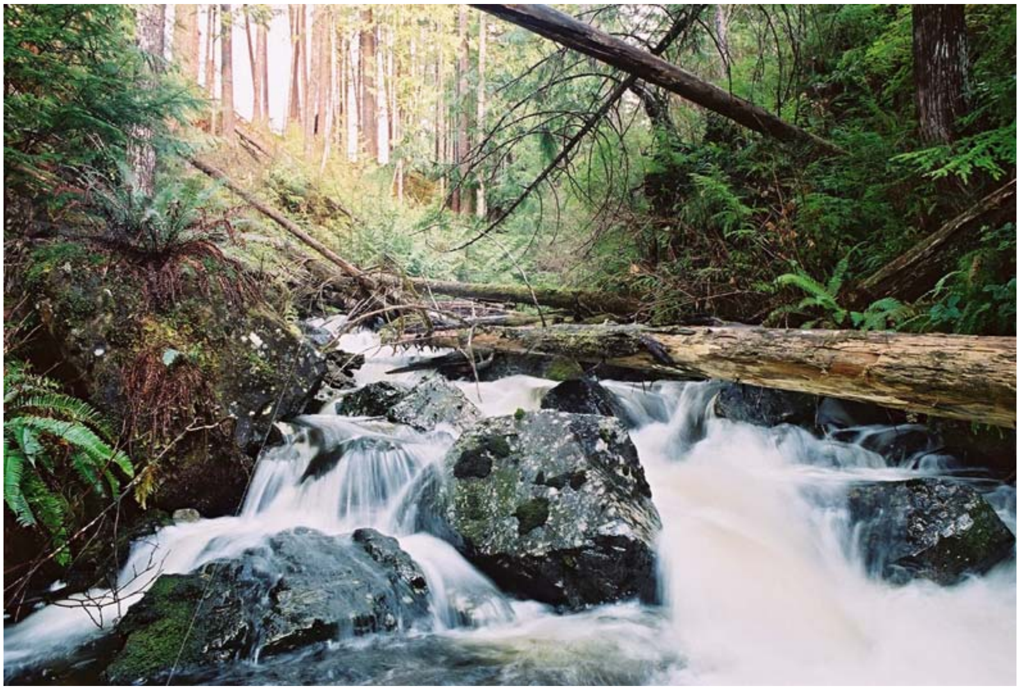
# An Effectiveness Monitoring Pilot Program for Riparian Area Forest Management on North-central Vancouver Island (TFL 37)



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# Executive Summary

The objective of this report is to describe a scientifically credible monitoring program for evaluating the effectiveness of riparian forest management practices on Canadian Forest Products Ltd. (Canfor) Tree Farm License (TFL) #37, situated on north-central Vancouver Island. A pilot program was initiated in the fall of 2003 to monitor both terrestrial and aquatic components of the riparian ecosystem that are known to respond in a predictable manner to adjacent forest harvesting. This program focuses on selecting and testing indicators of riparian forest structure, one wildlife component known to be a good indicator of riparian ecosystem health (terrestrial salamanders), and six physical characteristics within the stream channel.

Based on stream occurrence and current *Forest Practices Code* prescriptions regarding riparian forest management, we suggest that sampling locations include S2, S3 and S4 streams with (treatment) and without (control) adjacent harvesting, and that are accessible with relative ease from existing roads. Control plots appear rare and will likely be located outside TFL #37. Sampled streams should occur within the CWHxm2, CWHvm1 and CWHvm2 biogeoclimatic zones where the dominant habitats by area, and the recommended habitats of focus, are the Douglas fir – western hemlock, and the western hemlock – amabilis fir forest types. Due to limitations in stream availability (i.e., adequate sample size), stratified sampling beyond BEC zone and stream type is not recommended.

The selected indicators which were measured and evaluated at each stream included:

| **Terrestrial** | **Stream/Aquatic** |
|---|---|
| • Canopy Closure<br>• Shrub cover / composition<br>• Tree density (stems per ha) or basal area<br>• Coarse woody debris<br>• Windthrow (suspended tree index)<br>• Wildlife habitat use (terrestrial salamanders) | • Bank erosion<br>• Bar type and frequency<br>• Logjam frequency<br>• LWD cover and volume<br>• Fish cover types<br>• Streambank tree loss |

The Canfor pilot study assessed the suitability of the above indicators for effectiveness monitoring (i.e, will the indicator detect changes to the riparian ecosystem), as well as the feasibility (i.e., practicality and efficiency) of measuring each indicator in the field. Twenty potential S2 harvest streams, 13 potential S3 harvest streams, and 1 potential S4 harvest stream were identified for sampling. We were unable to locate, either on the ground or by GIS overlay, any appropriate riparian forest control streams within the TFL (i.e., mature or old riparian forest with no adjacent harvesting, and which is accessible and within the same BEC unit). Suitable control sites may have to be established in protected areas or at other locations on Vancouver Island with similar forest conditions.

Of the potential sample streams, 12 were visited and sampled using the full sampling methods. Initial coefficients of variation for each indicator were obtained, and required sample sizes necessary to achieve a desired statistical power were determined. Given the initial variability of indicators found during the pilot sampling, this monitoring program is designed to monitor changes over time and make comparisons between areas with a 90% probability of detecting change at a statistical significance of alpha = 0.10. To achieve this we recommend a conservative, **initial estimate of 6 streams (sites) per stream class** (i.e., S2, S3, etc., plus unharvested control streams) with an initial sample frequency of once per year, for at least 2-3 years. This sample size should provide enough rigour to detect change in the most variable indicator (suspended tree index), and be more than sufficient for indicators with less inherent variability (tree density or basal area, CWD, salamanders, etc.). Data analyses should rigorous and should review whether variability and statistical power are sufficient to warrant adjustments to sample size or sampling intensity/frequency. However, as in any study design, the pursuit of adequate sample size must be balanced with the availability of sufficient resources to conduct the work. It is anticipated that replication within this monitoring program will be limited by stream availability before sampling cost.

We recommend dropping tree height as a measurable variable at forest structure plots because it is redundant to measuring tree diameter (Pearson correlation coefficient = 0.85) and takes longer to measure in the field. We found a similar relationship between tree density and canopy closure. These relationships should be evaluated and indicators adjusted as required in future projects.

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## Acknowledgements

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*Photo credits:* Shawn Hamilton

## Introduction and Effectiveness Monitoring

Monitoring is the repetition of measurements over time for the purpose of detecting change. Effectiveness monitoring is used to assess whether a particular activity had a desired effect. This type of monitoring differs from other types of monitoring such as baseline, trend, implementation, project, validation, and compliance monitoring which are discussed elsewhere in detail (e.g. MacDonald *et al.* 1991). To be most meaningful, a monitoring program should provide insights into the relations between environmental stressors and anticipated ecosystem responses (Mulder *et al.* 1999). A functioning long-term monitoring program can become the key component for bringing together the efforts of management organizations, decision-makers, and researchers that intend to manage and conserve natural ecosystems (Wissmar 1993).

Monitoring programs can vary from detailed quantitative assessments, to subjective, informal observations, and both techniques have their advantages and pitfalls depending on the objectives and context of the monitoring project. Detailed assessments provide statistical accuracy to observations of change, but they are usually very expensive to initiate and maintain over time. Whereas, informal observations are extremely cost effective and can facilitate the design of more rigorous monitoring projects, but are limited in terms of defensible results. To maximize efficiency, whether qualitative or informal, a good monitoring program must identify the parameters of the system to be monitored that are:

1) Sensitive to the management activities of concern;
2) Responsive to these activities in a predictable manner; and
3) Capable of indicating management impacts (i.e. unlikely to be confused with natural variability in time and space; MacDonald and Smart 1993).

Furthermore, successful monitoring programs must be ecologically relevant, cost-effective and, if quantitative, they must be statistically credible (Hinds 1984). Thus it follows that the majority of the work invested in a monitoring program is required in the planning stages before implementation, and not after data collection.

The preliminary steps in developing a monitoring program are similar in all cases and apply to most monitoring situations regardless of specific goals or objectives. Decisions made at each step often have repercussions for the entire monitoring project, and sometimes will require reassessment at multiple steps. MacDonald *et al.* (1991) summarize the preliminary steps of monitoring program development as follows:

1) Define the general purpose and objectives of the program.
2) Define the approximate budget and personnel restraints.
3) Review existing data.
4) Select monitoring indicators, sampling locations, sampling procedures, and analysis techniques.
5) Evaluate hypothetical data or data from pilot studies and assess power to detect change.

6) Reassess monitoring objectives and compatibility with existing resources.
7) Adjust the monitoring program as necessary.

Once the monitoring program is initially developed, it is generally best to consider the first field session as a pilot study. This allows more flexibility to adapt the sampling design to the available resources, and to acquire knowledge of the variability inherent in the selected indicators. The initial monitoring program plan should not be considered as a final, fixed document, and there should be incentive to quickly analyze the initial data and adapt procedures accordingly. Designation of the first phase of data collection as a pilot study greatly enhances the potential for communication among all those involved – technicians, statisticians, managers, and technical specialists (MacDonald *et al.* 1991). However, prior to pilot studies, detailed research and review of indicator selection must be completed.

There are general principles that apply when selecting indicators to monitor. One of the most important steps in program development, indicator selection is initially based on the best available information linking management actions (stressors) to response variables. The selection of these variables is a critical component of program development because 'what is being measured' defines statistical, financial and logistical constraints. Indicators should be chosen based on a conceptual model clearly linking stressors and indicators, with pathways leading to effects on ecosystem structure and function. This process enables the effectiveness monitoring program to investigate the relations between anticipated stressors and environmental consequences, and provides the opportunity to develop predictive models to anticipate trends instead of waiting until trends have been demonstrated. Good indicators for any program should:

| | |
|---|---|
| • be easily measured | • predict changes that can be averted by management actions |
| • be sensitive to 'stresses' in the system | • have a known response |
| • respond to 'stress' in a predictable manner | • have relatively low variation in their response |
| • be anticipatory (signify change before it's 'too late') | • incorporate a suggested *Coefficient of Variation* from previous studies |

The ultimate success of any monitoring program depends on the validity of the assumed cause-effect relations between the stressor(s), their ecological effects, and the selected indicators of stress. Thus, **indicators should be selected based on their known relationship to the identified stress; the power to detect change in each indicator should be explicit**.

## Power to Detect Change

The power to detect change is a key component in any monitoring program. Statistical power refers to the probability of successfully concluding that a change has occurred or a difference between two groups exists when it actually *does* exist. In terms of monitoring, it is usually expressed as "having at least *x* % probability of detecting a *y* % change per year within an area for *z* years of surveying, and having less than *a* % probability of declaring a change or difference when there really was none." Several factors will have statistical implications on monitoring programs (Cohen 1988, Gibbs et al. 1998, Thomas and Krebs 1997). The result of any statistical comparison will be affected by 1) the type of test chosen, 2) the directionality of the hypothesis being tested (directional or non-directional, or one or two-sided p-values, or "tails"), and 3) the significance level chosen (alpha). Non-statistical factors will also ultimately have similar implications on the mathematical probability of detecting change. These include the sample size (e.g. number of monitoring plots or transects) and the inherent variability of the selected indicators over space and time (i.e. between study sites, sampling transects, and sampling seasons). Lastly, the quality of the data, or whether the indicators were sampled properly in the field, will have implications on data quality and statistical usefulness.

Generally, statistical power increases with increasing sample size, increasing effect or trend size, and greater sampling effort within sites. The power to detect change declines if indicators have a high variability, or if the significance level of a statistical test is increased (Cohen 1988, Thomas and Krebs 1997). For each indicator selected, it must be stated explicitly what the power to detect change will be over the long-term. This is generally done through approximations of statistical power in reference to the variability measured for each indicator either from a pilot study or from published literature or both. Control over statistical power is thus a combination of selecting a specific statistical test, controlling for variability over space and time (random, systematic or stratified sampling), ensuring indicators are measured properly, and effectively approximating variability (these factors are discussed in more detail in the results and discussion sections of this document).

The final step in the monitoring program is to produce interim reports, and these should feed back into the management process as collected information is summarized and interpreted. This step is key. If data are only collected but not interpreted at regular intervals, then the objectives of a monitoring program will not be achieved. Summaries should be presented clearly and in a format that is widely understood by all involved. Successful completion of this step will likely result in the information actually being incorporated into decision-making processes, which is the original purpose of any monitoring program.

## Riparian Area Management

Riparian zones are found adjacent to watercourses such as streams, rivers, wetlands, ponds and lakes, and represent the interface between the terrestrial and the aquatic ecosystem. The structure and function of riparian zones are determined by several key elements, namely topography, surface water, soils, microclimate, and vegetation (Cummins 1980, Brinson et al. 1981, Swanson et al. 1982, Oakley et al. 1985, Bilby 1988, Brosofske et al. 1997). In the Pacific Northwest, riparian areas are generally considered as:

1) important sources of large organic debris (Swanson et al. 1982, Keller and Swanson 1979, Bilby 1988);
2) areas whose vegetation has a key influence on stream chemistry via shading and nutrient assimilation; (Brinson et al. 1981);
3) areas where standing vegetation can play important roles in stream function and landscape hydrology (Bilby 1988); and
4) as relatively biodiverse areas important to the majority of vertebrate species in the Pacific Northwest (Raedeke 1989).

As such, riparian areas in managed landscapes are often left wholly or partially unharvested to protect water quality and riparian zone functioning (Martin and Pierce 1980, Hornbeck et al. 1986, Castelle *et al.* 1994). Consequently, in areas where the rotation age is shorter than that required for the stand to develop old-growth features, riparian areas could represent a significant portion of older forest, and can act as future reserves for wildlife specific to this seral stage (Hannon *et al.* 2002). The ecological role of riparian areas has become of importance in forest management, and there is a need to further our understanding of riparian ecology through effective monitoring of these areas. Forest harvesting around riparian areas in the Pacific Northwest is widespread, and various harvesting prescriptions exist in an attempt to preserve riparian ecosystem functioning. However, the ultimate effectiveness of these riparian harvest prescriptions remains largely unknown.

Canadian Forest Products Ltd. (Englewood Division) manages Tree Farm License (TFL) #37 near Woss, British Columbia on north-central Vancouver Island. A Sustainable Forest Management Plan (SFMP, Deal and Manning 2002) outlines the forest management indicators, objectives and practices for the TFL, through certification under Canadian Standards Association (CSA) Sustainable Forest Management System standard CAN/CSA-Z809-96. Objectives for riparian forest management to protect aquatic habitat, water quality and riparian forest structure are an important objective under this SFMP. These objectives are consistent with the current British Columbia Forest Practices Code regulations and the associated *Riparian Management Area Guidebook* (BC Forest Service and BC Ministry of Environment 1995), and broadly include:

1) To minimize or prevent impacts of forest and range uses on stream channel dynamics, aquatic ecosystems, and water quality of all streams, lakes, and wetlands;
2) To minimize or prevent impacts of forest and range uses on diversity, productivity, and sustainability of wildlife habitat and vegetation adjacent to streams, lakes, and wetlands with reserve zones, or where high wildlife values are present; and
3) To allow for forest and range use that is consistent with 1 and 2 above.

The BC Forest Practices Code defines riparian management areas (RMA) as consisting of a Riparian Management Zone (RMZ) and in some cases, a Riparian Reserve Zone (RRZ). The RMA is established to provide protective cover, stability, and structure for the stream, lake or wetland. For streams, the dimensions of the RMA are contingent on the stream size (i.e., bankfull width), and whether it contains fish or is part of a community watershed. Riparian reserve zones can be established inside the RMA, and again, this will depend on the stream width and whether fish or community water values are present. Timber harvest is not permitted in the RRZ. RMA classification is based on guidelines outlined in the *Riparian Management Area Guidebook*. These are described as follows:

**Table 1.** Stream types and riparian harvesting prescriptions as outlined in the BC Forest Practices code.

| Riparian Class | Fish Presence | Average stream width (m) | RMA (m) | RMZ (m) | RRZ (m) |
|---|---|---|---|---|---|
| | | | | | |
| S1 Large | Yes | ≥ 100 | 100 | 100 | 0 |
| S1 | Yes | >20-<100 | 70 | 20 | 50 |
| S2 | Yes | >5-<20 | 50 | 20 | 30 |
| S3 | Yes | 1.5-<5 | 40 | 20 | 20 |
| S4 | Yes | <1.5 | 30 | 30 | 0 |
| S5 | No | >3 | 30 | 30 | 0 |
| S6 | No | <3 | 20 | 20 | 0 |
| | | | | | |

The result of riparian forest harvesting can be the creation of excessive forest edge resulting from relatively narrow buffer strips left along stream channels. Areas of once interior forest are opened up to adjacent cutblocks and are exposed to new light, wind, and temperature regimes. This can have direct physical impacts along the buffer strips such as blowdown, gap creation, changes in edge species composition, or changes in bank stability and CWD input. These changes are contingent on local topography, buffer strip width and stream size or type, and they will change as cutblocks regenerate and succession proceeds.

We suggest that riparian effectiveness monitoring should focus on the **physical parameters of forest stand structure, along with some specific stream/aquatic variables that could change after riparian forest harvesting and buffer strip (RMA) establishment** (see Table 1). These parameters must have a predictable response and should have relatively low variability. Further, they should be easily measured through conventional means. As such, we have chosen not to focus on relatively highly variable indices such as local water quality, temperature, dissolved nutrients or sediment load. These are generally expensive to monitor, require resources that will likely not be available over the long-term, and respond in change to many other indirect landscape processes other than adjacent forest harvesting (e.g. upstream disturbances and watershed disturbances). The focus in this project will be on monitoring the primary physical characteristics of the stream channel and adjacent forest. That is, we are assuming that the maintenance of natural variability in local, riparian buffer strip forest structure will conserve most of the structural characteristics and function of the localized riparian zone adjacent to harvested areas.

## Riparian Area Management Effectiveness Monitoring in TFL 37

Canfor conducts forest management and timber harvesting consistent with the above-mentioned provincial guidelines. In order to evaluate the effectiveness of these management practices in or adjacent to RMAs, Canfor has initiated a pilot program **to select and evaluate suitable indicators for maintaining ecological integrity and function of riparian areas**. This report summarizes the findings of this pilot program.

Riparian management guidelines in British Columbia have been in place since 1995 (Riparian Management Area Guidebook; BC Forest Service and BC Ministry of Environment 1995). While there have been numerous compliance audits conducted with respect to implementation of these guidelines (i.e. implementation monitoring; see MacDonald *et al.* 1991), there has been very little evaluation of the effectiveness of these guidelines in meeting their intended objectives. Namely "to minimize or prevent impacts of forest and range uses on stream channel dynamics, aquatic ecosystems, water quality, …diversity, productivity and sustainability of wildlife habitat and vegetation adjacent to streams… with reserve zones, …and allow for forest and range use that is consistent with these objectives (Deal and Manning 2002)." Hence there is a growing interest to begin monitoring of riparian buffer strips to assess their ecological characteristics and role in reference to resource management activities (see Hannon *et al.* 2002 and O'Connel *et al.* 2000).

### Selection of Indicators

Choosing indicators to be monitored in this program involved evaluating (a) the expected responsiveness of the indicator to riparian harvesting; (b) the variability of the indicator and the predictability and direction of its anticipated response; and (c) the precision and cost of obtaining field data for the indicator. For many potential indicators, this information is not known, and thus must initially be approximated from the published literature and expert opinion. A better understanding of the suitability of the selected indicators is one of the main objectives of the Canfor pilot program, and is discussed subsequently in this report.

Forest harvesting within or adjacent to riparian areas can affect the forest structure in the riparian zone. Selecting indicators to monitor this situation will always be a trade-off between logistical and financial feasibility. Numerous parameters of the biotic and abiotic forest structure could be affected and could be monitored. Each potential monitoring parameter has a unique set of advantages and disadvantages depending on the interpreted or identified stresses on the system being monitored. The choice of indicators must be a balance between the ease of monitoring a parameter, and how relevant changes in that parameter are to riparian forest management. A good candidate indicator should be directly related to the potential impacts of riparian forest harvesting, should be relatively sensitive to management impacts, have a low-to-moderate sampling frequency, and have an associated sampling and analyses cost that is also low or moderate.

Clearly, more data from more indicators would facilitate a better understanding and a more precise evaluation of changes in riparian areas post-harvest, but no monitoring project is free from cost constraints, hence idealized data sets must be balanced with real-world constraints (MacDonald *et al.* 1991).

The following list of indicators were selected for testing in the field in order to evaluate their variability and cost-effectiveness for sampling. **Table 2 summarizes the indicators that were considered directly relevant to riparian forest management, and were judged as cost efficient to measure**.

**Table 2.** Selected indicators included in the TFL 37 riaprian effectiveness monitoring pilot program.

| **Indicators** | |
|---|---|
| ***Terrestrial – Forest Structure within buffer strip*** | |
| Canopy Closure | Crown canopy closure of dominants and subdominants |
| Shrub Cover and Composition | % cover and diversity of all tall and short shrubs |
| Tree Basal Area or Stem Density | Prism plot measurements and fixed radius plots (5.64 m) |
| Coarse Woody Debris | CWD volume, includes decay classes and piece diameters |
| Tree Characteristics | Height and dbh, tree species and tree class |

| **Table 1** (continued) | |
|---|---|
| Windthrow – suspended tree index (STI) | The falling or breakage of trees directly attributable to wind exposure in the riparian management area, resulting in trees which have fallen across the stream channel (measured as count data) |
| ***Stream Characteristics - Aquatic*** | |
| Bank Erosion | Undercut and/or overhanging, or vertical and/or sloping bank edges |
| Bar Type and Frequency | Presence and type of gravel bar within the channel approximately one bankfull width and one bankfull depth in size |
| Log Jam Frequency | Frequency of logjams tightly packed together that span, or nearly span the channel |
| Large Wood Debris (LWD) Volume | The volume (as percent cover with quantifier) of LWD at each survey interval |
| Fish Cover Types | Type and abundance of effective fish cover in the stream channel |
| Stream Bank Tree Loss | Trees fallen or removed that were rooted within 2 m of the stream bank edge |

Not all of the above indicators were measured in every sampling session. Some were measured more frequently than others (see *Methods*) depending on the effort required and the expected rate of change of each indicator. It is the intention of this program to provide a practical and theoretical justification for each indicator, and also to provide initial field data to assess or validate each for their practicality. *As more data are collected in the future, it will be necessary to reassess each indicator and in some cases redefine some as the monitoring program evolves*. Inevitably there will likely not be full agreement on the selected parameters depending on the researcher's own experience and values, but a monitoring program should adapt and evolve through constructive criticism and review, and should be altered as fieldwork progresses and initial data are analyzed.

## TERRESTRIAL INDICATORS

### Forest Stand Structure

Stand structure can consist of several variables such as site quality, history of disturbance, canopy species, stand density, and stand structural stage. Stand structure describes the characteristics of canopy (tree layer) as well as understory vegetation (shrub, herb or moss layers). Here, we are defining stand structure in terms of the stand components most likely to be impacted by riparian forest management adjacent to cutblocks. Selected indicators of stand structure include canopy closure, tree basal area (or stems per ha) within the RMA, coarse woody debris (size and decay class), understory shrub complexity (% cover and species), and windthrown trees suspended over the stream channel. Some of these variables are directly affected by riparian harvesting, while some are indirectly affected. These indicators were selected based on their relationship to potential changes in riparian areas post-harvest, as well as their ease and cost of measurement.

### Canopy Closure

Canopy closure is defined as shade cover from the branches and foliage formed collectively by the crowns of adjacent trees and other woody growth.

Riparian area harvesting and the formation of linear buffer strips along stream channels can increase the influence of edge forest (Hannon *et al*. 2002). The resulting change in the forest structure can result in changes in the overstory canopy closure either through direct tree loss (e.g. windthrow) or indirect changes in species composition as a result of changes in microclimate and species succession. Openings in the canopy can affect boundary permeability to light, wind, and invasive species (Wiens *et al.* 1985, Forman and Moore 1992). These openings are created through active management of large trees susceptible to windfall along the buffer strip edge. Inputs of light can change dramatically with changes in canopy closure, and hence organic matter, forest nutrients and species composition are all likely to show associated changes.

In forests on Vancouver Island, Klinka *et al.* (1996) found that canopy cover was influenced by canopy species and site quality, and that changes in canopy cover had a strong influence on the composition and abundance of understory shrubs. They noted that canopy cover was a good predictor of shrub cover (between 0.73 and 0.83 correlation). Consequently, our data analyses will test this relationship; **crown canopy cover has therefore been selected as a measure of forest structure and a potential indicator of riparian area integrity**.

### Understory Shrub Complexity and Cover

Understory shrubs are important components of many forests. They form in high densities in early successional forest commonly after major disturbances like forest harvesting, wildfire or catastrophic windthrow, and again in later successional stages in both conifer and hardwood forests (Tappeiner and John 1973; Whitney 1986; Balogh and Grigal 1987; Kurmis and Sucoff 1989; Tappeiner *et al.* 1991; Huffman *et al.* 1994; O'Dea *et al*. 1995; Ricard and Messier 1996; Tappeiner *et al.* 2001). Dense populations of shrubs, whether in an understory or open areas, affect the availability of light, thus affecting the establishment and growth of other plants (Strothman 1967; Henderson 1970; Maguire and Forman 1983; Bailey and Tappeiner 1998). If dense shrub populations persist, they may have a major effect on the development, structure, and species composition of future forest stands several years post-harvest (Tappeiner *et al.* 2001).

Changes in forest interior conditions from the creation of riparian buffer strips adjacent to harvest openings can alter site conditions for understory species. Immediately at the forest edge, greater sapling abundance is generally the result of increased light from canopy openings. However, this is often concurrent with development of the shrub layer (Constabel and Lieffers 1996). Understory shrub species common to coastal Douglas Fir forests (e.g., salmonberry) sprout vigorously (Tappeiner *et al.* 2001) and can form dense cover (>20 000 stems/ha) depending on site characteristics post-harvest (Tappeiner *et al.* 1991; Huffman *et al.* 1994). Significant changes in shrub cover will alter understory processes for both vegetation and wildlife that respond to microclimate conditions.

When gaps occur in the canopy of Douglas fir or red alder stands as a result of windthrow or other disturbances, the forest floor of these stands can be quickly occupied by understory shrubs (e.g. salal or salmonberry) (Schrader 1998). Shrub density and vigor may decline to very low levels as the overhanging conifer canopy closes, and may remain low for 50+ years if high overstory density is maintained (Tappeiner *et al.* 2001). Shade-tolerant species like western hemlock often become established in the understory, so that shrub cover is reduced throughout the stand or in patches as their crowns develop. If trees become established and overtop the shrubs, the cover and rhizome density of those shrubs may be reduced (Tappeiner *et al.* 2001).

Significant portions of the canopy will be opened through the removal of large trees in the RMA. Given the relationships discussed above, it is likely that shrub composition will be indirectly affected by riparian harvesting within and adjacent to the RMA. The initial relationship between shrub density and canopy closure will probably be correlated, but whether this relationship holds several years post-harvest is not clear. **Shrub composition is therefore included in the indicator list for the Canfor riparian effectiveness monitoring program.**

### Tree Basal Area within the RMA

Basal area (BA) is defined as the area covered per hectare by the bases of live trees that are not rooted within 2m of the streambank edge. Removal of trees between the streambank edge and the adjacent cutblock can occur from either blowdown or direct removal from harvesting prescriptions. An optimal level of tree removal from the RMA might be required to mitigate the effects of windthrow. The level of removal will affect stem densities, understory microclimate and light conditions near the stream channel, and will alter edge habitat structure (e.g., crown closure, shrub and understory cover). Consequently, **basal area (or stem density as a surrogate measure) has been chosen as a measure of forest structure and an indicator of riparian area integrity.**

### Coarse Woody Debris

Coarse woody debris (CWD) and large woody debris (LWD, instream woody material) are both known to be affected directly and indirectly by forest harvesting in both upland and riparian areas. McHenry *at al.* (1998) noted that riparian forestry practices, in-stream salvage, and stream cleanout, have altered CWD and LWD characteristics of Olympic Peninsula streams. They found that there was a correlation between the years since logging and the loss of larger pieces of downed wood. In southwest Washington streams, Bilby and Ward (1991) found that the majority of old-growth derived CWD was lost within 5 years immediately following logging and the magnitude of loss increased with stream size. This loss of LWD from streams in this manner may be related to flooding and decreased residence time of wood and sediment in channels modified by intensive logging (McHenry *et al.*1998). Hartman and Scrivener (1990) found little annual change in LWD abundance pre-logging in British Columbia (one study area only). They then noted that post-harvest, the total volume of CWD decreased by 30%. Similarly, in Washington State, Ralph *et al.* (1994) found a reduction in CWD size and distribution in harvested basins. McHenry *et al.* (1998) suggested one of three mechanisms for change in CWD post-logging: (i) direct loss due to destabilization during logging and yarding; (ii) channel destabilization and subsequent transport; and (iii) long-term decay processes. Thus, CWD change in Olympic Peninsula streams is thought to be strongly related to both initial logging disturbance and channel disturbance associated with watershed- level impacts. Finally, Bryant (1985) found post-harvest logjam destabilization for streams in southeastern Alaska.

Provincial guidelines (RIC 1999) define CWD as dead woody material in various stages of decomposition that is located above the soil, is larger than 7.5 cm in diameter at the crossing point of the sampling transect, and is not self-supporting. **Because CWD is a component of forest structure known to be directly affected by forest harvesting, it has been included in the Canfor pilot monitoring program**. CWD present within the forested riparian buffer and within the stream channel (as LWD cover) have been measured.

### Windthrow – Suspended Tree Index (STI)

Windthrow is the falling or breakage of trees directly attributable to wind force and exposure. Windthrow can occur as uprooting directly beneath the bole, or from breakage at varying heights along the tree stem. Stathers *et al.* (1994) describe two types of windthrow. *Catastrophic windthrow* occurs infrequently, over large areas, and is a result of strong storms. *Endemic windthrow* occurs more frequently as a result of numerous, lower-velocity, localized wind events and affects individual stems or small groups of trees. Endemic windthrow often spreads progressively from an abrupt or unstable boundary and is often an indirect result of forest management practices (e.g., clearcutting, Stathers *et al.* 1994). The Canfor monitoring program deals only with endemic windthrow.

The relationship between riparian systems and the recruitment of large woody debris (LWD) has received increasing attention during the last two decades, particularly in reference to the maintenance of stream function and the creation of fish and wildlife habitat (Keller and Swanson 1979; Bisson *et al.* 1987; Hicks *et al.* 1991a; Abbe and Montgomery 1996; McHenry *et al.* 1998). Mechanisms by which LWD can enter a stream can include bank cutting, flooding, collapse from ice and snow loading, and debris slides (Keller and Swanson 1979), but windthrow is considered the most common of the mechanisms.

Working in first through fifth order streams in mature forests of the Oregon Cascades, Lienkaemper and Swanson (1987) found that wind was the mechanism for roughly 66% of woody debris entry into the stream. In streams of the Oregon Coast and Cascade Ranges, McDade *et al.* (1990) and Long (1987) found that windthrow was the major mechanism (upwards of 80%) for LWD entry into channel systems. Windthrow is a primary mechanism for LWD creation; the magnitude and frequency of windthrow in a riparian area will directly affect canopy cover, which in turn indirectly affects understory light, understory vegetation community, stream temperature, and wildlife microhabitat. Depending on the buffer strip width and the structural edge dynamics along the cutblock, windthrow could be restricted to the cutblock edge, or occur within the RMA and provide LWD input directly into the stream channel.

Windthrow can be a direct result of forest harvesting activities. Trees at the edges of cutblocks are exposed to stronger winds making them more susceptible to windthrow (Moore 1977; De Walle 1983; Oke 1987). The amount of windthrow has been related to the ratio of exposed forest perimeter to the total area of a cutblock (Elling and Verry 1978; Fleming and Crossfield 1983). Thinning also increases wind penetration into the stand and can increase the amount of windthrow for some years after treatment (Somerville 1980; Cremer *et al*. 1982; Gardiner *et al.* 1997; Ruel 2000). Therefore, forest harvesting that increases the length of exposed stand edges can directly affect the amount of windthrow in an area. Riparian reserves are known to create such conditions because they generally result in linear forest patches that have a high perimeter to area ratio. Rowan *et al.* (2003) outline some common measures to reduce windthrow damage to acceptable levels. These include moving harvest boundaries to more windfirm locations and realigning boundaries with the damaging wind direction (e.g. Ruel 1995); modification of tree crowns (e.g., crown thinning, Hutte 1983); and feathering the stand edge along cutblock edges to remove trees considered vulnerable to windthrow.

In managed areas, the occurrence of riparian windthrow can be variable over the short term. In Oregon, Andrus and Froehlich (1992) found that wind had damaged from 0 to 72% of trees by basal area in stream buffer strips within 6 years of logging, with most losses being <20%. Steinblums *et al.* (1984) found roughly 2% annual volume loss in buffers <15 years old in Oregon buffer strips. Van Sickle and Gregory (1990) estimated the probability of tree fall to be 13% over 10 years, based on observed inputs from riparian buffers in this area. In Washington State, 82% of 91 stream buffers showed <10% windthrow two years after harvest (TFW Field Implementation Committee, 1994). Short-term windthrow losses in buffer strips in the Pacific Northwest do not seem to be large (Andrus and Froehlich 1992; Steinblums *et al.* 1984). In reference to forest management, Sherwood (1993) suggested that windthrow losses in this area were generally chronic (relatively frequent, random tree fall) and occasionally episodic (e.g. large, rare windthrow events), and Sinton (1996) found return intervals for windthrow-generating storms to be patchy.

Because windthrow is a key mechanism for CWD production and LWD entry into stream channels, and which may directly affect the physical structure of the channel, we have **chosen it as an indicator of riparian effectiveness monitoring in the Canfor pilot program** (using suspended tree index as a quantifier of windthrow in the RMA, of trees which have fallen across the stream channel).

### Wildlife Habitat Use – Terrestrial salamanders

In forest monitoring, measurement of structural components is appropriate because of their relationship to the habitat needs of many medium and large-sized wildlife species, and their dominant role in structuring the ecosystem (e.g., downed wood). Measuring forest structural components is advantageous because these elements are stationary and can be remotely sensed in many instances. A common assumption in management is that if structural components are well represented in an area, then wildlife use will follow accordingly. To incorporate a validation of this in our monitoring program, we wanted to include one measurement of wildlife species use. To do this we wanted an indicator guild that was well associated with both the terrestrial and aquatic components of the riparian area, and was known to be sensitive and respond to adjacent forest harvesting. Of particular interest are species, such as keystone species, whose life histories are tightly intertwined with other small life forms and other fine-scale forest ecosystem processes (Power *et al.* 1996; Simberloff 1998; Welsh and Droge 2001).

Unlike riparian structural components and vegetation, wildlife species have the ability to move in and out of the riparian management area. Thus, scale and the specialization of the indicator to the riparian area are key components to consider. Species that are highly mobile may not spend a significant amount of time within the riparian zone, and can recolonize areas more easily if local extinctions occur as a result of riparian management activities. Selection of an indicator for wildlife-species-use must account for scale, and must be indicative of local, terrestrial and aquatic riparian processes. **Forest amphibians, particularly salamanders, have a number of characteristics that make them excellent candidates for monitoring in this context** (Welsh and Droge 2001).

Generally, herpetofauna are considered one of the most abundant groups of vertebrates in many forest ecosystems (Burton and Likens 1975, Congdon *et al.* 1986). However, until recently they have not been well studied in reference to forest management practices (Bury and Corn 1988a, Gibbons 1988, Wigley and Roberts 1994, deMaynadier and Hunter 1995) despite their presumed role in forest food webs (Burton and Likens 1975, Vitt *et al.* 1990), and the recent controversy about widespread amphibian declines (e.g., Blaustein and Wake 1990, Vitt *et al.* 1990, Pechmann *et al.*

1991, Pechmann and Wilbur 1994). Because of their life history traits, amphibians are considered to have potential value as indicators of habitat quality (Corn and Bury 1989, Dunson *et al.* 1992; Welsh and Droge 2001). Forest salamanders are often well associated with many other forest plants and animals, they are often numerous, can be easily and cheaply sampled, are functionally positioned at mid-levels in the food web, and are highly sensitive to stressor-induced perturbations of many sorts (Vitt *et al*. 1990; Olsen 1991; Wake 1991; Blaustien *et al*. 1994; Welsh and Ollivier 1998).

The life history traits of salamanders make them good choices for indicators. Forest salamanders are functionally linked to other, smaller organisms through the food web and through micro-habitat use and, as such, are thought to be good indicators of fine-scale soil-level processes within a forest monitoring program (Wyman 1998). Although this link is speculative or at best correlative (Wyman 1998), most of these smaller organisms are poorly described taxonomically, their natural history poorly known, they are difficult to detect, and they cannot be reliably sampled. Thus the fine scale overlap with the forest micro-environment, shared with the lesser-known organisms at a fine, soil-level scale, make salamanders a good indicator candidate (Fleishman 1997).

Patchiness of habitats, combined with low fecundity, small home ranges, short dispersal distances (Mathis et al. 1995) and a propensity for geographic isolation, make salamanders relatively vulnerable to the potential effects of forest harvesting and associated fragmentation (Welsh and Droge 2001), and are therefore a good potential indicator species relative to the impacts of these management activities. Lastly, to assess the effort required for sampling and for sufficient statistical power for the detection of change between areas and over time, salamanders are a good indicator group because their abundance indices have low and known confidence intervals (Gibbs *et al.* 1998; Welsh and Droge 2001). As Welsh and Droge (2001) indicate, the practical result of this **is a high power to determine relative abundance across plots (or trends over time) compared with other groups of animals** which show much higher variation (e.g. salamanders less variability than forest songbirds < small mammals < Lepidoptera [Gibbs et al. 1998]).

Riparian habitats have historically been considered important to terrestrial vertebrates because they are generally structurally complex, and represent key transition areas between terrestrial and aquatic systems (Naiman *et al.* 1993). Raedeke (1989) estimated that 70% of all vertebrate species in parts of the Pacific Northwest, U.S.A., used riparian habitats in some way during their lives.

Riparian zones may provide optimal habitat for forest salamander species. Because these areas are the interface between terrestrial and aquatic ecosystems, unlike conditions found in upland forest, these areas have a unique set of environmental conditions that may favour amphibian species (Brosofske *et al.* 1997). Studies have shown that riparian areas may be very important to amphibian populations (e.g., McComb *et al*. 1993, Gomez and Anthony 1996) because of the presence of water, the microclimate temperature, and complex forest structure. Humidity and temperature are two important factors determining the distribution of many amphibian species; higher humidity and lower temperatures are often associated with stream presence (Brosofske *et al.* 1997; Maxcy and Richardson 2000). Thus, riparian forests may provide unique habitat for wildlife because these forests experience a lower frequency or intensity of natural disturbance (Suffling *et al.* 1982; Bergeron 1991; Denneler *et al.* 1999), have higher nutrient inputs than upland forests, and are ecotonal (Naiman and Décamps 1997). Because amphibians are inextricably linked to aquatic and terrestrial environments, they are of particular concern for riparian area management.

Increasingly, researchers have recognized the importance of amphibians in reference to forest management (Dunson *et al.* 1992; deMaynadier and Hunter 1995, 1998). Some data suggest that intensive forestry practices such as site preparation activities may have long-term effects on local amphibian populations, especially salamanders (see review by deMaynadier and Hunter 1995). Among salamanders in particular, the impacts of forest management may be species-specific (e.g., Foley 1994) or limited to distinct regions within the range of a species (Diller and Wallace 1994, Welsh and Lind 1995). These species are sensitive to environmental change and may be negatively impacted by habitat changes resulting from timber harvest and associated road construction (Bury and Corn 1988; Corn and Bury 1989; deMaynadier and Hunter 1995; Welsh and Lind 1996).

Studies from the Pacific Northwest and Northeast that have investigated relationships between stand age and herpetofauna, indicate that structural components of forests are important predictors of herpetofaunal abundance (deMaynadier and Hunter 1995). Specifically, structural components near ground level, such as coarse woody debris (e.g., Aubry *et al.* 1988, Bury and Corn 1988, Raphael 1988, Dupuis *et al.* 1995), leaf litter depth and moisture (e.g., Heatwole 1962, Pough *et al.* 1987, DeGraaf and Yamasaki 1992, Welsh and Lind 1995), and understory vegetation cover (e.g., Pough *et al.* 1987, Welsh and Lind 1995), are often correlated with the presence and abundance of these animals.

Changes in the forest that result in the drying of the upper soil and litter layer will decrease the capacity of the forest to support salamanders (Welsh and Droge 2001). Opening of the canopy, or the creation of large openings can affect temperature and moisture regimes, promoting drying in the litter layer (Chen *et al.* 1999) that in turn affects salamander populations (e.g. Harpole and Haas 1999). Managed forests tend to be low in large woody material, both downed logs and snags (Aubury *et al.* 1988; Bury and Corn 1988; Spies and Cline 1988) and in leaf litter (Covington 1981), which all are known to be related to salamander abundance (Welsh and Lind 1991).

All of the above factors suggest that **monitoring terrestrial salamander abundance can be a very useful indicator of the “health and functioning” of riparian ecosystems, and the effectiveness of forest management practices in these areas.**

## STREAM / AQUATIC INDICATORS

There is a spatial limitation to what can be learned from limiting monitoring activities to the stream channel and adjacent riparian forest areas. A full understanding of in-channel events requires a broader watershed perspective (see Simonson 1993). Knowing the type and location of management activities upstream, and how these activities might affect in-stream processes, will ultimately be required for effective monitoring and management recommendations. The potential for downstream effects to appear in monitored areas as a result of upstream activities, is a key to understanding data collected at each monitoring transect. **Therefore a broader, watershed perspective must be incorporated into riparian effectiveness monitoring programs by assessing channel characteristics (indicators) adjacent to and upstream from riparian forest harvesting**.

Channel Stability is defined in this report as a combination of the following variables: bank erosion, bar frequency and type, logjam frequency, LWD volume, fish cover types, and streambank tree loss. An assessment of these variables is used as an attempt to incorporate a watershed perspective to the measurements taken in local stream reaches.

The Riparian Task Force (RTF), consisting of biologists, foresters and hydrologists from the B.C. Ministry of Forests, Ministry of Water, Land and Air Protection, academia and the private sector, suggested several extensive indicators for riparian effectiveness monitoring (BC Min. Forests 2003). Of these, Canfor selected five for the 2003/2004 pilot study to develop a riparian effectiveness monitoring protocol for TFL 37. In addition, a sixth indictor was identified. These are listed below:

1. Bank erosion
2. Bar frequency and type
3. Logjam frequency
4. LWD volume
5. Fish cover types
6. Loss of streamside trees

The original list of extensive indicators (BC Min. Forests 2003) was reduced to enable a more cost effective approach to riparian effectiveness monitoring within TFL 37, but was maintained in part to provide some consistency with the RTF methodology. In addition to the terrestrial indicators (forest structure, salamanders) described above, the results of the 2003/2004 Canfor pilot study will be used to determine the efficacy of the stream/aquatic indicators in terms of:

1) sensitivity to riparian treatments;
2) ease and repeatability of measurement in the field;
3) relevance to regional measurements being compiled by the RTF; and
4) cost effectiveness.

A description of each of the six stream/aquatic indicators is provided below.

### Bank Erosion

Bank erosion is a naturally occurring event. However, when streambank integrity is anthropogenically disturbed (i.e. adjacent streamside logging), the rate of bank erosion can increase (e.g. Davies 1997; Mckergow *et al.* 2003). Direct linkages to soil erosion and factors correlated with soil erosion have been clearly linked with adjacent harvesting activities (Hartman *et al.* 1996; Stott *et al.* 2001; Shaw and Carter 2002; Hartanto *et al.* 2003). Typically, the presence of post-harvest bank erosion is easily observed in the field using either quantitative or qualitative methods (e.g. Simonson 1993). The cost-effective use of qualitative bank erosion assessment will be applied for this program.

### Bar Frequency and Type

The number, type and location of gravel bars is an indication of the amount of sediment and LWD in the channel. In particular, mid-channel bars are an indication of increased sediment supply to the stream. The frequency of small bars is typically an indicator of a disturbed channel, and within-stream vegetation communities can be influenced by the type and frequency of bars (e.g. McBride and Strahan 1984). Sediment forming new bar types may have entered the stream from an upstream source that may not be related to the riparian treatment that is being monitored. Therefore, a large spatial scale that incorporates potential upstream disturbances must be considered when assessing bar frequency and type (see *Recommendations* for further discussion of upstream disturbances). As an indication of post-harvest erosion and differential stream flow, the presence and type of gravel bars will be monitored for this program.

### Logjam Frequency and LWD Volume

The ecological role of LWD within stream channels is relatively well known (see Carlson *et al.* 1990). LWD is involved in the storage of sediment and fine organic matter (Nakamura and Swanson 1994; Bilby and Ward 1991), influences channel morphology (Robison and Beschta 1990; Nakamura and Swanson 1994), influences the composition of riparian vegetation (Malanson and Butler 1990), and affects stream nutrient dynamics (Aumen *et al.* 1990) and fish habitat (Carlson *et al.* 1990). The frequency of logjams and the volume and size of LWD entering the stream can change directly as a result of forest harvesting. Adjacent forest harvesting exposes once-interior forest to the effects of wind (i.e., blowdown) that has a clear relationship to LWD input into stream channels (Moore 1977; De Walle 1983; Oke 1987) and the resulting aquatic habitat. The number and size of logjams may increase after streamside logging, particularly in riffle-pool streams. In step-pool streams the number of logjams may increase, but logjam size may decrease. Because these ecological relationships are clear, **we have included both logjam frequency and LWD input, as indicators of stream channel function in the Canfor monitoring program.**

### Fish Cover Types

Healthy streams typically contain a diverse range of habitat types. When streams are impacted by forest harvesting, the number of habitat types can be significantly altered (e.g., Moring 1982; Holtby 1988). Protection of fish habitat types has been, and still is, a key rational for the creation of contemporary riparian, forested buffer strips (e.g. Barton & Taylor 1981; Bilby & Bisson 1992; Rabeni & Smale 1995; Stauffer *et al.* 2000). Historically, changes in post-harvest fish habitat have resulted in changes in harvesting techniques, or controls on harvesting activities (e.g. Murphy et al. 1986; Ice *et al.* 1989). The changes in aquatic habitat after logging are well studied (e.g. Hicks *et al.* 1991b; Hartman *et al.* 1996). Because fish habitat types can be easily identified and enumerated within stream reaches **we are including the presence and number of fish habitat types as indicators in this monitoring program**.

### Loss of Streamside Trees

Streamside Tree Loss is defined here as fallen or removed trees that were once rooted in the stream channel (roughly within 2 m of the stream). These would include trees that have fallen naturally and those that have been removed by harvest. The width of the RMA and the prescribed harvest practices therein will affect streambank tree loss both directly and indirectly. **Changes in tree abundance immediately adjacent to the stream will affect bank stability, erosion, stream shading, bank structure, and large woody debris input.** Consequently, streamside tree loss has been selected as an effectiveness monitoring indicator.

Much of the rational for including streamside tree loss as an indicator is similar to that discussed for LWD, however for monitoring purposes, streamside tree loss will include only those trees that had root matrices within the stream channel (i.e. within ~2-3m of the water's edge).

# TERRESTRIAL MONITORING METHODS

## Candidate Site Selection

Monitoring will be limited to S2, S3, and S4 fish-bearing streams. These stream types were chosen because they require riparian management prescriptions in the form of forested buffer strips, and occur frequently on the landscape. There are only a few S1 streams (rivers) in TFL 37 and, under the current Forest Planning and Practices Regulation (and previous Forest Practices Code), S5 and S6 streams do not require riparian management. Consequently, S1, S5 and S6 streams were not included in the monitoring program.

Site selection was based on a combination of access, BEC zone occurrence within the TFL, the location (and access) of fish-bearing streams with management prescriptions, and the dominant forest cover types based on leading tree species by area calculations. Because sampling resources are limited, effort was focused on the most frequently occurring forest types in the most common BEC zones that were reasonably accessible by active roads and that contained target management streams (i.e. S2-S4). This limited the number of strata to replicate but included the forest types and BEC zones most representative of the TFL.

***BEC Zone Occurrence*** -- Using GIS polygon coverages for TFL #37, the occurrence by area of each BEC zone was identified (Table 2). Potential sites were selected based on access, and thus a comparison of BEC zone representation between (1) the entire TFL and (2) the area within 300m of roads was conducted. The three dominant zones were similar for both metrics; CWHvm1, CWHvm2, and CWHxm2 dominate both within the entire TFL and within 300m of accessible roads. These were therefore the focus of sampling efforts, but effort within each was contingent on the occurrence of management streams therein (see Table 2 below).

**Table 3**. BEC zone occurrence by area within TFL#37 and within 300m of active roads.

| BEC Zone | % area of whole TFL | % area within 300m of roads |
|---|---|---|
| | | |
| ATc | 2.56 | 0.02 |
| CWHvh1 | 0.67 | 0.35 |
| CWHvm1 | 29.9 | 38.47 |
| CWHvm2 | 26.6 | 21.0 |
| CWHxm2 | 18.3 | 34.3 |
| GLACIER | 0.12 | -- |
| MHmm1 | 21.7 | 5.71 |
| | | |

***Forest type Occurrence*** -- Using TEM polygon coverages for TFL#37 the most frequently occurring forest types, or tree species combinations, were identified. Within accessible areas, the most common leading tree species by area are as follows: western hemlock (HW 40%) > Douglas-fir (FD 17%) > yellow cedar (YC) > amabilis fir (BA) > mountain hemlock (MH) > western redcedar (CW) > red alder > shore pine > sitka spruce > western white pine > grand fir > cottonwood. The top two leading tree species combinations (at least double anything else) by area are as follows: FD-HW (15%) and HW-BA (13%), with all others <5% or most commonly <1% occurrence.

Initial site selection and sampling effort, therefore, was directed at FD-HW and HW-BA forest types occurring within the CWHxm2, CWHvm1, and CWHvm2. The CWHvm1 and CWHvm2 represent roughly 60% of the areas accessible by active roads within TFL#37, however, management prescription steams (S2-S4) with post-FPC harvesting do not occur frequently at the higher elevations characteristic of these zones and these stream types. For this reason the CWHxm2 will likely be the focus BEC zone unless monitoring of S5 and S6 streams is incorporated in future efforts.

***Candidate Stream Locations*** – Using the abovementioned selection criteria, potential streams were located throughout the TFL. Potential streams were defined as streams with FPC harvest prescriptions on at least one side, with cutblock length of >250m adjacent to the stream that were accessible by a road or where the road was <1 km from the stream. We identified 20 potential S2 harvest streams, 13 potential S3 harvest streams, and 1 potential S4 harvest stream. We were unable to locate, either on the ground or on GIS, any appropriate control areas that are not mature, second-growth forest within the TFL. Of the potential harvest streams, 12 were visited and sampled using the full sampling protocol (outlined later in this methods section). These are listed below.

| Stream Type | Treatment | Canfor Block | MCA Stream # from maps | UTM | Sampled (Y/N) |
|---|---|---|---|---|---|
| S3 | Harvest | KT039 | #78 | 688143<br>5543947 (plot 1) | Y |
| S3 | Harvest | NS002 | | 650090<br>5534291 (plot 6) | Y |
| S3 | Harvest | | #64 | 644290<br>5592964 (plot 1) | Y |
| S2 | Harvest | R063 | | 661444<br>5567215 (plot 3) | Y |
| S3 | Harvest | MK019 | | 658406<br>5569100 (plot 3) | Y |
| S2 | Harvest | HT017 | | 682238<br>5570421 (plot 4) | Y |
| S3 | Control | SAMPLED BUT DROPPED | | 665326<br>5566452 (plot 6) | Y |
| S3 | Harvest | K304 | | 681329<br>5554091 (plot 4) | Y |
| S2 | Harvest | | #26 | 688075<br>5541938 (plot 6) | Y |
| S2 | Harvest | KT039 | #25 | 668207<br>5543496 (walk in) | Y |
| S2 | Harvest | HI036 | | 645802<br>5585426 (plot 6) | Y |
| S2 | Harvest | TS028 | | 681460<br>5571664 (plot 6) | Y |

The remainder of the potential, **unvisited sites** identified in the initial draft of the sampling program, dominated by S2 and S3 harvest streams, are listed below.
(* = sampled in pilot study; ** = field checked as suitable for future sampling; no asterisk=not visited because of access, but identified as future sites for sampling).

| **S2** | | | **S3** | | | **S4** | | |
|---|---|---|---|---|---|---|---|---|
| **Harvest** | | | **Harvest** | | | **Harvest** | | |
| **Map** | **Stream** | **UTM** | **Map** | **Stream** | **UTM** | **Map** | **Stream** | **UTM** |
| 1** | 5(S2) | 669831.22<br>5570010.14 xm2 | | | | 13 | 63(S4) | 646319.68<br>5595312.48 |
| 4* | 20(S2) | 677799.43<br>5551364.11 vm1 | 4** | 24(S3) | 681212.87<br>5554260.63 xm2 | | | |
| 4 | 22(S2) | 677808.86<br>5547505.04 vm1 | 5* | 27(S3) | 688164.46<br>5540466.39 vm1 | | | |
| 5** | 25(S2) | 688167.85<br>5543319.18 xm2 | 8 | 37(S3) | 649804.31<br>5576877.44 xm2 | | | |
| 5** | 26(S2) | 687825.83<br>5542231.31 vm1 | 8 | 38(S3) | 650123.21<br>5574652.84 xm2 | | | |
| 6** | 25(S2) | 681359.16<br>5571622.73 vm1 | 8** | 39(S3) | 650059.24<br>5574432.32 xm2 | | | |
| 6 | 32(S2) | 682527.10<br>5570648.29 vm1 | 13** | 64(S3) | 644572.18<br>5592723.24 xm2 | | | |
| 7* | 34(S2) | 691762.87<br>5557919.00 vm2 | 14* | 66(S3) | 645805.53<br>5600968.50 vm1 | | | |
| 7* | 35(S2) | 696418.18<br>5555032.33 vm2 | 16 | 71(S3) | 681183.72<br>5541210.42 vm1 | | | |
| 8 | 40(S2) | 652752.86<br>5572969.58 xm2 | 17 | 72(S3) | 696013.83<br>5531330.09 vm1 | | | |
| 8 | 41(S2) | 652683.46<br>5572700.43 xm2 | 5** | 78(S3) | 688117.26<br>5543845.24 xm2 | | | |
| 8 | 42(S2) | 647618.82<br>5579112.35 xm2 | 8* | 82(S3) | 650094.92<br>5573340.73 xm2 | | | |
| 10** | 51(S2) | 661242.21<br>5567516.37 xm2 | 10** | 83(S3) | 658489.76<br>5569192.46 xm2 | | | |
| 10* | 53(S2) | 663171.74<br>5566506.51 xm2 | | | | | | |
| 11* | 58(S2) | 662329.31<br>5557445.11 vm1 | | | | | | |
| 12* | 62(S2) | 645672.92<br>5585086.69 xm2 | | | | | | |
| 15 | 71(S2) | 633491.85<br>5585106.83 vm1 | | | | | | |
| 15 | 70(S2) | 636181.71<br>5584360.70 vm2 | | | | | | |
| 15 | 69(S2) | 638986.75<br>5582145.61 vm1 | | | | | | |
| 18* | 77(S2) | 694959.40<br>5544228.31 vm1 | | | | | | |

## Plot Locations

***Stream Nomenclature*** – In harvested areas sampled streams were named for the Canfor cutblock designation (e.g. KT039). This was generally sufficient as there were no instances where more than one FPC stream of the same class went through the same cutblock. For blocks with no CanFor designation the number designation from the candidate stream list (developed by MCA during pilot sampling – *see accompanying map book*) was used. Control sites were named using consecutive numbers and recorded as such in the GIS. The numbering system for control streams is currently arbitrary, as long as numbers do not overlap. Because much of the mature forest on the TFL is second growth, it is likely that old growth stream site control areas will be outside of the TFL in parks or other similar ecosystems. *There were no appropriate control areas identified during the pilot stage of this project.*

Some streams were visited and sampled during the pilot study in Nov-Dec 2003. These streams were named using the abovementioned rules. Many other sites were visited and confirmed as appropriate for monitoring and are designated as such in the candidate site list (above). *These should be used in future monitoring efforts*.

***Harvested Streams*** – Because the ultimate number of appropriate sites is limited, there will be a mixture in the final sample of streams that have been harvested on both sides, and streams that have only been harvested on one side. Final site selection should account for this (i.e. equal sample size of each) but, because of the limited number of sites, we have chosen not to approach these as separate treatments for site-selection stratification. The effects of one- or two-sided harvesting can be accounted for in statistical analyses post-sampling. The monitoring program will run low on candidate streams within the TFL if too many levels of stratification are used in final site selection.

For streams harvested on both sides, a coin toss was used to determine which side the sampling plots were to be established. Using a range finder, the midpoint of the stream running between the cutblock edges was approximated. Facing the stream, Plot #1 were established 25m to the left of the midpoint, and plot #2 established 25m to the right of the midpoint. Plot #1 will always be to the left of the midpoint when facing the stream for all sampling areas. The midpoint per se will serve only as the sampling point for the 'Suspended Tree Index sample #2' of 3 (see below) and no other vegetation sampling will be done at this point. Where logging roads intersect the stream, a coin toss determined which side of the road to sample and, cutblock space provided, either Plot #1 or Plot #4 (depending on the orientation of the road) was established 50m away from the road. If roads bisected the stream unevenly, then the sampling zone was established on the long side of the riparian buffer with the closest sampling point to the road at least 50m from the road edge.

***Control Plots*** – A coin toss was used to select the side of the stream to be sampled. Because access was an important factor in site selection, control plots were close to roads, but the establishment of plots was at least 50m from the access road and away from any obvious vegetation edge effects from road presence. As such, if the stream and the road were perpendicular, the midpoint between Plots #1 and #4 was at least 75m from the road edge.

During the pilot study in Nov-Dec 2003, we found no appropriate control streams within the TFL. Most 'unharvested' streams within the TFL were mature second-growth of significantly different structure relative to the old growth forest left behind in most harvest area buffer strips. Early season snow levels prevented complete reconnaissance of many potential control sites in nearby provincial park areas, but it should be assumed at the onset that most control sites will be outside the TFL, or possibly coordinated with other relevant riparian research in other similar areas on Vancouver Island.

***Plot Establishment*** – From the midpoint of the transect line, a logger's tape was run 25m along the stream in both directions (Figure 1). At a distance of 25m from the midpoint, the plot center for either Plot #1 or #4 was established 6m from the edge of the stream bank (typically the water's edge). The water's edge was defined as the last line of intact, rooted vegetation before either the water, or the high-water streambed (sometimes a dry area).

From these plot centers an aluminum tree tag were nailed to the nearest tree or large piece of CWD (noting bearing and distance from tag to plot center) and a UTM reading was taken using a hand-held GPS. To minimize spatial error in site location, an 'average' reading is recommended for the GPS waypoint.

Tree tags were fixed with aluminum nails or screws in a visible location. This was not necessarily on the uphill side as described in the RIC standards – uphill positioning was not always logistically safe to accomplish on steep creek banks during the pilot study. Nails or screws were put into tree trunks enough to secure the tag, but not completely to obstruct future radial growth of the tree. Often the exact plot center was close to, or at least within 30cm, from a tree or a large piece of CWD that the plot tag could be fastened onto. Sometimes this was not possible. If no proper fastening point was available for the tree tag, then a bearing and measurement was taken from a marked plot to the plot center. For repeated-measures vegetation monitoring it is important that resampling is done as close to the original plot center as possible, so these distances should be noted in sufficient detail. The bearing from the stream's edge to the center of plot #1 and Plot #4 will often be different. These are to be perpendicular to the stream edge at the location of the respective plot, and this will vary from Plot #1 to Plot #4 dependent on the variability in stream direction between points. These bearings were recorded and used to establish the other 4 plots.

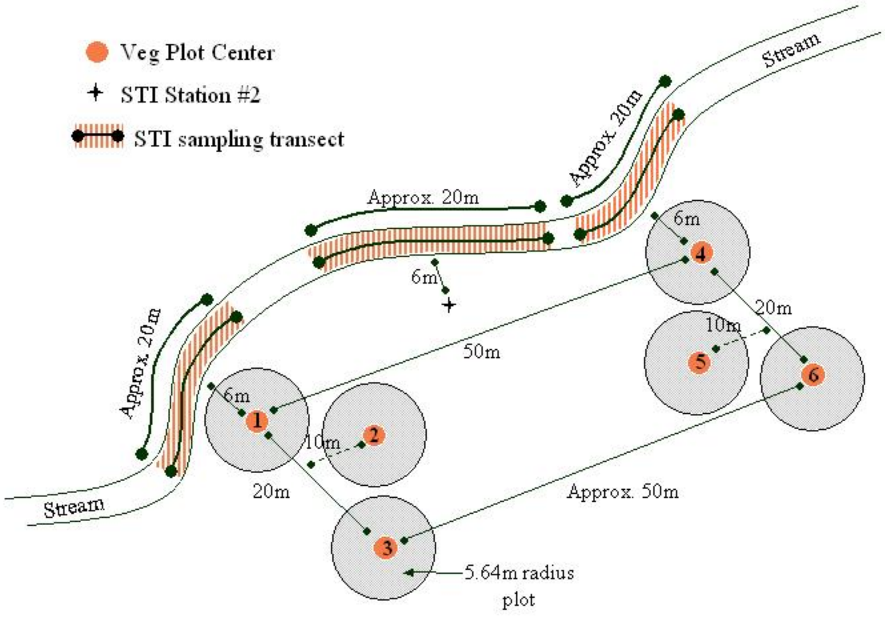


**Figure 1**. Forest structure sampling plot schematic (not to scale).

Using the respective bearing recorded when measuring the 6 meters perpendicular to the stream's edge, a logger's tape was run out away from the stream from Plot #1 and Plot #4. At 10m away from the stream from Plot #1 or Plot #4 a 90° bearing was taken off this line towards the center point, and either Plot #5 or Plot #2 was established 10m off this line. Using the respective bearing recorded when measuring the 6m distance perpendicular to the stream's edge, a logger's tape was run out 20m away from the stream from Plot #1 or Plot #4, the end of which marked the center location of either Plot #3 or Plot #6.

On harvested streams these plots were often out of the forested buffer strip and in the cutblock (at least 6 meters beyond the 20m RMZ). All plot centers were marked with an aluminum tree tag and GPS locations recorded in UTM's. Often Plots #1 and #4 were in ravine areas where GPS coverage was not available. These locations were recorded using systematic measurements from Plots #6 and #3 where, further out of the ravine, GPS coverage is more reliable. Around all plot centers, a 5.64m radius vegetation sampling plot was established, within and around which forest structure was measured.

## Terrestrial - Forest Structure Methods

Forest Structure measurements will include:

| | |
|---|---|
| • Canopy cover | • Windthrow (suspended tree index) |
| • Shrub cover | • Coarse woody debris |
| • Tree basal area (or tree density) | |

***Canopy Cover*** -- Canopy cover was estimated from each plot center. This measurement was an estimation of the overhead percent cover for dominant and intermediate tree crowns within the 5.64m sampling plot. The portions of tree crowns that entered the plot circle from trees that were rooted outside of the plot circle were included in the estimate. Crown from trees that were less than 20 m were not included in the canopy cover estimate. This estimate was done visually with no special equipment. Canopy cover was estimated using Figure 2 as a guide.

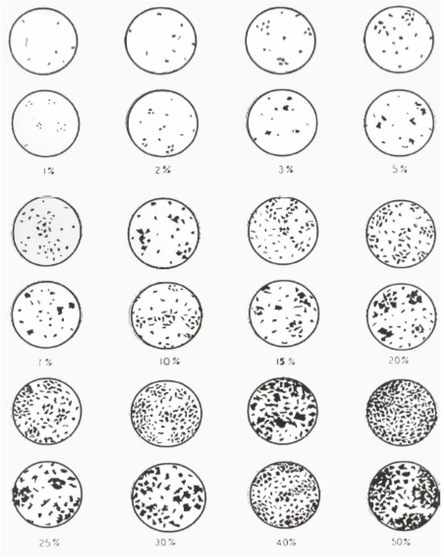


**Figure 2**. Sample Cover Estimates.

***Shrubs*** -- All species of shrubs found within the 5.64m radius plot circle were recorded. These were recorded as either “tall” (>2m height but < 10m height) or “short” (<2m height) for each species. Percent cover was estimated as the percent of ground covered by each species within the 5.64m radius plot. A percent coverage for each species, both tall and short, was estimated. A total coverage for all tall shrubs was estimated, and a total coverage for all short shrubs was also estimated.

***Basal Area and Trees*** -- Trees in this plot were defined as >10m in height and greater than 12.5 cm DBH. Tree species less than this height were recorded as tall shrubs. All trees that had *stem centers* within the 5.64m plot radius were recorded. For each tree that fell within the plot radius the species, DBH (cm), decay class (see below), canopy layer (see below), and height (in meters) were recorded.

Tree Decay Class

Tree decay class was assessed using Figure 3 as a guide.

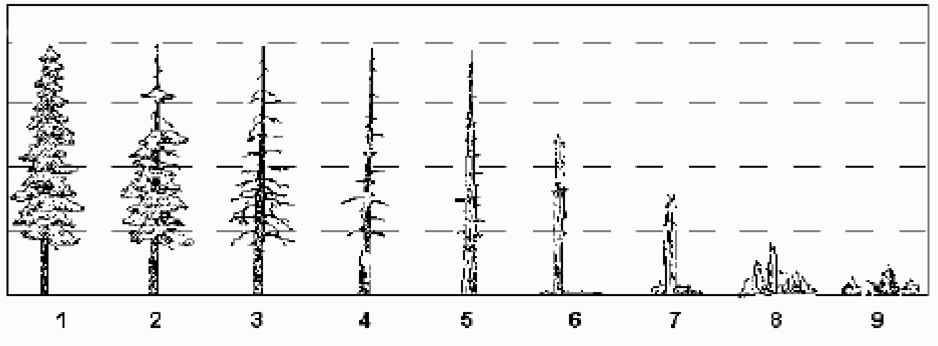


**Figure 3**. Standing tree decay class guide.

Canopy Layer

Canopy trees were classed as follows:

| **Class** | **Description*** |
|---|---|
| | |
| Dominant | *Trees with crowns that extend above the general level of the crown canopy (may include trees, shrubs or other obstructions) immediately around the measured trees. They are somewhat taller than the codominant trees, and have well-developed crowns, which may be somewhat crowded on the sides, receiving full light from above and partly from the side.* |
| | |
| Codominant | *Trees with crowns forming the general level of the crown canopy (may include trees, shrubs or other obstructions) immediately around the measured trees. The crown is generally smaller than those of the dominant trees and is usually more crowded on the sides, receiving full light from above and little from the sides.* |
| | |
| Intermediate | *Trees with crowns below, but extending into, the general level of the crown canopy (may include trees, shrubs or other obstructions) immediately around the measured trees. The crowns are usually small and quite crowded on the sides, receiving little direct light from above but none from the sides.* |
| | |
| Shrub | *Trees with crowns entirely below the general level of the crown canopy (may include trees, shrubs or other obstructions) around the measured trees, receiving no direct light either from above or from the sides.* Less than 10m in height. |
| | |

*Quoted directly from RICS 2003

Trees considered "wildlife trees" were specifically noted within all plots. For the purposes of this report, wildlife trees were defined as standing dead trees (classes 3-8, see Figure 3). Live trees that show some physical defects (e.g., dead or broken tops, large stem scars or cracks, evidence of internal decay) were considered class 2 wildlife trees, but were not differentiated from live, defect-free class 1 trees for purposes of analyses.

Prism sweeps were conducted from the plot centers of Plot #1 and Plot #4 at each riparian sampling site; sweeps were conducted in a semi-circle oriented away from the stream. A semi-circle was used in order to include only trees occurring within the riparian buffer strip, and therefore excluded those portions of the stream channel that may have been of varying widths (i.e., therefore adding a potential bias to these measurements). A 90° angle from the original random bearing used to establish the first plot was used to define the line beyond which trees were included in the 180 degree BAF sweep. Data were recorded for all trees that occurred within the prism sweep plot (see Figure 4). A #10 Basal Area Factor (BAF) prism was selected to optimize the number of trees recorded at prism plots. *However, because of some logistic difficulty in acquiring a range of BAF prisms, fixed radius plot (5.64 m) measurements were also used to calculate stem density/ha values*. Tree species, diameter at breast height [DBH (cm)], decay class, canopy layer, and height (m) were recorded for all trees that fell within the prism sweep. Tree diameters were measured using a diameter tape, and tree heights were measured using a clinometer.

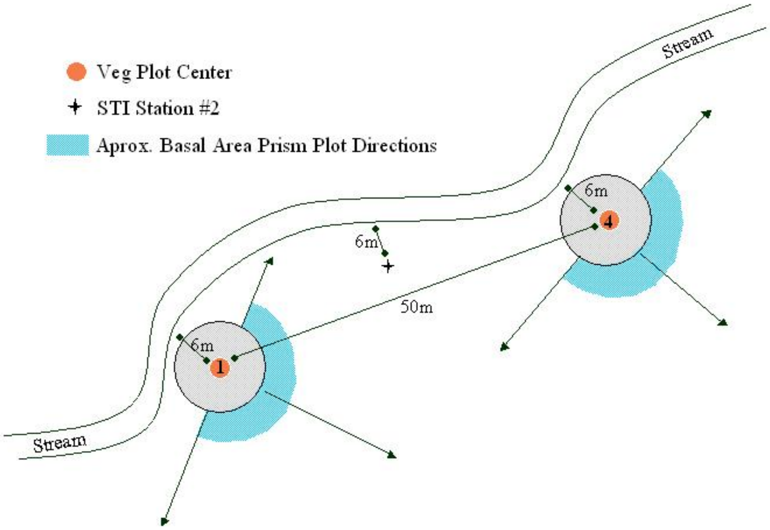


**Figure 4**. Prism Plot Example.

***Coarse Woody Debris*** – Four coarse woody debris transects, each 24m in length, were completed. Two started from the center of Plot #1 and two from the center of Plot #6.

### Plot #1 CWD Transect Establishment

Two 24m transects were run from the plot center of Plot #1. For the first transect a random compass bearing was chosen from within the 180° *facing the stream*. This 180° limitation was to acquire and specifically include CWD information along the stream bank that typically included blowdown events and streamside CWD accumulations that would be missed if transects frequently were directed away from the stream and into the forest or adjacent cutblock. If the first transect hit the stream edge, it “bounce back” 180° along the same line resampling the same pieces of cwd (see RIC 2003 Ground VRI Methods) until a total of 24m had been completed. Bounce-back’s are to be noted clearly in field notes. The second transect was established +90° from the first compass bearing with no directional limitations in reference to the stream. Again, if this transect hit the stream edge it bounced back 180° along the same line, resampling the same CWD, until 24m of sampling was completed.

### Plot #6 CWD Transect Establishment

Two 24m transects were run from the center of Plot #6. Both transects were 24m in length. The first transect was established by selecting a random compass bearing with no directional limitations. The second transect was established +90° from the bearing of the first transect. If obstructions were met (including unsafe terrain) then the same "bounce-back" rule applied along each transect as described above.

### CWD Sampling on all Transects

Pieces of CWD were counted following the VRI methods outlined in the 2003 RIC guidelines (see RIC 2003). All pieces of CWD greater than 7.5 cm in diameter (or an equivalent area at the crossing of 44.2 square centimeters measured perpendicular to their length for odd shaped pieces) where the center line of the wood piece was crossed by the vertically projected transect were measured. This included fallen or suspended (not self-supporting) dead tree boles (with or without roots attached), fallen trees with green foliage if they were no longer rooted to the ground, recently cut logs, uprooted (not self-supporting) stumps greater than 7.5 cm in diameter and less than 1.3 meters in length, and any exposed dead roots greater than 7.5 cm that had center lines crossing the sampling transect. To be measured the CWD piece must have been greater than 7.5 cm in diameter (or equivalent) at the point where the line transect crosses the center line. The center line was defined as the midline of any section of wood, and did not necessarily correspond to the pith.

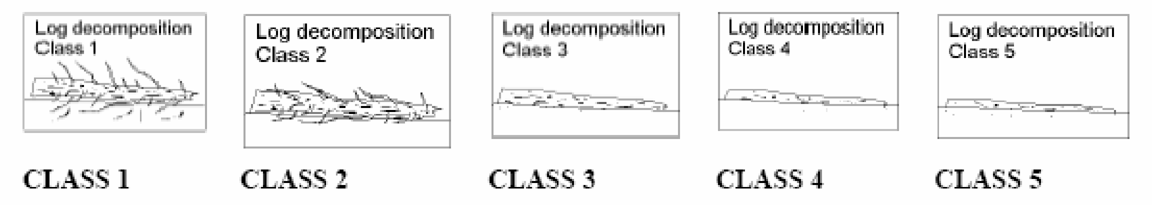
Coarse woody debris measurements did not include live or dead trees still rooted which were self-supporting, dead branches still connected to standing trees, exposed roots of self supporting trees, or self-supporting stumps or their exposed roots. A piece was no longer considered CWD when the wood had decomposed to the point where it was described as forest floor humus.

For each piece of CWD the following was recorded: tree species (if known – recorded as "unknown hardwood" or "unknown conifer" if not known), diameter perpendicular to the bole at transect line crossing (cm), and decay class. For irregular pieces of wood (non cylindrical) the length and depth of the wood touching the transect line was measured. CWD volumes were calculated as follows:

**Volume for regular (cylindrical) pieces ($m^3$/ha) = ($\sum$diameters)$^2$ x (1.234 / transect length)**

**Volume for irregular pieces ($m^3$/ha) = [$\sum$ (Length x depth)] / 48m**

Decay class for each CWD piece was described as follows:



***Suspended Tree Index*** (STI)– A count of suspended trees across the stream channel was completed at Plot #1, Plot #6 and at the midpoint 6m from the stream edge. *Suspended trees were defined as trees that had fallen and were currently suspended across the stream channel in bridge fashion* (see Figure 5**).** Trees included in this count must have been crossing the center of the stream. From each of the three sampling points (PLOT #1, PLOT #4 and the midpoint), observers estimated a 20m transect running along the center of the stream (i.e. for 10 meters on either side of the sampling point). All suspended trees greater than 12.5 cm dbh at the line crossing point were counted. These were typically recently fallen, blowdown trees (Figure 5). To separate old and newly fallen trees, counted trees were grouped into decay class 1 and 2 inclusive, and decay class 3 and 4 inclusive. Counted trees were limited to those that were visible from the plot centers without having to walk around to see past trees or shrubs that obstruct views of the stream channel. Observers arched and leaned to see past obstructions, but did not hike about to improve visibility. We assumed that any bias regarding foliage obstructions would be roughly equal across the study.

**Figure 5**. Example of suspended trees over a stream channel in TFL #37 for the Suspended Tree Index.

## Terrestrial Salamander Methods

Counts, indices, and estimates of animal populations, including animal sign and area use, form the basis for Wildlife Metrics in monitoring programs. The cornerstone of wildlife monitoring programs is an estimate of the abundance of a particular population. Abundance can be measured in two ways (Krebs 1989). *Absolute density* is the number of individuals per unit area, and *relative density* is the density of one population relative to another. Relative density is often calculated with some biological index that is correlated with absolute density, and it is much more efficient and inexpensive to obtain because it does not require the identification of every individual within a specified area. This monitoring program uses measures of relative density, and defines Wildlife Species Use as the number of individuals encountered or captured per unit of search or capture effort.

In TFL 37, three species of salamanders likely will be encountered: the Clouded Salamander (*Aneides ferreus*), the Northwestern Salamander (*Ambystoma gracile*), and the Western Redback Salamander (*Plethodon vehiculum*). Each of these species has a terrestrial adult life stage that will be the focus of abundance estimates. Davis (1996) suggests that these species have similar micro-habitat associations within the forests of Vancouver Island; under rocks and bark, under or within logs, on soil, in underground burrows, or within leaf litter at the base of vegetation. The BC RIC (1999) Inventory Methods for Salamanders summarizes their habitat associations as follows:

| Species | Habitat associations* |
|---|---|
| | |
| Clouded Salamander (*Aneides ferreus*) | *Can be found in moist terrestrial habitats such as under exfoliating bark and in cracks and cavities of decomposing logs, stumps, and snags, in talus, and occasionally in trees (Nussbaum et al. 1983; Stebbins 1985; Davis and Gregory 1993; Leonard et al. 1993; Davis 1996, 1998). Davis (1996, 1998) usually found A. ferreus in decay class 3 logs, and much less commonly in logs of the other decay classes. They are very rarely found under coarse woody debris (CWD) on the soil surface.* |
| | |
| Western Redbacked Salamander (*Plethodon vehiculum*) | *Are found within leaf litter and Sword Fern (*Polystichum munitum*) bases, under moss, rocks or CWD on the forest floor, and under or among rocks on talus and rock outcrops (Leonard et al. 1993). They favour damp, but not wet, shady areas of the forest and can be very abundant (Davis 1996, 1998).* |
| | |
| Northwestern Slamander (*Ambystoma gracile*) | *Terrestrial habitat poorly known. Underground burrows, occasionally under or within logs; some populations may be neotenic.* |
| | |

*Direct quotes from BC Resource Inventory Committee (1999).

Similar habitat requirements for the adult stage of these animals implies that one standardized sampling technique would be applicable to all. Because we expect the cover attributes among study areas to vary enough to affect constrained search efforts, we will use artificial cover objects (ACO's). ACO's are especially well suited to studies that require repeated sampling, are relatively easy to sample once in place, result in little or no damage to the natural habitat, and can attract species that are difficult to trap in pitfall traps, or locate via searches in areas where LWD is excessive (BC Resource Inventory Committee 1999). Generally, any piece of untreated board placed on the forest floor will attract *P. vehiculum*, but *A. ferreus* requires a more complex cover board. Davis (1996) suggests a design that consists of a baseboard and two cover boards and that is suitable for all three species (Figure 6). This design is outlined in detail in the salamander sampling methods (RIC 1999).

Artificial cover objects will be created from one 1.8 m long untreated rough cut 2 x 12 base board (maximum length), two 1.8 m long untreated rough cut 1 x 6 cover boards, and strips of cedar lath. The strips of cedar lath are nailed to the top surface of the baseboard with galvanized nails. The top cover boards are placed on top of the baseboards to create a wedge-shaped space between the cover boards and the baseboard. Rainwater can drip through the crack between the two top cover boards into this space. This creates a complex microhabitat so that a salamander could be found on the soil under the baseboard, or between the baseboard and the top boards.

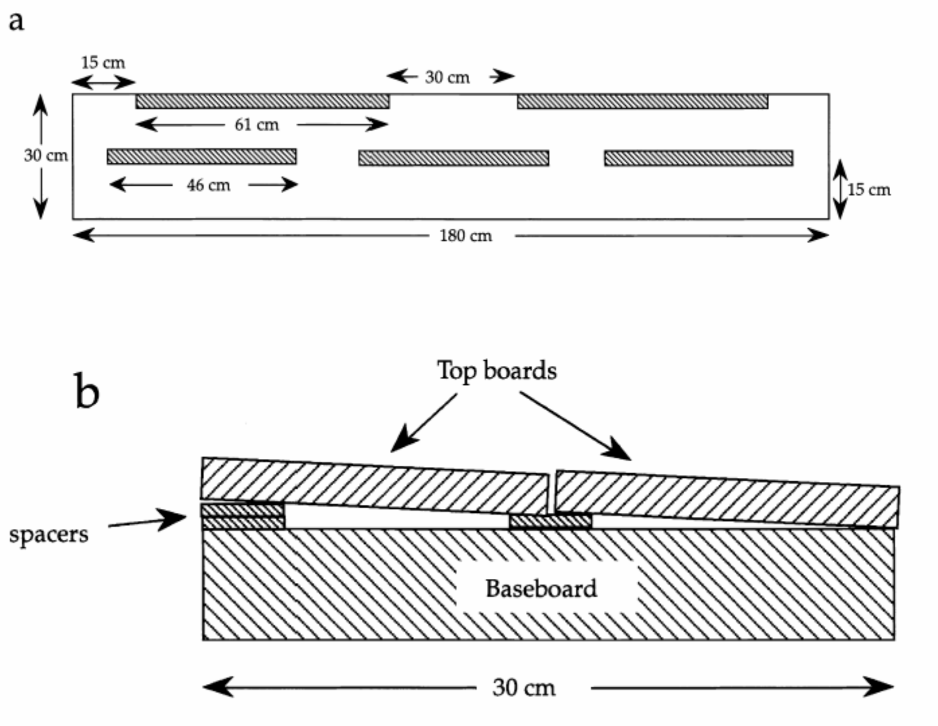


**Figure 6**. Top (a) and side (b) view of the artificial cover board (picture copied directly from the RIC Terrestrial Salamander Sampling Guide 1999).

Artificial cover boards will be placed within each vegetation sampling plot (see Figure 1 for reference). Baseboards will be cleared of vegetation and placed flat on the soil surface. The 2 cover boards will be placed on top as shown in Figure 6b creating a complex microhabitat for salamanders under the baseboard, or between the baseboard and the top boards. When checking the boards, the top boards will be removed to one side, and the upper surface of the baseboard will be checked immediately before any salamanders escape into the forest littler layer where they are difficult to relocate. Salamanders will be held in containers or plastic bags while setting the boards back in place. The number and species of salamanders will be recorded, and captured animals will be released at the point of capture. Detailed measurements will not be taken on each animal. The goal is to record the number of captures per trapping effort. Detailed information for each captured animal will not be necessary for monitoring purposes.

## AQUATIC SAMPLING METHODS - STREAMS

Stream sections were surveyed to assess the effectiveness of riparian management in maintaining the ecological function of the aquatic habitat. To accomplish this, six habitat indicators were measured at set intervals (see below) along each stream's channel. These included:

- Bank erosion
- Bar frequency and type
- Logjam frequency
- LWD volume
- Fish cover types
- Loss of streamside trees

### Plot Locations

Stream sampling areas were established near the terrestrial sampling plots (see *Plot Location* section under *Terrestrial Methods* above). Sampling areas began at the most downstream terrestrial sampling plot, and continued upstream for the required number of bankfull channel widths.

***Measurement of average bankfull channel width*** – The average width of the channel was determined and used as the set distance that separated each sampling point along the stream. The average channel width was determined by measuring the channel width at one location near the start of the survey (typically near the first terrestrial sampling plot). As much as possible, this measurement was taken at a location that was representative of the reach.

A number of standard criteria were used to determine bankfull width, and only those relevant to each field site should be used. The standard criteria are:

- a change in vegetation from bare ground with no trees to vegetated ground with trees, from no moss to moss covered ground or from bare ground to grass covered ground;
- a topographic break from vertical bank to flat floodplain;
- a topographic break from steep bank to more gentle slope;
- the highest elevation below which no fine woody debris (needles, leaves, cones or seeds) occurs; and
- a change in texture of deposited sediment from clay to sand, or sand to pebbles, or boulders to pebbles.

***Establishing the Survey Interval*** -- Once the average channel width was established, this measurement became the survey interval. For example, if the average channel width was 7.5 m, then the six indicator parameters are measured every 7.5 m along the channel for 50 survey intervals.

## Aquatic Indicator Methods

Once the survey interval was established, the channel was walked and observers used a hip-chain to record the distances covered on foot. Every survey interval was measured along the thalweg (the deepest portion of the channel) and from here data pertaining to the six indicator parameters was recorded in a field book. Standards for the stream sampling methods outlined below are detailed in RIC (1999).

***Bank Erosion*** -- At each survey interval (one bankfull channel width), the observer determined if the banks were either:

(a) undercut and/or overhanging, or
(b) vertical and/or sloping.

In a sinuous channel, bank erosion is expected to alternate from one side of the channel to the next. For the purpose of this procedure, one bank was chosen for assessment. If only one bank was adjacent to a cutblock then that side was assessed. However, both sides were assessed if there were management activities on both sides of the stream.

***Bar Frequency and Bar Type*** -- The presence and type of gravel bar was recorded at each survey interval using the characteristics outlined in Figure 7. Gravel bars were defined as areas within the channel that had an accumulation of sediment of approximately one bankfull width and one bankfull depth in size. If a single bar extended beyond one survey interval, it is counted at each interval (e.g., if a bar extended for five bankfull channel widths, it is counted and classified five times).

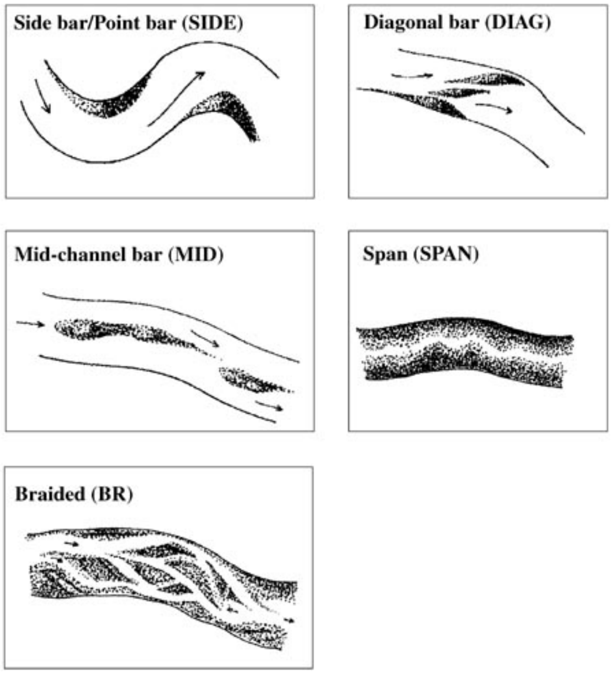


**Figure 7**. Classification system for gravel bars (from Figure 9 of the BC Forest Practices Code Guidebook Channel Assessment Procedure (RIC 1996))

***Logjam Frequency*** -- The frequency of logjams were recorded by noting the presence or absence of logjams at each survey interval. A logjam was defined as an accumulation of logs tightly packed together (i.e., there are no obvious holes in the structure) that span, or nearly span the channel.

***LWD Volume*** -- in (a) Small (bank width (Wb) < 3 m) , (b) Medium (Wb 3 – 6 m), and (c) Large (Wb > 6 m) channels.

The percent cover of LWD at each survey interval was recorded as a way to describe the volume of large woody debris (LWD) in the channel. Percent cover was estimated visually to the nearest 5 %.

***Number of Fish Cover Types*** -- The number of different fish cover types was enumerated over the length of the sampling zone (between the first and last sampling interval). Basic types of cover enumerated often include:

- overhanging vegetation within 1 m of the channel surface
- overhanging LWD
- in-channel LWD
- stable small woody debris
- stable undercut banks
- non-embedded boulders and cobbles that are stable at high flows
- deep quiet water
- aquatic vegetation

To be counted as cover, at least one dimension of the area encompassed by the cover should be equal to or greater than 10 % of the channel width.

***Stream Bank Tree Loss*** -- The loss of streamside trees was determined by counting stumps along the top of the stream bank, or fallen trees that originated from within the stream channel. Only those trees whose roots maintained stream bank stability (i.e., within ~2 m of the bank edge) were included in this count.

*Photographic Record* **--** Using a tripod-mounted 35 mm SLR camera with a 28 mm wide angle lens, photographs were taken for each stream channel. Photographs were taken from photo-stations established as close to mid channel as possible. Photographs were not taken at every survey interval. The number and location of all photo-stations were positioned to provide a photographic record for a 'significant portion' of the section of stream surveyed. The distance, or position within the survey reach, for each photo-station was recorded. Photographs looking upstream as well as downstream were taken at each photo-station. In subsequent years photographs should be re-taken from approximately the same position within the survey reach. An example of a photograph taken during the 2003 pilot monitoring program is provided in Figure 8. The average stream width in this photograph is 7.5 m and the channel gradient is approximately 5 %.

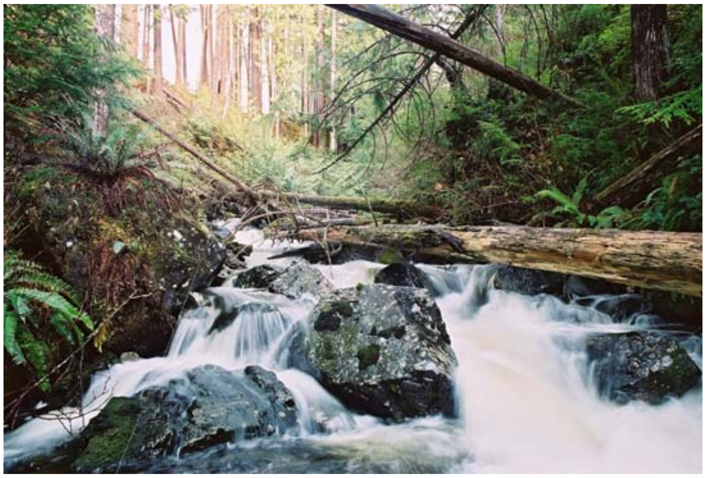


**Figure 8**. Representative photograph taken as part of the riparian effectiveness monitoring program in Dec. 2003.

# RESULTS – TERRESTRIAL SAMPLING

## Time required to complete sampling

Twelve stream sites were sampled in Nov.-Dec. 2003 using the terrestrial survey protocol described earlier (see *Methods* section) for forest structure. The time required to complete a site (i.e. 6 plots) ranged from 80 to 200 minutes with an average time of 121 minutes (Table 4). S3 harvest streams took on average 30 minutes longer to sample (mean = 137 minutes) than S2 harvest streams (mean = 106 minutes). Sites with high levels of blowdown and CWD required more time to sample.

Times and averages listed above do not include travel time or hiking time required to get to the survey plots. However, these times do include time required to take GPS locations and affix plot tags at the center of each vegetation plot. Once sites are established and plots are located and completely tagged, the time required to complete these surveys will be decreased by approximately 15 minutes.

## Forest Structure

The primary goal of the pilot study was to assess the variability of each indicator, and to ensure that the measuring standards selected were feasible given the field setting. We acquired data for all indicators selected. All methods used were deemed appropriate except for the BAF measurements. The BAF prisms we had available did not account for enough trees given the tree densities of most riparian stands in the area, therefore *basal area calculation were subsequently changed to calculations (using stems per hectare)* based on the circular fixed radius (5.64 m) vegetation plots, rather than using BAF calculations. Averages and associated errors were calculated for continuous and count variables measured at each site (Table 5).

Species-level data (i.e. diversity or community composition) were summarized by plot and site occurrence. Data on shrub species composition, separated by height, were summarized by counting the number of times each species occurred in a plot for each site (Tables 6 and 7). 20 shrub species were observed, distributed among all study sites, with the most common being *Vaccinium parvifolium, Gaultheria shallon, Rubus spectabilis* and *Tsuga heterophylla.* Similar summaries were done for tree species (Table 8), and were summarized as occurrence of tree species by site for all study sites. A total of eight tree species were observed; the maximum number of occurrences per site was six. The most common trees were western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*), which are consistent with our site selection parameters for general habitat type (i.e., BEC unit).

The site occurrences of standing trees by decay class were summarized for each study site (Table 9). Decay classes 1 and 2 (live healthy and live with defects, respectively) were most common throughout. Class 3-5 standing dead trees (hard snags, intact height) were found variably at the sample sites and at low numbers. Class 6 trees (dead spongy, about 1/3 of original tree height broken away) occurred with low counts (but roughly equal occurrences) at most sample sites.

Coarse woody debris volumes were summarized by decay class within stream type (Figure 9). CWD volumes for S2 streams ranged from 124 $m^3$/ha (decay class 4) to 324 $m^3$/ha (decay class 1). S3 streams had CWD volumes ranging from 79 $m^3$/ha for class 4 pieces, to 431 $m^3$/ha for class 1 pieces. For both stream types, decay class 1 and 2 CWD (i.e., hard, recently fallen and intact) were present in greater volumes than the more decayed class 3 and 4 CWD (i.e., not suspended above ground, partly decaying to decayed into blocky pieces).

The initial average values (Table 5) of forest structure indicators were tested for correlations between indicators (Table 10). Canopy cover was significantly positively correlated at alpha = 0.05 with tree density (stems per ha, *Pearson's Correlation Coefficient (PCC)* = 0.66, $P = 0.02$). Tree height and diameter (DBH) were highly correlated at the 0.01 level (PCC = 0.85, $P < 0.001$). STI was negatively correlated with tree density at the 0.05 level (PCC = -0.6, $P = 0.04$). None of the other indicators were significantly correlated with each other (e.g., canopy closure and shrub cover, etc.).

The six forest structure indicators measured in the pilot study produced a range in coefficients of variation (CV) from 0.3 (tree height) to 1.0 (STI counts) (Table 11). Using these overall CV values, along with initial plot values reported as averages and standard deviation (Table 5), simulations were run to approximate sample sizes required to detect different rates of change for the selected indicators. We simulated required sample sizes for: 1) the indicator that showed the highest coefficient of variation of all of the selected indicators, and 2) the indicator (tree density [stems/ha]) with the median CV of all indicators. Approximating sample size for the most variable (highest CV) indicator (STI counts) ensured that sample size, and thus power to detect change, for less variable indicators would be sufficient if not over-estimated.

Statistical power calculations required an estimate of the mean, standard deviation, and the CV for each indicator. From these values we ran simulations for different sampling intensities and frequencies using PROGRAM MONITOR (Gibbs et al. 1998) to approximate required sample size. Initial mean and standard deviation values are key to this approximation and were entered as plot averages (Table 5) into the simulation software as the initial values (see Gibbs et al. 1998).

Simulations were run for sample sizes of 12, 15 and 20 sites (6 plots per site as outlined in the methods section) for the most variable indicator (STI), and 6 and 10 sites for the indicator with median variability (tree density). It was assumed for simulation purposes that sites would initially be sampled once per year for 10 years and statistical tests would be 2-tailed with alpha = 0.1, to detect exponential changes between 1% and 3%. The exact statistical power presented here is only an approximation and will vary depending on the numerous and potential statistical tests that could be done on the indicator data.

Given the survey protocol outlined in this monitoring program, to have a 90% chance of detecting at least a 3% exponential change in the most variable indicator (i.e., STI) over 10 years of sampling, a minimum of 20 sampling sites are required (Figure 10). Only the "20 sites" line fell within both of the minimum limits indicated by dotted lines on the power graph. Power to detect positive changes will be higher than that required to detect negative changes, but this seems relevant only if <20 sites are surveyed. Given the initial variability noted in the data from the pilot study, a gain of 10% in statistical power requires the addition of roughly 8 sampling sites. **Power decreases significantly if sites are sampled less over longer periods of time**. Power will increase further if one-tailed tests are used in routine, future analyses. Therefore, **when only considering STI,** initial sample size should be set at 15-20 sites per stream type, and sampling should commence once per year for the first few years until temporal variability can be assessed. Because this is based on the most variable indicator, it is highly likely that actual sample sizes and sampling frequency will be less. Statistical power will be greater for the other indicators because their CV values are lower than those initially found for the STI indicator used in this simulation. This was examined using tree density as the simulated indicator.

We simulated statistical power to detect change in stem density (stems/ha counts) with 6 and 10 sampling sites sampled once per year for 10 years (Figure 11). **Six sampling sites (i.e. 6 streams per stream type) appear adequate to detect change in the indicators with average CV's** (i.e. essentially all indicators except shrub cover and STI). An increase to 10 sampling sites per stream type did not notably increase the power of the test.

The simulations presented here suggest that for each treatment type in the monitoring program (i.e. stream types – S2, S3, and control), approximately 15 sites be selected to have a 90% chance of detecting change in the most variable indicator, and **6 sites be selected to detect change in *most* indicators**. These simulations provide both a realistic (median) and conservative estimate of sample size. More extensive sampling in the first 2-3 years of the monitoring program might produce data with lower variability. At that time sample size can be adjusted accordingly or indicators can be dropped from the program if necessary. Alternatively, sampling frequency can be adjusted as required. To accurately evaluate statistical power it will be necessary to conduct several statistical tests with many data sets collected during the first 3 years of this monitoring program. The simulation presented here, however, sets a conservative, empirical rational for initial sample size determination, and can be adjusted as monitoring proceeds.

**Table 4.** Time required to complete the terrestrial survey protocol at each stream site sampled for riparian effectiveness monitoring in TFL 37.

| Site | Treatment | Time Required (min) |
|---|---|---|
| RO63 | S2 HARVEST | 136 |
| HT017 | S2 HARVEST | 147 |
| STREAM #26 | S2 HARVEST | 80 |
| KT039 | S2 HARVEST | 85 |
| HI036 | S2 HARVEST | 97 |
| TS028 | S2 HARVEST | 92 |
| | | |
| KT039 | S3 HARVEST | 181 |
| NS002 | S3 HARVEST | 169 |
| STREAM #64 | S3 HARVEST | 200 |
| MK019 | S3 HARVEST | 107 |
| K304 | S3 HARVEST | 82 |
| STREAM #38 | S3 HARVEST | 80 |
| | | |
| **Mean** | | **121** |

**Table 5.** Within-site averages and standard deviations for terrestrial indicators measured in TFL 37 (mean±SD). Sample size is indicated in brackets under each indicator title.

| site | **Canopy Closure(%) (6)** | **Shrub Cover(%) (6)** | **Tree Density (stems/ha) (6)** | **CWD Volume ($m^3$/ha) (2)** | **Tree Diam. (dbh, cm) (6)** | **Tree Height(m) (6)** | **STI (3)** |
|---|---|---|---|---|---|---|---|
| **HI036** | 47±12.4 | 6.8±11.3 | 43.3±17.5 | 380±55 | 71.3±29.1 | 33±10 | 3.3± 2.8 |
| **HT017** | 33.3±16.3 | 7.6±8.1 | 40.0±22.8 | 603±62 | 71.1±62.5 | 27.3±14.6 | 1.3±0.5 |
| **K304** | 26.6±19.4 | 13.3±20.0 | 20.0±8.9 | 1076±1056 | 20±3.3 | 10.5±3.6 | 3.3±3.5 |
| **KT039 S3** | 10±24.4 | 10.7±18.5 | 20.0±15.4 | 601±26 | 37.4±25.9 | 17.1±12.8 | (zero) |
| **MK019** | 18.6±23.0 | 4.9±5.5 | 16.0±8.9 | 897±997 | 57±26.2 | 32.3±16.7 | 8.6±3.5 |
| **NS002** | 36.6±23.3 | 2.5±2.7 | 46.6±43.6 | 309±15 | 30.5±15.2 | 21.8±9.7 | (zero) |
| **RO63** | 40±37.0 | 8.9±14.4 | 33.3±16.3 | 442±89 | 35.6±15.6 | 23.5±17.3 | 5.3±2.0 |
| **STREAM #26** | 14.3±8.8 | 15.3±20.2 | 33.3±16.3 | 551±252 | 37.6±28.1 | 24.7±9.9 | 1.3±1.5 |
| **STREAM #38** | 41.6±16.0 | 35.8±37.6 | 36.0±22.5 | 169±163 | 39.4±14.9 | 20.9±5.9 | 0.6±1.1 |
| **STREAM #64** | 18.2±18.2 | 16.2±30.7 | 15.0±12.2 | 407±195 | 59.3±19.2 | 32.1±6.7 | 8.6±4.6 |
| **TS028** | 30±24.6 | 12.6±16.1 | 50.0±30.9 | 543±371 | 38.6±39.8 | 21.0±9.9 | 1.6±1.5 |
| **KT039 (S2)** | 36.6±16.3 | 4.8±6.2 | 33.3±24.2 | 601±26 | 42.1±25.9 | 23.4±8.3 | 0.6±0.5 |

**Table 6.** Count occurrences of short shrub species (<2m in height) from terrestrial plots in TFL 37.

| | | *Species Code** | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Short Shrubs | | *a* | ***b*** | *c* | ***d*** | ***e*** | ***f*** | ***g*** | **h** | **i** | ***j*** | **k** | **l** | **m** | **n** | **o** | **p** | **q** | **r** | **s** | **t** |
| ***S2 Harvest*** | HI036 | | | | | | | | | | | | | | 5 | 4 | | | | 5 | |
| | HT017 | | 1 | | | 4 | | 1 | | | | 2 | | | 5 | 3 | 2 | | | 5 | |
| | KT039 (s2) | | | | | 2 | | 1 | | | 1 | | | | 4 | 1 | | | | 4 | |
| | R063 | | | 2 | | | | | 1 | | | | | 3 | 5 | 3 | 1 | | | 5 | |
| | STREAM #26 | | 1 | | 2 | | | 1 | 1 | | | | | | 3 | | 2 | | | 3 | |
| | TS028 | | | | | 1 | | | | | | | | | 4 | 3 | | | | 5 | |
| **S3 Harvest** | K304 | | | | | | | | 1 | | 4 | | | 3 | 3 | 2 | 1 | | 1 | 3 | |
| | KT039 | 1 | | | | 2 | | 1 | 1 | | | | | 1 | 6 | 6 | 4 | | | 6 | |
| | MK019 | | | | | | 1 | | 3 | 2 | | | | | 6 | 3 | 2 | | 1 | 6 | 2 |
| | NS002 | | | | | | | 1 | | | | | 1 | 1 | 5 | 4 | 1 | | | 1 | |
| | STREAM #38 | | | | | | | | | | | | | | 1 | | 2 | | | | |
| | STREAM #64 | | | | 1 | | | | 2 | | | | | 5 | 5 | 6 | 2 | | | 6 | |

*** (a)** Abies amabilis, **(b)** Vaccinium alaskaense **(c)** Alnus rubra **(d)** Menziesia ferruginea **(e)** Abies lasiocarpa **(f)** Rubus leucodermis **(g)** Oplopanax horridus **(h)** Peudotsuga menziesii **(i)** Abies grandis **(j)** Mahonia nervosa **(k)** Taxis brevifolia **(l)** Rubus ursinus **(m)** Thuja plicata **(n)** Vaccinium parvifolium **(o)** Gaultheria shallon **(p)** Rubus spectabilis **(q)** Picea sitchensis **(r)** Rubus parviflorus **(s)** Tsuga heterophylla **(t)** Pinus monticola*

**Table 7.** Count occurrences of tall shrub species (>2m in height) from terrestrial plots in TFL 37.

| | | *Species Code** | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Tall Shrubs | | *a* | ***b*** | *c* | ***d*** | ***e*** | ***f*** | ***g*** | **h** | **i** | ***j*** | **k** | **l** | **m** | **n** | **o** | **p** | **q** | **r** | **s** | **t** |
| ***<u>S2 Harvest</u>*** | HI036 | | | | | | | | | | | | | | 4 | | | | | 1 | |
| | HT017 | | | | | 3 | | | | | | | | | 3 | | 3 | | | 2 | |
| | KT039 (s2) | | | | | 3 | | | | | | | | | | | | | | 2 | |
| | R063 | | | 2 | | | | | | | | | | 3 | 3 | 1 | 1 | | | 5 | |
| | STREAM #26 | | | | 1 | | | | | | | | | | 5 | | 1 | | | 3 | |
| | TS028 | | | | | 1 | | | | | | | | | 6 | | | | | 5 | |
| **S3 Harvest** | K304 | | | | | | | | | | | | | | 4 | | | | | 4 | |
| | KT039 | 1 | | | 1 | 1 | | | | | | | | 1 | | | | | | 3 | |
| | MK019 | | | | | | | | | | | | | | 5 | | | | | 4 | |
| | NS002 | | | | | | | 1 | | | | | | 3 | 1 | | 3 | | | 2 | |
| | STREAM #38 | | | 1 | | | | | | | | | | | | | 5 | 1 | | | |
| | STREAM #64 | | | | 1 | | | | 3 | | | | | 1 | 4 | | 2 | | | 4 | |

** **(a)** Abies amabilis **(b)** Vaccinium alaskaense **(c)** Alnus rubra **(d)** Menziesia ferruginea **(e)** Abies lasiocarpa **(f)** Rubus leucodermis **(g)** Oplopanax horridus **(h)** Peudotsuga menziesii **(i)** Abies grandis **(j)** Mahonia nervosa **(k)** Taxis brevifolia **(l)** Rubus ursinus **(m)** Thuja plicata **(n)** Vaccinium parvifolium **(o)** Gaultheria shallon **(p)** Rubus spectabilis **(q)** Picea sitchensis **(r)** Rubus parviflorus **(s)** Tsuga heterophylla **(t)** Pinus monticola*

**Table 8.** Count occurrences of tree species (>5m in height) from terrestrial plots in TFL 37.

| **Treatment** | **Site** | *Species Code** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | **a** | **b** | **c** | **d** | **e** | **f** | **g** | **h** |
| S2 HARVEST | HI036 | | | | 1 | | 16 | | 12 |
| | HT017 | | | 7 | | | 3 | | 14 |
| | KT039 | | 1 | 7 | | | 2 | | 11 |
| | RO63 | | 3 | | 7 | | 3 | | 13 |
| | STREAM #26 | 2 | | 7 | | | 5 | | 10 |
| | TS028 | | | 17 | | | 5 | | 11 |
| S3 HARVEST | K304 | | | | 3 | | 1 | | 10 |
| | KT039 | 1 | | 6 | 1 | | 2 | | 2 |
| | MK019 | | | | 2 | | 5 | | 4 |
| | NS002 | | 2 | | 8 | | 5 | | 13 |
| | STREAM #38 | | 15 | | | 6 | 2 | 1 | 3 |
| | STREAM #64 | | | | | | 6 | | 2 |

* ***(a)*** *Abies amabilis* ***(b)*** *Alnus rubra* ***(c)*** *Abies lasiocarpa* ***(d)*** *Peudotsuga menziesii* ***(e)*** *Acer macrophyllum* ***(f)*** *Thuja plicata* ***(g)*** *Picea sitchensis* ***(h)*** *Tsuga heterophylla*

**Table 9.** Count occurrences of tree species (>5m in height) by decay class (see *Methods*) from terrestrial plots in TFL 37.

| **Treatment** | **Site** | **TREE DECAY CLASS** | | | | | |
|---|---|---|---|---|---|---|---|
| | | **1** | **2** | **3** | **4** | **5** | **6** |
| S2 HARVEST | HI036 | 3 | 2 | | | | 1 |
| | HT017 | 8 | 4 | | | | 1 |
| | KT039 | 7 | 5 | | | | |
| | RO63 | 5 | | | | | 1 |
| | STREAM #26 | 6 | 2 | | | | 1 |
| | TS028 | 10 | 1 | | | | 1 |
| S3 HARVEST | K304 | 3 | | | | | 1 |
| | KT039 | 4 | 3 | 3 | | | |
| | MK019 | 3 | | | | | |
| | NS002 | 17 | 3 | 1 | 2 | 3 | 1 |
| | STREAM #38 | 8 | 3 | | 1 | | |
| | STREAM #64 | 2 | 4 | | | | |

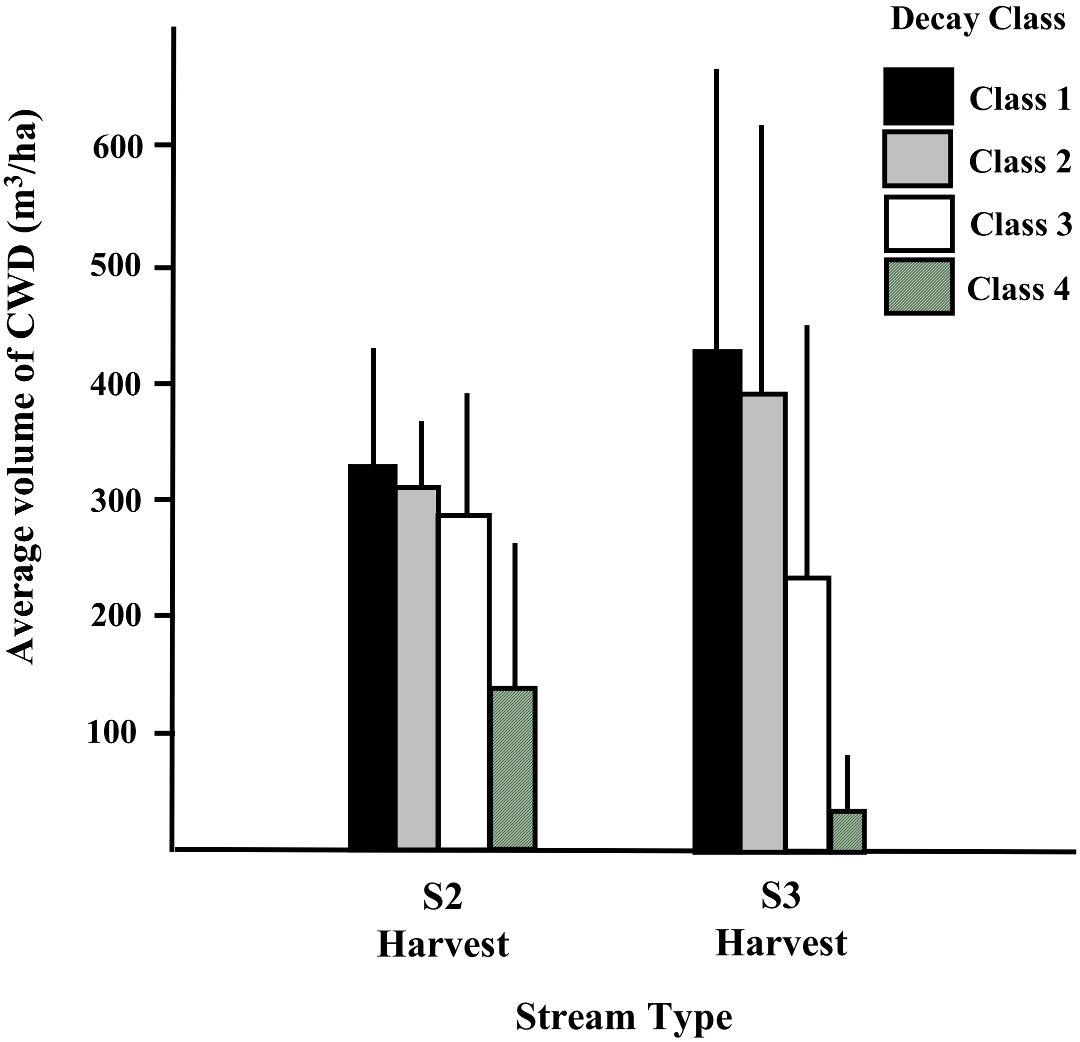


**Figure 9.** Average volume of CWD by decay class as measured in S2 and S3 study streams in TFL 37 (n = 6 for each stream type).

**Table 10.** Pearson correlation coefficients of initial site averages for terrestrial forest structure indicators sampled in TFL 37.

| | | CANOPY CLOSURE | SHRUB COVER | TREE DENSITY | CWD | TREE DIAM. | TREE HEIGHT |
|---|---|---|---|---|---|---|---|
| **SHRUB COVER** | Pearson Correlation | .027 | | | | | |
| | Sig. (2-tailed) | .935 | | | | | |
| | N | 12 | | | | | |
| **TREE DENSITY** (stems/ha) | Pearson Correlation | .658* | -.080 | | | | |
| | Sig. (2-tailed) | .020 | .805 | | | | |
| | N | 12 | 12 | | | | |
| **CWD** (volume) | Pearson Correlation | .308 | .313 | -.053 | | | |
| | Sig. (2-tailed) | .331 | .322 | .869 | | | |
| | N | 12 | 12 | 12 | | | |
| **TREE DIAM.** | Pearson Correlation | .147 | -.176 | .041 | -.256 | | |
| | Sig. (2-tailed) | .649 | .584 | .899 | .422 | | |
| | N | 12 | 12 | 12 | 12 | | |
| **TREE HEIGHT** | Pearson Correlation | .101 | -.222 | .004 | -.185 | .854** | |
| | Sig. (2-tailed) | .755 | .488 | .990 | .565 | .000 | |
| | N | 12 | 12 | 12 | 12 | 12 | |
| **STI** | Pearson Correlation | -.214 | -.106 | -.596* | .008 | .347 | .546 |
| | Sig. (2-tailed) | .505 | .744 | .041 | .981 | .268 | .066 |
| | N | 12 | 12 | 12 | 12 | 12 | 12 |

* Correlation is significant at the 0.05 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

**Table 11.** Overall averages for terrestrial forest structure indicator values (i.e. mean of plot means) for all sites. Sample size = 12 for each mean.

| Indicator | Average Value | Standard Deviation | Coefficient of Variation[a] |
|---|---|---|---|
| Tree Height (m) | 23.9 | 6.6 | 0.3 |
| Salamanders | NS | NS | 0.35 [b] |
| Tree Diam. (dbh, cm) | 45 | 16.1 | 0.35 |
| Tree Density (stems/ha) | 32.1 | 11.9 | 0.37 |
| CWD volume ($m^3$/ha) | 547.26 | 246.37 | 0.45 |
| Canopy Closure (%) | 29% | 22.5% | 0.7 |
| Shrub Cover (%) | 11.3% | 8.6% | 0.8 |
| Suspended Tree Index | 2.9 | 3.1 | 1.0 |

[a] listed in rank order
[b] from Table 1 in Gibbs et al. (1998), not sampled in TFL37 pilot study
NS = not surveyed

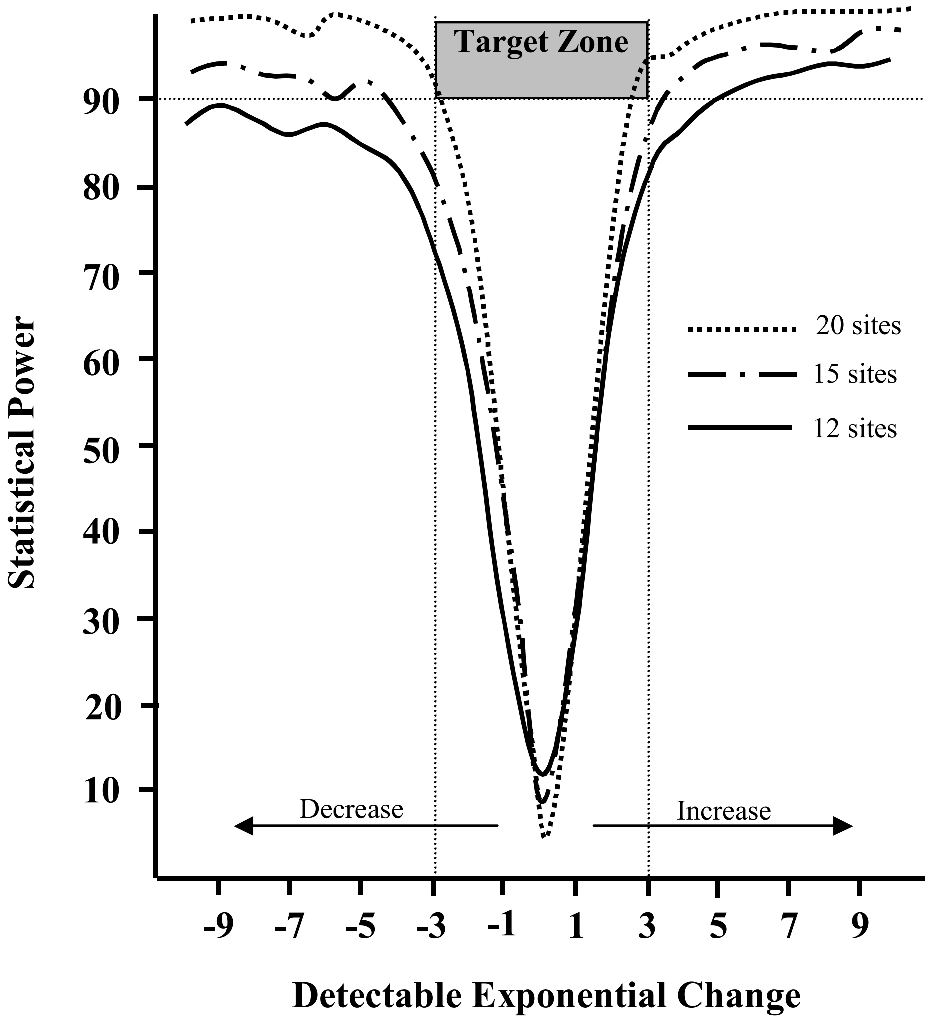


**Figure 10.** Statistical power to detect changes in the most variable indicator (STI counts, based on overall CV) with 12, 15 or 20 sample sites, at 6 plots per sample, and 10 years of sampling once per year, CV = 1.0. Horizontal and vertical dotted lines indicate the minimum statistical power (90%) and the minimum change detection limits.

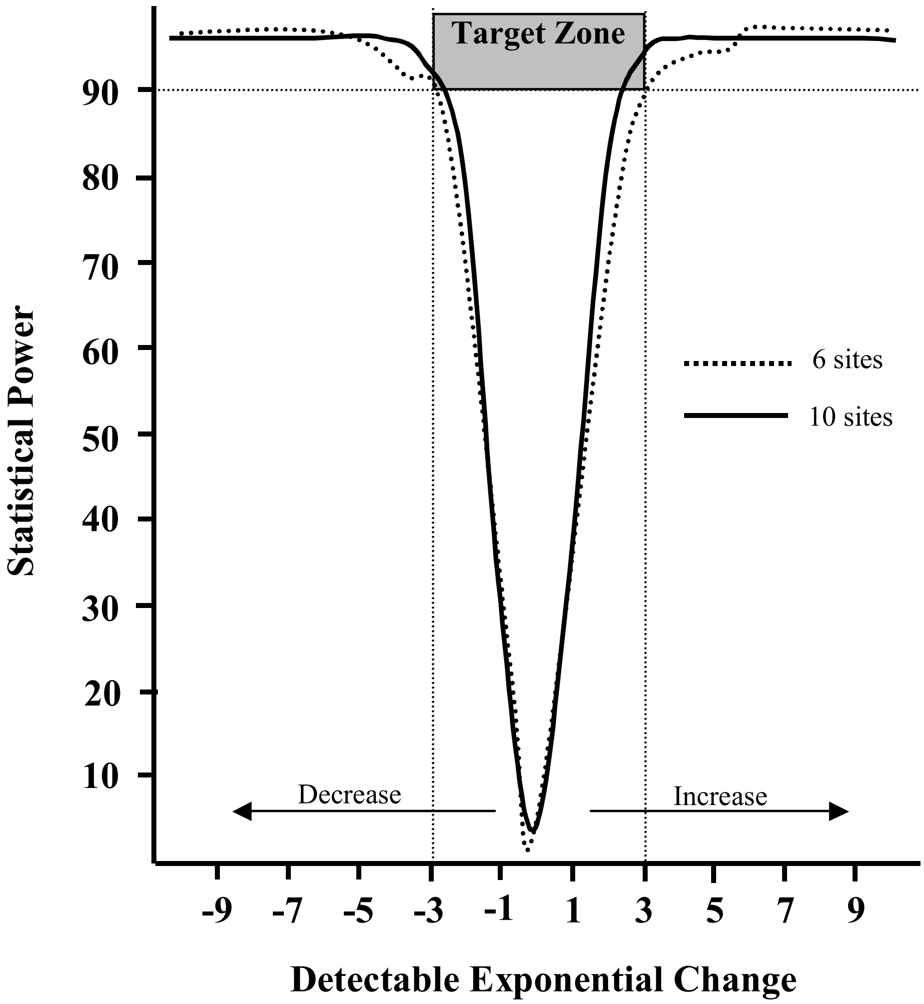


**Figure 11.** Statistical power to detect changes in a terrestrial indicator with average variability (stem density as stems/ha, based on overall Coefficients of Variation) with 6 or 10 sample sites, 6 plots per sample, and 10 years of sampling once per year, CV = 0.4 (median value of all indicator CV's). Horizontal and vertical dotted lines indicate the minimum statistical power (90%) and the minimum change detection limits.

## RESULTS – STREAM/AQUATIC SAMPLING

On December 6 and 7, 2003, stream reaches in four blocks in TFL 37 were surveyed.

The four streams surveyed were:

1. S2 stream - Stream #26;
2. S3 stream in Block KT 039;
3. S3 stream in Block HI 036; and
4. S3 stream in Block TS 028.

Average survey time for these streams was 69 minutes per stream, including the instream survey and the photographic survey (this does not include travel time to the block). This equated to a combined average stream survey rate of approximately 34 minutes/50 m of stream reach (i.e., measurement of instream indicators plus photographic survey). The main factor that significantly increased survey time was access along the stream bed or bank. Streams which had numerous large boulders, deep pools, logjams, suspended trees, or steep sidewalls, required longer survey times. The photographic survey took about as much time as the instream survey. Thus, average survey time could be reduced by approximately one-half if photo documentation is not required.

### Stream #26

The average channel width of Stream # 26 was approximately 12.5 m. and the channel gradient was between 6 and 7 %. The stream reach was surveyed for a distance of 137.5 m, or 11 channel widths. The survey was terminated at the downstream end of a canyon with deep pool habitat. The survey took 35 min to complete. If the survey were to continue upstream, the time necessary to complete the survey would increase significantly due to difficult access through this canyon.

Seven photographs were taken from three photo-reference-stations. The photographic survey took 20 minutes to complete.

The channel in the study reach appeared to be unaltered by adjacent forest harvesting. The presence of some areas of sloping banks did not appear to indicate that the rate of bank erosion had increased. The low number of log jams (one small one observed) and the low percentage cover for LWD was related to the high stream power. The high stream power during peak flows has the potential to transport LWD downstream. This likely accounted for the low volume of LWD in this stream section. Although two streamside trees had been removed, lateral channel stability was not affected due to the presence of bedrock outcroppings in the stream bank at this location.

### Block KT 039 S3 stream

The average channel width of the stream was approximately 3.0 m. and the channel gradient was 1 to 3 %. The stream reach was surveyed for 51 m, or 17 channel widths.

The survey took 23 min to complete. Six photographs were taken from six photo-reference-stations. The photographic survey took 24 minutes to complete.

This stream has been affected by the adjacent forest harvesting since several streamside trees had recently blown down during wind storms. Several of these trees were rooted in the stream bank and some small areas of accelerated bank erosion and sedimentation to the channel had occurred. Long term channel stability is not likely significantly affected although additional monitoring would be required to assess this.

### Block HI 036

The stream reach was surveyed for 105.0 m, or 14 channel widths. The average channel width of stream was approximately 7.5 m. and the channel gradient was 5 to 18 %. The survey took 43 min to complete. Fifteen photographs were taken from eight photo-reference-stations. The photographic survey took 53 minutes to complete.

With the exception of some effect from the road right-of-way and road crossing, channel morphology did not appear to be influenced by the adjacent forest harvesting.

### Block TS 028

The average channel width of the stream surveyed was approximately 11.0 m and the channel gradient was 10 to 15 %. The stream reach was surveyed for 112.0 m, or 11 channel widths. The survey took 37 min to complete. Seventeen photographs were taken from seven photo-reference-stations. The photographic survey took 41 minutes to complete.

Although sloping banks, particularly along the left bank were noted, no evidence of accelerated bank erosion was observed. The channel in this study reach did not appear to be affected by the adjacent forest harvesting.

# RECCOMMENDATIONS

## Terrestrial Sampling

### *Site selection*

Initially, only potential sampling areas within a 300m distance from active roads were considered for site selection. In TFL #37, the area within 300m of active roads effectively accounts for almost all S2, S3 and S4 streams in the TFL. By default, the area delineated by this road distance represents the three desired BEC units selected in the study design (CWHxm2, CWHvm1, CWHvm2). Therefore, **locating study streams based on access in these BEC subzones is recommended for future stream site selection.** Additional levels of stratification will likely not be possible due to limited resources (i.e. too much replication required). If levels of stratification beyond what is described here are deemed important in future efforts, we recommend that these be controlled statistically (i.e., using covariates, etc.) rather than physically (i.e. with larger sample sizes and excessive, categorized replication).

S4 streams with adjacent forest management post-1995 (i.e. post implementation of the *Forest Practices Code*) were not present in adequate number for analyses. These stream types will have to be added to the monitoring program if and when they become available on the landscape. To date it is not possible to incorporate S4 streams with sufficient replication (given the indicators selected). **If sufficient sample size is anticipated for S4 streams, then control-sampling efforts (pre-harvest) for this stream type should be initiated as soon as resources allow.**

It is anticipated that replication within this monitoring program will be limited by stream availability before sampling cost.

### *Site locations*

A key factor which affects sampling efficiency for this program will be the relative ease of locating the sampling plots among years. We suggest that recent air photos be used to mark plot locations and that these photos be used to relocate plots in future years. In the pilot study we used metal tree tags to mark plot centers, but these alone will not aid completely in efficient plot relocation, and the GPS coordinates will not provide sufficient grain to lead future observers directly to plot centers. Many riparian areas are ravine-like, and GPS coordinates are unreliable. Sometimes a GPS fix must be taken in adjacent uplands, and a bearing/distance recorded from that point to the nearest plot center. This type of information will be required in the field each time a crew samples an area. We suggest that UTM location data be organized into an electronic file that can be accessed while in the field within a palm data unit (i.e. UTM numbers, bearings, directions to plots), or be printed and accessible as hard copy data forms. Significant person-hour resources could be wasted searching for established plots, so effective mapping and plot flagging should be a priority in order to avoid this pitfall.

*Control stream sites*

We were unable to find appropriate control stream areas during the pilot study within TFL #37. Locating and sampling control areas will be critical in future monitoring programs to establish a natural range of variability for comparison to managed areas. The riparian areas within the TFL that are adjacent to FPC riparian management activities are generally old-growth riparian buffer strips that previously have not been logged. Areas of continuous forest in the TFL that we deemed as potential control plots were in all cases second growth forest that had been logged within the last 150 years. These sites were fundamentally different in structure compared to current FPC buffer strips. We **recommend looking outside the TFL, potentially into nearby parks, for comparable control areas**.

*Tree Density (Stems/ha) measurements*

The BAF prisms we used were found to be inadequate given the tree densities encountered in the pilot study area. Consequently, we calculated tree density from the fixed radius (5.64 m) vegetation plots. The BAF prism is likely a better method because it incorporates DBH, however, various prism sizes will need to be tested in future efforts in order to optimize the number of trees counted in each prism sweep.

*General field sampling*

We suggest that a **standardized data form (either hard copy or electronic) be used** while sampling. Unless field crews are relatively experienced, or were involved in the development of modification of the field techniques, it is likely that without the aid of a formal data form, certain metrics will be missed in the field. Formal data sheets also aid in data retrieval or data verification between years and crews.

*Indicators*

Tree height was highly correlated (0.85) with tree diameter (DBH). Only one of these parameters should be measured in the field. Many of the managed areas have significant amounts of CWD making hiking relatively difficult along the stream channel, and it is generally easier (physically) to measure the diameter of nearby trees instead of measuring tree height. Therefore, we would **suggest dropping tree height as a measurement** variable of forest structure unless this is specifically required for other monitoring objectives, and cannot be predicted with accuracy from growth-yield tables developed for tree species in the TFL. The height-DBH relationship is well known and accepted in forest mensuration.

CWD and the suspended tree index were negatively correlated (-0.596). Most of the CWD encountered in the terrestrial riparian area during the pilot study was from recent blow-down, indicating that in these plots, windthrow contributed substantially to STI and resultant lower standing tree density values. What is not necessarily clear, however, is whether this relationship will hold true for other stream types, and particularly in control areas. For this reason we hesitate to suggest choosing one of these two indicators over the other, and would **recommend measuring both CWD and the suspended tree index across all treatment types for the first couple of years** of any monitoring program to test whether this relationship is consistent.

Tree density was positively correlated with canopy closure (0.658) and we would expect this relationship to be consistent throughout. Because tree density is a more accurate measurement (i.e. relative to canopy cover estimations) **we suggest that canopy closure initially be estimated, but be dropped as an indicator in the future if its correlation to tree density is consistent over the first few years of monitoring.**

Of interest is the lack of a relationship between shrub cover and crown canopy closure. These two indicators have been reported as being correlated in other coastal forest studies (see Tappeiner et al. 2001), yet we found almost no correlation (PCC = 0.027) in our pilot study data. We suggest testing this relationship further when more data have been collected.

The ***estimate for required sample size*** (n = 15-20 streams per stream type) to detect change in STI *is high, and perhaps not logistically realistic*. The number of trees blown over the stream channel was a clear representation and a direct response to adjacent management practices; however, initial data suggest that the variability in this indicator is too high (CV=1.0) to be considered as an effective indicator. We view this as a difficult trade off. Our initial pilot data may have been too variable because of our relatively small sample size (n=12 stream sites). This could change when more sites are sampled. **We recommend initially keeping the STI as an indicator because it shows such a clear relationship to adjacent management activities, but its variability should be reassessed in future monitoring efforts**. If variability in this indicator is consistently high, then it should be dropped as an indicator.

### *Statistical analysis*

Within this monitoring program we have tried to keep the number of indicators and the methods used to measure them relatively simple. However, this will not translate into simple data analyses in future monitoring efforts. As more variables adjacent to riparian plots are considered (e.g. landscape-scale perspectives) the statistical analyses employed will become somewhat complex, and the interpretation of these data will require a qualified biostatistician. The data will require appropriate management and organization by someone familiar with advanced parametrics, multivariate analyses, ordination techniques, autocorrelation (both spatial and temporal), and data reduction (at a minimum), and the proper interpretation of each. This monitoring program is designed to allow comparisons of indicator values over time and between areas, and there are fundamental differences between these two approaches. Proper study design and statistical interpretation will be required at all stages of this program.

We have approached S2 and S3 streams as separate treatment types yet there is no empirical rational that this is necessarily so. **It may be that these stream types are similar in terms of forest structure before and after management prescription, and thus the required sample size would be reduced significantly by pooling these into one stream type.** A similar relationship may exist for S3 and S4 streams. This comparison should be done using sufficient replication within the first 2-3 years of the monitoring program, and sampling efforts adjusted accordingly. If stream types are found to be similar regarding forest structure, then sample stratification will have to be based on the "width of the RMA", as this will vary among stream class types.

### Stream/Aquatic Sampling

Since stream ecosystems are continuously changing, monitoring this process in a manner that will allow us to separate natural change from change associated with forest management practices is a challenge.

FPC-compliant forest management practices adjacent to streams, particularly those stream reaches with RRZs (i.e. S1, S2 and S3), will typically have only subtle effects on the stream ecosystem. These effects will be difficult, if not impossible, to measure simply by collecting data for physical variables such as log jam frequency, sediment variability, etc. In addition, logging-related effects may not manifest for a number of years, and therefore need to be predicted. Collecting and analyzing physical data related to channel form only might not tell us if changes are occurring, or if these changes are related to riparian management.

Consequently, we recommend developing a watershed-based approach that combines both the measurement of physical stream parameters and expert opinion. One suggested approach is outlined below:

1. utilize existing aerial photographs and/or take low level aerial photographs (e.g. 1:5,000) every 3-5 years;
2. ensure that on-the-ground surveys of the aquatic ecosystem are conducted by individuals qualified and experienced enough to be able to detect change or predict change based on professional opinion;
3. collect physical data and photograph the channel in a repeatable manner that will support professional opinion (reference sites, etc.); and
4. document the professional opinion process and describe why a particular decision was made.

#### *Watershed-based Approach*

A watershed-based approach is necessary if we hope to separate riparian management-related effects from external (upstream) effects. It would be advantageous to identify and describe how the upstream reaches are influencing the study reach. For example, a study reach may become unstable, or undergo a change in sediment composition. This change, however, may be related to upstream effects such as landslides or changes in flood magnitude, and may not have anything to do with riparian management adjacent to the study reach.

### *Aerial Photography*

The use of existing aerial photographs would help professionals analyze and describe the watershed and to identify upstream processes (e.i. landslides) that may influence the study reach. Conducting additional low-level aerial photography would provide an important database to monitor changes in riparian function and would help to support professional opinion.

Existing aerial photography will typically have limited application on the medium to smaller sized streams since the channel may not be visible through the forest canopy. Low level (i.e. 1:5,000) aerial photographs taken prior to leaf-out may be much more useful depending on the stream.

### *Field Survey Team*

It is anticipated that professional judgement will be required to assess riparian effects on the aquatic habitat. The ideal would be to create a Professional Team that would include a channel morphology expert (i.e. a fluvial geo-morphologist or fluvial engineer) and a fisheries biologist (experienced in stream habitat assessments) that would walk the stream together and possibly fly the watershed in a helicopter. This team approach is highly recommended.

### *Field Data*

Field data would be collected to verify professional opinion. It is recommended that on-the-ground photographs be taken from established photo-reference-stations. During each survey, photographs could be taken from these stations using the same type of equipment and setup, allowing inter-year comparisons.

Reference, or control study reaches, should be established where possible. Physical characteristics that are measured should be assessed after each field season to determine if they are appropriate. It would be desirable to measure parameters that are not influenced by stream flow, such as log jams, streamside trees, etc. Stream flows during the December 6 and 7, 2003 assessment were too high to accurately assess inventory gravel bars.

At this stage, the measurement of biological response (i.e., fish density) is not recommended. A possible exception is benthic invertebrates if methodologies currently in use in other areas of the province prove to be able to detect change and are cost-effective. Until then, it will be assumed that if the aquatic habitat is maintained within its range of natural variability, then the aquatic community structure and function will also be maintained.

*Documentation*

A common criticism of assessments that utilize profession judgement to reach conclusions, is that the assessment reports do not adequately describe the decision making process. Consequently, the following procedures are proposed:

1. data related to physical channel parameters (i.e. bank erosion, LWD, etc.) are collected, described and compared to regional standards where possible; and
2. each professional should write a description of the channel and describe if and how stream form and function have been effected by riparian management.

It is recommended that the stream assessment report describe the stream survey reach within a watershed context (i.e., consider potential upstream influences), and that all professional decisions be well documented.

## CONCLUSIONS

In conclusion, the forest structure, wildlife, and stream/aquatic variables selected and pilot tested on TFL 37 (see list below), are suitable ecologically as well as practical and relatively cost effective, for use in a monitoring program to evaluate the effectiveness of riparian forest management practices.

| Terrestrial | Stream/Aquatic |
|---|---|
| • Canopy Closure<br>• Shrub cover / composition<br>• Tree density (stems per ha) or basal area<br>• Coarse woody debris<br>• Windthrow (suspended tree index)<br>• Wildlife habitat use (terrestrial salamanders) | • Bank erosion<br>• Bar type and frequency<br>• Logjam frequency<br>• LWD cover and volume<br>• Fish cover types<br>• Streambank tree loss |

The Canfor pilot study assessed the suitability of the above indicators for effectiveness monitoring (i.e, will the indicator detect changes to the riparian ecosystem), as well as the feasibility (i.e., practicality and efficiency) of measuring each indicator in the field. 20 potential S2 harvest streams, 13 potential S3 harvest streams, and 1 potential S4 harvest stream were identified for sampling. No suitable control streams were found within the TFL (i.e., mature or old riparian forest with no adjacent harvesting, and which is accessible and within the same BEC unit). Suitable control sites may have to be established in protected areas or at other locations on Vancouver Island with similar forest conditions.

Of the potential sample streams, 12 were visited and sampled using the full sampling methods described herein. The recommended monitoring program is designed to monitor changes over time and make comparisons between areas with a 90% probability of detecting change at a statistical significance of alpha = 0.10. To achieve this we recommend a conservative, **initial estimate of approximately 6 streams (sites) per stream class** (i.e., S2, S3, etc., plus unharvested control streams) with an initial sample frequency of once per year, for at least 2-3 years. **This sample size should provide enough rigour to detect change in the most variable indicator (suspended tree index), and be more than sufficient for indicators with less inherent variability** (tree density or basal area CWD, salamanders, etc.). Data analyses should be rigorous and should review whether variability and statistical power are sufficient to warrant adjustments to sample size or sampling intensity/frequency. However, as in any study design, the pursuit of adequate sample size must be balanced with the availability of sufficient resources to conduct the work. It is anticipated that replication within this monitoring program will be limited by stream availability before sampling cost. We recommend dropping tree height as a measurable variable at forest structure plots because it is redundant to measuring tree diameter (Pearson correlation coefficient = 0.85) and takes longer to measure in the field.

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