needles, respectively. Needle inputs were ~6 times higher in streams with reserves compared to the clearcuts 7 years later. However, there was no significant difference between 10 m and 30 m buffers. BMIs in logged headwaters experience changes in community structure with shifts from coniferous to deciduous terrestrial inputs, and increased nutrient loading after clearcutting. These changes generally favour more disturbance-associated, generalist species (Kiffney et al. 2003; Kiffney and Richardson 2012). There is some evidence, however, that BMI communities recover more quickly from forestry practices in headwater streams than in larger streams and downstream reaches (Reid et al. 2010).

Relevant EBM LUOs

Section 12. Objectives for upland streams

Recommendations and Revisions

1. A riparian reserve buffer between 10 and 30 m has been shown to provide litter inputs similar to unlogged forest in 1st and 2nd order headwater streams (Kiffney and Richardon 2010). The functional relationship between terrestrial inputs and reserve width should differ depending on the steepness of valley walls bordering streams. Other studies have shown that reserves >30 m were necessary to maintain ecological functions, species diversity and animal populations (Kiffney et al. 2003 and references therein, Marczak et al. 2010).

2. In the LUOs, “upland streams” are defined as “streams with a slope greater than 5% that are classified as S4 to S6 streams in section 47 of the *Forest Planning and Practices Regulation*”; whereas the FPC used a 20% slope cutoff. The current FRPA also treats upland streams more conservatively than the upland stream LUOs (see Table 1 in the Haida Gwaii Forest Stewardship Plan Supporting Information 2011). It is important to assess whether this new classification system results in more streams being classified as upland and therefore afforded less protection.

Considerations for Experimental Watershed Design

1. One of the most urgent research priorities in forestry-freshwater research is to investigate the short and long-term response of headwater ecosystems to disturbances (Clapcott and Barmuta 2010).

2. There is a strong call from recent published studies to move away from site-specific studies to more landscape or watershed-scale studies that span longer time periods (e.g. Martel et al. 2007; Stephenson and Morin 2009; Clapcott and Barmuta 2010; Richardson et al. 2012).

3. Because of their larger maximum size, the spatial scale at which changes in tree community composition are detectable is greater than that for understory vegetation. Herbs and shrubs are better indicators for the compressed environmental gradients found in smaller riparian areas along confined upland channels (D’Souza et al. 2012).

b. How effective is 1.5X tree height for maintaining riparian function? And what are the impacts of wind damage in hydroriparian buffers on species diversity and site productivity and water quality?

Summary of Knowledge

The EBM LUOs 9(1) and 10(1) require reserve zone distances of 1.5 times the dominant tree height. However, the effectiveness of this management practice has not been studied in the primary literature. Literature that tests the effectiveness of various riparian buffer widths in maintaining a diversity of riparian functions make comparisons based on distance, irrespective of vegetation type or height (Table 3).

In the Coastal Western Hemlock Biogeoclimatic Zone, western hemlock and western redcedar are common. These two species may reach a maximum height of 50 and 60 m, respectively. Assuming that most trees would not be at maximum height, we could estimate the approximate maximum tree height would be about 20 to 40 m. Under the EBM LUOs, this would translate into a 30-60 m riparian reserve zone. However, the dominant tree height in any given stand depends on the productivity and disturbance history of a site, and this will vary substantially across and within watersheds. It is important to note, though, that these buffers are being applied to S1-S3 streams, which will typically be associated with the more productive valley bottoms of watersheds that have taller trees. Comparing this with the results presented by Broadmeadow and Nisbet (2004)'s review, a distance of 30-60 m could very well maintain many of the riparian functions listed. However, a significant concern with the effectiveness of the 1.5 times tree height LUOs is that the reserve zone distances are entirely dependent on riparian forest tree height, which may not always meet the distance that the literature has found is necessary for maintaining these functions.

Table 3. Broadmeadow and Nisbet (2004) summarize the range of riparian buffer widths reported in the literature as being required for various riparian functions.

Function	Buffer width (m)
Denitrification	5–30 m
Temperature moderation	15–70 m
Invertebrate diversity	10–50 m
Sediment removal	15–65 m
Modelled sediment control	15–100 m
LWD and leaf litter supply	25–100 m

Buffer widths at the minimum end of the ranges of values presented above may be adequate to maintain physical and chemical characteristics of streams, but wider buffers of 30 m or more are necessary to maintain ecological integrity (Kiffney et al. 2003, Broadmeadow and

Nisbet 2004, see also discussion of buffer width requirements for aquatic and terrestrial species in 2a).

Although some research has been conducted to investigate the role of forest management in wind damage of riparian habitats, little contemporary scientific literature has explicitly assessed the effects of wind damage on hydroriparian species and productivity. Zmihorski (2010) assessed changes in bird community after a large windthrow in Piska Forest, North-Eastern Poland. Bird counts were performed in three different habitats: 1) windthrow left for natural regeneration (where there were fallen logs, leaning trees and broken trunks); 2) managed windthrow (i.e. cleared windthrow where all the fallen, leaning and broken trees were removed and artificial replanting occurred); and 3) undisturbed managed forest. It was found that bird communities differed significantly among the three habitats. Several species, including some forest specialists, were more abundant in the natural windthrow; whereas, birds found in the managed windthrow were predominantly open habitat or edge-dwelling species. Other forest specialists were found primarily in the intact stand. For birds, forest wind damage of a magnitude within the range of natural disturbance can be associated with higher bird densities than in intact forest, likely because of decreased habitat homogeneity and increased spatial structure complexity.

Forestry practices often salvage wood downed by wind damage for human use before it rots on the ground, though this is less common in more remote areas such as the Central and North Coast. However, leaving the down wood can be beneficial to vertebrate, invertebrate, plant, and fungal communities. Down wood provides vertebrates with sheltered areas for reproduction, a modified microclimate, increases habitat diversity and aeration in water by forming riffles, and provides cover and foraging sites, among other uses (Bunnell and Houde 2010). For example, insects associated with down wood comprised about 30% of the diet of black bears (Bull et al. 2001). By extension, artificial disturbance can also have negative impacts by shifting community composition. Decay has pervasive effects on species present because it changes the structure, chemistry, moisture content, and foraging opportunities in wood (Bunnell and Houde 2010).

Increased wind damage is likely to reduce the functional width of riparian buffers due to edge effects such as increased light penetration. Edge effects, in particular, do not appear to be well considered by the 9(1) and 10(1) EBM LUOs. Areas under the influence of edge effects do not have the interior condition associated with old-growth forest. To evaluate the extent of edge effects, the distance of edge influence (DEI) metric can be calculated by measuring the distance from the edge into an old-growth forest over which there is a statistical difference in composition, structure, or function. Boucher et al. (2011) reviewed studies documenting logging-induced edge effects in Canadian and Fennoscandian old-growth boreal forests (8 studies) and found the median value of maximum DEI was 50 m (range of max DEI of 30-54 m). This is significant as most linear buffers were only 60–100 m wide (1–7 ha).

An agglomerated block strategy provides an alternative management practice to the traditional linear riparian buffer. Here, forest remnants are generally >50 ha in size and up to 200 ha, and riparian strips are still preserved. Boucher et al. (2011) compared the area under edge influence for two recently logged (<20 yrs) landscapes with the same area of residual old-growth forests but with either linear versus agglomerated residual scenarios. The area under edge influence was almost 2.5 times larger in the linear versus the agglomerated strategy (91.6% vs. 37.5%). However, this approach certainly has tradeoffs with some of the other functions listed above that require more consistent tree retention along streams. This emphasizes the need to refine and revise the EBM LUOs so as to reflect the importance of riparian width as well as shape and area covered. The greater efficacy of agglomerated blocks at reducing edge effects compared to linear buffers in this study is consistent with the conclusion by Olson and Burnett (2009) (discussed in response question 2a) that simple linear buffers are not sufficient.

Relevant EBM LUOs

Section 9.(1). Adjacent to high value fish habitat, maintain a reserve zone with a width, on average, of 1.5 times the height of the dominant trees, and do not alter or harvest the forest in the reserve zone unless there is no practicable alternative.

Section 10.(1). Adjacent to the following aquatic habitat a) S1 to S3 streams, as defined in the Forest and Land Practices Act … retain 90% of the functional riparian forest in management zones with a width, on average, of 1.5 times the height of the dominant trees.

Recommendations and Revisions

1. Allowing part of fallen, broken or damaged trees to remain after windthrow is crucial for persistence of forest-dwelling birds. It may also be economically justified not to salvage log. In the Zmihorski (2010) case study, damage to the wood caused by the windstorm lowered the quality of the timber and made it up to 40% less profitable. Bunnell and Houde (2010) make the following management recommendations with regards to down wood:

- Retention of wood as living trees is critical to ensuring future dead wood
- Sustain 50% of naturally occurring amounts of down wood at the landscape level
- Disturbance through forest management does not replicate natural disturbance or natural contributions of down wood
- Sustain a range of size and decay classes of down wood
- Ensure that some large pieces are retained
- Provide both aggregated and dispersed down wood
- There is no way to distribute logging residuals that benefits all taxa so the wisest approach is to use a variety of approaches within one area (e.g. aggregating logging residuals in piles and leaving them dispersed)

2. 1.5 times tree height is not likely to be effective in maintaining the diversity of riparian reserve functions if the dominant trees are 20 m or less in height. For basic physical riparian functions, forest management should maintain minimum riparian reserve zone buffers of ≥30 m. This width may be smaller or larger depending on the riparian functions of particular concern and the climatic and biophysical characteristics of the stream. For obligate riparian-associated species, dispersal-limited species, and interior old-growth-associated species of special concern (e.g. the Coastal tailed frog), these riparian buffers need to be much greater than the 1.5 times tree height rule, alone, will likely provide.

3. Following from recommendation no.2, Boucher et al. (2011) states that species closely associated with interior old-growth conditions, particularly dispersal-limited species, may experience substantial detrimental effects in linear buffers only 60-100 m wide.

4. Narrow linear residual forest structures cannot ensure functional connectivity and should not be considered efficient in preserving biodiversity (Boucher et al. 2011). Larger residual

old-growth blocks and connecting features must be maintained. The distance of edge influence (DEI) is good metric to incorporate in planning riparian widths.

Considerations for Experimental Watershed Design

1. To conduct an analysis specific to BC (or the Pacific Northwest) to determine if the median maximum DEI is different than reported by Boucher et al. (2011) for boreal forest.

2. Bryophytes and epiphytic lichens are good indicators of change in forest structure and composition, and therefore for calculating the DEI by harvesting.

3. Examine the extent of windthrow based on different widths of riparian buffers, as well as cutblock sizes and configurations. Quantify how different functions are affected by varying levels of wind throw. Compare these results to the effect on function of varying levels of harvesting in riparian buffers.

c. Does forest management under EBM LUOs affect High Value Fish Habitat (HVFH) channel morphology (or general aquatic habitat channel morphology)? And does HVFH channel morphology change (beyond RONV) in managed watersheds?

Summary of Knowledge

Channel morphology is dependent on three principal, interconnected factors (Hogan and Luzi 2010): a) sediment supply, b) riparian vegetation, and c) streamflow changes. Forest management that impacts any of these factors may change channel morphology.

Sediment supply

Forest harvesting and roads increase sediment supply to channels (the processes by which this occurs are discussed in depth in 2d). When sediment inputs exceed the transport capacity of a stream, sediments are deposited. This increases the width-to-depth ratio of the channel, which decreases its stability. The channel pattern becomes straighter (less morphologically complex) over time. Depending on the channel type, vertical shifts in the channel bed or lateral shifts of the channel may also occur (Hogan and Luzi 2010).

Headwater streams primarily receive sediments from adjacent hillslopes rather than by hydraulic transport, as in larger streams (Hassan et al. 2005). Therefore, headwater channel morphology is influenced by sediment and wood inputs from adjacent slopes. Large woody debris (LWD) along banks may reduce erosion, but LWD placed diagonally can increase bank erosion by diverting flow toward the bank. LWD inputs from slash left behind after harvesting can even transform the type of stream reach itself. Bedrock reaches, which are located in the steep, uppermost part of drainage basins, and lack alluvial deposits, can be transformed into colluvial (contain material derived from adjacent hillslopes and have insufficient stream flow to transport boulders, wood, debris, introduced into the channel) and alluvial (contain stream-deposited sediments) reaches by LWD inputs (Hassan et al. 2005).

Carnation Creek watershed on Vancouver Island is a case study that illustrates the long-term interaction between hillslope and channel processes and how disturbances are transferred downstream. Pre-logging, the channel had a complex morphology. After it was logged to the streambank during 1978-79, LWD entered the channel from slash left along the banks. A logjam resulted and it grew until it impeded downstream sediment transport. The channel widened. Removal of the riparian trees further contributed to weakening of the banks. In 1984, a large storm caused many gully failures, delivering sediment into a steep section of the creek. This material moved downstream until it encountered the logjam. This aggravated channel widening. 10 years later another storm generated a peak flow event that broke through the logjam and evacuated the sediment. The width of the stream in this area is almost at the pre-disturbance value (Hogan and Luzi 2010).

Riparian vegetation

Riparian vegetation influences channel morphology by providing stability to stream banks and through the provision of LWD. In Haida Gwaii, 75% of the Mosquito Creek watershed was logged and no riparian buffers were left around the stream. Loss of bank strength led to channel widening and an increase in sediment supply. At the same time, removal of the riparian vegetation reduced natural LWD input into the stream, which decreased the channel’s sediment storage ability. As a result, the logged watershed stream became geomorphically simple whereas the forested watershed is geomorphically complex (Hogan and Luzi 2010).

Riparian buffers can help mediate changes to channel morphology by decreasing the amount of sediment and woody debris transported to the channel from harvesting outside the buffer; however, they are not always effective at doing so. For example, in a Thunder Bay, Ontario study, there was no significant difference in stream widths between clearcut and buffer locations, but streams in harvested areas were wider than unharvested sites. Stream depth was lowest in harvest sites, intermediate in buffers, and deepest in reference sites. Likely because clearcut harvesting may increase slash loads in stream channels, resulting in slower stream flow, higher retention of fine sediments, and redirection of flow to create bank erosion (Mallik et al. 2011).

Stream flow changes

As discussed in question 1, canopy loss reduces evaporation and interception of precipitation, which increases groundwater recharge and elevates the water table (Iida et al. 2005). This is significant because groundwater is the source of most base flow in streams (Douglas 2008). A higher water table increases the potential for greater water yield and more rapid and higher peak flows, all of which may scour the streambed, transport stream sediments and debris or accelerate erosion of channel banks. However, it is important to note that these impacts may not be seen immediately after disturbance because this is a threshold process—water must fill storage before it can spill and generate runoff (Keim et al. 2006). Similarly, Moore et al. (2008) showed that lower peak flow magnitude but longer duration of high flows result from snow accumulation in clearcuts. Snow water equivalent (SWE) and ablation rates were 34% higher in clearcuts relative to mature forest.

Channel morphology can also be impacted by greater overland flow due to soil compaction by harvesting with skidders. This can increase the peak stream flows and channel erosion. But the significance of soil compaction and resulting overland flow depends on the degree of compaction and how much of the watershed area is disturbed (Putz et al. 2003).

Relevant EBM LUOs

Section 8.(1) Maintain an ECA of less than 20% in important fisheries watersheds
Section 9.(1) Adjacent to HVFH, maintain a reserve zone with a width, on average, of 1.5 times the height of the dominant trees, and do not alter or harvest the forest in the reserve zone.
Section 10. Objectives for aquatic habitat that is not HVFH.

Recommendations and Revisions

1. In BC, the foundation for determining riparian practices is largely fish-based, especially salmonids, yet there is no scientifically sound basis for managing riparian and aquatic values on the presence of game fish alone (Tschaplinski and Pike 2010).

2. Trees in a recovering riparian forest will likely have to be taller than the original stands to compensate for widened channels (Tschaplinski and Pike 2010). Consequently, the LUOs' prescriptive reserve zone width of 1.5 times the height of the dominant trees should be wider in cases where upstream or hillslope processes are anticipated to enlarge the channel.

Considerations for Experimental Watershed Design

1. There are very few published studies on the relationship between forestry and groundwater. To increase our understanding of the impact of changes in groundwater on channel morphology, groundwater monitoring needs to be integrated into watershed hydrology studies (Douglas 2008).

2. Depending on the magnitude of change in channel morphology between harvested and unharvested sites, channel recovery after disturbance could take 16 years (Mallik et al. 2011) to over 30 years (Carnation Creek channel; Hogan and Luzi 2010).

d. How does forest management under EBM LUO's affect sediment supply to HVFH (or general aquatic habitat) and impact HVFH (or general aquatic habitat) function? Specifically, what is the impact of harvesting and road building activities on:

i. Class IV terrain and around small steep streams with high potential for debris transport?

Summary of Knowledge

In the Central and North Coast Land Use Order (2009), "high value fish habitat" (HVFH) means critical spawning and rearing areas for anadromous and non-anadromous fish including:

- estuaries (including eel grass beds and salmonid and eulachon rearing areas);
- wet floodplains (including main channel salmonid and eulachon spawning habitats, and of channel habitat used for rearing and spawning); and
- marine interface areas (including shallow intertidal areas, kelp beds, herring spawning areas, and other nearshore habitats used by marine invertebrates for reproduction and rearing.

Logging and road construction on hillslopes, unstable or steep terrain may initiate landslides in the form of debris slides and debris flows, and cause erosion of soil and stream channels. Debris slides are the most common type of landslide in BC. Slope instability problems with organic soils occur during road construction at slope angles of 20-30° and may trigger debris slides when vibrating heavy machinery overloads and liquefies saturated organic material. In steep channels where logging and road construction cause hydrologic changes that lead to high streamflow, this may undercut streambanks and cause a dam, which when it fails, results in a debris flow (Geertsema et al. 2010). Similarly, when debris slides on hillslopes enter gullies they tend to become debris flows that come to rest downstream. The behaviour of debris flow in channels depends on the type of debris/fluid mixture (Hassan et al. 2005). Fine sediments hold water and flow further on quite low gradients. Sandy matrix material drains easily and tends to stop flowing on gradients of order 10° or more.

Three variables mediate sediment mobilization in headwater streams (Hassan et al. 2005): 1) channel gradients (determines stream power), 2) hillslope gradient (determines stability and magnitude of mass movements), 3) valley bottom width (influences flood hydrology and whether debris flows coming off slopes enter streams). 2) and 3) are elaborated on below:

Hillslope gradient

Steeper/less concave basins transport debris flows further into the drainage network, causing channel scouring, deposition of sediment and wood that can alter fish habitat. Results suggest fish populations residing in basins with high steepness and/or low concavity will have more severe fluctuations in population abundance and may be at greater risk of local extirpation because most tributaries are too steep to provide habitat, confining fish to main-stem channels (May and Lisle 2012). In contrast, fish distribution in basins with low steepness and/or high concavity can expand into the tributaries, allowing for a spatial spreading of risk that may enhance a population's ability to persist during adverse conditions.

Valley bottom width

In steep and unstable terrain on Haida Gwaii, small streams have narrow floodplains, and thus the stream channel is closely linked to upslope processes. In such cases, regardless of riparian management, hillslope processes will likely dominate (Tschaplinski and Pike 2010).

The occurrence of landslides from roads on steep terrain has decreased 4-fold since the introduction of the Forest Practice Code (FPC) (Horel 2006). Post-FPC, most landslides that originated at roads and cutblocks were small events (≤0.25 ha). However, while these may not affect forest productivity they may significantly impact fish streams.

The steep slopes on Haida Gwaii (>20°) reportedly have the highest post-clearcut landslide rate—about 1–1.7 Ls/km^2 per year. But landslide frequency has decreased on Vancouver Island with the introduction of the FPC (0.86 Ls/km^2 before and 0.49 Ls/km^2 after) (Jordan et al. 2010). The Forest Practices Board (2005) identified that under the FPC, the rate of landslides on Vancouver Island was less than in the pre-Code era but still 2-3 times greater than the rate of natural landslides in unlogged watersheds. Post-FPC landslides occur more frequently in clearcuts and on open slopes, and terminate less frequently in streams. The reason for this positive shift is that gullies are logged less frequently, stream escarpments are not logged at all anymore, and riparian reserves now extend to the top of the stream escarpments.

The Forest Practices Board review also concluded that terrain stability field assessments (TSFAs) correctly identified most cutblock areas that eventually experienced landslides as potentially unstable or unstable. However, 60% of the Code landslides in the study area had a potential “material adverse effect on a forest resource” (defined as a landslide of 200 m^3 of sediment that directly entered a watershed stream, fish-bearing reach, or tributary of a fish stream within 500 m of fish habitat; a landslide that caused debris flow that scoured a fish stream; a landslide that destroyed more than 0.25 ha of forest). Therefore, more precautionary practices are required.

Adverse effects on HVFH are also caused by sedimentation. Logging and roads can alter timing and magnitude of runoff events and make more sediment available for transfer by exposing soil. In the upper Kootenay River Watershed, relative abundance of cutthroat trout was significantly negatively correlated with road density and pool frequency (which is limiting for resident cutthroat). This trend was driven by road density within 100 m of streams (particularly on erodible soils). Therefore the link between roads and abundance is likely sedimentation (Valdal and Quinn 2011). Sediments can smother spawning habitat, reduce oxygen transmission to embryos, alter type of spawning substrate and pore size and permeability. Infilling of interstitial spaces and loss of vegetation can limit cover for juveniles and reduce foraging efficiency (Kemp et al. 2011).

Relevant EBM LUOs

Section 8.(1) Maintain an ECA of less than 20% in important fisheries watersheds
Section 9.(1) Adjacent to HVFH, maintain a reserve zone with a width, on average, of 1.5 times the height of the dominant trees, and do not alter or harvest the forest in the reserve zone.
Section 10. Objectives for aquatic habitat that is not HVFH.
Section 12.(1) Maintain 70% or more of the forest, in the portion of the watershed where upland streams occur, as functional riparian forest.

Recommendations and Revisions

1. The Forest and Range Practices Act (FRPA) states that primary forest activities must not cause landslides. To this end, planners rely heavily on TSFA stability ratings to identify unstable terrain (classes IV and V). However, TSFAs do not adequately account for the

connectivity between upslope processes and downslope processes in assessing landslide risk. The rating only refers to the likelihood of a landslide initiating in a particular polygon following forestry activity. Therefore, downslope areas that are at risk of upslope landslide activity are unlikely to be appropriately identified as of concern under TSFAs (Geertsma and Schwab 2006). The stability of a site will be overestimated if the ranking does not include an assessment of the upslope hazards (e.g. a rock slide upslope of the lower logged portion of the slope that was considered 'stable' can initiate a large movement of earth).

2. Based on work in the Oregon Coast Range on 3rd and 5th order fish-bearing streams, Bigelow et al. (2007) identify four attributes of debris flows in unmanaged forests that should be considered when managing forests:

- Dual nature of debris flows— they can destroy in the short term but construct habitat in the long term;
- Transport of large wood to fish-bearing streams can be an important natural process. Therefore, trees along channels that are likely to supply large wood should be protected;
- Logging and road construction can alter the spatial and temporal scales of debris flows (e.g. too frequent, or not during the season in which they might naturally occur); and
- The landscape context determines the effects (positive or negative).

3. A method of index of debris flow susceptibility is that the "Melton ruggedness number" (the ratio of elevation range to the square root of watershed area) be greater than 0.6 (Wilford et al. 2005b).

4. Unstable watersheds may actually be less sensitive to development impacts if they are highly active (i.e. frequently experience mass movement of sediments into streams) compared to those watersheds that have a low background level of disturbance. Jordan (2006) suggests an order-of-magnitude guideline for judging impact: If sediment input is 1% of background—impact is insignificant; 10% of background—impact is probably significant; 100% of background—impact is highly significant.

5. Deactivation of roads reduces the incidence of landslides (Forest Practices Board 2005).

6. Slope instability problems with organic soils occur during road construction at slope angles of 20-30° (Geertsema et al. 2010).

Considerations for Experimental Watershed Design

1. Theory suggests that “Snap-and-fly” single-stem harvesting or dispersed harvesting should have less effect on hillslope stability than clearcutting (e.g. by not damaging soil drainage pathways). However, there are no field studies to show the effect of decreased harvest levels on post-harvest landslide rates (Jordan et al. 2010).

2. Most post-harvest landslides occur 5-15 years or longer after logging due to the length of time for root strength deterioration and to experience a storm strong enough to trigger a landslide (Forest Practices Board 2005).

3. An important variable to consider in the design of an experimental watershed program to test harvesting impacts on debris flows will be the background levels of disturbance in a given watershed. One method to consider as an index of debris flow susceptibility is the “Melton ruggedness number” (Wilford et al. 2005b).

ii. Streambank stability, channel movement and sediment input for active fluvial units under differing forested buffer widths?

Summary of Knowledge

Leaving riparian buffers along stream channels has become a standard precautionary practice in forest management of large and fish-bearing streams post-FPC. However, in the last decade there has been little assessment of how wide riparian buffers must be to prevent negative impacts to streams and channel morphology. Those studies that have compared buffer treatments have found differing results, indicating that successful mediation of forestry impacts is very context-dependent.

Case study 1

A Prince George policy for riparian management of class S4 streams required a 5m wide machine-free zone next to streams, all non-merchantable vegetation retained, plus 10 merchantable conifers per 100 m of channel length, and maintenance of 50-70% of pre-harvest riparian shade. A study using a paired, before-after, control-impact experimental design found moderate-level concerns for long-term channel morphology integrity and benthic invertebrates. The B.C. Ministry of Forest and Range (2007) concluded that channel deterioration will likely occur over long term. Short-term findings show riparian practices limited fine-sediment generation but sediment entered streams from roads at stream crossings. There is high concern for long-term LWD supply, stream shade, and litter fall, given the limited width of riparian retention. Continuation of the policy is projected to result in ~60% reduction in in-stream LWD, resulting in channel simplification. More riparian retention was recommended.

Case study 2

To determine the effects of road building, logging, and slash burning on sediment production during the Alsea Watershed Study (AWS) in Oregon, a paired watershed study compared pre- and post-logging data on two channels, Deer Creek and Needle Creek. 25% of Deer Creek watershed was clearcut in 3 harvest units of 25 ha in size each. A forest buffer was left around the stream in the lowermost watershed harvest unit. 82% of Needle Creek watershed was logged and no forest buffer was left around the stream. Overall, road building, logging, and site preparation measurably increased sediment production (Beschta and Jackson 2008). Road building alone significantly increased sediment yield in Deer Creek. Logs yarded across Needle Creek channel caused streambank disturbance and left high levels of slash to accumulate in the stream, both of which contributed to increased sediment yields. Construction of mid-slope roads across steepened portions of hillslopes, clearcutting large portion of a watershed, and yarding across streams were all activities that contributed to increased sediment yield. Forested stream buffers appeared to prevent increases in sediment.

Case study 3

In the Takla region of BC, a paired-watershed design was conducted to examine the impacts of harvesting 40% of the watershed harvested with either 1) a riparian harvest that leaves limited trees or 2) retaining all trees within at least 10 m of the streambank (Beaudry In:

MacIsaac 2003). Four years later, it was found that maintaining a 30 m wide riparian buffers in many areas along each side of the stream minimized suspended sediment inputs (it increased by 21% and it was detectable for only 1 yr). However, a large amount of blowdown occurred in the stream, which may cause increases in sediments (not evaluated yet). In the watershed with no buffer, sediment inputs were large and detectable for all three years monitored after the logging.

Historically, only the influence of the riparian zone on the channel has been of concern to managers. However, it is increasingly emphasized in the literature that channels can, in turn, impact the composition of riparian zones (Hogan and Luzi 2010). For instance, Reeves et al. (2003) surveyed a relatively pristine 4th order stream in Oregon to determine the contribution of LWD from riparian versus upslope sources. They found that much of the LWD came from upslope (65% of all LW pieces and 46% of the LW volume). The upslope LWD originated largely from landslides, which indicates the need to consider these upslope processes in forest management and provides evidence for the importance of upslope disturbance as source for large wood in coastal North Pacific streams. The conclusion from studies such as these is that riparian management alone is insufficient without appropriate consideration for disturbances of upslope origin.

Relevant EBM LUOs

Section 8.(1) Maintain an ECA of less than 20% in important fisheries watersheds
Section 9.(1) Adjacent to HVFH, maintain a reserve zone with a width, on average, of 1.5 times the height of the dominant trees, and do not alter or harvest the forest in the reserve zone.
Section 10. Objectives for aquatic habitat that is not HVFH.
Section 13.(1) Adjacent to active fluvial units, retain 90% of the functional riparian forest in a management zone with a width, on average, equal to 1.5 times the height of the dominant trees.

Recommendations and Revisions

1. Based on few case studies in the last decade, a 30 m wide riparian reserve zone may be sufficient to minimize suspended sediment inputs from logging adjacent to the buffer. However, this is dependent on the percentage of the watershed harvested (See 2e, Beaudry In: MacIsaac 2003) and the amount of forest loss around headwaters. It is increasingly emphasized in the literature that requiring riparian reserve zones only for S1 to S3 streams is not sufficient to protect these streams from negative impacts of upstream harvesting. Increasing riparian buffer width for S4 to S6 streams (fish-bearing or not), will reduce downstream impacts of forest harvesting.

2. Riparian management needs to consider the interaction between stream channels and hillslope processes. For example, before delineating riparian buffers, managers need to account for the ability of the channel to shift laterally over time with input of logging/road-derived sediment. Otherwise, the buffers will become ineffective as the channel migrates (Hogan and Luzi 2010).

3. The CIT Hydroriparian Planning Guide (2004) precautionary guideline is to reserve active fluvial units of unknown activity from harvesting and road construction.

Considerations for Experimental Watershed Design

1. There is a need to conduct a large scale across watershed study to investigate the impacts of different riparian buffer reserve and management zone widths around stream channels. Treatments should include varying buffer widths around streams of different size and location in the drainage network, in particular S4 to S6 streams. Another treatment should be the percent of the total watershed cut (or ECA).

e. Does 70% retention of the forest around small streams maintain function? Can there be some partial harvest and to what level?

Summary of Knowledge

Few studies have been conducted to evaluate what percentage of forest is needed to maintain hydrological function. We expect this to be quite case-dependent. There is evidence that retention of 70% of the forest around small streams may still impact the hydrological function of the watershed. In Case study 3 (Question 2dii), 60% of the watershed was retained. Yet, this level of harvest still resulted in substantial increase in peak flows of small headwater streams, accelerating channel erosion. These headwaters can be important for fish and protection requires maintaining peak flows within natural range of variability (Beaudry In: MacIsaac 2003).

In a different system, at the White River Riparian Harvesting Impacts Project (WRRHIP) in Ontario, the frequency and magnitude of rainfall events were concluded to be more dominant controls on sediment transport to streams than area disturbed (Kreutzweiser et al. 2009). In this study, three stream sites were clearcut and the riparian buffers (30–100 m wide) were partially harvested:

- WR1: Area of watershed harvested (AWH) = 9% and Average Basal Area harvested in the buffer (BA) = 21%
- WR2: AWH = 53% and BA = 28%
- WR6: AWH = 88% and BA = 10%

At the highest riparian logging (WR2 with 28% BA), sedimentation rates were 5 times higher than reference sites and 3 times higher than the highest pre-logging rates at this site. However, this occurred only in the first post-logging year. Sediments appeared to be from ground disturbance to the soil rather than from the road. No differences were noted the next two years.

Hudson (2001) suggested that watersheds with a valley wide enough to absorb the amount of sediment contributed from road-sources might be able to tolerate greater harvesting. For example, in Russel Creek, Coastal B.C., 30% of the watershed was logged, but watershed morphology had a greater influence on sediment budget of Russell Creek than forest harvesting. Harvested sources averaged 30% of the Russell Creek sediment budget, mostly from landslides.

Relevant EBM LUOs

Section 8.(1) Maintain an ECA of less than 20% in important fisheries watersheds
Section 10. Objectives for aquatic habitat that is not HVFH.
Section 12.(1) Maintain 70% or more of the forest, in the portion of the watershed where upland streams occur, as functional riparian forest.

Recommendations and Revisions

1. To identify the appropriate harvest targets and thresholds, this question needs to be phrased more specifically. Does 70% retention of the forest around small streams mean a) 70% of the forest around headwaters (where small streams are abundant), b) 70% of the forest in the watershed, or c) 70% of the forest adjacent to small streams (i.e., riparian forest)? Also, “small streams” is an overly vague term as an S5 stream can be just as wide as an S3 stream, despite management guidelines classifying these differently in the LUOs and FRPA.

2. The precautionary guidelines for meeting hydroriparian objectives discussed in the CIT Hydroriparian Planning Guide (2004) may be thought to represent the most current knowledge in relation to this question.

Considerations for Experimental Watershed Design

1. There is insufficient recent scientific literature to answer this question. Research is needed.

Priority Question 3.

Issues of range of natural variation in amounts of forested systems, habitat supply thresholds and landscape level conservation.

a. How effective at maintaining ecological integrity and biodiversity across scales are the current targets for ecosystem representation?

i. What amount of habitat across scales maintains ecological integrity and biodiversity?

Summary of Knowledge

Habitat loss can have detrimental and at times irreversible effects on biodiversity. Some of these impacts are realized as gradual shifts in ecological responses (e.g. species abundance, richness and behaviour), whereas others become evident only after a sudden shift in ecological responses. Recently, sudden shifts have come to be thought of as evidence that there may be ‘‘critical threshold’’ levels of habitat. A critical threshold is ‘‘an abrupt, nonlinear change that occurs in some parameter across a small range of habitat loss” (Swift and Hannon 2010). For example, the abundance of a species may decline abruptly with habitat loss once the habitat area falls beyond some percentage of the total landscape area.

Identifying what critical threshold levels of habitat may be for species has been a topic of increasing research focus. Further, policy makers and land managers are increasingly employing the threshold concept in setting targets for ecosystem representation (Lindenmayer and Luck 2005), such as in the EBM LUO Section 14 and the CIT Hydroriparian Planning Guide. However, several recent reviews have examined the theoretical and empirical evidence for critical thresholds in species’ responses to habitat loss (Lindenmayer and Luck 2005; Swift and Hannon 2010; Ficetola and Denoël 2009). These studies agree that there are a number of significant issues with the use of critical thresholds for this management purpose:

1. Researchers have very different operational definitions of habitat thresholds.

2. Inconsistent and inappropriate use of methods for identifying habitat thresholds.
Ficetola and Denoel (2009) identified that three main groups of methods are used to estimate thresholds: a) Logistic regression is the most popular method. Here, authors generally consider a ''threshold'' the value for a given level of habitat above which the probability of species presence rises above a given value (e.g. 0.5); b) Nonlinear models. These are not able to explicitly evaluate whether an abrupt transition is present. As a result, authors visually assess the position of a threshold in plots. Visual inspection leads to especially subjective conclusions (Squires 2013); and c) methods such as piecewise/broken stick regression that test both whether there is nonlinearity in a relationship, and whether abrupt transitions (thresholds) explain this nonlinearity.

To investigate how results generated by these methods differ, Ficetola and Denoel (2009) constructed simulated datasets with known properties and used these to test the performance of each method. They found that logistic regression (the most widely used method) falsely detected abrupt thresholds in datasets without a threshold. Moreover, logistic regression can fail to correctly identify the value of the threshold. Piecewise regression was shown to be the most suitable method for this purpose. Use of the first two methods above are inappropriate for detecting thresholds and this prevents researchers from correctly identifying how much habitat is needed at the landscape-scale to maintain biodiversity.

3. Variability and context-dependency of thresholds.
Using critical thresholds in habitat area to make broad management decisions has also been criticized because thresholds vary by species, landscape type, and spatial scale (Lindenmayer and Luck 2005; Swift and Hannon 2010; van der Hoek et al. 2013). Moreover, research often overlooks the influence of habitat quality on species persistence (Lindenmayer and Luck 2005).

The spatial scale at which studies are conducted is often not reflective of the large scales at which managers make decisions. In a review by Squires (2013), over half of 32 empirical habitat threshold studies were patch-occupancy studies. In these, species presence-absence data is related to patch-scale habitat use, which makes it difficult to infer landscape-scale habitat use. Similarly, most studies that investigate the effects of habitat fragmentation on species do so at the individual not population scale, yet interpret these results at the population scale (Andrén 1994; Fahrig 2003). Study designs that adequately relate species responses to landscape-scale processes are also commonly lacking in studies of forest retention. A recent meta-analysis by Rosenvald and Lohmus (2008) found only one forest retention study conducted at the landscape scale.

Two common flaws in study design result in weak inferences about species responses to landscape-scale patterns and processes: a) the use of over-lapping landscape units, and b) the failure to use landscape units large enough to encompass species' population processes (Eigenbrod et al. 2011; Squires 2013). An appropriate-sized landscape unit can be inferred from dispersal distances, as it is commonly found in simulated and real landscapes that the dispersal ability of a species strongly influences their response to habitat loss (Jackson and Fahrig 2012). Jackson and Fahrig (2012) offer the guideline that the radius of a landscape unit from a central sampling point should be 4-9 times the median dispersal distance. Applying this to estimated dispersal distances for woodland-dependent birds, Garrard et al. (2012) found that appropriate sizes of landscape units for birds are 10x10 km to 124x124 km. However, most 'landscape' scale studies are conducted at scales smaller than 10x10 km. For example, just three of 32 empirical tests for habitat thresholds were conducted across multiple, appropriately-sized, non-overlapping landscapes (Squires 2013).

4. Lack of evidence that thresholds exist

Squires (2013) conducted a review of 32 empirical studies and generally found weak evidence for habitat thresholds, when defined as minimum habitat needed for species persistence. Swift and Hannon (2010) reviewed landscape-scale studies, and where thresholds were apparent, most fell within Andrén's (1994) proposed range of 10–30% habitat cover. It is important to note that because responses to landscape habitat representation are species-specific, a threshold will likely not be observed if aggregate response measures such as species richness are used (Lindenmayer et al. 2005; Becker et al. 2012). For example, Lindenmayer et al. (2005) found no evidence of threshold responses in bird or lizard species richness relative to the area of native vegetation.

Relevant EBM LUO’s

Section 14. Objectives for landscape level biodiversity

Recommendations and Revisions

1. Species loss and population declines will take place above threshold levels. Often, thresholds identify the point where a substantial number of species are lost from the landscape, whereas the focus should be on maintaining habitat area at the point where species are able to maintain viable populations for many generations (Lindenmayer and Luck 2005).

2. Tests for linear relationships between habitat loss and species abundance may in some cases support more risk-averse forest protection targets than thresholds derived from nonlinear relationships (also see Swift and Hannon 2010). For example, in Homan et al. (2004), linear models predicted a 50% probability of spotted salamander occurrence at 40–50% forest cover, whereas the threshold relationships they found at smaller spatial scales showed 50% probability of occurrence at about 20–30% forest cover.

Considerations for Experimental Watershed Design

1. Studies are needed to investigate the relationship between landscape-scale processes and species responses. Which species are likely to exhibit critical thresholds, and in what types of landscapes? (Swift and Hannon 2010). These studies should also test for unimodal or humped relationships between species responses and habitat area. For example, for some species, abundance may be greatest at intermediate habitat levels (Mortelliti et al. 2012). For forest-dependent species, it is important to identify species for which forest cover does not decline linearly with declines in habitat (e.g., Cushman and McGarigal 2003; Schmidt and Roland 2006).

2. Species richness of birds has generally proven to be a poor predictor of threshold responses to habitat loss (Lindenmayer and Luck 2005; Becker et al. 2012). Species richness often does not decline abruptly until habitat area reaches very low levels. For example, Radford et al. (2005) found a high rate of decline in forest-dependent birds below a threshold

of 10% forest cover. Yet, species richness was greatest where forest cover was 10–20%, just above the threshold. Species richness of bird communities often increases with initial loss in forest cover, due to colonization by early-seral species, and remains consistent until low levels of old forest (< 20%) are reached, because most late-seral species are able to persist in mid-seral stands (e.g. Cushman and McGarigal 2003).

ii. What are appropriate definitions of risk associated with habitat loss at a landscape scale?

Summary of Knowledge

Calculating risk in ecological systems is a relatively new field of study. In contemporary decision analysis literature, "risk" is defined as the product of two components: 1) the magnitude of the undesirable consequences that arise from uncertain events, and 2) the probability of those undesirable consequences occurring (Peterman 2004). *Risk assessment* (or risk analysis) refers to the process of estimating both components of risk, not one or the other. Risk assessment feeds into a *decision analysis,* where decision makers quantitatively evaluate how the risks will affect the ability of each management option to meet management objectives. *Risk management* is the process in which decision makers select actions after considering advice from risk assessment, decision analysis, and trade-offs with other external factors (Figure 5; Peterman 2004).

It is evident that this risk management framework is consistent with the EBM vision and could be useful for standardizing considerations of risk in forestry management. Here, "risk" would be defined as the expected level of undesirable consequences given a level of habitat loss, where the undesirable consequences could be declines in species abundance or biodiversity.

Currently, explicit calculations of ecological risk and risk assessment are not incorporated in the EBM LUOs. Instead, the EBM LUOs rely on two concepts to determine whether the level of undesirable consequences from habitat loss is 'acceptable' or not:

1. To what extent would the impacts of habitat loss exceed the range of natural variability (RONV)?

Often inferences about acceptable levels of habitat loss are derived from the RONV in forest vegetation. However, this does not accurately estimate effects on species persisting in habitat that exceeds the RONV. In estimating risk, managers should consider the species' life histories and whether species are evolutionarily adapted to conditions that only fall within the RONV. For example, the Coastal tailed frog has a life cycle that is highly specialized to small headwater streams and adjacent forests and therefore would be particularly sensitive to habitat loss.

The relevance of the RONV standard in estimating risk of adverse impacts is also called into question in the context of intermediate disturbance. Sensitive species generally show declines even at intermediate levels of forest cover. However, the intermediate disturbance hypothesis predicts that some species will experience positive responses at intermediate levels of forest cover loss due to increased habitat heterogeneity. Though less studied, there is some support for this hypothesis (e.g. Preston and Harestad 2007; Cushman and McGarigal 2003).

Perhaps of more use to land managers are measures of cumulative disturbance that cause adverse changes in species communities. For example, Scrimgeour et al. (2008) found fish communities were moderately to highly impacted across sites in sub-watersheds where more than 40% of the area was disturbed from land use activities (forestry, oil and gas, roads, and land converted to agriculture). Edge effects due to forest loss may not be encompassed by a RONV measures. Addition of disturbance indices such as road density to species-habitat models (Poulin and Villard 2011; St. Laurant et al. 2011) may do a better job of approximating the level of risk to species than % RONV alone. These types of studies are also more likely to capture the effect of the habitat matrix on species abundance and richness, which has in some cases been found to influence species more than characteristics of remnant habitat (e.g. Brady et al. 2011).

2. What is the expected level of habitat loss that will cause rapid population decline (i.e. the 'thresholds' paradigm)?

As discussed in 3.a.i, habitat thresholds are increasingly being employed in resource management as a means of defining tipping points in biodiversity loss or loss of ecological function. This approach could be an appropriate way of defining risk of species decline if targets for forest harvest are set well below thresholds of abrupt species decline. However, defining risk using thresholds in species population responses to habitat has often proven difficult if not problematic (Lindenmayer and Luck 2005; Ficetola and Denoel 2009; Swift and Hannon 2010).

In some cases, post-harvest data provide a record of both species disappearances and the proportion of habitat within a landscape that remained when species disappeared. In the absence of finer-scale data to estimate habitat thresholds, this level of habitat can be used to define “occurrence thresholds”. For example, regardless what the relationship between habitat loss and species response looks like (nonlinear or linear), the data in Homan et al. (2004) show that amphibians rarely occurred where surrounding forest cover was less than 20%. Similarly, one of few studies that specifically evaluated for occurrence thresholds found that bird species most strongly associated with trees were nearly absent below 20–40% tree cover (Cunningham and Johnson 2012). Conversely, these points can be used to identify when forest protection will not yield further benefits for species, allowing for optimal allocation of resources between stakeholder values.

Additional potential guidelines for defining risk of adverse impacts at the stand scale can be found in literature on variable-retention forestry. The aim of this harvest strategy is to retain live trees, snags, and downed wood that can provide habitat elements of mature harvest, such as landscape connectivity and moderation of microclimate (Linden et al. 2012) (Retention is discussed in greater depth in question 4). Removal of 85% or more of tree cover within a stand has consistently been found to result in loss of communities that were present prior to harvest (Aubry et al. 2009; Otto and Roloff 2012). In contrast, a number of studies have found that the abundance of sensitive bird and mammal species is maintained in stands with high (>70 %) retention levels (Vanderwel et al. 2009; Le Blanc et al. 2010; Holloway et al. 2012). Thus, the literature appears to support definitions of high risk at the stand scale as 15% retention, and low risk as retention that exceeds 70%. As usual, however, responses to levels of retention are species-specific.

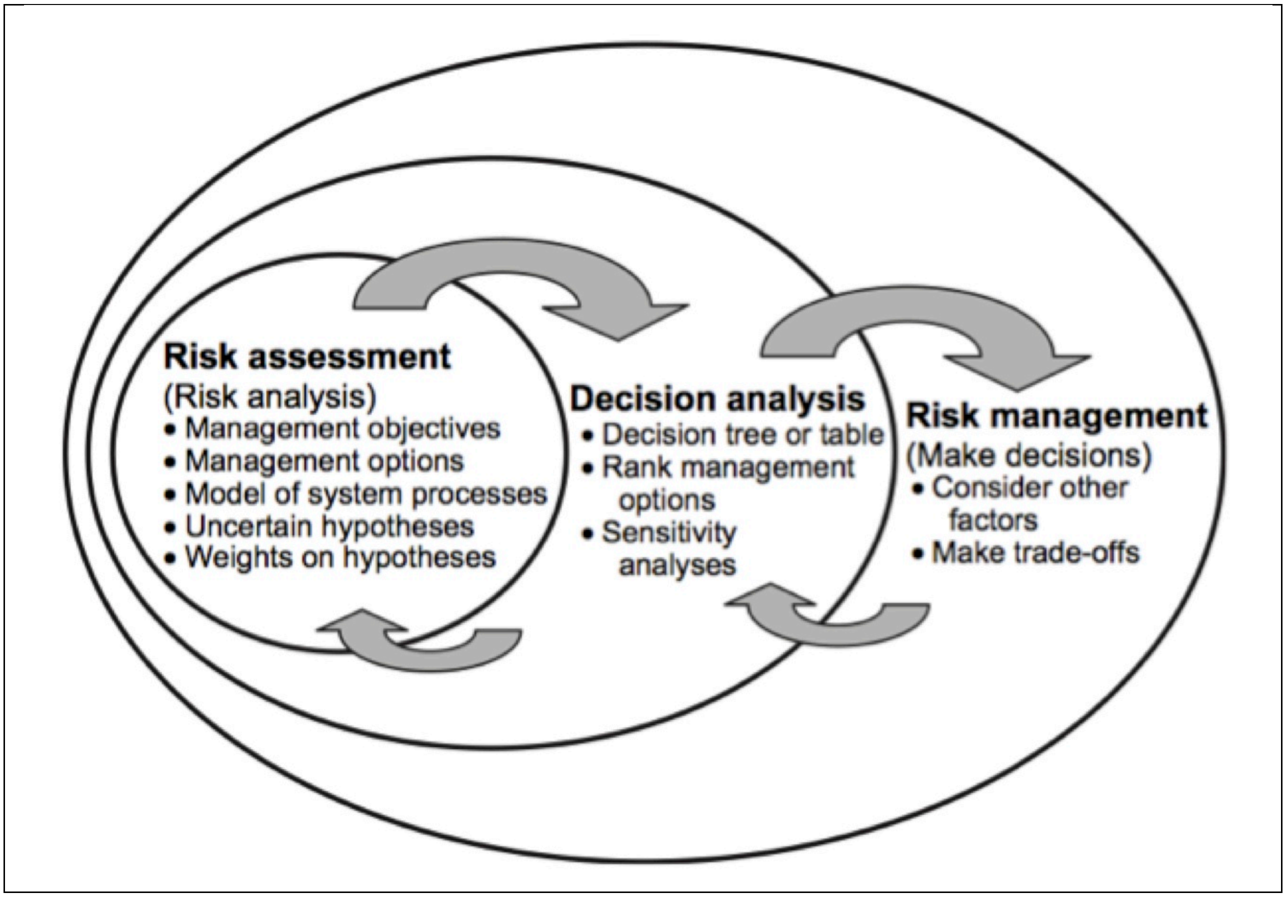

Figure 5. ***Risk assessment is a component of decision analysis, which considers risks to rank management options in the context of a stated management objective. In a forestry context, results from these analyses would provide advice to policy makers, who also consider other factors. Arrows indicate flows of information, including iterative feedback that is consistent with EBM's adaptive management strategy (Fig. 2 from Peterman (2004))***

Relevant EBM LUO's

Section 14. Objectives for landscape level biodiversity

Recommendations and Revisions

1. To avoid misunderstandings, policy makers, managers, and stakeholders should always clearly state what they mean by the term ''risk'' (Peterman 2004).

2. Tests for threshold responses may be more relevant in terms of quantifying risk when defined as thresholds in occurrence. It would be helpful to know at what levels of forest cover sensitive species disappear. Regardless of the shape of responses, it would also be helpful to know at what levels of forest cover sensitive species begin to decline.

3. A landscape scale study relating species responses to environmental gradients would be very valuable, since landscape scale studies are consistently rare, across three broad types of literature - habitat loss-fragmentation, habitat thresholds, forest retention.

4. At the persistence end of the gradient, quantification of risk from changes in the responses of sensitive species may be more relevant if species are related to landscape-scale 'disturbance' (e.g. road density, industrial footprints), rather than habitat (i.e. forest cover) *per se*.

Considerations for Experimental Watershed Design

1. A landscape scale study requires measuring species responses across multiple, independent, non-overlapping landscapes, each of a size relevant to the life histories of the studied species (Eigenbrod et al. 2011; Jackson and Fahrig 2012).

2. Linking changes in species responses to measures of landscape disturbance, rather than just amount of forest cover, will likely provide a more relevant measure of risk.

iii. How do sensitive species respond to low levels of landscape-level reserves?

Summary of Knowledge

Sensitive species are those species that have been found to respond rapidly and/or strongly to loss of forest cover or to other vegetation gradients relevant in forest management (e.g. brown creeper sensitive to forest edges; D'Astous and Villard 2012). Identifying these species is critical for directing research, management, and monitoring efforts and an important first step in conserving biodiversity and ecological integrity. As discussed in 3.a.i., few studies have related species responses to landscape-scale habitat areas. Here, we focus specifically on reviewing the responses of vertebrate species sensitive to logging. The relative sensitivity of invertebrates, fungi, bryophytes, and lichens is discussed in response to Question 3.a.v.

Two bird species have consistently been identified as sensitive to logging in the Pacific Northwest: the brown creeper and the Pacific slope flycatcher. The brown creeper rarely occurs, if at all, in stands with less than 30% forest retention (Preston and Harestad 2007; Vanderwel et al. 2009). Even at 70% retention, populations have been observed to decline in abundance by 25% and show lower nesting productivity (Vanderwel et al. 2010; D'Astous and Villard 2012). It has been found to avoid forest edges (Brand and George 2001) and reproduces more successfully when more than 100 m from forest edges (Poulin and Villard 2011). Kissling and Garton (2008) found it did not occur in sea-side forest buffers less than 250 m wide. The abundance of the Pacific slope flycatcher has likewise consistently been found to increase with forest patch area (Brand and George 2001; Preston and Harestad 2007; Kissling and Garton 2008).

The Northern flying squirrel appears to be particularly sensitive to loss of forest cover (Lehmkuhl et al. 2006), as well as to spatial configuration of remnant forest (Smith et al. 2011). Tree canopy cover was the strongest predictor of squirrel density—squirrels were twice as dense (2.2 squirrels per ha) when forest cover was above a threshold of 55% versus below this threshold (1.1 squirrels/ha; Lehmkuhl et al. 2006). Less than 40% stand level retention does not provide suitable habitat (Vanderwel et al. 2009; Holloway et al. 2012). Under simulated harvest regimes, Northern flying squirrels were one of a few species whose habitat was altered beyond the natural range of variability, particularly due to the loss of arboreal lichen on which they forage (Doyon et al. 2008). Shanley et al. (2013) found that they were not found in old forest patches less than 73 ha. Smith and Person (2007) predicted that reserves at least 79,000 ha in size would be needed to ensure 85% probability of persistence of northern flying squirrels for at least 100 years.

Tilghman et al. (2012) conducted a meta-analysis of 24 studies across North America that evaluated the effects of canopy cover on terrestrial salamanders. Salamanders almost always declined in abundance in response to logging: by 29% over the long term in response to retention harvesting, and by 62% in short-term responses to clearcutting. Salamanders have been found to be the most abundant vertebrates in forests of the Pacific Northwest (reviewed in Tilghman et al. 2012) therefore this represents a substantial loss.

Relevant EBM LUO's

Section 14. (7) To the extent practicable, include within old forest retention areas, stands of monumental cedar for future cultural cedar use, red and blue-listed plant communities, habitats important for species at risk, ungulate winter ranges, and regionally important wildlife.

Section 15. Objectives for red-listed and blue-listed plant communities.

Recommendations and Revisions

1. Studies consistently find a relatively small suite of vertebrate species that are sensitive to logging (e.g. brown creeper, northern flying squirrel). The species that are sensitive to logging are not always those prioritized for other reasons (e.g. species at risk).

Considerations for Experimental Watershed Design

1. Assessments of population density, distribution and habitat relationships are needed for species that are sensitive to harvesting but are not listed as priority species. Two species we highlight are the brown creeper and the northern flying squirrel. Other species may also need to be considered.

iv. For priority species/ecosystems what is critical habitat and what level of stewardship will result in low risk management?

Summary of Knowledge

Priority species are listed in the Land Use Order and include red and blue-listed plant species, and regionally important wildlife including (but not limited to) mountain goats, grizzly bears, northern goshawks, tailed frogs, and marbled murrelets. Priority species will be discussed here. Please refer to Question 2a.i. for literature regarding tailed frogs.

Mountain goats

Relatively little is known about the North American mountain goat (*Oreamnos americanus*). They have been found to generally select habitat that is close to escape terrain (classified as any area with slope greater than 40 degrees) and avoid valleys (Shafer et al. 2011). Shafer et al. (2011) identified that two hot spots of diversity occur in North America, one of which is located in Northern BC. Comparing Fig. 6 with Fig. 1, it appears that areas of greatest diversity lie largely outside the Central and North Coast EBM zones. However, long-distance dispersal and movement across sub-optimal habitat is essential to gene dispersal (Shafer et al. 2011). Further, peripheral populations are of evolutionary importance as they may have unique traits (e.g. climate tolerance) or be locally adapted (Hampe and Petit 2005; Shafer et al. 2011).

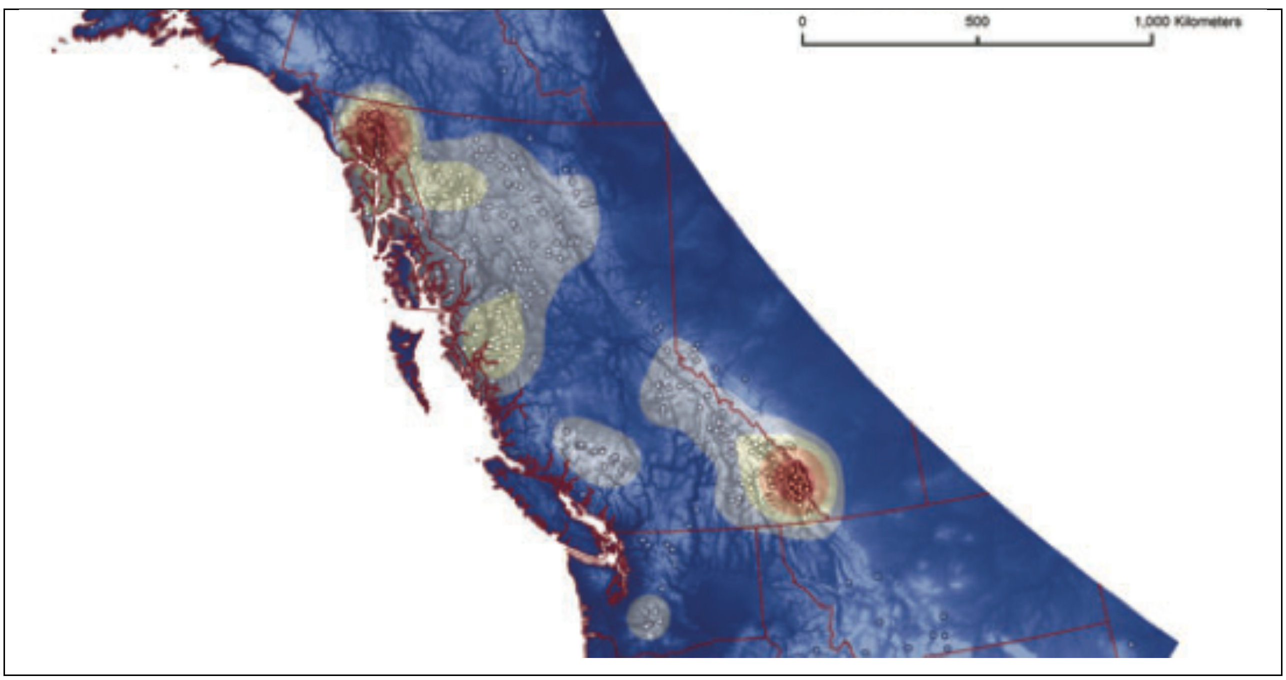

Figure 6. ***Map of hot spots of mountain goat genetic diversity (Figure 1B from Shafer et al. 2011).***

Grizzly Bears

Grizzly bears (*Ursus arctos*) are Blue-listed in BC and listed as of *Special Concern* under COSEWIC. A full report on this species can be found under the Ministry of Environment "Accounts and Measures for Managing Identified Wildlife" (Gyug et al. 2004). Critical habitat for grizzly bears varies seasonally and has recently been mapped in the Central and North coasts of British Columbia to inform the EBM LUOs (Ministry of Environment 2008; Figure 7). Effective management and conservation of grizzly bears requires the preservation of a wide range of habitat classes across elevations because the species' habitat use varies seasonally. Mowat et al. (2005) suggest the ecoregion scale is appropriate for estimating grizzly bear densities (also see Artelle et al. in review).

Northern Goshawks

The coastal subspecies of Northern Goshawk (*Accipiter gentilis laingi*) is Red-listed in BC and is listed as *Threatened* under COSEWIC. The goshawk is sensitive to forest development. The forest characteristics that make up its habitat are largely similar to suitable forest-harvesting cutblocks (old-growth trees, low elevations) (Mahon et al. 2009). Mitchell et al. (2008) surveyed 19 landscape units almost entirely in the Coastal Western Hemlock zone to locate critical goshawk habitat on the Central Coast of BC. They have proposed seven Wildlife Habitat Areas (WHA) in four landscape units (the Bella Coola, Nusatsum, Saloompt, and Talchako/Gyllenspetz landscape units). The dominant tree species at most nests was Western hemlock, followed by Douglas fir. These species offer key structural characteristics for goshawk nesting, such as branches that form platforms. Over 70% (10 of the located nests) had a canopy closure greater than 45% and an understory ≤35%. Further surveying is needed to locate nests in the Central Coast, especially in areas with more difficult accessibility (Mitchell et al. 2008).

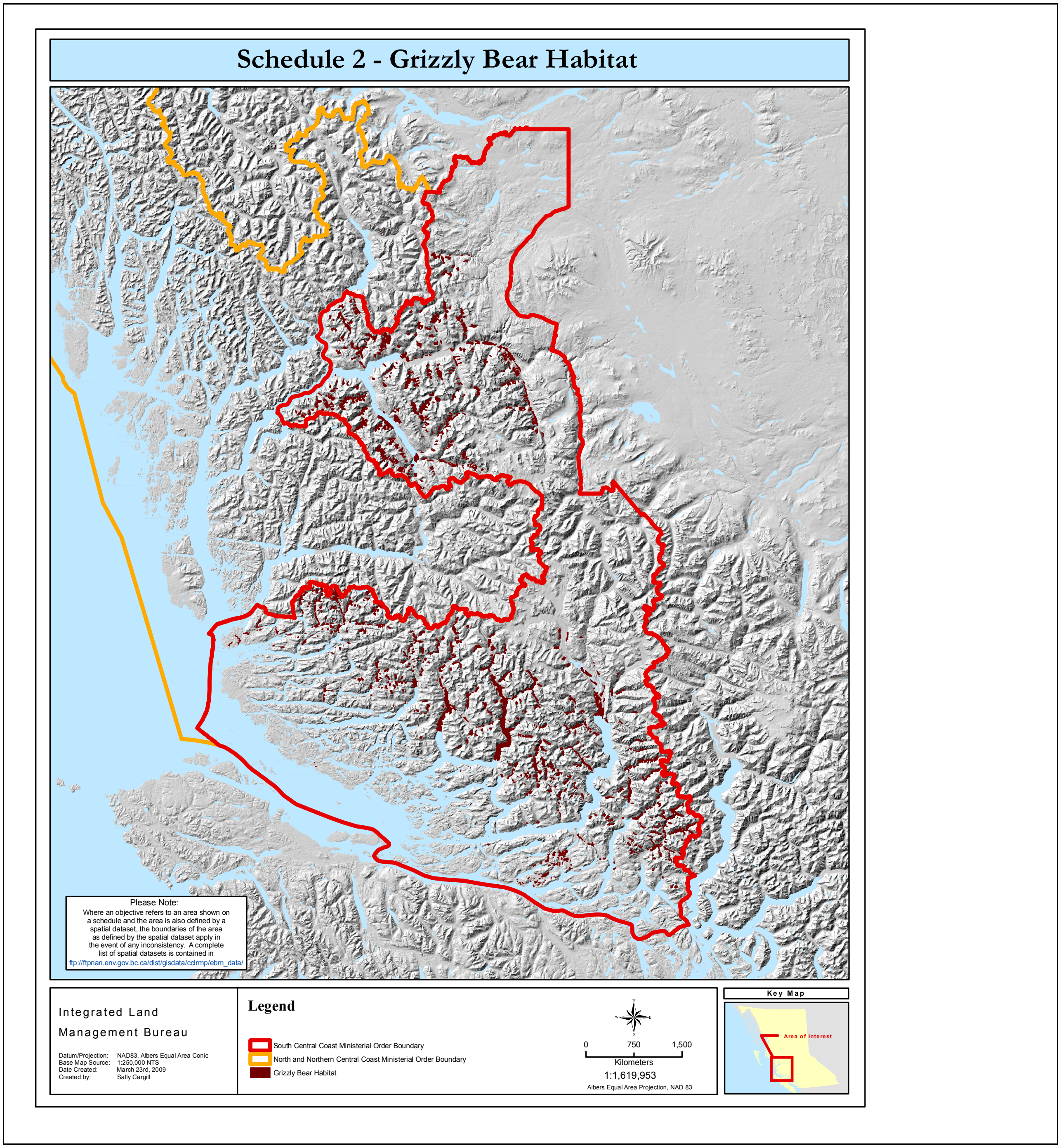

Figure 7. Map of critical grizzly bear habitat (Ministry of Environment 2008).

Marbled Murrelets

The Marbled Murrelet (*Brachyramphus marmoratus*) is listed as *Threatened* under COSEWIC. Burger et al. (2010) undertook a large scale study over 170 watersheds in coastal BC and found that tree diameter was the most important predictor for Marbled Murrelet nesting sites, with tree height, species, elevation, slope and latitude also predictors at a lesser extent. Nesting trees are 20% taller on average (Silvergieter and Lank 2011). This bird is very sensitive to forest harvesting; edge effects can cause higher intensities of predation (Malt and Lank 2009; Golightly and Schneider 2011), as well as an altered microclimatic environment less favorable for epiphytic growth (van Rooyen et al. 2011). Over the past 30 years, it is estimated that 20% to 24% of potential habitat was lost from harvesting activities (Long et al. 2011). Further, Miller et al. (2012) estimate a 30% decline in Marbled Murrelet population in Washington, Oregon, and Northern California over the past 10 years. This decline coincides with the loss of nesting habitat and possible other marine and terrestrial factors. Hazlitt et al. (2010) suggest incorporating marine values in the planning process for reserve designs.

Relevant EBM LUO's

Section 14. (7) To the extent practicable, include within old forest retention areas, stands of monumental cedar for future cultural cedar use, red and blue-listed plant communities, habitats important for species at risk, ungulate winter ranges, and regionally important wildlife.
Section 15. Objectives for red-listed and blue-listed plant communities.

Recommendations and Revisions

1. Critical habitat for priority species has in most cases been identified and in some cases mapped. However, more research is necessary to identify further active nesting territory for both the Marbled Murrelet and Northern Goshawk.

2. The preservation of a wide range of habitat classes across elevations is necessary for effective management and conservation of grizzly bears.

3. The Land Use Orders highlight the conservation of red and blue-listed plant communities, habitats important for species at risk, ungulate winter ranges, and regionally important wildlife. We encourage the broadening of this definition to include maintaining the historical diversity, distribution, and composition of species present on the landscape.

Considerations for Experimental Watershed Design

1. Collecting baseline information on how grizzly bears interact with the landscape in relatively disturbance-free regions is important for informing management decisions (Milakovic et al. 2012).

2. The Central Coast First Nations, in partnership with Dr. Chris Darimont at the University of Victoria and others, have recently formed a working group for grizzly bear management. The BC Government should partner with First Nations in grizzly bear management, as this has been identified as an important priority for local communities on the Central Coast.

3. In general, the level of stewardship needed to produce low risk management is not well known for priority species and ecosystems. The Coast Area Forest Research team should partner with academia and with local First Nations to monitor key species in priority watersheds. This should include population censuses, habitat relationships and distributions to improve or build species-habitat relationships (e.g. Fenger et al. 2009).

v. What extent and spatial arrangement of mature forest habitat is necessary for maintaining cryptogams, fungi, and arthropods?

Summary of Knowledge

An extensive literature search remains to be carried out on this topic. Bryophyte, fungi, and lichen responses to loss of forest cover due to logging are well documented in European forests, where generally these groups have been found to be among the most sensitive of all taxa to forest harvesting (Brunet et al. 2010; Paillet et al. 2010).

In the Pacific northwest, bryophytes experienced large declines in abundance at 40% forest retention, and were much more sensitive to forest loss than late seral herbs (Arsenault et al. 2012; Halpern et al. 2012). In old inland BC rainforests, the growth of cyanolichen was at least 15% higher in unharvested, uneven-aged versus even-aged stands, suggesting that growth may be similarly impeded in harvested even-aged stands (Coxson and Stevenson 2007). Studies have found that the size of retention patches also has an effect on bryophytes and lichens. In temperate rainforests of coastal BC, bryophyte diversity was retained in patches of old growth larger than 3.5 ha (Baldwin and Bradfield 2007). In Sweden, patches less than 0.05 ha were too small to serve as refugia for bryophytes and lichens (Perhans et al. 2009).

Two recent studies of arthropod responses to stand retention in Oregon showed bark arthropods densities increased by 2.6 times at 15% retention level relative to at 40% retention, but that old-growth species declined in abundance by 57–84% at both retention levels (Halaj et al. 2008; 2009). For all arthropods except ants, the spatial arrangement of retention has been shown to have little effect on species abundance (Halaj et al. 2008; 2009). Ant densities were higher in aggregated 1 ha patches than at sites with dispersed retention. In a European forest, about five more species of saproxylic beetles on average were found at sites where 11% of the surrounding landscape (within 1 km) was old forest, compared to sites surrounded by less than 2% cover of old forest (Olsson et al. 2012).

Relevant EBM LUO's

Section 16. (1) Maintain forest structure and diversity at the stand level:
(a) By establishing stand retention equal to or greater than 15% of the cutblock; and
(b) In cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.

Recommendations and Revisions

1. Given the history of research in European forests on these taxonomic groups, a more extensive literature review will likely offer more conclusive information.

2. Fungi, lichens, bryophytes, and arthropods are likely to be among the most sensitive to logging. As stated under 'Considerations for EWD' for 2.b.i, relative to most herbaceous plants and vertebrate species, monitoring these taxonomic groups is likely to provide more

sensitive tests of the impacts of the spatial arrangement of logging at both stand and landscape scales.

Considerations for Experimental Watershed Design

1. See 'Recommendations and Revisions' no. 2, above.

b. How does configuration of habitat influence ecological integrity as measured using suites of response variables?

Summary of Knowledge

Habitat fragmentation due to habitat loss involves alterations to all aspects of habitat configuration, such as the number of habitat fragments, edge density, and patch shape (Swift and Hannon 2010). When populations become isolated across habitat patches, they become more vulnerable to extinction through the interaction of demographic, environmental, and genetic factors. However, there are two topics of ongoing debate in the habitat configuration literature: 1) whether habitat configuration is a more important driver of population dynamics than habitat area; and 2) whether the effects of changes in configuration that result from habitat fragmentation can be adequately disassociated from changes in habitat area.

1. Are the impacts of habitat configuration more important than those of habitat area?

Habitat fragmentation can be especially detrimental to the ecological integrity of forests because species persistence depends on matching dispersal ability with habitat configuration. Dispersal is fundamental to determining patch colonisation and extinction rates in metapopulation and source-sink dynamics; and facilitates gene flow across landscapes (Harrison et al. 2012). In modeling genetic exchanges between patches in simulated landscapes, Cushman et al. (2012) found that habitat configuration is more important than habitat area in driving genetic differentiation. This was likely because changes in configuration directly impact genetic differentiation by affecting the spatial pattern of dispersal and mating.

In a number of studies of forest birds, however, habitat area was more important that fragmentation in determining forest bird species distributions (Harrison et al. 2012; Smith et al. 2011b). Harrison et al. (2012) found that even species that tend to decline disproportionately as landscape level habitat decreases continued to display high inter-generational genetic connectivity in fragmented habitats (Harrison et al. 2012). This suggests that for some species negative effects of changes in configuration may only become important when the amount of habitat is low (Andrén 1994; Betts et al. 2007; Smith et al. 2011b). Such threshold responses might be evidenced by a rate of change in ecological responses that is greater than expected from habitat loss alone (Swift and Hannon 2010).

Configuration effects are undoubtedly very species-dependent. They may primarily be important to consider for species that have low mobility (Harrison et al. 2012) and/or are sensitive to disturbance (Aubry et al. 2009), such as amphibians (see question 2 for amphibian-specific management recommendations) or the northern flying squirrel. For example, Smith et al. (2011) found that the proportion of connectivity in surrounding landscapes, measured as the distance between remnant forest patches, strongly influenced the homing ability of translocated northern flying squirrels.

The effect of habitat fragmentation and habitat loss may also be quite dependent on landscape size (Smith et al. 2011b). For many species, biological processes such as reproduction, dispersal success, mortality in the matrix, foraging, as well as nest predation and parasitism for birds, occur at different rates and span different habitat types. Consequently, if fragmentation has positive effects on some processes and negative effects on others, then the net effect of fragmentation may be scale dependent. For example, Smith et al. (2011b) found that the effects of fragmentation on forest birds tended to be change in magnitude and direction depending on the habitat size. However, the amount of habitat in the landscape was positively associated with forest birds and the strength of this effect increased with the landscape size.

2. Dissociating effects of changes in configuration from changes in habitat area

Separating the effects of habitat fragmentation and loss is challenging as habitat amount and configuration are inextricably linked. For instance, it is not possible to have high habitat fragmentation in a landscape that has a high percentage of habitat, given that there are only so many ways to separate patches that cover the entire landscape (Cushman et al. 2012). Cushman et al. (2012) found it difficult to assess the relative effects of habitat area and configuration because of high confounding between the metrics. Similar confounding exists in retention studies, which have generally found that the spatial arrangement of retention is less important than the amount of retention (Rosenvald and Lohmus 2008; Aubry et al. 2009).

Relevant EBM LUO's

Section 14. Objectives for landscape level biodiversity

Section 16. (1) Maintain forest structure and diversity at the stand level:

(a) By establishing stand retention equal to or greater than 15% of the cutblock; and

(b) In cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.

Recommendations and Revisions

1. For mobile bird species, reconnecting isolated habitat patches appears secondary in importance to increasing habitat extent and/or quality (Harrison et al. 2012).

2. Smith et al. (2011b) make two broad management recommendations related to habitat configuration:

- If fragmentation effects are only apparent at some scales then these should be managed at that scale; and
- If fragmentation has positive effects at one scale but negative effects at another, then managing fragmentation may not be the most efficient use of conservation resources.

3. As stated in the 'Recommendations and Revisions' for 3.a.v., monitoring sensitive species, such as fungi, lichen, and bryophytes, may be an effective litmus test for compromised ecological integrity across different habitat configurations.

Considerations for Experimental Watershed Design

1. Cushman et al. (2012) identified that patch cohesion, correlation length and aggregation index are strong predictors of genetic differentiation and are therefore among the best metrics for studies aiming to quantify the impacts of habitat configuration on movement and gene flow.

2. In addition to levels of disturbance, habitat amount and configuration, other measures to consider are the quality of retained habitat patches (for example, seral stage), as well as characteristics of the matrix (Rosenvald and Lohmus 2008; Kupfer 2012). Habitat quality is a key predictor of species distributions; therefore, not accounting for it in studies of species responses to habitat loss and configuration may lead to inaccurate conclusions (Bollmann et al. 2011; St. Laurant et al. 2011).

c. What are current levels of natural disturbance by ecosystem type? How does natural disturbance change over time?

Summary of Knowledge

Currently, the information presented in the CIT Hydroriparian Planning Guide (2004) taken from Price and Daust (2003), shown below in Table 4, still represent the best available knowledge of rates of natural disturbances on the Central and North Coast. However, two further studies by Parish and Antos (2004, 2006) and one by Pearson (2010) provide further evidence that natural stand-replacing disturbances are rare on the Central Coast. There seems to be no evidence for stand replacement due to disturbance in a coastal montane stand (Parish and Antos 2004; 2006). Pearson (2010) used aerial photos and GIS to examine stand-replacing disturbances on the Central Coast over the past 140 years. At the regional scale, evidence of natural disturbances was found for 3.1% of the forested area, whereas disturbance from logging accounted for 5.1%. Within valley bottoms, however, logged areas were 10 times the size of natural disturbances (Pearson 2010).

A relatively recent topic in the literature is that of how climate change will impact forest disturbance regimes. For example, Daniels et al. (2011) found that a doubling of tree mortality rates in old forests in western North America was associated with warming-related water deficits, as well as increases in the frequency of insect, wind, and pathogen disturbances. An in-depth discussion of the effects of climate change on forests is outside the scope of this report but should be addressed in response to FLNRO priority question 6.

Table 4. Range of natural variability in proportion of old forest in upland and wetland, fluvial, and ocean spray ecosystems.[a] (Table 2 in CIT Hydroriparian Planning Guide 2004)

Region	Ecosystem[b]	Return interval for disturbance[c]	Proportion of forest >250 years (%)
Hypermaritime	upland and wetland	4,500–33,300	95–99
	fluvial	2,200–11,100	89–98
	ocean spray	1,000–5,600	78–96
Outer Coast North	upland and wetland	1,800–10,000	87–98
	fluvial	500–2,100	61–89
Outer Coast South	upland and wetland	900–2,500	76–90
	fluvial	400–1,400	54–84
Inner Coast	upland and wetland	500–5,600	61–96
	fluvial	300–900	43–76

[a] Price and Daust (2003).

[b] Based on groups of site series listed in small-scale Predictive Ecosystem Mapping.

[c] To nearest 100 years.

Relevant EBM LUO's

Section 16. (1) Maintain forest structure and diversity at the stand level

Recommendations and Revisions

1. The CIT Hydroriparian Planning Guide (2004) remains the best available source for information pertaining to levels of natural disturbance.

Considerations for Experimental Watershed Design

1. An assessment should be carried out to understand the state of knowledge and level of uncertainty associated with the range of natural variability in each BEC zone. The range of natural variability in disturbance will be an important watershed-scale covariate in quantitative models that test how forestry may impact species or ecological integrity.

2. One new way to assess disturbance regimes is through the use of LiDAR technology. The Hakai Institute on Calvert Island (www.hakai.org) has flown LiDAR for the entire Calvert Island and there is interest in expanding to nearby mainland watersheds and beyond. LiDAR will be very useful for determining forest size structure as it provides data on tree height for every single tree in the landscape.

Priority Question 4.

Interactions between stand-level retention and landscape-level representation/conservation

a. What is the effectiveness (for various processes and measures of function) of stand level retention?

i. In the context of different levels of landscape level representation?

Summary of Knowledge

The effectiveness of stand-level retention depends on the amount and configuration of habitat retained. Generally, the amount of retention is more important than the configuration of retention, although the relative importance of these factors varies by species (Rosenvald and Lohmus 2008; Bollman et al. 2011; Cushman et al. 2012).

As yet, the results of stand-scale studies have rarely been extrapolated to make landscape-level inferences about habitat representation (see Cushman et al. 2012 on landscape-scale fragmentation in question 3b). Rosenvald and Lohmus (2008) reviewed 181 peer-reviewed studies on species responses to retention and only one study was conducted at the landscape scale. Furthermore, few studies actually relate retention levels to desired outcomes or specific management objectives and therefore it is difficult to say how much retention is sufficient (Gustafsson et al. 2012).

Literature on the efficacy of stand-level retention has primarily focused on changes in abundance and diversity of small mammals, birds, arthropods, and vegetation (See Table 5 below; Rosenvald and Lohmus 2008). The efficacy of stand-level retention in maintaining forest structure and function is another topic of research focus, and is discussed below in question 4aii.

Table 5. Review of 181 retention studies ordered by type of forest, geographical region (A- North America, E- Europe), and species studied (Table 1 from Rosenvald and Lohmus 2008).

Study issues and objects	Boreal[a]		Temperate[a]		Subtropical[a]	Total
	A	*E*	*A*	*E*	*A*	
Biodiversity effects[b]	49	22	60	3	10	144 (28)
Epiphytes	1	4	4	1		10 (3)
Ground vegetation	7	6	12			25 (5)
Ectomycorrhizal fungi	5	1	5			11 (2)
Arthropods	5	11	3		1	20 (2)
Amphibians and reptiles			6		1	7
Birds	22		19	2	5	48 (10)
Mammals	9		11		3	23 (6)
Effects on retention trees and regeneration	38	16	45	2	8	109 (28)
Stand regeneration	13	6	18	2	5	44 (13)
Growth of retention trees	3	2	12		2	19 (10)
Stand structure[c]	4	5	5			14 (5)
Wind-resistance	15	2	3			20
Logging damage on trees and soil	3	1	7		1	12
Other ecological and biological effects[d]	10	4	5		1	20
Total	97	42	110	5	19	273 (56)

In brackets—long-term studies (at least 20 years post-cut).
[a] Region.
[b] Incl. five studies on bird nest predation and two on mammal diets (all in North America).
[c] Incl. three studies on GTR as a source of coarse woody debris (all in boreal Europe).
[d] Incl. six studies on the production, germination, dispersal or predation of tree seeds; four studies on GTR effects to genetic structure of regeneration; five studies on the changes in microclimate and five on the changes in soils.

Small Mammals

In general, small mammal species composition becomes increasingly dissimilar to that of uncut forest stands with decreasing levels of retention (Gitzen et al. 2007; Lindenmayer et al. 2010). The red-backed vole has been found to be particularly sensitive to logging, and shows declines in abundance in clearcut and low retention sites (Sullivan et al. 2007; Klenner and Sullivan 2009). Other species do not decline in abundance until retention level falls below some level. For example, Hodson et al. (2012) found no difference in snowshoe hare abundance between cut and uncut stands when retention was greater than 50%. Holloway et al. (2012) found that flying squirrels did not show lower abundance if retention exceeded 75%; however, abundance was much lower at stand retention levels of 40% and lower.

Birds

Many recent studies report that if harvesting is kept at moderate levels, bird composition in forests stands will remain more similar to uncut stands than clearcuts (Kissling and Garton 2008; LeBlanc et al. 2010; Astous and Villard 2012; Linden et al. 2012). However, early- and late-seral-associated species may respond quite differently to disturbance. Generally, early-seral species are abundant in clearcuts and in stands with low retention, whereas late-seral species are more abundant in moderate to high retention harvests (Becker et al. 2012). For some sensitive species, even high levels of retention (70%) may not be enough. The Brown Creeper and the Ovenbird are late-seral-associated species and are very sensitive to logging. In eastern Canada, it was found that Brown Creeper nest densities decreased by 50% from harvested sites that retained 60–70% basal area, despite that the abundance of invertebrates (their food source) did not change (D'Astous and Villard 2012). Similarly, a meta-analysis predicted a 25% decline in Brown Creeper and Ovenbird populations with 70% retention, and a 75% decline with 50% retention (Vanderwel et al. 2010).

Cushman and McGarigal (2003) conducted one of the only studies examining bird response to retention at the landscape scale. They examined 30 landscapes in 3 sub-basins in the Oregon Coast Range, and found that species richness increased dramatically with a decrease in mature forest cover from 100% to 80%, due to the addition of early-seral bird species. However, the authors note that this result should not be taken out of context—most species of conservation concern tend to be associated with old-growth and late-seral forest.

Arthropods

A number of studies show declines in arthropods with forest harvest. Halaj et al. 2008 found that in comparison to unharvested controls, old forest-associated spider species were 57%–84% less abundant and carabid beetle 60% less abundant in retention patches of 15% and 40%. Furthermore, the same study concluded that none of their treatments (15% and 40% retention, aggregated and dispersed) were able to support the abundance and diversity of late-seral-associated arthropod species.

Vegetation

Understory vegetation cover generally increases with increased levels of harvesting (Craig and Macdonald 2009). In Alberta, plant species richness was unaffected by retention, and decreased as tree density increased (Craig and Macdonald 2009). However, many studies show significant declines in late-seral associated species. In the DEMO study in western Oregon and Washington, Halpern et al. (2012) reported that 10 years after harvesting, late-seral herbs were less abundant, and bryophytes occurred at much lower abundance in 40% retention patches. Variable retention does maintain downed wood however, and therefore epixylic bryophyte communities are not as affected (Arsenault et al. 2012).

Overall, the literature agrees on two points: 1) higher levels of retention maintain greater species abundance and richness (See Fig 8; Rosenvald and Lohmus 2008), and retention-harvesting does lessen species loss as compared to clearcutting; but 2) high retention levels do not maintain all species at unharvested levels.

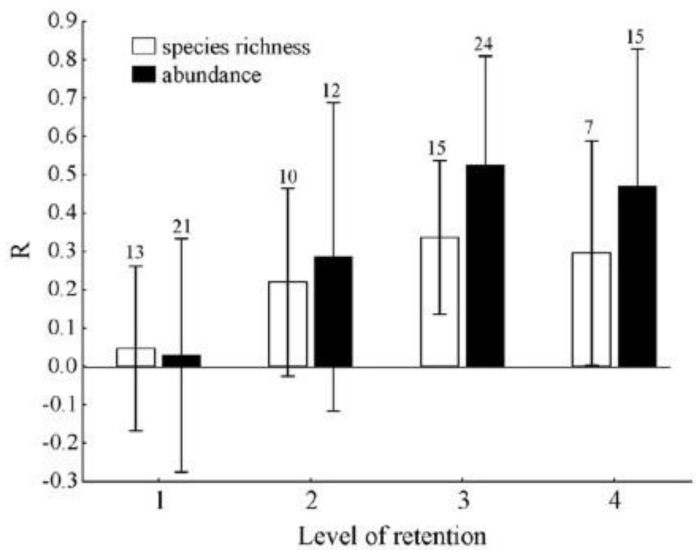

Figure 8. ***Mean differences between retention cuts and clearcuts in species richness and abundance for eight taxa in relation to four levels of retention: 1- solitary trees; 2- group-retention; 3- two-storey retention; 4- shelterwood. R indicates the natural logarithm of the ratio of values between retention to clearcut (positive values indicate a higher value of retention cuts). Number of studies are indicated at the top of each bar. Lines are 95% confidence intervals for the mean (Figure 3 in Rosenvald and Lohmus 2008).***

Relevant EBM LUOs

Section 16.(1) Maintain forest structure and diversity at the stand level:

(a) By establishing stand retention equal to or greater than 15% of the cutblock; and

(b) In cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.

Section 16.(2) To the extent practicable, include the following within stand retention:
(a) Habitat elements important for species at risk, ungulate winter range, and regionally important wildlife;
(b) Representation of ecosystems and plant communities that are red-listed or blue-listed in the watershed and landscape;
(c) Functional riparian forest adjacent to active fluvial units, forested swamps, fen and marsh wetlands and upland streams with unique climate and other characteristics;
(d) Western red cedar and yellow cedar, in a range of diameters representative of the preharvest stand, and important for future cultural cedar use; and
(e) Wildlife trees and coarse woody debris.

Recommendations and Revisions

1. The retention limit set in Section 16.1(a) should be revisited. The literature agrees that most species, especially late-seral species decrease in abundance and richness with less than 15% retention.

Considerations for Experimental Watershed Design

1. Late seral-associated forest species, in particular, are good indicators of retention effectiveness (Vanderwel et al. 2009).

2. Landscape level representation is currently measured using site series designations in BC, however the literature does not measure effectiveness of stand-level retention in relation to levels of ecosystem representation.

3. With BC's BEC classification system, a unique opportunity exists to link studies to BEC Site Series in order to measure landscape-scale representation.

ii. What habitat elements are critical for ecological integrity in long-term retention within harvested cutblocks?

Summary of Knowledge

Habitat elements critical to ecological integrity have been well researched (Bauhus et al. 2009, Gustafsson 2012). These elements include riparian features (refer to Questions 1 and 2), large woody debris (LWD), snags, big trees, vertical and horizontal structural diversity, and a healthy diverse understory vegetation community. Bauhus et al. (2009) states that these elements are gradually lost in the homogenous structure of managed stands. They discuss silvicultural strategies to maintain old growth attributes:

Large Woody Debris

Variable retention has been shown to increase the availability of downed wood and promote microclimatic conditions suitable for epixylic bryophytes (Arsenault et al. 2012). Furthermore, increased soil nitrogen with heavy debris retention or bole-only harvest is beneficial to tree growth later in development (Slesak et al. 2010). Dispersed retention (retaining trees spread out throughout the cutblock) often maintains a higher level of LWD after harvest, and therefore is preferable to organisms relying on LWD habitat; those with low mobility and small home ranges (Bauhus et al. 2009).

Snags

Creating snags within cutblocks increases structural diversity and habitat for a variety of species such as insects and cavity-nesting birds (Kroll et al. 2012). Kroll et al (2012) found that as mature forest declined from 58% to 13% after 40 years, use of created snags increased from 7% to 17%. Species that depend on dead wood, or saproxylic organisms, prefer dispersed retention as it creates more habitat (Bauhus et al. 2009).

Big trees

In the DEMO study in Oregon and Washington, dispersed retention significantly changed the diameter distribution of trees due to selective retention of large trees (Maguire et al. 2007). Current literature shows that long rotations are needed to retain big trees (Bauhus et al. 2009) and that the species of retained trees matters (Rosenvald and Lohmus 2008).

Vertical and horizontal structural diversity

Stand structure accounted for most of the variability in abundance of birds and small mammals in a study comparing regenerating forest to uncut forest (Hughes St-Laurent et al. 2007). The DEMO study in Oregon and Washington found that although dispersed retention maintains a higher mean density, due to the selective retention of large trees, aggregated treatments maintain a greater level of vertical structure and complexity as well as reduce windthrow, facilitate slash burning, and reduce overstory competition (Maguire et al. 2007; Bauhus et al. 2009).

Healthy diverse understory vegetation community

More than 10% green tree retention is beneficial in maintaining understory communities more similar to uncut forests (Craig and Macdonald 2009). The pattern of retention makes little difference for vegetation communities. However, 1 ha aggregates do provide short-term refugia for certain vegetation such as bryophytes and late-seral herbs. These benefits would be expected to decline with time for species that respond negatively to edge effects such as increased light, wind, and temperature (Aubry et al. 2009).

Relevant EBM LUOs

Section 7.(3) Within a cutblock, for the first 15% of the pre-harvest stand retained in stand level retention, as specified in section 16(1), design aggregate and dispersed stand retention to maintain a range of diameters of mature and old western red cedar and yellow cedar representative of the pre-harvest stand.
Section 16.(1) Maintain forest structure and diversity at the stand level:
(a) By establishing stand retention equal to or greater than 15% of the cutblock; and
(b) In cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.
Section 16. (2) To the extent practicable, include the following within stand retention:
(a) Habitat elements important for species at risk, ungulate winter range, and regionally important wildlife;

(b) Representation of ecosystems and plant communities that are red-listed or blue-listed in the watershed and landscape;
(c) Functional riparian forest adjacent to active fluvial units, forested swamps, fen and marsh wetlands and upland streams with unique climate and other characteristics;
(d) Western red cedar and yellow cedar, in a range of diameters representative of the preharvest stand, and important for future cultural cedar use; and
(e) Wildlife trees and coarse woody debris.

Recommendations and Revisions

1. "Regionally important wildlife" has been defined and steps are being taken to research habitat needs and map critical habitat for these species (See Question 3a.iv.). However, further research is needed to determine whether the habitat elements represented by regionally important wildlife are broad enough to encompass the needs of other species. In other words, the brown creeper is an important component in the ecosystem, but are its needs also included in the LUOs? How do we ensure that the necessary elements of all species are maintained?

2. Consider pattern of retention (aggregated versus dispersed) and linkage areas (Olson and Burnett 2009) as there are associated obstacles for future forestry operations and safety concerns as well as spatial requirements to retain certain habitat elements (Bauhus et al. 2009; Bunnell 2008).

3. A mixed approach to variable retention is recommended (Bunnell 2008).

Considerations for Experimental Watershed Design

1. An outstanding research question for managing for old-growth concerns the quantity, spatial arrangement and temporal dynamics of forest structural attributes required to meet various management objectives (Bauhus et al. 2009).

2. What is the current average use of natural snags? (Kroll et al. 2012).

3. Responses to retention are species specific, making it critical to define what species are considered 'regionally important wildlife' and research their associated important habitat elements.

iii. Does 15% retention retain important structure or ecological diversity?

Summary of Knowledge

All current research is in agreement that 15% retention is not enough to retain important structure or ecological diversity in coastal temperate rainforests. A review of all literature from the DEMO study concluded that 15% retention was not significantly different from clearcuts (Aubry et al. 2009). Current guidelines of 3-10% retention levels in Michigan, Washington, are not enough to maintain mature-associated songbirds (Otto and Roloff 2012). Similarly, more than 10% retention was required to maintain understory vegetation more similar to uncut forests (Craig and Macdonald 2009). See above section (Question 4a.i.) for the effect of different retention levels on small mammals, birds, amphibians, and vegetation.

The broad context of natural disturbance regimes can also help to clarify the reasons why 15% retention is not enough in temperate rainforests. Coastal BC is dominated by small-scale gap dynamics, with low to moderate intensity fire intervals of 750-1000 years (Daniels and Grey 2006), whereas parts of the Canadian boreal forest can have large stand replacing fires every 200 years. Coastal temperate forests, although sometimes ravaged by wind storms, are often undisturbed for very long periods of time, creating the habitat elements mentioned above (CWD, snags, big trees, etc). These habitat elements are not retained in great enough quantities with 15% retention to retain the species that depend on them (Bauhus et al. 2009).

Relevant EBM LUOs

Section 16.(1) Maintain forest structure and diversity at the stand level:
(a) by establishing stand retention greater than or equal to 15% of the cutblock

Recommendations and Revisions

1. The retention limit set in Section 16. 1(a) should be revisited. The literature agrees that most species, especially late-seral species, decrease in abundance and richness with less than 15% retention.

Considerations for Experimental Watershed Design

1. The level of retention should be included as a management treatment at both the stand and landscape scale for analyses of how retention affects biodiversity. Fenger et al. (2009) suggest that the level of retention should be assigned as categorical treatments (e.g. high vs. low retention). We recommend that the level of retention be considered as a continuous variable if many stands and watersheds are included in the analysis.

iv. In cutblocks 15 hectares or greater in size, what does distributing 50% of the retention within the cutblock do for structure and diversity?

Summary of Knowledge

The literature does not address the difference of treatments within different sizes of cutblocks specifically, however it does discuss aggregated versus dispersed retention and how these patterns of retention impact forest structure and diversity. Generally, the literature suggests that aggregated retention maintains both vertical structure and complexity and habitat elements of mature forest better than dispersed retention (Maguire et al. 2007; Bunnell 2008; Rosenvald and Lohmus 2008). On the other hand, dispersed retention may selectively retain larger trees and more CWD (Bauhus et al. 2009). Dispersed retention may also benefit fungi dispersal but has not yet been studied (Rosenvald and Lohmus 2008).

Other studies did not find a large effect of retention pattern, only that 1ha aggregates provided refugia for bryophytes and late-seral herbs, where dispersed retention did not (Aubry et al. 2009). Even these refugia were noted as short-term, since edge effects such as light, wind, and temperature were expected to cause decline in these species over time (Aubry et al. 2009). This addresses the importance of siting retention *within* the block (see below Question 4b.i.). Furthermore, some preliminary research suggests that the size of the retained patch may be more important than the overall level of retention; this may be an important question to pursue (Preston and Harestad 2007).

Relevant EBM LUOs

Section 16.(1) Maintain forest structure and diversity at the stand level:
(b) in cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.
Section 16.(2) To the extent practicable, include the following within stand retention:
(a) Habitat elements important for species at risk, ungulate winter range, and regionally important wildlife;
(b) Representation of ecosystems and plant communities that are red-listed or blue-listed in the watershed and landscape;
(c) Functional riparian forest adjacent to active fluvial units, forested swamps, fen and marsh wetlands and upland streams with unique climate and other characteristics;
(d) Western red cedar and yellow cedar, in a range of diameters representative of the preharvest stand, and important for future cultural cedar use; and
(e) Wildlife trees and coarse woody debris.

Recommendations and Revisions

1. This is an important Order from a management perspective since maintaining retention *within* the cutblock restricts the ability for companies to use adjacent stands as “retention.” Although, the specificity is curious, when the literature has neither examined this question in detail nor come to conclusion on a specific size, amount, and configuration of retention best suited to “maintain forest structure and diversity at the stand level.”

2. The literature suggests that, although both aggregate and dispersed retention have certain benefits, aggregate retention may maintain forest structure and diversity better than dispersed retention. If so, these subsections should be amended to reflect this knowledge.

Considerations for Experimental Watershed Design

1. More research is needed to answer this question. A number of permutations of cutblock size, amount and configuration of retention should be used in order to reach a reasonable conclusion as which best maintains forest structure and diversity.

b. What constitutes "excellent retention" from an ecological perspective?

Summary of Knowledge

Characteristics of stand-level retention necessary to be considered "excellent retention" are discussed in Section 3 of Kremsater et al. (2008). They suggest that to be considered "excellent retention,"

- In-stand retention should be greater than 30%, match the site series and species composition of the pre-harvested stand, and match the structure of the pre-harvested stand, or be biased towards larger structures;
- Must be within the block, or adjacent to the block, and mapped and permanent, and;
- Must not be double-counted.

In regards to the first point, other literature examining effects of retention levels on species abundance would argue that 30% retention is still too low for many late-seral associated species (refer to Question 4a). Sensitive species such as the brown creeper need 70% or more, and certain arthropods need more than 40% (Halaj et al. 2008). We did not find any further literature discussing this issue.

Relevant EBM LUOs

Section 16.(1) Maintain forest structure and diversity at the stand level:
(a) by establishing stand retention greater than or equal to or greater than 15% of the cutblock
(b) in cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.

Recommendations and Revisions

1. In-stand retention of 15% is not sufficient to be considered “excellent retention;” Kremsater et al. (2008) would recommend 30%, however other literature recommends greater than 50% for certain sensitive species.

2. “Excellent retention” as defined in Kremsater et al. (2008), does not follow from more recent findings of retention effectiveness (see Question 4a). Although current literature does agree that ‘greater than 30%’ retention is needed to maintain ecological integrity, it is also true that the same literature would suggest perhaps greater than at least 50% should be maintained.

Considerations for Experimental Watershed Design

1. See *Considerations for Experimental Watershed Design* no. 1 under Question 4a.iii, above.

2. In experimental watershed analyses, it may be necessary to consider not only the amount of retention, but also the quality of that retention, for predictions of how forest management affects biodiversity. One suggestion is to compare the forest structure (e.g. size-abundance relationships) using LiDAR technology before and after harvesting to directly measure the quality of retention.

c. Are current recommended accounting strategies for stand level retention contributions to landscape level ecosystem representation targets valid? And are there combinations of stand level and landscape level retention levels that pose high risk to species populations within watersheds/ landscapes?

Summary of Knowledge

Current recommended accounting strategies for stand-level retention contributions to landscape-level representation are given by Kremsater et al (2008). They suggest that when “excellent retention” conditions are met:

- Retention between 30 and 80% should count at half or quarter value (i.e. 2 ha of 50% retention = 0.5 ha of old forest), and
- Retention between 80 and 100% counts as proportional value (i.e. 1 ha of 80% retention = 0.8 ha of old forest).

We did not encounter any literature to suggest that these targets are not valid.

Combinations of stand level and landscape level retention levels that are too low, or double counted, could pose risks to species. Above, we discussed the various definitions of risk associated with species. High risk is defined in the literature as 15% retention, and low risk as retention that exceeds 70% at the stand scale (Aubry et al. 2009; Vanderwel et al. 2009; Le Blanc et al. 2010; Holloway et al. 2012; Otto and Roloff 2012). High risk would be particularly evident if low retention levels are used across a large portion of a landscape or watershed.

Relevant EBM LUOs

Section 16.(1) Maintain forest structure and diversity at the stand level:
(a) by establishing stand retention greater than or equal to or greater than 15% of the cutblock
(b) in cutblocks 15 hectares or greater in size, by distributing 50% of the stand retention within the cutblock, except in second growth stands where a windthrow hazard assessment indicates a high biophysical hazard for windthrow.

Recommendations and Revisions

1. Accounting strategies as indicated by Kremsater et al (2008) would ensure low risk to species populations and should be integrated into the Land Use Orders.

Considerations for Experimental Watershed Design

1. No literature currently exists that compares different combinations of stand-level and landscape-level retention. This could be explored with the Experimental Watersheds Programme.

d. What impact do stand-level retention targets have on western redcedar regeneration and growth on the landscape?

Summary of Knowledge

Klinka and Brisco (2009) conducted a comprehensive literature review on Western redcedar (*Thuja plicata*), which provides extensive information that is relevant to this question. This report should be incorporated into the research design. Briefly, the review states that Western redcedar has a high potential for natural regeneration in the open, and less so in low light conditions. When regeneration failures occur, they are not due to low seed production. Rather, they are often due to mortality during germination, related to the seed-bed quality and microclimate. Further, the review simply states that clearcutting, variable retention, patch-cutting, and group selection are all appropriate methods for facilitating the establishing the growth of redcedar. There does not seem to be further literature within this review discussing retention effects on redcedar regeneration and growth.

The report by Utzig and Holt (2009) provides an overview of factors influencing Western redcedar in the past, present, and future on Haida Gwaii. Supplies of old growth Western redcedar and yellow cedar have been drastically reduced over the past two centuries. Regeneration of redcedar has been severely affected by deer browsing on Haida Gwaii. This report provides examples of past research projects on redcedar, and provides some ideas about future studies.

The Forest Practices Board (FPB) conducted an investigation into “High Retention Harvesting” on the coast of BC and produced a report in 2007 (updated in 2009). A large proportion of sampled sites showed that high value redcedar was targeted. The FPB found that stand productivity and the associated timber supply were adversely affected in over half of the sites. A background report to this investigation by Symmetree Consulting Group Ltd (2008) provides an overview of issues arising from the management of Western redcedar under varying retention levels. They report that different management techniques should be used depending on the desired wood characteristics (less taper, limited lower branches, and durability). The following findings were noted:

- Opening size is important. Low light levels increase lower branches and a tapered bole;
- Thinning from below, rather than removal of overstory, increases growth on stem wood rather than high branches and a spreading crown;
- Recommend high initial stocking levels (>2000sph) with delayed density control;
- When natural regeneration is used, much of the regeneration will be vegetative and have internal rot;
- In mixed stands, redcedar will most likely be outcompeted by western hemlock; and
- High quality timber can not be expected for a minimum of 140 years.

This report also provides a useful decision tool for management.

Relevant EBM LUOs

Section 7.(1) Maintain a sufficient volume and quality of western red cedar and yellow cedar to support the applicable First Nation's cultural cedar use of western red cedar and yellow cedar, to the extent practicable.
Section 7.(3) Within a cutblock, for the first 15% of the pre-harvest stand retained in stand-level retention, as specified in section 16(1), design aggregate and dispersed stand retention to maintain a range of diameters of mature and old western red cedar and yellow cedar representative of the pre- harvest stand.

Recommendations and Revisions

1. The phrase "sufficient volume and quality" in LUO Section 7.1 should be clarified.

2. Need a better understanding of the rationale for why only the first 15% of pre-harvest stand retention should maintain representative redcedar and yellow cedar.

Considerations for Experimental Watershed Design

1. Examine the extent to which redcedar and yellow cedar regeneration is being affected by deer browse.

2. Identify the biophysical factors that influence redcedar and yellow cedar regeneration and growth. Examine whether growth forms needed for First Nation cultural purposes (eg. monumental cedar) are being promoted under current silviculture and retention strategies.

Summary of Recommendations

We divide our recommendations into three main sections: 1) Hydroriparian ecological integrity; 2) Terrestrial landscape ecological integrity; and 3) Broad experimental watershed considerations.

Hydroriparian ecological integrity

Maintain stream and riparian function

"Properly functioning condition" is defined by the B.C. Government FREP program (summarized in Tschaplinski and Pike 2010) as the ability of a stream and riparian area to:

- Withstand normal peak flood events without experiencing accelerated soil loss, channel movement, or bank movement;
- Filter runoff;
- Store and safely release water;
- Maintain connectivity of fish habitats;
- Maintain an adequate riparian root network and LWD supply; and
- Provide shade and reduce bank microclimate change.

The literature suggests that to maintain ecological integrity, the definition of the term 'functional' in functional riparian forest as defined by the Central and North Land Use Orders should be expanded to include those outlined above. Consideration should also be made for including a broader component of biodiversity other than fish to maintain riparian function. Research needs to be conducted on the feasibility of using this expanded definition of functional riparian forest.

Impacts to stream and riparian function can be substantially reduced if riparian practices:

- Limit introduction of logging debris and sediment into channels;
- Limit physical contact with streambanks and streambeds when falling and yarding around streams; fall and yard trees away from the channel wherever possible; and

- Retain more vegetation around S4 to S6 streams (Tschaplinski and Pike 2010).

Do a more complete analysis of the FREP program data

The Government of BC should prioritize a more thorough analysis of the FREP program data. With so many streams assessed (>1400), there is substantial power to build across-watershed predictive models of how forest harvesting treatments and environmental features of watersheds interact to affect stream and riparian function. The analysis could provide significant insight into how an experimental watersheds program could be designed for the central coast, although findings will be most relevant to the watersheds in the B.C. interior where the surveys have been conducted.

Increase retention in headwater streams

The literature highlights that retaining more vegetation, including wider buffers, in S4 to S6 streams decreases adverse effects to stream functioning (reviews in Moore and Richardson 2010, Tschaplinski and Pike 2010; Winkler et al. 2010). Forestry-related impacts on streams and aquatic habitats can occur over two decades or more, particularly where impacts are related to mass wasting events in the headwaters and propagated over time down the stream channel network (Tschaplinski and Pike 2010). Thus, riparian reserve zones for S1 to S3 streams are not always sufficient to protect these streams from the negative impacts of harvesting in the headwaters. A specific priority should be to retain more vegetation beside S4 to S6 streams that are a part of the drainage network of important fisheries watersheds. This is particularly relevant since increased headwater protection often presents operational challenges and, therefore, increased costs for forest companies. Therefore, one of the highest research priorities in forestry-freshwater research is to investigate the short and long-term responses to different retention and harvest rates in headwater ecosystems and class IV and V terrain (Clapcott and Barmuta 2010; Tschaplinski and Pike 2010; Moore and Richardson 2012). In addition, the LUOs classification of S4 streams as “upland” streams should be re-evaluated based on data from this research. These streams are associated with fish and, sometimes, lower positions within watersheds compared to S5 and S6 streams.

Increased retention in headwater areas leads to an interesting operational question: if small streams are managed more conservatively, can some harvesting opportunities be increased by re-allocating riparian retention from reserve zones of larger streams to headwater streams?

Refine Equivalent Clearcut Area to reflect major environmental gradients

Hydrologic recovery will vary considerably within and across watersheds on the central and north coast of B.C. For example, we predict a strong southwest to northeast gradient in the elevational impacts of forest harvest on the flow regime as systems move from rain, to rain-on-snow, to snow and glacial melt dominated runoff. Significant impacts to stream structure and function have been found when equivalent clearcut area (ECA) is lower than 20% (e.g. Hudson 2002). Thus, the ECA framework would benefit from integrating the diverse factors that affect stream flow such as climate, vegetation, elevation, topography, soil types, presence of lakes, glaciers, etc. (e.g. Hudson and Horel 2007; Winkler et al. 2010). We recommend that more specific guidelines for "hydrologically effective greenup" be set in the Land Use Orders that reflect major environmental gradients such as climate, elevation, topography and watershed size. Determining these guidelines will require large-scale and long-term research.

There is no easy answer when it comes to buffer widths

Other than preserving whole watersheds, there is no one buffer width that is going to maintain all aspects of stream and riparian function, including biodiversity. Strategies for riparian planning really depend on tradeoffs between different objectives. For example, one approach is to apply a mix of riparian buffers depending on function: 10m for bank stability, 15–30 m for water quality and habitat attributes, and 40–100 for riparian-dependent species (Broadmeadow and Nisbet 2004; Olson et al. 2007; Olson and Burnett 2009; Marczak et al. 2010). For obligate riparian-associated species, dispersal-limited species, and interior old-growth-associated species of special concern (e.g. the Coastal tailed frog), riparian buffers may even need to be greater than 100 meters wide (e.g. Boucher et al. 2011). A few considerations:

- The literature highlights the need for large scale studies to investigate the impacts of different riparian buffer reserve and management zone widths around stream channels (e.g. Cockle and Richardson 2003; Deschenes et al. 2007).
- Treatments should include varying buffer widths around streams of different size and location in the drainage network, in particular S4 to S6 streams.
- Incorporating linkage areas between riparian buffers better preserves habitat connectivity for species dependent on both stream and forest (Olson et al. 2007).

Similarly, an agglomerated block strategy (Boucher et al. 2011) has been shown to increase the effectiveness of standard linear riparian buffers.

- One and a half times tree height is not always going to be effective in maintaining the diversity of riparian reserve functions, particularly when the dominant trees are 20 m or less in height.
- For basic physical riparian functions, forest management should consider minimum riparian reserve zone buffers of 30 m. Currently, S1-S3 streams will have this level of protection but S4-S6 streams usually will not.
- The distance of edge influence (DEI) could be a good metric to incorporate into planning riparian buffer widths (Boucher et al. 2011).

Expand definition of ‘important fisheries watershed’

In BC, the foundation for determining riparian practices is largely fish-based, especially on salmonids, yet there is no scientifically sound basis for managing riparian and aquatic values on the presence of game fish alone (Price et al. 2009; Tschaplinski and Pike 2010). For example, is it only important fisheries watersheds that should keep ECA’s less than 20%? What defines an important fisheries watershed? Almost all small streams on the Central Coast without impassable barriers support anadromous salmon (Harvey and MacDuffee 2002; Price et al. 2008), which provide a portfolio of salmon population diversity (e.g. Schindler et al. 2010) of relatively unknown significance for regional fisheries and ecosystem processes. Furthermore, stream-associated amphibians are generally strongly dependent on both the stream and forest for completion of their life cycles and research continues to support that monitoring their populations provides a better index of riparian forest functioning and biodiversity (Olson et al. 2007).

Large-scale experimental watershed study

One of the fundamental findings of our literature review is that the impacts of forest management practices on stream and riparian function vary considerably across the diverse environmental gradients that are present both within and across watersheds. For example, it is notoriously difficult to make inferences about hydrological processes from paired-watershed experiments because storms in control and treatment watersheds do not always coincide in time, duration, intensity, or spatial extent (Alila et al. 2009). In contrast, meta-analyses, as well as information-theoretic, Bayesian, and mixed-effects analyses across watersheds offer compelling advantages to single- or paired-watershed studies. We recommend a large-scale across watershed study of the effects of forest harvesting on stream and riparian functioning on the Central Coast of BC where EBM is being implemented. A few key points on the design of this study:

- We recommend a minimum of 50 S1 to S3 streams and 100 to 1000 S4 to S6 streams, with differences in the intensity and frequency of surveys depending on stream type.
- These experimental watersheds should be sited across the range of watershed types from the low hypermaritime to interior high-elevation systems on the central coast. The experimental design should also consider spatial and temporal harvesting intensity such as landscape level representation targets and different levels of stand retention.
- This design is an expansion on what Fenger et al. (2009) recommend for this ecosystem. Fenger et al. (2009) provide suggestions for many ‘treatments’ or ‘fixed-factors’, including the range of habitat conditions. We support many of their ideas but suggest that a more continuous variable or ‘covariate’ approach be used.
- This design will allow for quantitative modeling of the effects of forest harvesting on stream functioning across the range of watershed attributes. This includes how habitat variables interact with forest harvesting treatments and with each other to mediate impacts to watershed function (see Hocking and Reynolds 2011; Sheldon et al. 2012).
- Flow and temperature gauges should be installed into all 50 S1 to S3 streams and a portion of S4 to S6 streams.
- A network of weather stations should be established.
- An important variable to consider will be the background levels of disturbance in a given watershed. One method to consider as an index of debris flow susceptibility is the “Melton ruggedness number” (Wilford et al. 2005b).
- Monitoring of watershed hydrology should be coupled with standardized monitoring of stream and riparian structure and functioning.

Terrestrial landscape ecological integrity

Management suggestions for stand- and landscape-scale retention

Stand-level retention can be as low as 15% as defined by the LUOs. However, most species, especially late-seral species, decrease in abundance with 15% retention (e.g. Kremsater et al. 2008). This indicates that the retention limit set in LUO Section 16. 1(a) should be revisited. The literature supports definitions of high risk at the stand scale as 15% retention, and low risk as retention that exceeds 70%. Kremsater et al. (2008) recommend a minimum stand-level retention of 30%. The level of stand-level retention required for certain ecosystem functions is dependent on the level of landscape-level retention. Unfortunately, there is little data on this topic in the literature, highlighting the need for large-scale research. Landscape-level retention in the LUOs is currently based on maintaining ecosystem representation; however, stand-level retention is not similarly linked to ecosystem representation targets other than in the context of cedar forest. Additionally, most literature does not discuss stand-level retention in the context of ecosystem representation. Due to BC's BEC classification system, a unique opportunity exists to link stand-level retention studies to BEC Site Series in order to measure landscape-scale representation. A few additional points for consideration:

- At the landscape scale, consider connectivity of channels to uplands, headwaters to ridgelines, and linkage areas.
- The amount of retention has generally been found to be more important than the spatial arrangement of retention (Rosenvald and Lohmus 2008; Aubry et al. 2009). However, configuration effects may be quite important for some species, particularly those that have low mobility (Harrison et al. 2012) and/or are sensitive to disturbance (Aubry et al. 2009).
- Smith et al. (2011b) make two broad management recommendations related to habitat configuration: 1) If fragmentation effects are only apparent at some scales then these should be managed at that scale; and 2) If fragmentation has positive effects at one scale but negative effects at another, then managing fragmentation may not be the most efficient use of conservation resources.
- Habitat quality is a key predictor of species distributions; therefore, not accounting for it in studies of species responses to habitat loss and configuration may lead to inaccurate conclusions (Bollmann et al. 2011; St. Laurant et al. 2011).

Plan harvesting to emulate natural disturbance

LUO targets focus on representing forests based on the Range of Natural Variability (RONV) within ecosystems. How could forest management emulate natural disturbance regimes in coastal forests of the Great Bear Rainforest? An experimental watersheds program should include a number of "control" watersheds that exist in conservancies and do not have a significant history of harvesting. This will allow comparisons between treatment watersheds at different levels outside of RONV and streams that are only experiencing natural range of variability in hydrologic functioning. More flexible/variable harvesting including selective harvesting in riparian zones may better approximate disturbance patterns (Kardynal et al. 2009). An assessment should be carried out to understand the state of knowledge and level of uncertainty associated with the RONV in each BEC zone. For example, there is more research on the disturbance regime in hypermaritime areas of BC's coast compared to the very dry areas dominated by Douglas fir.

Habitat thresholds may not be practical for managing biodiversity

Policy makers and land managers are increasingly applying the threshold concept in setting targets for habitat representation. However, identifying the thresholds associated with levels of critical habitat for different species is an ongoing area of study. Thresholds are dependent on a number of factors (e.g., species, habitat quality, habitat configuration) and are frequently calculated using different methods. Therefore, without species- and context-specific objectives, thresholds are likely not useful metrics for preserving biodiversity. Perhaps most importantly, as currently employed, thresholds identify the point where a substantial number of species are lost from the landscape, whereas the focus should be on maintaining habitat area at the point where species are able to maintain viable populations for many generations (Lindenmayer and Luck 2005).

Decide on formal definition of “risk”

Policy makers, managers, and stakeholders should always clearly state what they mean by the term “risk”. As calculated in risk assessment literature, “risk” is the product of two components: 1) the magnitude of the undesirable consequences that arise from uncertain events, and 2) the probability of those undesirable consequences occurring (Peterman 2004). Currently, such explicit calculations of ecological risk and risk assessment are not incorporated in the EBM LUOs. Instead, the EBM LUOs rely largely on the RONV concept to determine whether the level of undesirable consequences from habitat loss is acceptable or not. However, RONV inadequately captures risks to biodiversity. It may not accurately predict effects on species persisting in habitats that have been physically altered beyond the RONV. The relevance of the RONV standard is further called into question in the context of intermediate disturbance. Whereas sensitive species show declines even at intermediate levels of forest cover, some species will experience positive responses at intermediate levels of forest cover due to increased habitat heterogeneity. Linking changes in species responses to measures of cumulative landscape disturbance, rather than just %RONV or amount of forest cover, will likely provide a more relevant measure of risk.

Conduct landscape scale study across experimental watersheds

The literature highlights the need for large-scale studies to investigate the relationship between landscape-scale processes and species responses (Cushman and McGarigal 2003; Cushman et al. 2012). Such a study on the central coast would be very valuable, since landscape-scale studies are consistently rare in the literature relating to habitat loss, fragmentation, thresholds, and forest retention. Some considerations for design:

- Cushman et al. (2012) identified that patch cohesion, correlation length and aggregation index are strong predictors of genetic differentiation and are therefore among the best metrics for studies aiming to quantify the impacts of habitat configuration on movement and gene flow.

- A landscape scale study requires measuring species responses across multiple, independent, non-overlapping landscapes, each of a size relevant to the life histories of the studied species (Jackson and Fahrig 2012; Eigenbrod et al. 2011).

- Fungi, lichens, bryophytes are among the most sensitive to logging. Monitoring these taxonomic groups is likely to provide sensitive tests of the impacts of the spatial arrangement of logging at both stand and landscape scales (Boucher et al. 2011; Halpern et al. 2012).

- The level of retention should be included as a continuous variable at both the stand and landscape scale for analyses of how retention affects biodiversity. Fenger et al. (2009) have good suggestions for the design and analysis of a landscape scale experimental watersheds study. However, we recommend a more continuous variable approach.

- Link the design and analysis of landscape-scale study to the across-watershed study focused on harvesting impacts to stream and riparian function. For example, what is the relationship between ECA, % functional forest, and landscape level habitat?

- Establish five long-term monitoring plots in each of the 50 watersheds.

- What is the relationship between TEM and LiDAR data? Integrate data from long-term monitoring plots with remote sensing data such as LiDAR.

- Use LiDAR technology to map and create metrics for the whole central coast. For example, LiDAR will give tree size distributions for watersheds or stands within watersheds. What is the relationship between stand structure (i.e. size-abundance relationship) and biodiversity?

- Set up an array of wildlife cameras to monitor key species (e.g. grizzly bears).

- Set up long-term bird monitoring programs in key watersheds, with a focus on sensitive species (e.g. Marbled murrelet, Northern goshawk, Brown creeper). More research is necessary to identify active nesting territory for both the Marbled Murrelet and Northern Goshawk.

- Conduct specific studies (supported by graduate students) on species of concern (e.g. tailed frog).

- Partner with First Nations to monitor species of concern. Create monitoring programs that build capacity and offer employment to First Nation communities

Broad experimental watershed considerations

Long-term experimental watershed study

There is a strong call from recent research to move away from site-specific studies to more landscape or watershed-scale analyses that span longer time periods. This will enable tests of the full impacts of forestry on watershed function and landscape ecological integrity (e.g. Martel et al. 2007; Stephenson and Morin 2009; Clapcott and Barmuta 2010; Richardson et al. 2012). For example, channel recovery after disturbance can take over 30 years (Hogan and Luzi 2010; Mallik et al. 2011). The length of the study should also span several times the generation time of the species of concern (e.g. a five year study is not adequate to detect a population decline in an amphibian species with a ten year lifespan). Understanding the potential effects of climate change will also be a key component of any long-term study. This means that adequate, long-term funding is critical to the success of this project.

Partnership with First Nations

The BC Ministry of Forests, Coast Forest Region should strongly consider a partnership with the Central Coast First Nations (www.ccira.ca) in monitoring priority watersheds on the Central Coast including the Heiltsuk, Kitasoo/Xai'Xais, Nuxalk and Wuikinuxv Nations. These coastal First Nations are participating in a Coastal First Nations program called the Coastal Guardian Watchmen Network (www.coastalguardianwatchmen.ca) that is developing a standardized regional monitoring system for streams, salmon and forest practices. These data protocols are similar to but simpler than the FREP system but could be integrated with government protocols (Figure 9). A critical piece will be to identify key personnel from each community to help lead monitoring nodes in high priority local watersheds (possible target = 10-15 S1 to S3 streams per community). Partnerships with First Nations should support broad capacity building within each community and allow for better integration of First Nation perspectives on how EBM should be implemented (e.g. human well-being). This relationship should provide increased opportunities for long-term funding for both partners.

Stream Transect (v.2) **Stream Name:**________________ **Reach #**_____ Entered in DMS:_______

Transect #	Location (m)	BF width (m)	Wet width (m)	BF depth (cm)	Max depth (cm)	Habitat type
						☐pool ☐log jam ☐riffle ☐cascade ☐glide ☐other:
Canopy closure (m)		Moss on rocks		Undercut bank?		
Left: Right:		☐little/no ☐occasional ☐frequent ☐abundant		Left: ☐Yes ☐No	Right: ☐Yes ☐No	
Pebble size: int axis (cm)						
Transect #	Location (m)	BF width (m)	Wet width (m)	BF depth (cm)	Max depth (cm)	Habitat type
						☐pool ☐log jam ☐riffle ☐cascade ☐glide ☐other:
Canopy closure (m)		Moss on rocks		Undercut bank?		
Left: Right:		☐little/no ☐occasional ☐frequent ☐abundant		Left: ☐Yes ☐No	Right: ☐Yes ☐No	
Pebble size: int axis (cm)						
Transect #	Location (m)	BF width (m)	Wet width (m)	BF depth (cm)	Max depth (cm)	Habitat type
						☐pool ☐log jam ☐riffle ☐cascade ☐glide ☐other:
Canopy closure (m)		Moss on rocks		Undercut bank?		
Left: Right:		☐little/no ☐occasional ☐frequent ☐abundant		Left: ☐Yes ☐No	Right: ☐Yes ☐No	
Pebble size: int axis (cm)						

Figure 9. ***Current Coastal Guardian Watchmen Network stream transect card (front only). There are five standardized data cards associated with streams for use across the central and north coast First Nations: a) Stream Assessment card designed to monitor habitat, and intended mainly for S2 and S3 streams; b) Stream Transect card designed to be paired with the Stream Assessment card and focused on measuring variation within reaches; c) Stream Visit card focused on water quality and quick visit data collection; d) Spawner Survey card focused on salmon and is similar to DFO Stream Inspection Log; e) Forest cards, which are still being designed, but could be prioritized for assessments of S4 to S6 streams during mandatory site visits with forest companies.***

Build broad stakeholder support

Due to the high costs of such an ambitious across-watershed monitoring and research project, The BC Ministry of Forests, Coast Forest Region, should continue to build coalitions and working groups across numerous stakeholders in addition to First Nations. Possibilities for partnership include with Federal Departments such as DFO and Environment Canada, forest companies, and academia (e.g. the Hakai Network for Coastal People, Ecosystems and Management (http://www.sfu.ca/hakai.html). Academic partnership will be a key way to share the cost burden of such a project, as well as to advance and communicate research results. The Hakai Beach Institute (Tula Foundation) located in the hypermaritime zone on Calvert Island will also be a key partner (www.hakai.org), and could represent a fifth node on the central coast, plus a location for training and technological development.

Conduct economic and human well-being analyses

An important consideration for human well-being is the social and economic opportunities created through forestry activities. We do not discuss many of the tradeoffs present for the economic viability of forestry companies if more rigorous ecological standards are required under EBM. Similarly, we do not discuss the social and economic opportunities that may be created with more rigorous ecological standards. We strongly recommend that economic analyses be conducted to analyze the tradeoffs of different forest practices, including the possibilities surrounding carbon credits, more local processing of wood products, and tourism. With more rigorous ecological standards, and challenges for forest companies in terms of economic viability, the BC Government should consider supporting the development of community-based forest economies in the Great Bear Rainforest.

Consider marine-terrestrial interactions

Many of these questions relate to freshwater-terrestrial interactions, but none relate to marine-terrestrial interactions. Examining connections between the ocean and terrestrial environments is important, including how forestry may affect the ecological role of salmon in streams or the export of carbon and other nutrients to the marine food web. In another example, substantial thought is going into the design of riparian reserves around streams, but no mention is made of buffers along the marine interface, as was done with EBM planning in Clayoquot Sound. More integration across disciplines is necessary.

Consider modifications to the Land Use Orders

Certain LUOs are phrased quite vaguely and need to be clarified. A more thorough analysis of the differences between the Coast Information Team recommendations and the LUOs needs to be undertaken. The LUOs discuss issues around human well-being and First Nations, and these dimensions cannot be thought of in isolation from the ecological and biophysical research.

An experimental watersheds program at this scale will attract global interest

An experimental watershed program on the Central Coast would help address the global need for assessing the ecological integrity of watersheds under environmental change (Millennium Ecosystem Assessment 2005). As discussed by Arthington et al. (2010) in a Special Issue of

Freshwater Biology, river systems not only sustain diverse communities of aquatic and terrestrial biota, but provide essential ecological goods and services to humans. Promoting sustainable watershed management is key to maintaining these functions. The information gathered from experimental watersheds can increase the accuracy with which we model relationships between hydrological patterns, fluvial disturbance, and ecological responses in hydroriparian ecosystems.

References

Adams, P.W. 2007. Policy and Management for Headwater Streams in the Pacific Northwest: Synthesis and Reflection. Forest Science 53(2): 104:118.

Alila, Y., Kuras, P.K., Schnorbus, M., Hudson, R. 2009. Forest and floods: a new paradigm sheds light on age-old controversies. Water Resources Research 45. doi:10.1029/2008WR007207

Arsenault, J., Fenton, N.J., Bergeron, Y. 2012. Effects of variable retention harvest on epixylic bryophytes in boreal black spruce- feathermoss forests. Canadian Journal of Forest Research 42: 1467–1476.

Artelle, K.A., Anderson, S.C., Cooper, A.B., Paquet, P.C., Reynolds, J.D., Darimont, C.T. In review. Confronting uncertainty in wildlife management: performance of grizzly bear management in British Columbia, Canada. Conservation Letters.

Arthington A.H., Naiman, R.J., McClain, M.E., Nilsson, C. 2010. Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. Freshwater Biology 55: 1–16.

Aubry, K.B., Halpern, C.B., Peterson, C.E. 2009. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. Forest Ecology and Management 258: 398–408.

B.C. Ministry of Forest and Range. 2007. Riparian Management and Natural Function of Small Streams in the Northern Interior of British Columbia. Course Manual Prepared by: P. Beaudry & Associates Ltd. Prepared for: BC. Ministry of Forest and Range Northern Interior Forest Region, Prince George.

Baker, S.C., Grove, S.J., Forster, L., Bonham, K.J., Bashford, D. 2009. Short-term responses of ground-active beetles to alternative silvicultural systems in the Warra Silvicultural Systems Trial, Tasmania, Australia. Forest Ecology and Management 258: 444-459.

Baldwin, L.K., and Bradfield, G.E. 2007. Bryophyte responses to fragmentation in temperate coastal rainforests: A functional group approach. Biological Conservation 136: 408-422.

Banner, A., P. LePage, J. Moran and A. de Groot (editors). 2005. The HyP3 Project: pattern, process, and productivity in hypermaritime forests of coastal British Columbia – a synthesis of 7-year results. B.C. Min. For., Res. Br., Victoria, B.C. Spec. Rep. 10. URL: <http://www.for.gov.bc.ca/hfd/pubs/Docs/Srs/Srs10.htm>

Bauhus, J., Puettmann, K., Messier, C. 2009. Silviculture for old-growth attributes. Forest Ecology and Management 258: 525-537.

Beaudry, P.G. 2003. Effects of Riparian Management Strategies on the Hydrology of Small Streams in the Takla Region of British Columbia – Fourth Year Results (Part III: Small Stream Studies). In MacIsaac, E.A. (editor). 2003. Forestry Impacts on Fish Habitat in the Northern Interior of British Columbia: A Compendium of Research from the Stuart-Takla Fish-Forestry Interaction Study. Can. Tech. Rep. Fish. Aquat. Sci. No. 2509. 143-162.

Becker, D.A., Wood, P.B., Keyser, P.D., Wigley, B., Dellinger, R., Weakland, C.A. 2011 Forest Ecology and Management 262: 449-460.

Beschta, R.L., and Jackson, W.L. 2008. Chapter 4: Forest Practices and Sediment Production in the Alsea Watershed Study. In: J. D. Stednick (ed.), Hydrological and Biological Responses to Forest Practices.

Bigelow, P.E., Benda, L.E., Miller, D.J., Burnett, K.M. 2007. On Debris Flows, River Networks, and the Spatial Structure of Channel Morphology. Forest Science 53(2) : 220- 238.

Bollmann, K., Graf, R.F., Suter, W. 2011. Quantitative predictions for patch occupancy of capercaillie in fragmented habitats. Ecography 34: 276-286.

Brady, M.J., McAlpine, C.A., Possingham, H.P., Miller, C.J., Baxter, G.S. 2011. Matrix is important for mammals in landscapes with small amounts of native forest habitat. Landscape Ecology 26(5): 617-628.

Brand, L.A., and George, T.L. 2001. Response of passerine birds to forest edge in coast redwood forest fragments. The Auk 118(3):678–686

Broadmeadow, S., and Nisbet, T.R. 2004. The effects of riparian forest management on the freshwater environment: a literature review of best management practice. Hydrology and Earth System Sciences, 8(3), 286-305.

Bunnell, F.L. 2008. Indicators for sustaining biological diversity in Canada's most controversial forest type—Coastal temperate rainforest. Ecological Indicators 8:149–157.

Burger, A.E., Ronconi, R.A., Silvergieter, M.P., Conroy, C., Bahn, V., Manley, I.A., Cober, A., Lank, D.B. 2010. Factors affecting the availability of thick epiphyte mats and other potential nest platforms for Marbled Murrelets in British Columbia. Canadian Journal of Forest Research 40: 727–746.

Buttle, J.M. 2011. Ch. 33: The Effects of Forest Harvesting on Forest Hydrology and Biogeochemistry. In: D.F. Levia et al. (eds.), Forest Hydrology and Biogeochemistry: Synthesis of Past Research and Future Directions, Ecological Studies 216.

Carr, A.E., Loague, K., VanderKwaak, J.E. 2013. Hydrologic-response simulations for the North Fork of Caspar Creek: second-growth, clear-cut, new-growth, and cumulative watershed effect scenarios. Hydrological Processes. Published online in Wiley Online Library (wileyonlinelibrary.com). doi: 10.1002/hyp.9697.

Clapcott, J.E., and Barmuta, L.A. 2010. Forest clearance increases metabolism and organic matter processes in small headwater streams. Journal of North American Benthological Society 29(2):546–561.

Clews, E., and Ormerod, S.J. 2010. Appraising riparian management effects on benthic macroinvertebrates in the Wye River system. Aquatic Conservation: Marine and Freshwater Ecosystems 20: S73–S81.

Clinton, B.D. 2011. Stream water responses to timber harvest: Riparian buffer width effectiveness. Forest Ecology and Management 261: 979–988.

Clinton, B.D., Vose, J.M., Knoepp, J.D., Elliott, K.J., Reynolds, B.C., Zarnoch, S.J. 2010. Can structural and functional characteristics be used to identify riparian zone width in southern Appalachian headwater catchments? Canadian Journal of Forest Research 40(2): 235-253.

Cockle, K.L., and Richardson, J.S. 2003. Do riparian buffer strips mitigate the impacts of clearcutting on small mammals? Biological Conservation 113: 133–140.

Coe, D. 2004. The Hydrologic Impacts of Roads At Varying Spatial and Temporal Scales: A Review of Published Literature as of April 2004. Prepared for: the Upland Processes Science Advisory Group of the Committee for Cooperative Monitoring, Evaluation, and Research (CMER).

Cohen, W.B., Spies, T.A., Alig, R.J., Oetter, D.R., Maiersperger, T.K., Fiorella, M. 2002. Characterizing 23 Years (1972–95) of Stand Replacement Disturbance in Western Oregon Forests with Landsat Imagery. Ecosystems 5: 122–137.

Cole, M.B., Russell, K.R., Mabee, T.J. 2003. Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon Coast Range mountains. Canadian Journal of Forest Research 33: 1433–1443.

Compton, J.E., Church, M.R., Larned, S.T., Hogsett, W.E. 2003. Nitrogen Export from Forested Watersheds in the Oregon Coast Range: The Role of N2-fixing Red Alder. Ecosystems 6: 773–785.

Coxson, D.S., and Stevenson, S.K. 2007. Growth rate responses of Lobaria pulmonaria to canopy structure in even-aged and old-growth cedar–hemlock forests of central-interior British Columbia, Canada. Forest Ecology and Management 242: 5–16.

Craig, A., and Macdonald, S.E. 2009. Threshold effects of variable retention harvesting on understory plant communities in the boreal mixedwood forest. Forest Ecology and Management 258: 2619–2627.

Crawford, J.A., and Semlitsch, R.D. 2006. Estimation of Core Terrestrial Habitat for Stream-Breeding Salamanders and Delineation of Riparian Buffers for Protection of Biodiversity. Conservation Biology 21(1):152-158.

Cunningham, M.A., and Johnson, D.H. 2012. Habitat selection and ranges of tolerance: how do species differ beyond critical thresholds? Ecology and Evolution 2(11): 2815–2828.

Cushman, S.A., and McGarigal, K. 2003. Landscape-level patterns of avian diversity in the Oregon Coast Range. Ecological Monographs 73(2): 259–281.

Cushman, S.A., Shirk, A., Landguth, E.L. 2012. Separating the effects of habitat area, fragmentation and matrix resistance on genetic differentiation in complex landscapes. Landscape Ecology 27:369–380.

Cyr, D., Gauthier, S., Bergeron, Y., Carcaillet, C. 2009. Forest management is driving the eastern North American boreal forest outside its natural range of variability. Frontiers in Ecology and the Environment 7(10): 519–524.

D'Astous, E., Villard, M. 2012. Effects of Selection Harvesting on Bark Invertebrates and Nest Provisioning Rate in an Old Forest Specialist, the Brown Creeper *Certhia americana*. Ecoscience 192:106-112.

D'Souza, L.E., Six, L.J. Bakker, L.J., Bilby, R.E. 2012. Spatial and temporal patterns of plant communities near small mountain streams in managed forests. Canadian Journal of Forest Research 42: 260–271.

Daniels, L., and Gray, R. 2004. Disturbance regimes in coastal B.C. Forrex - Forest Research Extension Partnership 6(2): 3.

Daniels, L., and Gray, R. 2006. Disturbance regimes in coastal British Columbia. BC Journal of Ecosystems and Management 7(2): 44-56.

Daniels, L.D., Maertens, T.B., Stan, A.B., McClosky, S.P.J., Cochrane, J.D., Gray, R.W. 2011. Direct and indirect impacts of climate change on forests: three case studies from British Columbia. Canadian Journal of Plant Pathology 33(2): 108-116.

Deschenes, J., Rodriguez, M.A., Bérubé, P. 2007. Context-dependant responses of juveile Atlantic salmon (*Salmo salar*) to forestry activities at multiple spatial scales within a river basin. Canadian Journal of Fisheries and Aquatic Sciences 64: 1069-1079.

Dorner, B., and Wong, C. 2003. Natural disturbance dynamics in coastal British Columbia. Prepared for: Coast Information Team – North Coast LRMP.

Douglas, T. 2008. Groundwater in British Columbia: Management for Fish and People. Streamline Watershed Management Bulletin 11(2): 20-24.

Doyon, F., Yamasaki, S., Duchesneau, R. 2008. The use of the natural range of variability for identifying biodiversity values at risk when implementing a forest management strategy. The Forestry Chronicle 84(3): 316-329.

Drever, R.C., Peterson, G., Messier, C., Bergeron, Y., Flannigan, M. 2006. Can forest management based on natural disturbances maintain ecological resilience? Canadian Journal of Forestry Research 36: 2285-2299.

Eigenbrod, F., Hecnar, S.J., Fahrig, L. 2011. Sub-optimal study design has major impacts on landscape-scale inference. Biological Conservation 144(1): 298-305.

Ficetola, G.F., and Denoël, M. 2009. Ecological thresholds: an assessment of methods to identify abrupt changes in species habitat relationships. Ecography 32: 1075-1084.

Ficetola, G.F., Padoa-Schioppa, E., De Bernardi, F. 2008. Influence of Landscape Elements in Riparian Buffers on the Conservation of Semiaquatic Amphibians. Conservation Biology 23(1): 114–123.

Forest Practices Board. 2005. Managing Landslide Risk from Forest Practices in British Columbia. Special Investigation Report No. FPB/SIR/14.

Garrard, G.E., McCarthy, M.A., Vesk, P.A., Radford, J.Q., Bennett, A.F. 2012. A predictive model of avian natal dispersal distance provides prior information for investigation response to landscape change. Journal of Animal Ecology 81(1): 14-23.

Gavin, D.G., Brubaker, L.B., Lertzman, K.P. 2003. Holocene fire history of a coastal temperate rainforest based on soil charcoal radiocarbon dates. Ecology 84(1): 186-201.

Geertsema, M., Schwab, J.W., Jordan, P., Millard, T.H., Rollerson, T.P. 2010. Hillslope Processes (Chapter 8). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of forest hydrology and geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 213-273.

Geertsma, M., and Schwab, J.W. 2006. Challenges with Terrain Stability Mapping in Northern British Columbia. Streamline Watershed Management Bulletin 10(1): 18-26.

Gitzen, R.A., West, S.D., Maguire, C.C., Manning, T., Halpern, C.B. 2007. Response of terrestrial small mammals to varying amounts and patterns of green-tree retention in Pacific Northwest forests. Forest Ecology and Management 251: 142-155.

Grant, G.E., Lewis, S.L., Swanson, F.J., Cissel, J.H., McDonnell, J.J. 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-science report for western Oregon and Washington. Gen. Tech. Rep. PNW-GTR-760. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 76 p.

Groom, J.D., Dent, L., Madsen, L.J., Fleuret, L.J. 2011a. Response of western Oregon (USA) stream temperatures to contemporary forest management. Forest Ecology and Management 262:1618–1629.

Groom, J.D., Dent, L., Madsen, L.J. 2011b. Stream temperature change detection for state and private forests in the Oregon Coast Range. Water Resources Research 47. 12 p.

Guillemette, F., Plamondon, A.P., Prévost, M., Lévesque, D. 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. Journal of Hydrology 302: 137-153.

Gustafsson, L., Baker, S.C., Bauhus, J., Beese, W.J., Brodie, A., Kouki, J., Lindenmayer, D.B., Lohmus, A., Pastur, G.M., Messier, C., Nayland, M., Palik, B., Sverdrup-Thygeson, A., Volney, W., Wayne, A., Franklin, J.F. 2012. Retention Forestry to Maintain Multifunctional Forests: A World Perspective. Bioscience 62(7): 633-645.

Gyug, L., Hamilton, T., Austin, M. 2004. Grizzly Bear Ursus arctos. Ministry of Evironment. Accounts and Measures for Managing Identified Wildlife.

Halaj, J., Halpern, C.B., Yi, H. 2009. Effects of green-tree retention on abundance and guild composition of corticolous arthropods. Forest Ecology and Management 258: 850–859.

Halaj, J., Halpern, C.B., Yi, H. 2008. Responses of litter-dwelling spiders and carabid beetles to varying levels and patterns of green-tree retention. Forest Ecology and Management 255: 887–900.

Halpern, C.B., Halaj, J., Evans, S.A., Dovciak, M. 2012. Level and pattern of overstory retention interact to shape long-term responses of understories to timber harvest. Ecological Applications 22(8) 2049–2064.

Hampe, A., and Petit, R.J. 2005. Conserving biodiversity under climate change: the rear edge matters. Ecology Letters 8: 461–467.

Harrison, K.A., Pavlova, A., Amos, J.N. Takeuchi, N., Lill, A., Radford, J.Q., Sunnucks, P. 2012. Fine-scale effects of habitat loss and fragmentation despite large-scale gene flow for some regionally declining woodland bird species. Landscape Ecology 27(6): 813-827.

Harvey, B., and MacDuffee, M. 2002. Ghost Runs: The future of wild salmon on the north and central coasts of British Columbia. Raincoast Conservation Society. Victoria, BC.

Hassan, M.A., Church, M., Lisle, T.E., Brardinoni, F., Benda, L., Grant, G.E. 2005. Sediment transport and channel morphology of small, forested streams. Journal of the American Water Resources Association (JAWRA) 41(4): 853-876.

Hawkes, V.C., and Gregory, P.T. 2012. Temporal changes in the relative abundance of amphibians relative to riparian buffer width in western Washington, USA. Forest Ecology and Management 274: 67–80.

Hazlitt, S.L., Martin, T.G., Sampson, L., Arcese, P. 2010. The effects of including marine ecological values in terrestrial reserve planning for a forest-nesting seabird. Biological Conservation 143: 1299–1303.

Healey, S.P., Cohen, W.B., Spies, T.A., Moeur, M., Pflugmacher, D., Whitley, M.G., Lefsky, M. 2008. The Relative Impact of Harvest and Fire upon Landscape-Level Dynamics of Older Forests: Lessons from the Northwest Forest Plan. Ecosystems 11: 1106–1119.

Hocking, M., and Reynolds, J.D. 2011. Impacts of Salmon on Riparian Plant Diversity. Science 331: 1609-1612.

Hodson, J., Fortin, D., Bélanger, L., Renaud-Roy, É. 2012. Browse History as an Indicator of Snowshoe Hare Response to Silvicultural Practices Adapted for Old-Growth Boreal Forests. Ecoscience 19(3):266-284.

Hogan, D.L., and Luzi, D.S. 2010. Channel Geomorphology: Fluvial Forms, Processes, and Forest Management Effects (Chapter 10). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of forest hydrology and geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 331-371.

Holloway, G.L., Smith, W.P., Halpern, C.B., Gitzen, R.A., Maguire, C.C., West, S.D. 2012. Influence of forest structure and experimental green-tree retention on northern flying squirrel (*Glaucomys sabrinus*) abundance. Forest Ecology and Management 285: 187–194.

Holmes, K.L., and Goebel, P.C. 2011. A Functional Approach to Riparian Area Delineation Using Geospatial Methods. Journal of Forestry 109:233-241.

Homan, R.N., Windmiller, B.S., Reed, J.M. 2004. Critical thresholds associated with habitat loss for two vernal pool-breeding amphibians. Ecological Applications 14(5): 1547-1553.

Horel, G. 2006. Summary of Landslide Occurrence on Northern Vancouver Island. Streamline Watershed Management Bulletin 10(1): 9p.

Hudson, R. 2000. Snowpack recovery in regenerating coastal British Columbia clearcuts. Canadian Journal of Forest Research 30: 548–556.

Hudson, R. 2001. Storm-Based Sediment Budgets in a Partially Harvested Watershed in Coastal British Columbia. Forest Research Technical Report TR-009 (Hydrology). Vancouver Forest Region, Nanaimo BC.

Hudson, R. 2002. Effects of Forest Harvesting and Regeneration on Peak Streamflow in a Coastal Watershed. Forest Research Technical Report TR-022 (Hydrology). Vancouver Forest Region, Nanaimo BC.

Hudson, R., and Horel, G. 2007. An operational method of assessing hydrologic recovery for Vancouver Island and south coastal BC Forest Research Technical Report TR-032 (Hydrology). Vancouver Forest Region, Nanaimo BC.

Huggard, D.J., Dunsworth, G.B., Herbers, J.R., Klenner, W., Kremsater, L.L., Serrouya, R. 2006. Monitoring ecological representation in currently non-harvestable areas: Four British Columbia case studies. The Forestry Chronicle 82(3): 383-394.

Iida, S., Tanaka, T., Sugita, M. 2005. Change of interception process due to the succession from Japanese red pine to evergreen oak. Journal of Hydrology 315(1): 154-166.

Jackson, H.B., and Fahrig, L. 2012. What size is a biologically relevant landscape? Landscape Ecology 27(7): 929-941.

Jones, M.D., Twieg, B.D., Durall, D.M., Berch, S.M. 2008. Location relative to a retention patch affects the ECM fungal community more than patch size in the first season after timber harvesting on Vancouver Island, British Columbia. Forest Ecology and Management 255(3): 1342-1352.

Jordan, P. 2006. The use of sediment budget concepts to assess the impact on watersheds of forestry operations in the southern interior of British Columbia. Geomorphology 79(1): 27-44.

Jordan, P., Millard, T.H., Campbell, D., Schwab, J.W., Wilford, D.J., Nicol, D., Collins, D. 2010. Forest Management Effects on Hillslope Processes (Chapter 9). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of Forest Hydrology and Geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 275-329.

Kardynal, K.J., Hobson, K.A., Van Wilgenburg, S.L., Morissette, J.L. 2009. Moving riparian management guidelines towards a natural disturbance model: An example using boreal riparian and shoreline forest bird communities. Forest Ecology and Management 257: 54-65.

Keim, R.F., Tromp-van Meerveld, H.J. McDonnell, J.J. 2006. A virtual experiment on the effects of evaporation and intensity smoothing by canopy interception on subsurface stormflow generation. Journal of Hydrology 327(3): 352-364.

Kemp, P., Sear, D., Collins, A., Naden, P., Jones, I. 2011. The impacts of fine sediment on riverine fish. Hydrological Processes 25(11): 1800-1821.

Kennedy, R.S.H., and Spies, T.A. 2004. Forest cover changes in the Oregon Coast Range from 1939-1993. Forest Ecology and Management 200: 129-147.

Kiffney, P.M., and Richardson, J.S. 2010. Organic matter inputs into headwater streams of southwestern British Columbia as a function of riparian reserves and time since harvesting. Forest Ecology and Management 260(11): 1931-1942.

Kiffney, P.M., Richardson, J.S., Bull, J.P. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. Journal of Applied Ecology 40(6): 1060-1076.

Kissling, M.L. and Garton, E.O. 2008. Forested buffer strips and breeding bird communities in southeast Alaska. Journal of Wildlife Management 72(3): 674-681.

Klein, R.D., Lewis, J., Buffleben, M.S. 2012. Logging and turbidity in the coastal watersheds of northern California. Geomorphology 139: 136-144.

Klenner, W., and Sullivan, T.P. 2009. Partial and clearcut harvesting of dry Douglas-fir forests: Implications for small mammal communities. Forest Ecology and Management 257(3): 1078-1086.

Kreutzweiser, D., Capell, S., Good, K., Holmes, S. 2009. Sediment deposition in streams adjacent to upland clearcuts and partially harvested riparian buffers in boreal forest catchments. Forest. Ecology and Management 258(7): 1578-1585.

Kreutzweiser, D.P., Sibley, P.K., Richardson, J.S., Gordon, A.M. 2012. Introduction and a theoretical basis for using disturbance by forest management activities to sustain aquatic ecosystems. Freshwater Science 31(1): 224-231.

Kroll, A.J. 2009. Sources of uncertainty in stream-associated amphibian ecology and responses to forest management in the Pacific Northwest, USA: A review. Forest Ecology and Management 257(4): 1188-1199.

Kroll, A.J., Duke, S.D., Hane, M.E., Johnson, J.R., Rochelle, M., Betts, M.G., Arnett, E.B. 2012. Landscape composition influences avian colonization of experimentally created snags. Biological Conservation 152: 145-151.

Kupfer, J.A. 2012. Landscape ecology and biogeography rethinking landscape metrics in a post-FRAGSTATS landscape. Progress in Physical Geography 36(3): 400-420.

Lacki, M.J., Baker, M.D., Johnson, J.S. 2010. Geographic variation in roost-site selection of Long-Legged Myotis in the Pacific Northwest. Journal of Wildlife Management 74(6): 1218-1228.

Lacki, M.J., Baker, M.D., Johnson, J.S. 2012. Temporal dynamics of roost snags of Long-Legged Myotis in the Pacific Northwest, USA. Journal of Wildlife Management 76(6): 1310-1316.

Le Blanc, M.L., Fortin, D., Darveau, M. Ruel, J.C. 2010. Short term response of small mammals and forest birds to silvicultural practices differing in tree retention in irregular boreal forests. Ecoscience 17(3): 334-342.

Leach, J.A., Moore, R.D., Hinch, S.G., Gomi, T. 2012. Estimation of forest harvesting-induced stream temperature changes and bioenergetics consequences for cutthroat trout in a coastal stream in British Columbia, Canada. Aquatic Sciences 74(3): 427-441.

Lee, P., Smyth, C., Boutin, S. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70(2): 165-180.

Lees, A.C. and Peres, C.A. 2008. Conservation value of remnant riparian forest corridors of varying quality for Amazonian birds and mammals. Conservation Biology 22(2): 439-449.

Lehmkuhl, J.F., Kistler, K.D., Begley, J.S., Boulanger, J. 2006. Demography of Northern Flying Squirrels informs ecosystem management of western interior forests. Ecological Applications 16(2): 584-600.

Lencinas, M.V., Pastur, G.M., Gallo, E., Cellini, J.M. 2009. Alternative silvicultural practices with variable retention improve bird conservation in managed South Patagonian forests. Forest Ecology and Management 258(4): 472-480.

Linden, D.W., Roloff, G.J., Kroll, A.J. 2012. Conserving avian richness through structure retention in managed forests of the Pacific Northwest, USA. Forest Ecology and Management 284: 174-184.

Lindenmayer, D.B. and Luck, G. 2005. Synthesis: Thresholds in conservation and management. Biological Conservation 124(3): 351-354.

Lindenmayer, D.B., Fischer, J., Cunningham, R.B. 2005. Native vegetation cover thresholds associated with species responses. Biological Conservation 124(3): 311-316.

Lindenmayer, D.B., Knight, E., McBurney, L., Michael, D., Banks, S.C. 2010. Small mammals and retention islands: An experimental study of animal response to alternative logging practices. Forest Ecology and Management 260(12): 2070-2078.

Long, J.A., Hazlitt, S.L., Nelson, T.A., Laberee, K. 2011. Estimating 30-year change in coastal old-growth habitat for a forest-nesting seabird in British Columbia, Canada. Endangered Species Research 14: 49–59.

Macdonald, J.S., Beaudry, P.G., MacIsaac, E.A., Herunter, H.E. 2003a. The effects of forest harvesting and best management practices on streamflow and suspended sediment concentrations during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. Canadian Journal of Forest Research 33(8): 1397-1407.

Macdonald, J.S., MacIsaac, E.A., Herunter, H.E. 2003b. The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia. Canadian Journal of Forest Research 33(8): 1371-1382.

Maguire, D.A., Halpern, C.B., Phillips, D.L. 2007. Changes in forest structure following variable-retention harvests in Douglas-fir dominated forests. Forest Ecology and Management 242(2): 708-726.

Mahon, T. 2009. Northern Goshawks in West-Central British Columbia 10-Year Project Summary. Prepared for: Bulkley Valley Centre for Natural Resources Research and Management.

Mallik, A.U., Newaz, S., Mackereth, R.W., Shahi, C. 2011. Geomorphic changes of headwater systems 3-23 years after forest harvesting by clearcutting. Ecosphere 2(4): article 43.

Malt, J.M., Lank, D.B. 2009. Marbled Murrelet nest predation risk in managed forest landscapes: dynamic fragmentation effects at multiple scales. Ecological Applications 19(5): 1274–1287.

Marczak, L.B., Sakamaki, T., Turvey, S.L., Deguise, I., Wood, S.L.R., Richardson, J.S. Are forested buffers an effective conservation strategy for riparian fauna? An assessment using meta-analysis. Ecological Applications 20(1): 126-134.

Martel, N., RodrÍguez, M.A., Bérubé, P. 2007. Multi-scale analysis of responses of stream macrobenthos to forestry activities and environmental context. Freshwater Biology 52(1): 85-97.

Masek, J.G., Huang, C., Wolfe, R., Cohen, W., Hall, F., Kutler, J., Nelson, P. 2008. North American forest disturbance mapped from a decadal Landsat record. Remote Sensing of Environment 112(6): 2914-2926.

May, C.L. and Lisle, T.E. 2012. River profile controls on channel morphology, debris flow disturbance, and the spatial extent of salmonids in steep mountain streams. Journal of Geophysical Research: Earth Surface (2003-2012), 117(F4).

Mellina, E. and Hinch, S.G. 2009. Influences of riparian logging and in-stream large wood removal on pool habitat and salmonid density and biomass: A meta-analysis. Canadian Journal of Forest Research 39(7):1280-1301.

Metzger, J.P., Martensen, A.C., Dixo, M., Bernacci, L.C., Ribeiro, M.C., Teixeira, A.M.G., Pardini, R. 2009. Time-lag in biological responses to landscape changes in a highly dynamic Atlantic forest region. Biological Conservation 142(6): 1166-1177.

Millennium Ecosystem Assessment. 2005. Millennium Ecosystem Assessment Synthesis Report. Island Press, Washington DC.

Miller, S.L., Raphael, M.G., Falxa, G.A., Strong, C., Baldwin, J., Bloxton, T., Galleher, B.M., Lance, M., Lynch, D., Pearson, S.F., Ralph, C.J., Young, R.D. 2012. Recent Population Decline of the Marbled Murrelet in the Pacific Northwest. The Condor 114(4):771-781.

Ministry of Environment. 2008. Grizzly Bear Habitat Mapping for Parks and Conservancies for the North and Central Coast.
URL:

Ministry of Forests, Lands, and Natural Resource Operations, Research Division, Fish-Forest Interaction Program. 2008. Queen Charlotte Islands Project. Fish-Forest Interaction Program website. URL: http://www.for.gov.bc.ca/hre/ffip/QCIslands.htm

Ministry of Forests, Lands, and Natural Resource Operations, Research Division. 2009a. Carnation Creek Project. Fish-Forest Interaction Program website. URL: http://www.for.gov.bc.ca/hre/ffip/CarnationCrk.htm

Ministry of Forests, Lands, and Natural Resource Operations, Research Division, Fish-Forest Interaction Program. 2009b. Bowron River Watershed Project: A Landscape-level Assessment of the Post-salvage Change in Stream and Riparian Function. Fish-Forest Interaction Program website. URL: http://www.for.gov.bc.ca/hre/ffip/Bowron.htm

Ministry of Forests, Lands, and Natural Resource Operations, Research Division, Fish-Forest Interaction Program. 2009c. Prince George Small Streams Project. Fish-Forest Interaction Program website. URL: http://www.for.gov.bc.ca/hre/ffip/PGSSP.htm

Mitchell, A.K., Koppenaal, R., Goodmanson, G., Benton, R., Bown, T. 2007. Regenerating montane conifers with variable retention systems in a coastal British Columbia forest: 10-Year results. Forest Ecology and Management 246(2): 240-250.

Mitchell, A., Michaelfelder, V., van der Marel, R. 2008. Proposed Wildlife Management Areas for the Northern Goshawk ((*Accipiter gentilis laingi*) on the Central Coast of British Columbia. Prepared for: Ministry of Environment

Monserud, R.A. 2002. Large-scale management experiments in the moist maritime forests of the Pacific Northwest. Landscape and Urban Planning 59(3): 159-180.

Moore, R.D. and Richardson, J.S. 2003. Progress towards understanding the structure, function, and ecological significance of small stream channels and their riparian zones. Canadian Journal of Forest Research 33(8): 1349-1351.

Moore, R.D., and Richardson, J.S. 2012. Natural disturbance and forest management in riparian zones: Comparison of effects at reach, catchment, and landscape scales. Freshwater Science 31(1): 239-247.

Moore, R.D., and Wondzell, S.M. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. Journal of the American Water Resources Association 41(4): 763-784.

Moore, R.D., Richards, G., Story, A. 2008. Electrical conductivity as an indicator of water chemistry and hydrologic process. Streamline Watershed Management Bull 11(2): 25-29.

Mortelliti, A., Sozio, G., Boccacci, F., Ranchelli, E., Cecere, J.G., Battisti, C., Boitani, L. 2012. Effect of habitat amount, configuration and quality in fragmented landscapes. Acta Oecologica 45: 1-7.

Mowat, G., Heard, D.C., Seip, D.R., Poole, K.G., Stenhouse, G., Paetkau, D.W. Grizzly *Ursus arctos* and black bear *U. americanus* densities in the interior mountains of North America. Wildlife Biology, 11(1): 31-48.

Müller, J., and Bütler, R. 2010. A review of habitat thresholds for dead wood: A baseline for management recommendations in European forests. European Journal of Forest Research 129: 981-992.

Naylor, B.J., Mackereth, R.W., Kreutzweiser, D.P. Sibley, P.K. 2012. Merging END concepts with protection of fish habitat and water quality in new direction for riparian forests in Ontario: A case study of science guiding policy and practice. Freshwater Science 31(1): 248-257.

Nelson, C.R., and Halpern, C.B. 2005. Edge-related responses of understory plants to aggregated retention harvest in the Pacific Northwest. Ecological Applications 15(1): 196-209.

Newsome, T.A., Heineman, J.L., Nemec, A.F.L., Comeau, P.G., Arsenault, A., Waterhouse, M. 2010. Ten-year regeneration responses to varying levels of overstory retention in two productive southern British Columbia ecosystems. Forest Ecology and Management 260(1): 132-145.

Nonaka, E. and Spies, T.A. 2005. Historical range of variability in landscape structure: A simulation study in Oregon, USA. Ecological Applications 15(5): 1727-1746.

Ohman, J.L., Gregory, M.J., Spies, T.A. 2007. Influence of environment, disturbance, and ownership on forest vegetation of coastal Oregon. Ecological Applications 17(1): 18-33.

Olson, D.H., and Burnett, K.M. 2009. Design and management of linkage areas across headwater drainages to conserve biodiversity in forest ecosystems. Forest Ecology and Management 258: S117-S126.

Olson, D.H., Anderson, P.D., Frissell, C.A., Welsh Jr, H.H., Bradford, D.F. 2007. Biodiversity management approaches for stream-riparian areas: Perspectives for Pacific Northwest headwater forests, microclimates, and amphibians. Forest Ecology and Management 246(1): 81-107.

Olsson, J., Johansson, T., Jonsson, B.G., Hijältén, J., Edman, M., Ericson, L. 2012. Landscape and substrate properties affect species richness and community composition of saproxylic beetles. Forest Ecology and Management 286: 108-120.

Otto, C.R.V. and Roloff, G.J. 2012. Songbird response to green-tree retention prescriptions in clearcut forests. Forest Ecology and Management 284: 241-250.

Paillet, Y. Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römermann, M., Bijlsma, R.J., de Bruyn, L., Fuhr, M., Grandin, U., Kanka, R., Lundin, L., Luque, S., Magura, T., Matesanz, S., Mészáros, I., Sebastià, M.-T., Schmidt, W., Standovár, T., Tóthmérész, B., Uotila, A., Valladares, F., Vellak, K., Virtanen, R. 2010. Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. Conservation Biology 24(1): 101-112.

Parish, R., and Antos, J.A. 2004. Structure and dynamics of an ancient montane forest in coastal British Columbia. Oecologia 141(4): 562-576.

Pearson, A.F. 2010. Natural and logging disturbances in the temperate rain forests of the Central Coast, British Columbia. Canadian Journal of Forest Research 40(10): 1970-1984.

Pengelly, C.J., and Cartar, R.V. 2010. Effects of variable retention logging in the boreal forest on the bumble bee-influenced pollination community, evaluated 8-9 years post-logging. Forest Ecology and Management 260(6): 994-1002.

Perhans, K., Appelgren, L., Jonsson, F., Nordin, U., Söderström, Gustafsson, L. 2009. Retention patches as potential refugia for bryophytes and lichens in managed forest landscapes. Biological Conservation 142: 1125-1133.

Peterman, R.M. 2004. Possible solutions to some challenges facing fisheries scientists and managers. ICES Journal of Marine Science, 61: 1331-1343.

Poff, N.L., and Zimmerman, J.K.H. 2010. Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. Freshwater Biology 55(1): 194-205.

Potter, C., Tan, P.N., Kumar, V., Kucharik, C., Klooster, S., Genovese, V., Cohen, W., Healey, S. 2005. Recent history of large-scale ecosystem disturbances in North America derived from the AVHRR satellite record. Ecosystems 8(7): 808-824.

Poulin, J.F., and Villard, M.C. 2011. Edge effect and matrix influence on the nest survival of an old forest specialist, the Brown Creeper (*Certhia Americana*). Landscape Ecology 26(7): 911-922.

Preston, M.I., and Harestad, A.S. 2007. Community and species responses by birds to group retention in a coastal temperate forest on Vancouver Island, British Columbia. Forest Ecology and Management 243(1): 156-167.

Price, K., Suski, A., McGarvie, J., Beasley, B., Richardson, J.S. 2003. Communities of aquatic insects of old-growth and clearcut coastal headwater streams of varying flow persistence. Canadian Journal of Forest Research 33(8): 1416-1432.

Price, M.H.H., Darimont, C.T., Temple, N.F., MacDuffee, S.M. 2008. Ghost runs: Management and status assessment of Pacific salmon (Oncorhynchus spp.) returning to British Columbia's central and north coasts. Canadian Journal of Fisheries and Aquatic Sciences 65(12): 2712–2718.

Putz, G., Burke, J.M., Smith, D.W., Chanasyk, D.S., Prepas, E.E., Mapfumo, E. 2003. Modelling the effects of boreal forest landscape management upon streamflow and water quality: Basic concepts and considerations. Journal of Environmental Engineering and Science 2(S1): S87-S101.

Radford, J.Q., Bennett, A.F., Cheers, G.J. 2005. Landscape-level thresholds of habitat cover for woodland-dependent birds. Biological Conservation 124(3): 317-337.

Reeves, G.H., Burnett, K.M., McGarry, E.V. 2003. Sources of large wood in the main stem of a fourth-order watershed in coastal Oregon. Canadian Journal of Forest Research 33(8): 1363-1370.

Reid, D.J., Quinn, J.M., Wright-Stow, A.E. 2010. Responses of stream macroinvertebrate communities to progressive forest harvesting: Influences of harvest intensity, stream size and riparian buffers. Forest Ecology and Management 260(10): 1804-1815.

Rex, J.F., Maloney, D.A., Krauskopf, P.N., Beaudry, P.G., Beaudry, L.J. 2012. Variable-retention riparian harvesting effects on riparian air and water temperature of sub-boreal headwater streams in British Columbia. Forest Ecology and Management 269: 259-270.

Richardson, J.S. 2008. Aquatic arthropods and forestry: Effects of large-scale land use on aquatic systems in Nearctic temperate regions. The Canadian. Entomologist 140(4): 495-509.

Richardson, J.S., and Moore, R.D. 2010. Stream and Riparian Ecology (Chapter 13). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of Forest Hydrology and Geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 441-460.

Richardson, J.S., Hoover, T.M., Lecerf, A. 2009. Coarse particulate organic matter dynamics in small streams: Towards linking function to physical structure. Freshwater Biology 54(10): 2116-2126.

Richardson, J.S., Taylor, E., Schluter, D., Pearson, M., Hatfield, T. 2010. Do riparian zones qualify as critical habitat for endangered freshwater fishes? Canadian Journal Fisheries Aquatic Science 67(7): 1197-1204.

Richardson, J.S., Naiman, R.J., Bisson, P.A. 2012. How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? Freshwater Science 31(1): 232-238.

Richter, A., and Duncan, A. 2012. Riparian functional assessment: Choosing metrics that quantify restoration success in Austin, Texas. City of Austin Watershed Protection Department, Environmental Resource Management Division. 43 pp.

Rosenvald, R., and Lõhmus, A. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management 255(1): 1-15.

Sajedi, T., Prescott, C.E., Seely, B., Lavkulich, L.M. 2012. Relationships among soil moisture, aeration and plant communities in natural and harvested coniferous forests in coastal British Columbia, Canada. Journal of Ecology 100(3): 605-618.

Sakamaki, T., and Richardson, J.S. 2011. Biogeochemical properties of fine particulate organic matter as an indicator of local and catchment impacts on forested streams. Journal of Applied Ecology 48(6): 1462-1471.

Schindler, D.E., Hilborn, R., Chasco, B., Boatright, C.P., Quinn, T.P., Rogers, L.A., Webster, M.S. 2010. Population diversity and the portfolio effect in an exploited species. Nature 465(7298): 609–612.

Schmera, D., Baur, B., Erős, T. 2012. Does functional redundancy of communities provide insurance against human disturbances? An analysis using regional-scale stream invertebrate data. Hydrobiologia 693(1): 183-194.

Schmidt, B.C., and Roland, J. 2006. Moth diversity in a fragmented habitat: Importance of functional groups and landscape scale in the boreal forest. Annals of the Entomological Society of America 99(6): 1110-1120.

Schnackenberg, E.S., and MacDonald, L.H. 1998. Detecting cumulative effects on headwater streams in the Routt National Forest, Colarado. Journal of the American Water Resources Association 34(5): 1163-1177.

Schnorbus, M., and Alila, Y. 2004. Forest harvesting impacts on the peak flow regime in the Columbia Mountains of southeastern British Columbia: An investigation using long-term numerical modeling. Water Resources Research 40(5): W05205.

Scrimgeour, G.J., Hvenegaard, P.J., Tchir, J. 2008. Cumulative industrial activity alters lotic fish assemblages in two boreal forest watersheds of Alberta, Canada. Environmental Management 42(6): 957-970.

Shafer, A.B., Coté, S.D., Coltman, D.W. 2011. Hot spots of genetic diversity descended from multiple Pleistocene refugia in an alpine ungulate. Evolution 65-1: 125–138.

Shanley, C.S., Pyare, S., Smith, W.P. 2013. Response of an ecological indicator to landscape composition and structure: Implications for functional units of temperate rainforest ecosystems. Ecological Indicators 24: 68-74.

Sheldon, F., Peterson, E.E., Boone, E.L., Sippel, S., Bunn, S.E., Harch, B.D. 2012. Identifying the spatial scale of land use that most strongly influences overall river ecosystem health score. Ecological Applications 22(8): 2188-2203.

Sheridan, C.D., and Olson, D.H. 2003. Amphibian assemblages in zero-order basins in the Oregon Coast Range. Canadian Journal of Forest Research 33(8): 1452-1477.

Sibley, P.K., Kreutzweiser, D.P., Naylor, B.J., Richardson, J.S., Gordon, A.M. 2012. Emulation of natural disturbance (END) for riparian forest management: Synthesis and recommendations. Freshwater Science 31(1): 258-264.

Silvergieter, M.P., and Lank, D.B. 2011. Marbled Murrelets Select Distinctive Nest Trees within Old-Growth Forest Patches. Avian Conservation and Ecology 6(2): 3 p.

Simanonok, M.P., Anderson, C.B., Pastur, G.M., Lencinas, M.V., Kennedy, J.H. 2011. A comparison of impacts from silviculture practices and North American beaver invasion on stream benthic macroinvertebrate community structure and function in *Nothofagus* forests of Tierra del Fuego. Forest Ecology and Management 262(2): 263-269.

Slesak, R.A., Harrington, T.B., Schoenholtz, S.H. 2010. Soil and Douglas-fir (*Pseudotsuga menziesii*) foliar nitrogen responses to variable logging-debris retention and competing vegetation control in the Pacific Northwest. Canadian Journal of Forest Research 40(2): 254-264.

Smith, W.P., and Person, D.K. 2007. Estimated persistence of northern flying squirrel populations in temperate rain forest fragments of Southeast Alaska. Biological Conservation 137(4): 626-636.

Smith, M.J., Betts, M.G., Forbes, G.J., Kehler, D.G., Bourgeois, M.C., Flemming, S.P. 2011a. Independent effects of connectivity predict homing success by northern flying squirrel in a forest mosaic. Landscape Ecology 26(5): 709-721.

Smith, A.C., Fahrig, L., Francis, C.M. 2011b. Landscape size affects the relative importance of habitat amount, habitat fragmentation, and matrix quality on forest birds. Ecography 34(1): 103-113.

Spies, T.A., Johnson, K.N., Burnett, K.M., Ohmann, J.L., McComb, B.C., Reeves, G.H., Bettinger, P., Kline, J.D., Garber-Yonts, B. 2007. Cumulative ecological and socioeconomic effects of forest policies in coastal Oregon. Ecological Applications 17(1): 5-17.

Spribille, T., Thor, G., Bunnell, F.L., Goward, T., Björk, C.R. 2008. Lichens on dead wood: Species-substrate relationships in the epiphytic lichen floras of the Pacific Northwest and Fennoscandia. Ecography 31(6): 741-750.

Staus, N.L., Strittholt, J.R., DellaSala, D.A., Robinson, R. 2002. Rate and pattern of forest disturbance in the Klamath-Siskiyou ecoregion, USA between 1972 and 1992. Landscape Ecology 17(5): 455-470.

Stephenson, J.M., and Morin, A. 2009. Covariation of stream community structure and biomass of algae, invertebrates and fish with forest cover at multiple spatial scales. Freshwater Biology 54(10): 2139-2154.

Studinski, J.M., Hartman, K.J., Niles, J.M., Keyser, P. 2012. The effects of riparian forest disturbance on stream temperature, sedimentation, and morphology. Hydrobiologia 686(1): 107-117.

St-Laurent, M.H., Dussault, C., Ferron, J., Gagnon, R. 2009. Dissecting habitat loss and fragmentation effects following logging in boreal forest: Conservation perspectives from landscape simulations. Biological Conservation 142(10): 2240-2249.

St-Laurent, M.H., Ferron, J., Haché, S., Gagnon, R. 2008. Planning timber harvest of residual forest stands without compromising bird and small mammal communities in boreal landscapes. Forest Ecology and Management 254(2): 261-275.

Sullivan, T.P., Sullivan, D.S., Lindgren, P.M.F. 2008. Influence of variable retention harvests on forest ecosystems: Plant and mammal responses up to 8 years post-harvest. Forest Ecology and Management 254(2): 239-254.

Swift, T.L., and Hannon, S.J. 2010. Critical thresholds associated with habitat loss: A review of the concepts, evidence, and applications. Biological Reviews 85(1): 35-53.

Symmetree Consulting Ltd. 2008. Western redcedar – issues for managing for desirable characteristics under retention of varying levels. Management to promote desired characteristics. Prepared for: the Forest Practices Board.

Thorpe, H.C., and Thomas, S.C. 2007. Partial harvesting in the Canadian boreal: Success will depend on stand dynamic responses. The Forestry Chronicle 83(3): 319-325.

Tiegs, S.D., Chaloner, D.T., Levi, P., Rüegg, J. Tank, J.L., Lamberti, G.A. 2008. Timber harvest transforms ecological roles of salmon in southeast Alaska rain forest streams. Ecological Applications 18(1): 4-11.

Tilghman, J.M., Ramee, S.W., Marsh, D.M. 2012. Meta-analysis of the effects of canopy removal on terrestrial salamander populations in North America. Biological Conservation 152: 1-9.

Tschaplinski, P.J., and Pike, R.G. 2010. Riparian Management and Effects on Function (Chapter 15). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of Forest Hydrology and Geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 479-525.

Urgenson, L.S., Halpern, C.B., Anderson P.D. 2013. Twelve-year responses of planted and naturally regenerating conifers to variable-retention harvest in the Pacific Northwest, USA. Canadian Journal of Forest Research 43(999): 46-55.

USDA Forest Service. 2010. Meeting Current and Future Conservation Challenges Through the Synthesis of Long-Term Silviculture and Range Management Research. Gen. Tech. Report WO-84. 82 p.

Vaidya, O.C., Smith, T.P., Fernand, H., Leek, N.R.M. 2008. Forestry best management practices: Evaluation of alternate streamside management zones on stream water quality in Pockwock Lake and Five Mile Lake watersheds in Central Nova Scotia, Canada. Environmental Monitoring and Assesessment 137(1-3): 1-14.

Valdal, E.J., and Quinn, M.S. 2011. Spatial analysis of forestry related disturbance on Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*): Implications for policy and management. Applied Spatial Analysis 4(2): 95-111.

Valett, H.M., Crenshaw, C.L., Wagner, P. 2002. Stream nutrient uptake, forest succession, and biogeochemical theory. Ecology 83(10): 2888-2901.

van der Hoek, Y., Renfrew, R., Manne, L.L. 2013. Assessing regional and interspecific variation in threshold responses of forest breeding birds through broad scale analyses. PloS one 8(2): e55996.

Vanderwel, M.C., Malcolm, J.R., Mills, S.C. 2007. A meta-analysis of bird responses to uniform partial harvesting across North America. Conservation Biology 21(5): 1230-1240.

Vanderwel, M.C., Mills, S.C., Malcolm, J.R. 2009. Effects of partial harvesting on vertebrate species associated with late-successional forests in Ontario's boreal region. The Forestry Chronicle 85(1): 91-104.

van Rooyen, J.C., Malt, J.M., Lank, D.B. 2011. Relating Microclimate to Epiphyte Availability: Edge Effects on Nesting Habitat Availability for the Marbled Murrelet. Northwest Science, 85(4): 549-561.

Verschuyl, J., Riffell, S., Miller, D., Wigley, T.B. 2011. Biodiversity response to intensive biomass production from forest thinning in North American forests – A meta-analysis. Forest Ecology and Management 261(2): 221-232.

Villard, M.A., D'Astous, É., Haché, S., Poulin, J.F., Thériault, S. 2012. Do we create ecological traps when trying to emulate natural disturbances? A test on songbirds of the northern hardwood forest. Canadian Journal of Forest Research 42(7): 1213-1219.

Vose, J.M., Sun, G., Ford, C.R., Bredemeier, M., Otsuki, K., Wei, X., Zhang, Z., Zhang, L. 2011. Forest ecohydrological research in the 21st century: What are the critical needs? Ecohydrology 4(2): 146-158.

Wilk, R.J., Raphael, M.G., Nations, C.S., Ricklefs, J.D. 2010. Initial response of small ground-dwelling mammals to forest alternative buffers along headwater streams in the Washington Coast Range, USA. Forest Ecology and Management 260(9): 1567-1578.

Winkler, R.D., Moore, R.D., Redding, T.E., Spittlehouse, D.L., Smerdon, B.D., Carlyle-Moses, D.E. 2010. The Effects of Forest Disturbance on Hydrologic Processes and Watershed Response (Chapter 7). In Pike, R.G., T.E. Redding, R.D. Moore, R.D. Winkler, and K.D. Bladon (editors). 2010. Compendium of forest hydrology and geomorphology in British Columbia. B.C. Ministry of Forest and Range, Forest Science Program, Victoria, B.C. and FORREX Forum for Research and Extension in Natural Resources, Kamloops, B.C. Land Management Handbook 66 pp. 179-212.

Wong, C., Sandmann, H., Dorner, B. 2004. Historical variability of natural disturbances in British Columbia: A literature review. FORREX series 12: 74pp.

Wootton, J.T. 2012. Effects of timber harvest on river food webs: Physical, chemical and biological responses. PLoS one 7(9): e43561.

Yi, H., and Moldenke, A. 2008. Responses of litter-dwelling arthropods to four different thinning intensities in Douglas-fir forests of the Pacific Northwest, USA. Annales Zoologici Fennici 45(3): 229-240.

Yi, H., and Moldenke, A.R. 2011. Diversity of shrub-dwelling arthropods to thinning of young Douglas-fir forests in the Pacific Northwest, USA. Forest Science 57(2): 134-144.

Zenner, E.K., Olszewski, S.L., Palik, B.J., Kastendick, D.N., Peck, J.E., Blinn, C.R. 2012. Riparian vegetation response to gradients in residual basal area with harvesting treatment and distance to stream. Forest Ecology and Management 283: 66-76.

Zhang, Y., Richardson, J.S., Pinto, X. 2009. Catchment-scale effects of forestry practices on benthic invertebrate communities in Pacific coastal streams. Journal of Applied Ecology 46(6): 1292-1303.