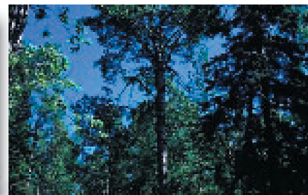
NORTHERN WATERSHED PROJECT

Project Report #1

RIPARIAN FOREST MANAGEMENT: Paradigms for Ecological Management and Practices in Alberta

The Northern Watershed Project is a collaborative research venture between the Alberta Research Council and the Alberta Conservation Association.

RIPARIAN FOREST MANAGEMENT:

PARADIGMS FOR ECOLOGICAL MANAGEMENT AND PRACTICES IN ALBERTA

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EXECUTIVE SUMMARY

This report reviewed the qualitative and quantitative data dealing with the ecological retention of treed riparian buffers in timber harvest cutblocks. The review focused primarily on the Boreal and Rocky Mountain ecoregions with applications to Alberta. Studies outside of the Boreal or Rocky Mountain were considered when demonstrating broader concepts or when data was not available for these regions. The report was organized into nine sections with two appendices, but deals with four major issues:

1) defining riparian areas,
2) qualitative and quantitative review for the ecological-basis of riparian buffers,
3) quantitative review of management guidelines for riparian buffers, and
4) natural disturbance-succession model (based on wildfires) for management of riparian.

Definitions for riparian areas reflect their use in administrative or discipline settings. The primary difficulty in defining riparian areas for management is clearly translating the changes in multiple functions, structures, and biota along the gradient, i.e. ecotone, from aquatic to terrestrial upland into administrative boundaries. Currently, the most widely accepted definition is based on a probabilistic delineation. Areas closer to the water's edge are more likely to be riparian. This definition implies that a hard boundary based on a single criterium may not be appropriate. Rather, impacts should be assessed for their effects across the entire gradient. This definition implies a transverse view of riparian areas moving from aquatic to upland. Longitudinal, i.e. downstream and upstream, and vertical aspects of the riparian are being currently studied but are less well-developed in terms of both conceptual or management models.

A review of the ecological-basis for riparian buffers suggest that their original purpose in preventing pollution and nutrient inputs, and sedimentation and erosion may have given way to provision of aquatic and terrestrial habitat. Buffers are relatively ineffective at mitigating increases in water yield, peak flows, and nutrient yields immediately after disturbance. These are catchment-level effects and are a function of the overall proportion of disturbance in the catchment. Buffers are effective for the reduction of sediment flows into waterbodies from overland flows created by disturbance of the forest floor. They are less effective for sediment transport by channelized flow. Current harvest practices that minimize soil disturbance particularly in low gradient boreal forests may forego the need for wide buffers. At present, most of the sediment transport from harvesting occurs from skid trails, landing areas, and roads.

For aquatic biota and habitats, shade from riparian canopy cover and inputs of allochthonous organic debris into streams are likely the factors that will be most affected by harvesting of treed buffers. In other forested systems, buffer widths of a single tree length (20 - 30 m) were recommended to maintain these features. Western boreal systems lack sufficient short- and long-term

data to evaluate the modification of riparian areas on both water temperature and organic inputs and their subsequent effects on invertebrates and fish communities.

Buffer recommendations for terrestrial biota range from the tens to hundreds of metres wide. However, the current objectives for maintaining terrestrial habitat and biota within riparian buffers after harvest are unclear. Use of riparian buffers can vary from temporary cover in order to prevent cutblock avoidance after harvest to core habitat for long-term sustainability of population. Larger riparian areas are likely required for temporary refugia that will re-colonize disturbed areas or core habitat for long-term population maintenance. Further complicating this issue is our lack of knowledge of which species need the combination of trees and proximity to water as a critical component of their life history. Most of the current research is based on presence/absence data and/or relative density data. Though habitat use can be implied from either data type, the use of buffers as dispersal corridors, temporary refugia, or core habitat cannot. Riparian management plans should contain clear objectives for the maintenance of species. The appropriate use of buffers in combination with upland areas should be used to attain these objectives. Research should be based on these objectives.

The retention of treed riparian buffers in Canada and the United States is the primary management tool applied throughout all reviewed provincial and state jurisdictions (N=60). Canadian jurisdictions have wider riparian buffers for small and large permanent streams, small and large lakes, and water sources than jurisdictions in the United States. Riparian management areas from Boreal jurisdictions are generally wider than all other regions. Rocky Mountain jurisdictions were intermediate in width. Alberta's riparian buffer widths are comparable or wider than most other jurisdictions for all waterbodies except intermittent streams.

Most jurisdictions use at least one modifying factor to stratify riparian areas. The most common modifying factors were (ordered form highest to lowest); waterbody type, slope, waterbody size, fishbearing, human water supplies/aesthetics, drainage basin area, shoreline forest management, saltwater flow, shoreline vegetation, upstream of fishbearing waters, downstream sediment threat, and flow rates. Jurisdictions that do not have specific guidelines for modifiers (e.g. slope, presence of fish, and human drinking water) usually have wider baseline buffers than jurisdictions with specific guidelines.

Alberta's guidelines are relatively simple, mostly focusing on waterbody type (e.g. streams, lakes) and size (e.g. channel width). Alberta's buffer widths on large streams, small streams, non-fishbearing and fishbearing lakes are wider than other jurisdictions with specific modifiers for presence of fish, human consumption of water, and slope. Most other Boreal jurisdictions have presence of fish as a modifier while Rocky Mountain jurisdictions have presence of fish and slope as modifiers. However, most of the riparian buffers in Alberta areas are equivalent to jurisdictions with modifiers. For Alberta's riparian guidelines the width of the management area associated with wetlands are poorly developed compared to some other jurisdictions.

Also, Alberta should clarify the criteria requirements for buffers on intermittent streams.

A relatively high percentage of jurisdictions (>80%) permit selective harvest within riparian management areas. Usual rates of harvest were less than 50% of merchantable trees usually with size restrictions to prevent “hi-grade” harvesting, i.e. selection of largest trees. Restrictions were also placed on shoreline harvesting and opening sizes. In many forest types, regeneration of tree species in riparian areas is a problem because of competition with shrubs or grasses.

Given the review of ecological data and guidelines for other jurisdictions, there appears to be no great urgency for changes in the current guidelines. However, there are some long-term changes that may be worth considering (in no particular order):

1. Widths for lakes are larger than other jurisdictions. At these widths, their objectives as habitat for terrestrial biota should be made clear.
2. Further development of riparian management objectives and practices for wetlands and intermittent streams should be considered.
3. Selective riparian harvest should be approached cautiously with an assessment of regeneration problems in riparian areas and changes to terrestrial habitat and biota.
4. Assessment of longitudinal, i.e. upstream and downstream, effects of management actions and incorporation into guidelines, i.e. cumulative watershed approach.

The previous three parts underscores a preservation-protection paradigm attached to the management of riparian areas. Human-caused disturbances in upland areas are to be isolated from the aquatic ecosystem. This traditional management philosophy views treed buffers as an important tool in protecting the riparian ecotone from forestry operations.

An alternative management philosophy argues disturbance-succession as a part of the temporal and spatial dynamics associated with forested landscapes and that riparian zones and aquatic ecosystems are part of those dynamics. For most forest regions in Alberta, this means using the dynamic patterns set by a natural disturbance-succession such as wildfire to set the range of variation for human disturbances such as timber harvest. Under this approach, timber harvest would operate within the dynamic range as natural disturbance-succession. The resultant landscape patterns and dynamics would be similar to natural patterns and dynamics, hence, it would be assumed to maintain natural habitats and biota.

In order to further understand the implications of a disturbance-succession model for riparian management, a single watershed was examined as a case study. We selected the Notikewin River and Rambling Creek watersheds which burned in 1982. The natural disturbance-succession pattern, as measured by the amount of residual or unburned forest, varied with stream type.

Basing riparian management on a natural disturbance-succession model would be quite different than Alberta's current 1994 guidelines. These differences include: a) removal of streamside forest around large permanent and small permanent streams, b) retention of streamside forest around some intermittent streams, c) retention of forest at distance from stream (i.e. upland) exceeding current buffer distances including large polygon-sized patches, and d) integration of green tree retention patterns between riparian and upland areas.

Further research is required to assess the economic, social, and ecological impacts of using a natural disturbance-succession model for riparian management. Despite being able to produce a similar landscape pattern, there are significant differences in many other aspects of wildfire ecology and timber harvest. These include; deadwood resources, understory communities, soil properties, erosion and sedimentation patterns and nutrient flows. It remains unclear whether similarities in landscape-level patterns retain the sufficient benefits to outweigh differences at the stand-level between wildfire and harvest. Also, it is unclear whether alternative landscape planning scenarios based on other criteria such as old growth or targeted species groups produce more ecologically sustainable landscapes. The lack of clear objectives on the range of acceptable variability and alternative models makes large-scale implementation of the disturbance-succession model unwarranted at this time. For these reasons, we recommend continued exploration and comparison of natural disturbance-succession through site-specific modeling leading to active adaptive management in the field through pilot projects.

1.0 GENERAL INTRODUCTION

1.1 Report scope

This document reviews the ecological and management literature on riparian buffer widths in Canada and the United States. In particular, it focuses on data that is applicable to Alberta's forested Boreal and Rocky Mountain ecoregions.

This document reviews and analyzes the ecological rationale underlying the management of riparian areas with a focus on the application of lateral, restricted- or no-harvest buffer zones. The review highlights studies from the Boreal and Rocky Mountains ecoregions and jurisdictions. However, other ecological regions and jurisdictions in North America are also included to provide either a context or suggest potential outcomes of management actions for topics which there is little or no data from Boreal or Rocky Mountain ecoregions. Some original research is presented on the impact of natural disturbance, i.e. wildfires, on riparian landscapes in a case study of the Notikewin River, Alberta.

The document is a resource for managers focusing on ecological issues surrounding riparian management.

This document is meant as a reference for the resource manager but is by no means an exhaustive review of riparian issues surrounding forestry. Important topics such as road construction, water impoundments/diversion, and chemical pollution are not covered. Nor do we discuss cultural, aesthetic, material, recreational, or economic values. These topics are worthy of consideration and are often considered trade-offs with ecological functions in conflicts over the management of riparian areas. No consideration of riparian management would be complete without inclusion of these important topics.

We attempted to qualitatively and quantitatively summarize ecological studies and management options from a rapidly changing field. Analysis looked for trends amongst studies and management guidelines. Based on quantitative and qualitative analysis, we offer some synthesis and suggest areas of future research and management effort. The sidebars comments are meant to provide readers with summaries through sections that they wish to move quickly through.

Section 1.0 defines the scope of the report and offers a review of definitions for riparian areas. Section 2.0 presents a qualitative review on the utility of riparian buffers for a number of ecological structures, functions, and biota in Boreal and Rocky Mountain forests. Section 3.0 presents a quantitative review and analysis of riparian buffer width recommendations based on the ecological literature. Section 4.0 reviews current buffer width guidelines from Canadian and American jurisdictions with an emphasis on Boreal and Rocky Mountain ecoregions. Section 5.0 examines the impact of wildfires on riparian zones and offers a case study for extrapolating a landscape patterns left after wildfire to both upland and riparian areas. Section 6.0 extrapolates the key findings from each section into management recommendations. Lastly, Section 7.0 discusses knowledge gaps.

1.2 Defining Riparian Areas

Riparian definitions usually reflect their intended application.

Typically, the definition for riparian areas varies according to its intended use in research or management. Examples of academic definitions include:

> "Riparian zones are the interfaces between terrestrial and aquatic ecosystems. As ecotones, they encompass sharp gradients of environmental factors, ecological processes, and plant communities. Riparian zones are not easily delineated but are comprised of mosaics of landforms, communities, and environments within the larger landscapes."
>
> Gregory *et al.* (1991)

> "Riparian ecosystems are the complex assemblages of organisms and their environment existing adjacent to and near flowing water."
>
> American Society of Fisheries (2000)

Definitions from regulatory bodies tend to focus on the boundaries of agency responsibility.

> "Riparian areas are plant communities contiguous to and affected by surface and subsurface hydrologic features of perennial or intermittent lotic and lentic water bodies (rivers, streams, lakes, or drainage ways). Riparian areas have one or both of the following characteristics: (1) distinctly different vegetative species than adjacent areas, and (2) species similar to adjacent areas but exhibiting more vigorous or robust growth forms. Riparian areas are usually transitional between wetland and upland".
>
> United States Fish and Wildlife Service (1997)

Under the Fisheries Act (Section 34), fish habitat is defined as:

> "spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly in order to carry out their life processes.
>
> Fish habitat therefore refers to:
>
> freshwater, estuarine and marine environments that directly or indirectly support fish stocks or fish populations that sustain, or have the potential to sustain, subsistence, commercial or recreational fishing activities.
>
> Department of Fisheries and Oceans,
> Environment Canada (2000)

The Canadian Council of Forest Ministers defines riparian zones as,

> "A strip of land of variable width adjacent to and influenced by a body of freshwater."
>
> Canadian Council of Forest Ministers 2000

The major challenge in defining riparian areas is to spatially delineate start and end points of multiple ecological processes.

Definitions are often tied to the need to delineate riparian zones within a landscape for management purposes. This task has proven challenging because it forces managers to clearly and unambiguously state where one set of ecological characteristics ends and another begins, namely the boundary between riparian and upland areas. In this regard, a number of key features may be used. Many examples focus on a "zone of influence" created by the presence of water. These include the outer limit of flooding (Gregory *et al.* 1991), a change in plant species composition (United States Fish and Wildlife Service 1997), or the influence of elevated water tables on vegetation (Naiman *et al.* 1993). Not surprisingly, different species groups or components have different responses (e.g. Harper and MacDonald 2001, Lyon and Sagers 1998). The variability in determining widths of riparian zones is unavoidable since specific physical or ecological processes have a unique spatial and temporal pattern, therefore, a unique zone of influence (Gregory *et al.* 1991). A single boundary may provide adequate protection for some ecosystem components while putting other components at risk.

An alternate approach is to apply a probabilistic definition to the size of the riparian area.

An alternate approach to delineating riparian zones focuses on a probabilistic distribution of riparian elements in the transition zone between aquatic and terrestrial components (Figure 1). In a review of definitions, Ilhardt *et al.* (2000) noted that despite diverse objectives most definitions were characterized by the following three properties:

- inclusion of the water and the feature (e.g. stream) that contains or transports water for at least part of the year
- an interactive, transition zone between aquatic and terrestrial ecosystems
- variable widths and boundaries

Based on these characteristics, they developed a probabilistic definition:

> "Riparian areas are three-dimensional ecotones of interactions that include terrestrial and aquatic ecosystems, that extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at a variable width."
>
> Ilhardt *et al.* 2000

Recently, researchers have taken a longitudinal, i.e. upstream and downstream, view of riparian areas.

This definition considers the distribution of riparian elements in the transition zone between aquatic and terrestrial components. Riparian function, structure, and biota extend from the shoreline laterally outwards (Figure 1). In areas closer to the shoreline, the functional, structural, and biotic elements associated with the riparian ecotone become more pronounced.

In recent years, a longitudinal view, i.e. upstream and downstream, of riparian zones has also gained support (Figure 2). Researchers focusing on stream biota (e.g. benthic macroinvertebrates and fish) and physical processes such as hydrology and fluvial geomorphology have traditionally delineated areas based on the longitudinal pattern of riparian areas (e.g. Vannote *et al.* 1980; Naiman *et al.* 1987; reviewed in Malanson 1993). Currently, this approach is finding greater interest among terrestrial ecologists (e.g. Naiman *et al.* 1993; Jonsson 1997). The longitudinal view considers the relationship of species among stream orders or across elevational changes within a watershed, as well as the relationship between longitudinal positions and upslope areas. The maximum interface between water and upland areas occurs within headwaters. These catchments are typically small with nearshore areas exhibiting relatively little difference with upland areas (Table 1). In contrast, the greatest differentiation between riparian and upland areas occurs in the higher stream orders. These differences should lead to differences in the management objectives of lower and higher order streams.

Aside from the spatial dimensions, riparian landscapes also feature temporal dimensions. Ward *et al.* (2002) argued that the engineering perspective has contributed to a view of riverine systems as regulated and homeostatic ecosystems. In contrast, their view is that the riverine landscape is dynamic and characterized by landscape evolution, ecological succession and turnover rates of its constituent elements. These elements include; surface waters, alluvial aquifers, riparian systems, and geomorphic features such as bars, islands, and terraces. Fluvial action is the predominant disturbance agent of landscape change. Changes in the overall landscape and connectivity of the riverine system are driven by ongoing natural cycles of natural disturbance-succession. Despite, these ongoing changes the overall suite of elements should remain relatively predictable and constant over time. The high diversity of faunal elements within riverine landscapes has been attributed to the high diversity and rates of turnover for habitats (Robinson *et al.* 2002). Life cycles of species adapt and come to rely on the turnover of riverine habitats (e.g. Schlosser and Kallemeyn 2000). However, the riverine landscape may retain a biotic imprint of the “ghost of landscapes past”, particularly those of anthropogenic disturbances. As an example, Harding *et al.* (1998) found that invertebrate and fish communities in western North Carolina were better explained by land practices of the 1950’s then current practices.

The longitudinal network of streams and rivers are an important component of a regional biodiversity.

Naiman and Rogers (1997) acknowledged the significance of riparian areas along main stem rivers as important contributors to regional biota. They argue that a significant portion of the biota in most ecosystems is represented in areas near streams, hence, the conservation value of streams and riparian areas can be immense. Despite, these arguments, the knowledge-base of biota associated with riparian areas or their longitudinal distribution remains relatively low in most ecoregions and jurisdictions including Alberta. Comprehensive surveys of terrestrial biota along riparian corridors are relatively rare. Where biota has been assessed, the diversity has been found to be relatively high. For example, the presence of vascular plants surveyed along a single river corridor in Sweden accounted for 13% of the total number of vascular plant species in country (Nilsson *et al.* 1994).

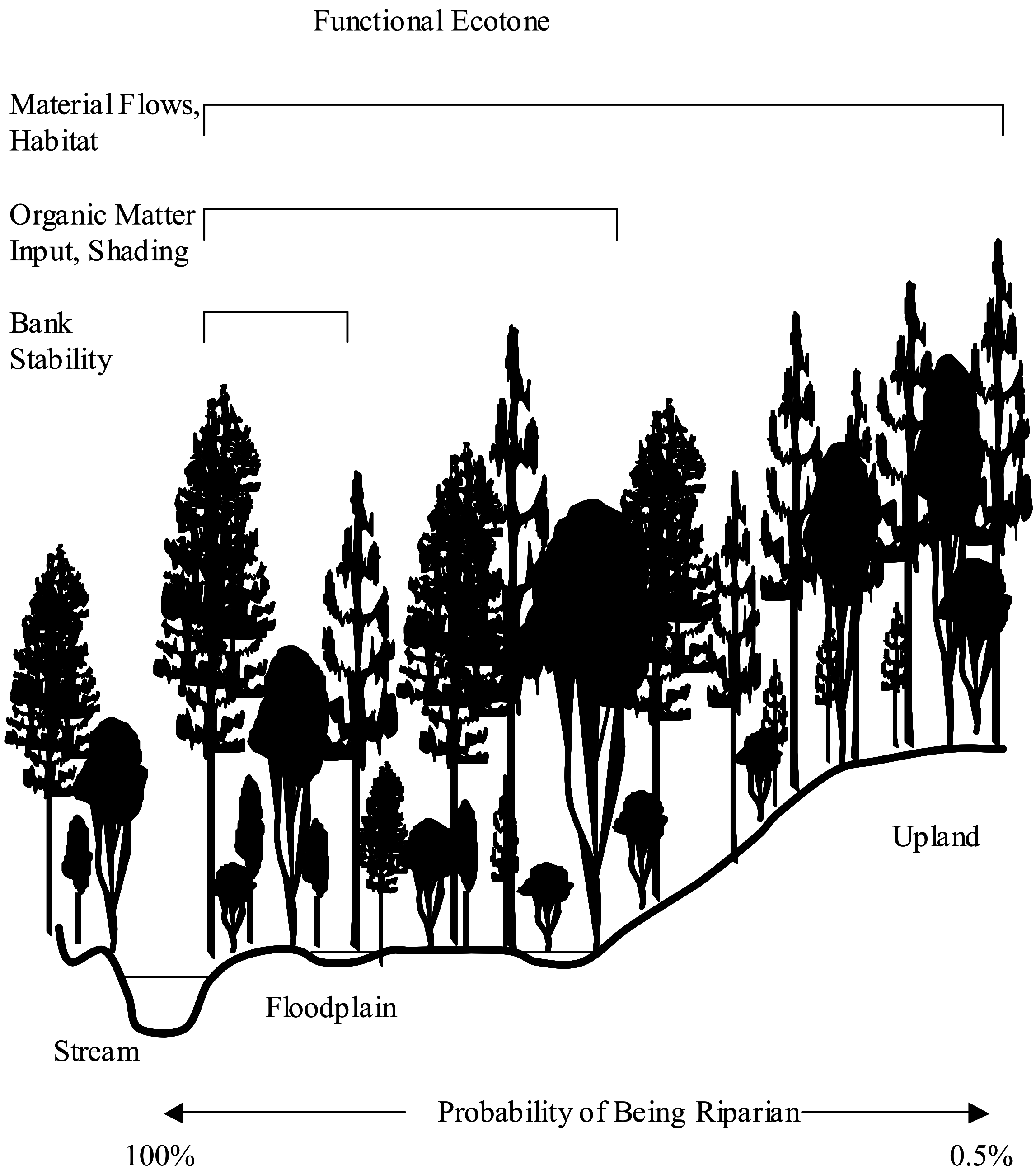

Figure 1. A lateral view of a typical riparian ecotone featuring probabilistic distributions of function, structure, and biota.

Table 1. Watershed sizes for western Boreal and Foothill regions from Kakwa, Notikewin, and Simonette River systems in Alberta. Mean catchment area (ha) and standard deviation in brackets.

Catchment Order	**Simonette River**	**Notikewin River**	**Kakwa River**
1	95 (162)	136 (427)	38 (62)
2	438 (615)	618 (758)	184 (259)
3	2,074 (3,232)	2,506 (2,635)	873 (944)
4	7,405 (5,562)	10,557 (7,705)	3,826 (3,306)
5	37,148 (34,092)	63,705 (34,673)	10,712 (8,635)
6	155,935 (113,169)	466,700 (207,733)	34,237 (7,249)
7	536,804 (0)	979,919 (0)	348,964 (0)

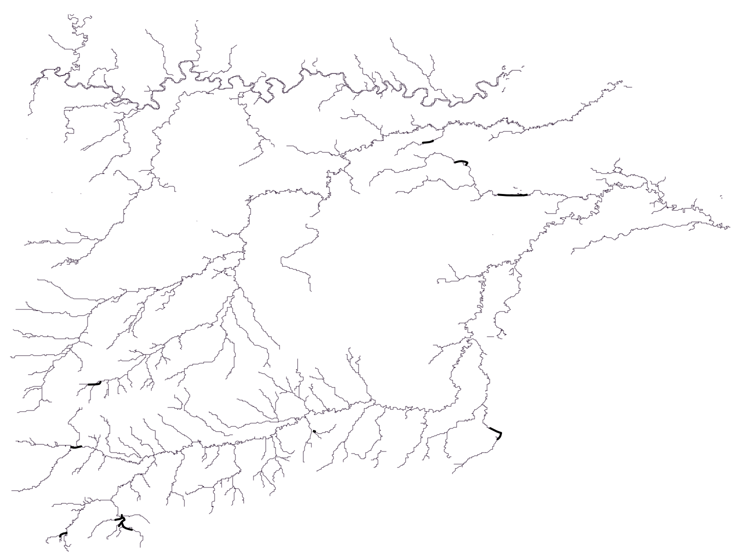

Figure 2. Longitudinal view of a typical northern riparian ecotone features connected watersheds both upstream and downstream of any point along the network.

2.0 QUALITATIVE REVIEW OF RIPARIAN FUNCTIONS, STRUCTURES, AND BIOTA

2.1 Aquatic Components

The complex and frequent disturbances (e.g. flooding, erosion, deposition, fire) associated with riparian areas results in a dynamic ecosystem defined by a high diversity of habitats and species (Gregory *et al.* 1991). Riparian areas provide ecological functions such as stream shading, control of nutrient export, retention of organic and inorganic material, regulation of quantity and type of organic input, and streambank stabilization (Gregory *et al.* 1991, O'Laughlin and Belt 1993).

This section provides a short primer on riparian ecology focusing on disturbance and mitigation, through the use of buffers, in Boreal and Rocky Mountain ecoregions. Research from other regions may be included if examples are lacking from these forest types. For further reading there are a number of excellent, extensive reviews in the literature (e.g. Gregory *et al.* 1991, Wenger 1999, Verry *et al.* 2000). Although this section artificially separates structure, function and biota, in nature these elements are integrated. As a consequences the management of a single element may have unforeseen consequences on other aspects of the riparian ecotone.

2.1.1 Water Yield and Peak Flows

Generally, the loss of surface vegetation, by either natural or anthropogenic disturbance, decreases evapo-transpiration by reducing available leaf area. In turn, this leads to increased soil saturation and results in more water being discharged as stream flow or seepage into groundwater. Elevated surface and stream flow can increase the potential for erosion and sedimentation, alter stream channelization, increase run-off from roads, and place pressure on culvert and other stream crossings.

Loss of vegetation either by harvest or wildfire decreases the evapo-transpiration through loss of available leaf area. This potentially leads to increases in water yield and changes in peak flow patterns.

Short-term results of forest canopy removal by wildfire or harvest suggest an increased water yield that lasts for at least 10 years, until the reestablishment of a continuous tree and shrub canopy. Beaty (1994) demonstrated an increased runoff associated with storm peaks in watersheds burned in 1974 and 1980 fires, within the Experimental Lakes Area in northwestern Ontario. Similarly, Helvey (1980) measured a 50% increase in the water yield of three Cascade Mountain watersheds in central Washington the year after wildfire, with elevated yields continuing for seven years post-fire. Hornbeck *et al.* (1994) found that clearcutting could increase water yield up to 40% for the first few years after harvest in eastern forests in the United States. This corresponds to similar, relatively short-term increases in water yields that have been documented in a number of other studies of eastern forests (Bosch and Hewlett 1982; Sahin and Hall 1996).

The percentage of disturbance in watersheds often determines the magnitude and duration of increased water yields. Increased yields can generally be expected for at least 10 years after disturbance.

In some studies, the amount of forest canopy removal played an important role in determining the magnitude and duration of elevated water yields. Keenan and Kimmins (1993) suggested that the amount of run-off varies with the proportion of basal area harvested within a watershed. Martin *et al.*

(2000) found a 3 to 9% and 23% increase in water yield from clearcut and strip cut stands, respectively, in the White Mountains of New Hampshire. These elevated water yields lasted for only 2 years after clearcutting but lasted for 7 years after strip cutting. Hicks *et al.* (1991) reported a similar pattern in Oregon, where clearcut and burned stands exhibited a 159% increase in water yield over reference watersheds. The increase diminished after the first year but was still evident 8 years after harvesting. Other authors have suggested that changes in water yield would be difficult to detect at harvest levels below 20 to 25% canopy removal (Bosch and Hewlett 1982; Hornbeck *et al.* 1994, but see Buttle 1994).

The relative impact of disturbance on water yield and peak flows may be influenced by run-off processes, local soils, climate, snowpack, season, topography, and watershed size (Buttle and Metcalf 2000). In studies from the northeastern United States, Hornbeck *et al.* (1993) and Martin *et al.* (2000) reported increases in peak flows during the late summer months. Conversely, in Oregon, Harr and McCorsin (1979) found decreases in peak flows after clearcut logging while Thomas and Megham (1998) found no significant change in peak flows after clearcut logging. The Sundance fire along Pack River, Idaho produced no increase in annual water yield but resulted in the peak flow occurring earlier in the spring (Campbell and Morris 1988). The change in peak flow was attributed to decreased transpiration losses and earlier melting of the snowpack in the newly cutover areas.

In Boreal forests, Buttle and Metcalf (2000) showed no definitive changes in runoff, peak flow, or timing in response to 5 to 25% harvesting of medium to large watersheds (<11,900 ha). In contrast, Verry *et al.* (1983) demonstrated that increases in the significant peak flows were associated with storm events and snow melt in clearcut aspen stands in Minnesota. In part, the scale of the area under consideration is critically important in assessing the relative impact of cutting on water yield (Buttle and Metcalf 2000). Large areas, particularly those with relatively low topographic relief such the Boreal forest, are likely able to buffer impacts such as local increases in peak flows. Paterson *et al.* (1998) further argued that temporal variation in climatic conditions over the long-term would overshadow impacts of harvest or wildfires within small boreal basins.

Riparian buffers have little direct impact on the magnitude of water yields or peak flows particularly in low gradient systems such as the boreal. They may serve mitigate erosion or sediment transport effects in extreme cases.

On the specific question of whether riparian buffers reduce changes to water yields and peak flow, there is no consistent answer. In part, many of the previous experiments on harvested areas were done with forested buffers around streams; hence, mitigation by buffers was not directly tested. In one of the few studies that looked at water yield without the buffers, Steedman (2000) found no changes to yield with harvest of riparian buffers in Boreal Shield lakes. This is not surprising given that the mechanism for increased water yield is the decline of evapo-transpiration due to loss of vegetation within the contributing area. The area of buffers is relatively small in most Boreal watersheds (~2% of area). Mitigation of increased water yield and peak flows in itself is not likely to be controlled by buffers; but buffers may play a role in alleviating secondary problems created by increased water yield and peak flows such as erosion and sedimentation.

2.1.2 Erosion and Sedimentation

Natural disturbances such as wildfires can lead to significant amounts of sediment transport.

Historically, two of the early applications for riparian buffers were to prevent sedimentation of streams from adjacent timber harvest areas and to protect and stabilize stream banks (Smith 1972). Erosion of upland sites depends on a number of factors but primarily on wind, litter and organic soil layers, infiltration rates of the soil, topography, and harvest practices (reviewed in Hornbeck and Kochenderfer 2000). In general, sediment transport is greatest immediately after disturbance and decreases thereafter. For years following the 1988 Yellowstone fires, snowmelt and stormflows created pulses of sediment transport (Minshall *et al.* 1989), however, these decreased in amplitude as burned areas revegetated. Helvey (1980) reported that the effects of high snowpacks or storms compound sedimentation of streams after wildfires, leading to an increased probability of mass soil movement, debris torrents, and stream scouring. They reported sedimentation rates 8 to 10 times greater than pre-burn levels even 10 years after the burn and the return of vegetative cover. Morris and Moses (1987) found that sediment flux rates were three orders of magnitude greater immediately after wildfires in Colorado's front range. These rates dropped to a single order of magnitude difference after four years, due largely to reduced water repellency of the soil and the preceding loss of looser, smaller-sized particles. There is little data on the impacts of wildfire on sedimentation for the Boreal forest. In one of the few studies after wildfires, a 20-fold increase in bedload transport was recorded after a wildfire in Ontario's Boreal forest, (Schindler *et al.* 1980; Bayley *et al.* 1992; Beaty 1994). Most of the increase was attributed to storm flows.

This can be compounded by ground disturbance caused by salvage logging after wildfires.

Two special cases worth considering are the impacts of wildfire on recently harvested areas and the impact of salvage logging. The combination of ground vegetation removal by fires, the removal of trees, and potential ground damage created by harvesting, could cause these sites to be potentially more susceptible to erosion and sedimentation. Even amongst early surveys, the damage associated with combinations of harvest and fire was potentially greater than either disturbance alone (Connaughton 1935). Salvage logging after wildfires may have exacerbated erosion and sedimentation in the Stanislaus National Forest (Chou *et al.* 1994). This study demonstrated that the total basin area, roading, skid trails, and bare ground were important in explaining accumulated sedimentation rates in steeper sloped areas while the watershed shape was the most important variable in gentler sloped salvage areas. Despite salvage logging being widely practiced and the incidental burning of many cutblocks by wildfire, there is relatively little information on the impacts of multiple disturbances on sedimentation and erosion in most forest types in Alberta.

The effectiveness of buffers for controlling sediment transport depends on efficiency of removal by vegetation type, volume entering buffer per unit of time, and erodability of upslope surface, and internal buffer characteristics.

Like wildfires, timber harvest can have a major impact on erosion and sedimentation. The factors that lead to erosion and sedimentation in harvest systems are similar to those in wildfire systems. In general, the recorded amounts of sediment transport are smaller for harvest than for wildfire and significantly decline in the first few years after harvest (e.g. Megahan 1975; Thomas and Megahan 1998). One important difference is that watersheds may experience a chronic increase in sediment loading under harvest regimes, whereas in wildfire systems the initial impact may be more severe but there is a long cycle between disturbances. The lack of long-term impact and recovery data for either disturbance makes it difficult to compare their relative impacts.

The use of buffers for controlling erosion and sedimentation from human disturbances is intended to reduce the potential for near-stream soil erosion and to filter sediments from upslope surface flows. Tree canopies and ground litter disperse the physical impact and flow of heavy rainfall and snowfalls. Roots physically bind soil along streambanks and stabilize channels. Large organic debris and increased surface roughness created by fine organic debris can moderate the high stream velocities associated with spring melt and stormflows. Removal of the riparian buffers may potentially decrease litterfall and expose soils to greater impacts of rainfall. This could lead to an increased potential for erosion, particularly for sandy soils (France 1997d). Furthermore, debris can temporarily trap some stream sediment. The width of buffers required to reduce erosion and surface sediment flows are dependent upon 1) efficiency of sediment removal, 2) volume of surface runoff entering the buffer per unit time, 3) erodability of the upslope surface, and 4) internal buffer characteristics, e.g. slope, infiltration rate, and surface obstructions (e.g. leaf litter, vegetation, downed woody materials). In general, the greater the amount of desired sediment removal, surface flow, and erodability of upslope areas, the wider the buffer must be to be effective.

A number of models have attempted to integrate influential environmental factors and propose general widths for buffers (Wong and McCuen 1982; Cook College Department of Environmental Resources 1989). Wong and McCuen (1982) suggested that under most circumstances buffers less than 60 m were sufficient to control sedimentation. Similar results were inferred from a model built by the Cook College Department of Environmental Resources (1989). Researchers suggested that buffer widths of 15 m for slopes less than 1% to 61 m for a 15 % slope were required for sediment removal of overland flows. Areas with a slope >15% or those impervious to water should not be counted as part of the buffer width. Other factors which reduce the efficiency of buffers include presence of perpendicular channels (Dillaha *et al.* 1989) and reductions in natural impediments to ground flow (e.g. vegetation, wood debris, rocks). Floodplain buffers or buffers amongst braided streams were much less effective because of their tendency to flood and become submerged during heavy flows (Barfield *et al.* 1979). In this regard, buffers measured from terraces or from the top of an upslope break were much more effective than buffers of the same width measured from streamside.

Empirical evidence on the effectiveness of riparian buffers suggests variable results.

Results of field studies involving the effect of buffer widths on sediment removal are variable. For example, Moring (1982) reported that 30 m buffers were unable to prevent increases in stream sedimentation after partial clearcutting of small watersheds in Oregon. Whereas other field studies have shown that in areas with little relief and minimal forest floor disturbance, buffers required to prevent sedimentation can be relatively narrow (Patric 1978). Well-drained, coarse-textured sandy loams in the eastern United States have relatively low erosive overland flow, hence, sediment transport from timber harvest with minimal forest floor disturbance, can be mitigated with relatively small buffers (Hornbeck and Kochenderfer 2000). Haupt and Kid (1965) found that 9 m strips were sufficient to remove sediments from cutblock features in Rocky Mountain areas of central Idaho. Low relief Boreal systems are unlikely to yield large amounts of sediment (Steedman and France 2000), even without buffer strips, Steedman and France (2000) found relatively little sediment deposition from lakeside harvesting in northwestern Ontario. Keutzweiser and Capell (2001) compared three types of harvest; selection-cut (40% canopy removal), shelterwood-cut (50% canopy removal), and diameter limit cut (85% canopy removal) with undisturbed watersheds for sugar maple-dominated forests in northern Ontario. Most of the fine sediment deposition was due to the ground disturbance created by trails and roads for accessing and removal of timber. All harvest treatments produced less fine sediment than measurements associated with an unharvested road improvement in a similar watershed type. The shelterwood-cut produced the least amount of fine sediment and in the author's opinion could be harvested without riparian buffers.

A possible complicating feature in the efficacy of buffers is that temporal variation in sedimentation rates is likely to be high. In one of the few long-term datasets, annual sediment yields from undisturbed forests within the Hubbard Brook Experimental Forest varied from 0.9 to 125 lbs. per acre over 25 years of measurement (Martin and Hornbeck 1994). The higher values were attributed to individual storm events during the early years (Bormann *et al.* 1974). Similarly, Kochenderfer *et al.* (1997) found that a single large storm with a likely recurrence interval of 50 years was responsible for 35% of the sediment yield over an 11 year period in Virginia. In setting the width of buffers, managers must decide on the relative amount of risk they are willing to accept by leaving buffers wide enough to handle average or atypical conditions.

Much of the sediment transport after timber harvest is due to the proximity of roads to waterbodies. In low gradient ecosystems such as Boreal, careful road planning and harvesting, minimizing soil disturbance during harvest, and application of specific buffers to prevent channel development from roads to waterbodies may prevent much of the erosion and sediment transport.

Much of the data from harvest systems indicates that most of the sediment comes from roads and trails used in forest harvest. Packer (1967) surveyed cutblocks and logging roads in the Rocky Mountains and concluded that most of the sediment from streams in harvested lands originated from logging roads. Road cuts were associated with 66% of the landslides on granitic soils in Idaho (Megahan 1978). Sources of sedimentation originating from cutblocks, such as exposure of mineral soil by inblock traffic and inorganic and organic debris flow, are relatively minor in comparison to roads (reviewed in Karr and Schlosser 1978; Reid 1981). The amount of traffic, type of traffic, and type of road was critically important in determining the amount of sediment generated. A study in the Pacific Northwest demonstrated that areas which had more than four large trucks (i.e. logging trucks) passing per day on gravel roads produced more than 500 tonnes of sediment per km per year (Reid and Dunne 1984). Moderate (1 to 4 large trucks) and light use (small vehicular traffic) gravel roads produced 42 and 3.8 tonnes of sediment per km per year, respectively. Under any traffic load, paved roads yielded much less sediment (2.0 tonnes of sediment per km per year). Steedman and France (2000) found that clearcut areas, roads, and skid trails generated aeolian sediments but these were relatively small. Increased littoral sediment was evident in one of three treatment lakes (two lakes with clearcut/no buffers; one lake clearcut/buffer) but this area had a major haul road near the shoreline.

Current forestry practices attempt to minimize the amount of exposed soil, hence, sheet erosion from the cutover areas is not generally a problem (Robert Anderson pers. comm). Whereas, channelized flow through buffers, particularly those originating from roads, is likely to be the greatest source of sediment from forestry operations. In this case, universally applied buffers are relatively ineffective at preventing sediment transport. Planning of roads and clearings to avoid potential channels to streams and specifically applied buffers to these features to are likely more effective strategies in controlling sediment transport.

2.1.3 Nutrient Transport

Like water yield and sediment transport, nutrient yields increase immediately after natural and anthropogenic disturbances in forested systems. These decrease as the time since disturbance increases.

In most forest ecosystems, the transfer of nutrients from upland to waterbodies reaches a maximum immediately after disturbance. This is caused by the large amounts of fine litter created by disturbances and increased decay rates facilitated by elevated soil temperatures and moisture, and a lack of vegetation to sequester available nutrients. The amount and duration of increased transport are controlled by a number of factors including: climate, mineral weathering, soil characteristics, hydrological characteristics, disturbance intensity, vegetation, and biological processes.

In general, the increase in nutrients following disturbances declines relatively quickly. Minshall *et al.* (1997) recorded a 20% increase in the level of nitrates in burned catchments of lodgepole pine after the Yellowstone fires. Furthermore, there was a direct relationship between the transport of nitrogen and the proportion of the catchment burned. In a study of three burned watersheds in the eastern Cascade Range of Washington state, Helvey *et al.* (1985) reported that average solution and sediment losses of total N rose from 0.004 kg per year to 0.54 kg per year in the second year following fire. Losses of total N dropped to 0.04 kg per year by the fifth year after the fire. This relatively quick drop is reflected in a number of other studies. Losses of nutrients including nitrogen, phosphorous, and potassium were detected for only four years after clearcutting and burning of slash in western larch and interior Douglas fir stands in Montana (DeByle and Packer 1972).

For Boreal ecosystems, McEachern *et al.* (2000) reported 2-, 1.5-, and 1.2-fold increases in phosphorous, nitrogen, and carbon, respectively, in Alberta's Boreal lakes after a severe fire. Using a retrospective design, they further demonstrated that these levels may remain elevated for decades after disturbance. Carignan *et al.* (2000) documented changes in most water chemistry parameters to Boreal Shield lakes after harvest or wildfires. Total phosphorous (two- to three fold), total organic nitrogen (two-fold), and K^+, Cl^-, and Ca^{2+} (up to sixfold) were significantly higher compared to undisturbed reference lakes. Lakes within burned watersheds experienced 60- and 6-fold increases in NO_3^- and SO_4^{2-} concentrations, respectively. The degree of change for most of these parameters was related to the ratio of watershed area disturbed to lake area or volume. They further suggested that the dominant mobile ions from wildfire (K^+, Cl^-, SO_4^{2-}, NO_3^-) or harvesting

(K^+, Cl^-, some dissolved organic carbon) were rapidly flushed from the system; while total phosphorous, total organic nitrogen, dissolved organic carbon, Ca^{2+}, and Mg^{2+}showed little change or were still increasing three years after harvest. Clearcutting of northern hardwoods in New Hampshire resulted in an increase of nitrate from >5 ppm to 25 ppm in streams (Martin *et al.* 1986). However, these increases were relatively short-lived and declined to pre-cut levels within 10 years.

Similar results were demonstrated in European Boreal watersheds. Rask *et al.* (1998) attributed the increase in total phosphorous and iron in small Finnish lakes with harvested catchments (15 to 33%) to increased inputs of inorganic material. In addition, darker water colour and higher chemical oxygen suggested increased organic inputs. Furthermore, small streams exhibited increased levels of suspended solids (10x), total phosphorous (4x), phosphate phosphorous (5x), and total nitrogen (2x) after harvest and soil scarification of adjacent, unbuffered harvest areas. Total phosphorous and phosphate phosphorous declined to levels in reference streams after six years. Suspended solids and total nitrogen also declined but were still greater than reference streams. Areas that were buffered did not exhibit increases in either suspended solids or nutrients after harvest and site preparation. Contrary to these results, Evans *et al.* (2000) found no differences in the amount of daily total dissolved phosphorous exported between harvested and non-harvested subcatchments in Boreal forests in Alberta. They attributed the lack of differences to the confounding effects of confining clay layers and surface topography on hydrology within harvested and unharvested subcatchments.

Aside from increased nutrients, disturbance may play a role in the transport of other compounds or chemicals. For example, the amount of mercury found in northern pike from Boreal catchments on the Canadian Shield in central Quebec varied with disturbance type (Garcia and Carignan 2000). Fish from logged catchments exhibited higher average mercury levels than uncut lake catchments. Fish from burned catchments were intermediate to logged and uncut catchments. This follows a similar pattern detected for zooplankton from these same lakes (Garcia and Carignan 1999). They argued that the higher mercury levels in lakes were at least partially explained by higher inputs of terrestrial dissolved organic carbon (DOC). Though not generally considered in other nutrient transport studies, Telang *et al.* (1981) found that clearcutting resulted in increased amounts of tannins, lignins, humic acid, and fulvic acid found in adjacent streams in Alberta's Rocky Mountains. This occurred despite the use of 200 to 300 m buffer strips along adjacent streams. Elevated concentrations of humic and fulvic acids were relatively short-lived (~1.5 years) while tannins and lignins were still elevated 4 years after clearcutting.

In agricultural and urban systems, riparian buffers are relatively effective at removal of nutrients from upland flows. However, there is little evidence of their efficacy in forested systems. Furthermore, the greater complexity of subsurface hydrology and geology in western Boreal systems may negate the application of results from other forested ecosystems.

The question of whether riparian buffers are able to prevent or ameliorate the degree of nutrient mobilization is still debated. Much of the data on the utilization of riparian areas for reduction of flows from upland to waterbodies is based on their use as filters for urban runoff (e.g. Schueler 1987), agricultural pollutants (e.g. Peterjohn and Correll 1984) or pesticides (e.g. Nriagu and Lakshminarayana 1989). In these situations, buffers seem to be quite effective. However, in forested systems buffers may be relatively ineffective at removing nutrients originating from forest harvest practices (Norris 1993). For example, Steedman (2000) found that dissolved organic carbon, chlorophyll, total nitrogen, K^+, Cl^-, and Si all increased (10 to 40%) by the second and third year after logging within the headwaters of boreal Precambrian Shield lakes. The same level of change occurred whether riparian buffers were present or not. Both buffered and unbuffered results were similar to those of Carignan *et al.* (2000) whose harvest sites had 20 m buffers. Further, elevated nitrate levels were detected in run-off from clearcut areas in Pennsylvania watersheds despite the presence of a selectively harvested 31 m buffer (Lynch and Corbett 1990); although some of the nitrate discharge may have been caused from blowdown within the buffer itself and from inflows from unbuffered intermittent streams.

Other broad landscape level factors such as area harvested, topographic positions and geologic features may influence lake and stream chemistry to a greater degree than the presence of forested riparian buffers. For example, Devito *et al.* (2000) demonstrated that lake position relative to groundwater flow and surface hydrology connections between lakes and wetlands were the primary determinants (57% variation) explaining the change in pre- and post-harvest concentrations of open water total phosphorous. The width of riparian buffers was not a significant factor or at least was not significant relative to these landscape factors. In comparison to the relatively simple hydrogeology of most humid forest ecosystems, the Western Boreal Plains feature a drier climate, deeper surficial glacial deposits, and larger and more complicated groundwater flow systems (Tóth 1970). Devito *et al.* (2000) argues that differences in physiographic features among regions makes extrapolation of riparian research and management from other systems to the Western Boreal Plains difficult and perhaps inappropriate. Therefore, research and management practices should be based on research specific to a particular region (e.g. HEAD Project 2002).

2.1.4 Temperature Control

The temperature of waterbodies greatly influences the productivity and biotic communities that inhabit the aquatic and terrestrial portions of riparian areas. Direct solar radiation on the surface of the water is the primary natural source of thermal energy (Brown 1980). Shading created by overhead vegetation in riparian areas is the key modifier of solar radiation and, hence, the temperature of streams. Brown (1969) demonstrated that the removal of overhead shading could increase the amount solar radiation reaching streams by as much as six times. Other factors such as flow rate, surface area, and bank elevation can also affect stream temperature. There are few studies for the Boreal or the Rocky Mountain regions, which document the impacts of streamside vegetation removal on stream temperature.

The increase in water temperature created by removal of riparian vegetation by disturbances has been shown to vary among different systems. In burned Ponderosa pine stands in north-central Washington, Helvey (1972) reported increased water temperatures in early July through to late August. At its peak, temperatures were ~5 ^{o}c greater than would be expected without the loss of streamside vegetation due to wildfire. Streams from partially harvested (7% to 33% removal) Douglas fir watersheds on the Olympic Peninsula in Washington exhibited an increase of 3.5^{o}c in the average maximum summertime temperature. Furthermore, stream temperature was more seasonally variable in harvested watersheds compared to streams in nearby uncut old growth stands. Such temperature increases were considered not sufficiently high enough to affect salmonid species (Murray *et al.* 2000). Young *et al.* (1999) found that nonanadromous cutthroat trout streams harvested to streamside margins reached a maximum summer temperature of 30^{o}c but had recovered to pre-harvest levels 10 years after harvest; although an initial four-fold decline in fish density was attributed to the increased water temperature. Removal of buffers from patch cut and clearcut/burned basins in the H.J. Andrews Experimental Forest in the western Cascades, Oregon, resulted in an increase of the maximum summer temperature, which occurred earlier in the season (Johnson and Jones 2000). Stream temperatures returned to pre-harvest levels after about 15 years. The relationship between buffer width and buffer length on stream temperature was explored by Barton *et al.* (1985) for streams in southern Ontario. Both longer and wider buffers led to lower stream temperatures, although longer buffers could be narrower and achieve the same degree of temperature control as wider buffers.

Water temperature is regulated in small waterbodies by the shade cast of shoreline vegetation. Significant increases in water temperature can made created by harvesting riparian vegetation.

On large waterbodies such as lakes, harvest of riparian areas has little effect on water temperature. Steedman *et al.* (1998, 2001) demonstrated that the average littoral water temperature was generally not warmer in two lakes with clearcut shorelines than buffered (~30 m) lakes in northwestern Ontario. Only during the early summer was the maximum littoral water temperature (1 to 2^{o}c) and average diurnal temperature range (0.3 to 0.6^{o}c) greater along clearcut shorelines. Less than two degree-days were accumulated over the three month summer period. Hourly patterns of solar energy largely drove the hourly littoral water temperature patterns, while day-to-day water temperature patterns were associated with air temperature.

Selective harvest, i.e. thinning, of buffers has little effect on their ability to provide shade and regulate stream temperature.

Partial harvest or thinning along streams had a lower impact on stream temperature than clearcutting. For example, removal of 44% of the basal areas of trees (>2.54 cm DBH) within riparian buffers did not significantly increase peak water temperatures in streams running through harvested Appalachian forests in West Virginia (Kochenderfer *et al.* 1997). Similarly, Aubertin and Patric (1974) found that partially cut (~50% retention) buffers (10 to 20 m wide) were still able to sufficiently shade streams and prevent temperature increases.

Combinations of harvest and silvicultural activity can have synergistic impacts on stream temperature. Lynch *et al.* (1984) compared the impacts on stream temperature of clearcut and clearcut-herbicided watersheds in Pennsylvania. They found that average summer maximum temperatures were 1^{o} and 9^{o}c higher in clearcut and clearcut-herbicided watersheds, respectively, when compared to unharvested areas. Similarly, a comparison of clearcutting and clearcut/slash burning treatments in southwestern British Columbia indicated that both treatments increased stream temperature relative to forested controls (Feller 1981). Increased temperatures lasted seven years in clearcut stands, but were more persistent in clearcut/slash burned stands.

Elevated stream temperatures can be lowered by flowing through shaded areas. Ground water flows can increase or decrease stream temperature depending on the cover in source areas.

A number of studies have examined changes in water temperature as stream channels pass from open to shaded areas. Streams running through clearcuts in western Oregon exhibited slight increases in temperature but these decreased to base temperatures after traversing fully forested areas for 150 m (Zwieniecki and Newton 1999). Similarly, in North Carolina, a 12 m buffer was effective in moderating increases in stream temperature in clearcut areas. They further noted that the stream temperatures fell upon entering neighbouring uncut forests and attributed the temperature decrease to cooler groundwater entering the stream. Decreased stream temperature due to the influx of cooler groundwater from forested areas has also been documented in other forested systems (e.g. Brown *et al.* 1971; Patton 1980; Swift and Baker 1973). In a reverse application, Woodall (1985) demonstrated that the shading effects of streams may be circumscribed by warmer groundwater entering from an upstream clearcut. Input from groundwater flow within 4 m of a clearcut surface could be sufficiently warmed to alter the stream temperature. Hewlett and Fortson (1982) also suggested this as a possible explanation for stream temperature increases despite the use of relatively wide buffers in southeastern Piedmont forests.

Most models of shading indicate that buffer widths of 30 m provide sufficient shading to maintain water temperature in small bodies. However, there is a lack of data for Alberta systems.

In a review of forest practices, Binkley and Brown (1993) concluded that almost universal use of intact or partially harvested buffers has significantly reduced increases in stream temperature after harvesting. A number of models exist for prescribing buffer widths to control solar radiation (e.g. Adams and Sullivan 1989; Beschta and Weatherred 1984). The important variables included; stream width and volume, buffer forest height and density, amount of watershed cut, solar inputs, and groundwater temperatures. Most of these models indicate that buffers of ~30 m wide were sufficient to prevent stream temperatures from rising. There is, however, a dearth of data from Alberta's forest systems. Given the importance of latitudinal angle, local topography, soil characteristics, hydrological characteristics, and responses by regional biota to stream temperature, it would be worthwhile to initiate and support ongoing research in this area. In particular, there is a lack of baseline knowledge and data to evaluate the impacts of stream temperature on aquatic and terrestrial riparian communities within the Boreal forest.

2.1.5 Organic Matter

The function of organic debris is determined by its size. Hence, classification schemes often focus on size.

Since this document looks at the role of riparian buffers in forested ecosystems, this section will focus primarily on the impact of allochthonous[1] inputs of organic material. Allochthonous inputs include: organic soil sediments, leaves, needles, bark, fruits, branch wood, and boles. The function of organic debris is largely dependent upon its size (Platts *et al.* 1987); therefore, debris is generally divided into three or four size classes (e.g. Tables 2 and 3).

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[1] Allochthonous input refers to materials derived from the surrounding lands. Autochthonous refers to the instream production of organic matter primarily through accumulations of algae or vascular plants.

Table 2. Classification system prescribed by Platts *et al.* (1987).

Abbreviation	Category	Size Range (cm dia.)
DOM	Dissolved organic material	<0.05
FPOM	Fine particulate organic material	0.05 to 0.10
CPOM	Coarse particulate organic material	0.10 to 10
LOD	Large organic debris	>10

Table 3. Classification system prescribed by Webster *et al*. (1990).

Abbreviation	Category	Size Range (cm dia.)
FBOM	Fine benthic organic matter	<0.1
CBOM	Coarse benthic organic matter	0.1 to 1
	Small wood	1 to 5
	Large wood	>5

Small particles are food for microbes and macroinvertebrates. Its loss can produce declines in these groups.

Smaller organic particles serve as a food source for microbes and macroinvertebrates. There loss of fine organic debris can result in declines of both types of organisms particularly macroinvertebrates (e.g. Wallace *et al.* 1997). These smaller particles also play significant roles in determining levels of lake carbon (Odum and Prentki 1978), light penetration (Bolgrien and Kratz 2000), and transport of metals (Kortelain 1993). For stream and possibly small lake systems, most fine debris comes from shoreline vegetation. France *et al.* (1996) found that harvesting of the riparian canopy around Boreal Shield lakes reduced the allochthonous inputs of leaf litter and small woody debris by 90%. The loss of allochthonous leaf litter resulted in subsequent declines in the input of dissolved carbon and total phosphorous. By tracking the transport of stable carbon isotopes, France (1997b) was able to further demonstrate that substantial amounts of detritus were derived from riparian trees. Similar declines in allochthonous materials have been noted in a number of forested, small stream systems in which treed riparian buffers have been removed (Bilby and Bisson 1992; Duncan and Brusven 1994; Webster and Waide 1982; Webster *et al.* 1990). Despite this, the loss of allochthonous material through harvesting was not thought to be significant since the reduced material has a relatively long-life relative to its input rates. France (1997c, 1998) argued that riparian cutting around lakes is not likely to have a detrimental effect on the taxa richness or abundance of macroinvertebrates.

The source of debris depends upon waterbody size or stream order. Smaller stream orders and waterbodies rely on external, i.e. allochthonous, sources while larger stream orders and waterbodies rely on internal, i.e. autochothonous, sources.

The relative input of allochthonous and autochthonous materials appears to be dependent on stream order. Connors and Naiman (1984) assessed the input of allochthonous inputs according to stream order in eastern Quebec. In first and second order streams, approximately 81 to 95% of organic carbon was derived from allochthonous inputs, whereas in fifth and sixth order streams 85 to 95% of inputs were autochothonous. Thus, a significant portion of the organic materials appears to enter stream systems from the headwater reaches. They also reported that although the surrounding forests were primarily coniferous, deciduous leaves dominated the annual budget and seasonal input patterns. Similar results from other Boreal forests indicate that allochthonous inputs dominated (>75%) second order streams and autochthonous inputs dominated fourth order streams (Junger and Planas 1994). Measurements of stable carbon isotope transport confirmed that the invertebrate food base was dominated by autochthonous inputs (>80%).

Aside from direct input, breakdown of large debris is potentially an important source of fine organic debris.

A second source of fine organic materials is derived from the breakdown of coarse or large organic debris. In most forest types, it is unclear what proportion of instream fine material comes from the breakdown of larger materials. Ward and Aumen (1986) estimated that in Oregon's western Cascades, the amount of fine material derived from large pieces could be significantly greater than the allochthonous input of fine materials.

Anthropogenic inputs of large debris has had a checkered past. At times its input was a favoured management practice at other times it was not.

Historically, the management of large woody debris in North American streams has been controversial. Early forestry used waterways to transport or "drive" logs to mills. In process a called "carding the ledges[2]" or "snagging", streams were cleared of debris including fallen logs and streamside vegetation to allow for the easy passage of logs (Smith 1972). As this method transport gave way to rail and truck transport. Many streams were left littered with the debris of past drives. Later many of these rivers would be "cleaned" again in an effort to correct past management practices and improve their aesthetic appeal.

Current ecological management sees large debris input as critical habitat feature for aquatic organisms.

Current forest management considers the input of large woody debris as a critical component of stream ecosystems (reviewed in Sedell *et al.* 1990). Large pieces of woody debris create pools, trap sediments, alter channelization, retard scouring of streambeds during high flows, and provide cover and substrate for fish and invertebrates. Root wads, small and large branches, and boles spatially partition the water column and channel span to provide ecological separation and habitat diversity. Areas of slack water, such as pools, created by debris act as refuges for aquatic and semi-aquatic organisms during periods of high or low flow. The surface of woody debris offers a substrate for the completion of many life functions of organisms, e.g. attachment for egg masses.

In part, the impact and function of woody debris depends upon its size relative to that of the waterbody. Gurnell *et al.* (2002) argued that streams should be categorized according to the size of pieces entering the channel. Small streams would be defined as having a channel width < median wood piece length; medium streams would have channel width > upper quartile piece length; and large streams would have channel width > all wood piece lengths. Stream flow regime, morphology, and bottom substrate are important factors in determining the movement and stability of woody pieces. Woody debris in small streams tends to remain where it falls and acts

[2] The origin of the term is unclear but one suggestion is that it derives from "carding" wool where a brush with short steel bristles is combed over wool to straighten the strands (Verry And Dolloff 2000).

to facilitate channel development. In medium streams, large pieces are important for the controlling the in-stream movement of other woody pieces and sediment transport. In both small and medium streams, woody pieces greatly enhance the interaction between aquatic and terrestrial interface. Lastly, large streams can transport pieces long distances depending upon channel geometry and distribution of flow velocities within the channel. Riparian areas such as islands and large floodplains are relatively enduring storage sites for large amounts of woody debris that can be released during periods of high flow. In these areas the supply of woody debris is often sufficiently constant to provide habitat for small mammals and birds (e.g. Steel *et al.* 1999).

Inputs of coarse woody debris are highly varied over time with large pulses of input after natural disturbances such as wildfires.

The input of coarse woody debris is highly variable over time. Relatively rare events such as hurricanes, volcanoes, and severe flooding can have long lasting effects on stream ecosystems (e.g. Covich *et al.* 1991; Dolloff *et al.* 1994; Lisle 1995). On shorter time scales, insect outbreaks (but see Hedman *et al.* 1996), wind storms, ice storms, and wildfires contribute less dramatic pulses of woody debris into streams. Immediately following the 1988 wildfires in Yellowstone National Park, the amount of debris entering streams peaked, but declined rapidly as lodgepole pine stands re-established (Minshall *et al.* 1989). The impacts of debris input were greater on smaller streams (<3rd order) than higher stream orders. Debris dams formed along many of the watercourses, greatly retarding and backing up the flow of water. These served to trap further accumulations of inorganic and organic debris. The breakdown of large debris in first few years after wildfire maybe accelerated by increased stream temperatures, nitrogen levels, channel instability, and greater invertebrate abundance (Golladay and Webster 1989). In a comparison of burned and unburned watersheds, Young (1994) found that the debris in the burned watershed was three times more likely to be transported than in an unburned watershed. Increased debris input and flows, and decreased bank stability attributed to the greater transport of debris after fire.

Unlike natural disturbances, timber harvest, particularly of riparian areas, is likely to lead to losses in inputs of large woody debris to streams .

Unlike streams affected by fire, the impact of harvesting on debris input depends on the management of streamside buffers. Clearcut areas with sufficiently wide buffers could contribute large debris to streams in amounts similar to uncut forests (Toews and Moore 1982). Based on data from six Alaskan streams, Murphy and Koski (1989) found that 95% of woody debris in streams originated from within 20 metres of the stream bank. Similarly, other studies have found that >99% of the coarse woody inputs were derived from within a single tree length of the stream bank (McDade *et al.* 1990, Robinson and Beschta 1990; VanSickle and Gregory 1990). In Washington and Oregon, a 30 m strip on one side of a stream provided 85% of the input of intact forests but a 10 m strip provided less than half of the naturally occurring input of debris (McDade *et al.* 1990). Reid and Hilton (1998) recommended an additional three tree lengths be added to buffer widths in an effort to decrease the accelerated fall rates noted after implementation of narrow buffers.

Aside from the amount of debris, the types and distribution of debris varies after harvest. Ratios of large:small organic debris, attached:unattached large debris, and pieces with rootwads:pieces without rootwads were relatively evenly dispersed amongst undisturbed bull trout spawning reaches in Montana (Hauer *et al.* 1999). Distributions of these same elements were much more varied amongst logged watersheds. This indicated that the delivery, storage, and transport of large debris had changed in logged habitats in the northern Rocky Mountain streams.

Though little data exists for harvest systems in the Boreal or Rocky Mountain ecoregions, the trajectories for large woody debris inputs are likely to be the same as upland systems. There is a decline after harvest that would last at least 20 years and possibly longer for the input of very large pieces of debris.

There exists little field data for riparian areas on the long-term temporal changes in debris dynamics from disturbance, through regeneration, to mature forest. In the Pacific Northwest of the United States post-harvest development has been described through direct field measurements and retrospective chronosequences (Swanson *et al.* 1976; Swanson and Lienkaemper 1978). In general, a pulse of autochthonous inputs occurred for a short period immediately after harvest, primarily due to increased algal growth stimulated by increased amounts of sunlight and nutrient inputs. Within five years, herbs and shrubs began to shade over the water surface reducing algal growth and autochthonous inputs. These were replaced by fine particulate debris primarily from leaves. Shrubs and small hardwood trees dominated the next seral stage of riparian succession. This sere was relatively productive for fine debris because of the annual input of deciduous leaves. Deciduous vegetation was slowly replaced by regenerating conifers with their inputs of branches, small boles, and needles forming the basis of inputs by about ten years. As these trees grow and die, inputs of large downed woody debris began to impact the stream by sixty years. In general, patterns were thought to be similar to upland debris input.

Relatively little data is available from the Boreal or Rocky Mountain regions on the amounts of downed woody inputs into waterbodies after harvest or wildfires. If the trajectory of riparian areas is similar to upland areas after disturbance (Lee *et al.* 2001), then we can hypothesize on its pattern after disturbance. In most fire-dominated systems, there is an initial pulse (~0 to 20 years) of coarse and large debris from the falldown of fire-killed trees, then a period of relatively little input of large debris during the middle seral stages (~20 to 50 years). As the watershed continues to age and the regenerating stands grow, there should be an increasing input of larger-sized pieces. Input of larger pieces into streams is concurrent with growth and mortality in regenerating stands. This continues until older seral stages, when the input rate plateaus or declines slightly. One important difference is the storage of large woody debris in riparian systems. Large woody pieces may reside for an order magnitude longer in Boreal freshwater systems than comparable above-ground terrestrial riparian and uplands (Guyette *et al.* 2002). Estimated mean residence times ranged from 261 years for small oligotrophic lakes and 350 to 800 years for major stream reaches.

Harvest systems which leave only narrow riparian buffers or harvest to streamside are likely to have a very different developmental trajectory than either wildfires or harvest systems that retain wider buffers. Loss or thinning of buffers in harvest stands is could result in early and mid-seral stage declines in medium and large woody debris. There is no initial pulse of debris into these systems following harvest as there is with wildfires. The lag between the regeneration of the harvested buffer extends the period without large debris inputs for potentially 40 to 50 years. Even at these older ages, the diameter of fallen debris is just above 10 cm (Lee *et al.* 1997). Combined with a relatively short rotation (~70 to 100 years), the total period of input for large debris is relatively short, possibly less than a half of the rotation length. Under this scenario, the harvesting of riparian areas would lead to an initial decline in coarse and large debris inputs and a potentially long-term loss of large debris with each successive rotation.

2.1.6 Invertebrates and Other Taxa

Invertebrates form a large and diverse group and can be categorized by their functional feeding relationships.

Invertebrates cover a large and diverse group of organisms including; protozoans, turbellaria, oligocheate worms, leeches, water mites, molluscs, crustaceans, and insects. Most of the research and management focuses on the impacts of forestry on macroinvertebrates. Within this group the insects have received the most study. Macroinvertebrate groups are often described by their functional attributes. The major functional feeding groups include: shredders, grazers/scrapers, predators, collectors, and scrapers/raspers (Cummins 1973; Cummins and Klug 1979; Cummins and Merritt 1984) (Table 4, Figure 3). Data on other invertebrates are generally lacking despite their importance in stream ecosystems (Giller and Malmquist 1998; Cushing and Allan 2001).

Wildfires can greatly alter macroinvertebrate communities but these recover after changes in stream temperature and organic inputs subside.

Driven primarily by increases in stream temperature and organic inputs after wildfires, macroinvertebrate densities can change greatly but tend to recover relatively quickly afterwards. After the Yellowstone fires (1988), there was a significant decrease in taxonomic richness prior to the first spring snowmelt in smaller streams but not in larger or reference streams (Minshall and Robinson 1996). By two to four years after the fire, there were no differences in richness amongst all burned and reference streams (Minshall *et al.* 1997). However, the densities of most species grew for the first few years after fire for both small and large streams. In particular, opportunistic species such as chironomids and *Baetis* increased. Their life histories feature drift dispersal and short generation times, traits well suited to conditions after wildfires.

In Alberta's Boreal systems, total biomass of macroinvertebrates from burned watersheds (average of 90% burned area) was much greater than reference watersheds (Scrimgeour *et al.* 2000). Total faunal densities were twice as much in burned lakes compared to reference lakes. Oligochate biomass was greater in harvested compared to burned or reference lakes but no differences were recorded for chironomid biomass. Biomass of the five most abundant macroinvertebrate groups were related to colour, soluble reactive phosphorous, total inorganic nitrogen, and epilithic chlorophyll a.

Table 4	Brief descriptions of instream functional trophic groups (modified from Giller and Malquist 1998).

Functional Group	Primary Food Source	Feeding Mechanism	Groups
Shredders/Gougers	Leaf detritius, wood, living aquatic plants	Chewing of detritus and macrophytes, mining of macrophytes, and gouging wood	Amphipods, isopods, stoneflies, caddisflies, moths/butterflies, beetles, flies
Collectors	Fine particulate organic matter (FPOM)	Filter feeding of suspended and deposited FPOM	Mayflies, caddisflies, beetles, flies
Scrapers/Grazers	Attached algae and biofilm	Removal of biofilm and algae affixed to instream mineral and organic surfaces.	Mayflies, caddisflies, moths/butterflies, beetles, flies
Macrophyte piercers	Cell and tissue fluids of living plants	Piercing and sucking of plant tissues	Caddisflies,
Predators	Tissue of living animals	Engulfing or piercing of prey	Alderflies/dobsonflies, beetles, caddisflies, dragonflies/damselflies, flies, stoneflies, water bugs
Parasites	Tissue or fluids of living animals	Internal or external parasitism	Parasitic flies, parasitic wasps

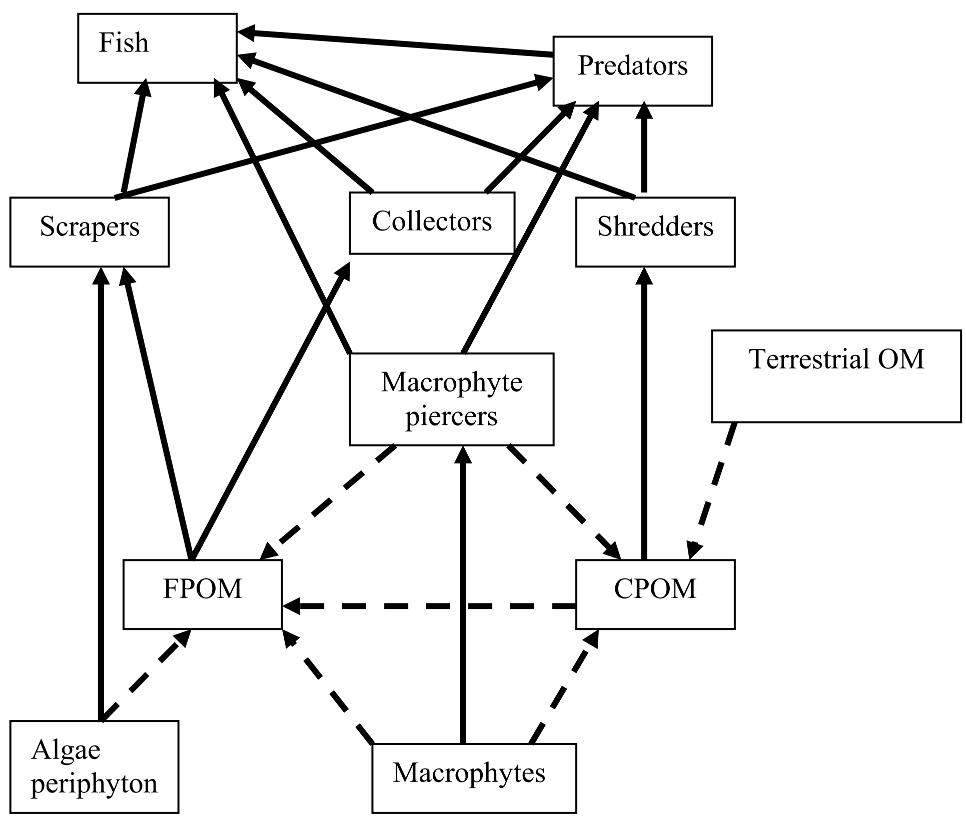

Figure 3	General trophic diagram of functional feeding groups. Solid arrows indicate feeding pathways. Broken arrows indicate decomposition of detritus.

There is often a change in macroinvertebrate communities following forest harvest. In particular, increased water temperature and light triggers increases in plant productivity and in growth and reproductive rates in macroinvertebrates. In a comparison of undisturbed and partially harvested (26 to 54% tree removal, 6 to 17 years previously) stream segments in northeast Oregon, similar amounts of woody debris were reported in streams and pools (Carlson *et al.* 1990). However, macroinvertebrate densities were 20 to 113 times greater in logged sites, although diversity was the same between logged and undisturbed sites. The increased densities were particularly notable in lower elevation streams and those less shaded by vegetation. Similarly, greater amounts of light after logging increased the density of both invertebrate and periphyton communities in small buffered (8 to 9 m) streams in northern New England (Noel *et al.* 1986). The impact of upstream detritus input on downstream macroinvertebrate communities was examined from streams in Alaska (Piccolo and Wipfli 2002). Young alder-dominated riparian areas which developed after immediately after harvest provided significantly more detritus and produced a greater macroinvertebrate biomass downstream than did young growth conifer riparian areas. Old growth and clearcut areas were intermediate between these forest types. Haapala *et al.* (2001) noted that alder detritus after harvesting in Central Finland supported more macroinvertebrates than either willow or birch leaves. They further suggested that shredders in Boreal streams face a bottleneck in late spring when leaves from the previous year's leaf fall were depleted. In a comparison of mixed, deciduous and conifer forests in Denmark, Friberg *et al.* (2002) found that shredder biomass was 4.5 times greater in streams running through deciduous forest over those in conifer forests. Mixed forests were intermediate. Conifer forests had streams with lower temperatures, less favourable water chemistry, and a lower quality of available food resource. They concluded that the shift to intensive (sic. enhanced) forestry had the potential to alter higher order trophic relationships and ecosystem function.

Harvest can also alter macroinvertebrate communities particularly if it results in changes to riparian buffers that alter stream temperature, organic inputs, or increases in sediment transport.

In a series of pioneering studies, changes in macroinvertebrate assemblages were examined in harvested and unharvested watersheds in the southern Appalachians of the eastern United States (Gurtz and Wallace 1984; Wallace and Gurtz 1986). They demonstrated that the number of taxa with increased densities were associated with moss-covered rock face, cobble riffles, pebble riffles, and sand (ordered highest to lowest) increases. Conversely, the number of taxa with decreased densities were highest on sand, pebble, cobble, and rock face. They suggested that substrate stability offered refugia for species and was a primary determinant of community structure after disturbance. Harvesting shifted the assemblage of macroinvertebrates from shredders to scrapers and collectors. Furthermore, species that maybe relatively infrequent in undisturbed systems may become dominant for long periods of time after disturbance.

Impacts on the macroinvertebrate community are often associated with secondary effects caused from harvesting. Channelized flow of forest ditches through buffers was a source of inorganic sediments which altered the invertebrate community in Swedish boreal systems (Vuori and Joensuu 1996). Specifically, sedimentation decreased the number and abundance of shredder-feeding stonefly species but increased the number and abundance of filter-feeding blackfly species. Culp *et al.* (1986) suggested that the damage caused by sediments was due largely to their physical movement (i.e. scouring) along the stream channel. The impact of sedimentation was immediate for species residing on substrate surfaces but delayed (6 – 9 hours) for species residing in deeper areas.

Micro-organisms can provide some insights into the long-term impacts of disturbance in watersheds. As an example, Scully *et al.* (2000) were able to develop insights into the long-term effects of harvest and regrowth on a small boreal lake. They found annual laminations of sediments ceased concomitant with logging from 1870 to 1890, indicating a change in the lake's mixing regime. Other changes included a loss of deepwater anaerobic photosynthetic bacteria and a reduction in metalimnetic chrysophytes. Deepwater bacteria remained absent for 100 years while chrysophytes returned slightly earlier. No changes were noted in cyanobacteria, chlorophytes, and cryptophytes. Changes were not indicative of either eutrophication or increased deposition of organic matter. Rather the researchers argued that wind stress after harvest led to deep water column mixing in the fall and reduction in deepwater anoxia. The authors further suggested that long-term effects after disturbances such as near-by timber harvest may be common in small, stratified boreal lakes that lack physical shelter due to landforms.

Harvest can also produce changes in microinvertebrate and algal communities.

Relatively few studies have measured the impact of disturbance on microinvertebrate and algal communities. Small streams in Boreal Finland exhibited a bloom in the biomasses of Volvocales and Cryptophyceae, which reached a maximum corresponding to times of year with the greatest open water (Holopainen and Huttunen 1998). Algal biomass declined within five years after harvest due to a decline of nutrient inputs and temperature. Impacts of harvest on algal abundance and diversity were largely due to increased nutrient content, light, temperature, and water colour. The authors argued that the use of buffers was important to moderating algal increases by providing shade. Patoine *et al.* (2000) assessed zooplankton in Boreal Shield lakes in Quebec after harvesting and wildfires. They found that most changes occurred within the first few years after disturbance. The rotifer-sized fractions increased in the first and second years after wildfire, but by the third year, densities had dropped to levels similar to undisturbed and logged watersheds. No differences amongst disturbances for any year were recorded in the crustacean-sized (largest), copepodite, or algal-sized (smallest) fractions. Rask *et al.* (1998) found no changes to phytoplankton densities but that periphyton growth (chlorophyll a concentrations) and zooplankton densities had increased after timber harvest.

The presence of undisturbed refugia within the watershed network is critical to the timely recovery of disturbed areas of the watershed.

The presence of instream heterogeneity, as a source of key refugia in streams and lakes, is important for the long-term presence of all aquatic organisms. For example, waters around abandoned beaver lodges in Boreal headwater lakes in northeastern Ontario were found to have a greater richness and abundance of benthic macroinvertebrates compared to other areas (France 1997c, 1998). These lakes were generally characterized by sand, rock, and sparse macrophyte growth. Beaver lodges provided important structural accumulations of coarse woody debris and sediment in an otherwise poor habitat. Clifford *et al.* (1993) found that the invertebrate assemblages around beaver dams differed from assemblages in main stem Boreal streams in Alberta. Assemblages associated with beaver dams were typical of fast flowing, lotic habitats whereas main stream and beaver pond assemblages were more typical of slow-flowing or lentic habitats. Furthermore, the recovery of macroinvertebrates after 12- to 18-year cyclical floods in second order Appalachian streams were largely attributed to the presence of refugia including debris dams, deep interstitial habitat, and first order tributaries (Angradi 1997).

Riparian buffers have reduced the impacts of upland disturbance in other forested ecosystems. At the current time, there is little data from Boreal and Rocky Mountain ecosystems.

The effectiveness of forested buffers in reducing impacts of harvesting on lake and stream biota has been documented in a relatively small number of studies. Newbold *et al.* (1980) studied the impact of stream buffer width on invertebrate communities in northern California steams. Streams with buffers ≥30 m exhibited no impact of harvest on invertebrates while streams with buffers <30 m experienced changes in species diversity. Amongst the narrower buffers, changes in diversity were positively correlated to buffer width. Post-harvest sampling (6 to 10 years after logging) indicated relatively little evidence of community recovery. However, the impacts on invertebrate communities were reduced when careful logging was practiced. Rask *et al.* (1998) found an increase in macroinvertebrate diversity but no changes in biomass of small, buffered lakes (50 m buffers) within harvested catchments.

2.1.7 Fish Communities

Fish are important indicators of the riparian ecotone.

Fish are an important indicator of aquatic ecosystem health (Karr 1991). Furthermore, for most of the public they are the organism most readily identified with the aquatic environment. Adult fish occupy the upper trophic levels in many aquatic communities. In the temperate zone, fish are predatory on macroinvertebrates and other fish. Changes in their prey base due to shifts in habitat or direct mortality tend to show quickly in their abundance. Zalewski *et al.* (2001) hypothesized that freshwater fish diversity, recruitment, and production depended on riparian ecotones. According to the authors, riparian inputs included organic matter important for the fish prey base of invertebrates, feeding places, shelter for fish particularly for reproduction and young fish. The authors further argued that self-regulating nature of riparian ecotones, i.e. resilience to disturbance and recovery after disturbance, were important features in determining their ability to maintain robust fish populations and diverse communities. Geology, physiography, climate, vegetation, human age, and successional stage should impact the energy and material transfer from the riparian ecotone. In turn, these factors should partially regulate fish populations.

In general, fish are adapted to cyclical disturbance patterns. Like macroinvertebrates, access to undisturbed refugia in stream network sare an important feature in facilitating recovery.

In a review of wildfire affected aquatic ecosystems, Gresswell (1999) found that aquatic biota in these areas had evolved strategies to survive disturbance cycles of greater than 100 years. Despite conditions that resulted in local disturbances, rapid recovery, i.e. years, (e.g. Rieman and Clayton 1997) was often possible if fish were able to recolonize from undisturbed parts of the stream network. Not surprisingly, mobile species were less impacted by disturbances than sedentary species. However, lasting effects from disturbance were most notable in native species whose populations had declined and become isolated through anthropogenic activities.

Protection of fish populations from the direct effects of forestry include: 1) management of the proportion of watershed harvested, 2) protection of headwater streams, 3) elimination of fine sediment, 4) provision of woody debris, and 5) provision of undisturbed refugia.

The impacts of timber harvest on fish populations reported in the literature are highly variable. Richards and Hollingsworth (2000) identified five important aspects of timber harvest that related directly to fisheries management. These included a) proportion of watershed harvested, b) protection of headwater streams, c) elimination of fine sediment, d) provision of coarse woody debris and habitat diversity, and e) provision of undisturbed refugia. Some studies have found no impact of harvest on fish populations. St. Onge and Magnan (2000) found no difference among burned (50 to 100% cleared), logged (8.5 to 73.2% harvested), and uncut lakes in the Laurentian Shield lakes of Quebec for catch per effort of white sucker, northern pike, yellow perch, lake whitefish, fallfish, brook trout, and burbot. The only difference was that the proportions of smaller bodied fish, i.e. yellow perch and white suckers, were smaller in burned and logged lakes. They hypothesized that this effect could be due to an increased post-emergent mortality or a shift of individuals to the pelagic zone. Similarly, no changes were found in the population structure or growth of perch in small Finnish lakes after harvesting (Rask *et al.* 1998).

The impact of timber harvest on fish populations is highly variable. The literature is complete with: 1) losses of fish populations from direct and indirect effects, 2) increases in some species, and 3) no effects.

In other studies, a reduction of fish populations was noted after timber harvest. Direct effects include the loss of streambank cover, increase in stream temperature, and/or loss of large woody debris. A reduction in brook trout was reported after clearcutting in Mastigouche Wildlife Reserve, Quebec (Bérubé and Lévesque 1998). Less obvious potential impacts of timber harvest have also been found to affect fish populations. For example, spawning and incubation habitats for brook trout were manifestations of coarse overburden lenses in the nearshore zone (Curry and Devito 1996). These lenses directed and accelerated groundwater flows to these areas. The authors suggested that a 90 m buffer in the terrestrial margin of these nearshore zones would only protect 9 to 55% of the recharge areas for these lenses. Declines were attributed to damage of spawning and nursing habitats. Wesche *et al.* (1987) found that overhead bank cover provided by riparian vegetation explained the greatest amount of variation in trout population size in Wyoming streams. Based partially on this and other results, Covington and Hubert (2000) developed a riparian vegetation index using measurements taken from aerial photographs. In one of the few studies to quantify changes in fish habitat over a relatively long period of time, McIntosh *et al.* (1994) compared >8,000 km of stream data from 1934 to 1942 and 1990 to 1992 from eastern Washington and Oregon. In general, they found streams in developed (timber harvest, livestock grazing, mining, and stream channelized) areas experienced a decline in both large woody debris and large pools whereas both these features were either similar or had increased in undeveloped watersheds.

Other negative effects of forestry were not directly caused by harvest activities but were the indirect side-effects of access created by logging roads. Impacts of forestry roads on lake trout populations were investigated by Gunn and Sein (2000) in three lakes in Ontario's Boreal Shield ecozone. They found no direct effects of harvesting through the loss of spawning habitat due to siltation. Instead, they argued that the more significant negative impact occurred from the increased access that led to over-fishing of lakes.

A number of studies have indicated an increase in fish abundance or condition with the removal of riparian vegetation. Dale Jones III *et al.* (1999) argued that removal of streamside vegetation in southern Applachian streams favoured species that guard pebble or pit nests or that live slow deep water. Their study focused on permanent deforestation of riparian areas. It is unclear how the repeated cycles of harvest and regeneration would shift fish assemblage.

Increased water temperature and fine organic inputs have been demonstrated to raise stream productivity. In some cases this has lead to increases in the size and abundance of fish populations. Wipfli (1997) showed that young forest canopies (alder-dominated) in the riparian zone of coastal Alaska stream reaches produced more terrestrial invertebrates in the diet of of coho salmon, cutthroat trout and Dolly Varden. In particular, the additional terrestrial invertebrates were important juvenile salmonid prey and may increase their abundance. A similar result was demonstrated by Kawaguchi and Nakano (2001). Forested riparian vegetation provided a greater (x 1.7) input of invertebrates than headwaters bordered by grassland vegetation.

Rowe *et al.* (2002) compared the impact of harvesting pine plantations with and without buffers alongside streams in New Zealand. Two of four common species (*Anguilla dieffenbachia* and *A. australis*) were not affected by either timber harvest treatment. However, the two other species (*Galaxias fasciatus* and *Gobiomorphus hutton*) declined in sites harvested without buffers but increased sites harvested with buffers. They attributed the greater densities to increases in insect abundance from harvested sites with a retention of streamside cover provided by buffers.

In part, responses of fish populations to disturbance are due to complex interactions between disturbance characteristics, life history parameters, and post-disturbance spatial diversity. The impact of timber harvest on fish populations have been demonstrated to run the entire gamut. No predictable generalizations are currently available. Cases for riparian harvesting would have to be evaluated on a case-by-case basis and effects may change through time. Other reports in this project, entitled “Cumulative Effects of Watershed Disturbance on Fish Communities in the Kakwa and Simonette Watersheds.” and “Empirical Relationships Between Watershed Characteristics and Fish Communities in the Notikewin watershed” (Scrimgeour *et al.* 2002 a,b) discuss the impacts of cumulative effects on fish communities in boreal Alberta.

Terrestrial riparian habitats are diverse, dynamic, and complex ecosystems that link the aquatic and upland areas. Much of their internal complexity is due to the ecotonal gradients and disturbances that span the width and longitude of the terrestrial riparian.

2.2 Terrestrial Components

Terrestrial riparian habitats serve as a dynamic link between terrestrial and aquatic ecosystems; they also potentially function as core habitat and dispersal corridors for plants and animals (Ewel 1979; Gregory *et al.* 1991). The diversity of biota in riparian areas reflects a spatially and temporally heterogeneous environment created by a number of different forces including fluvial disturbances (flooding, erosion, sedimentation, geomorphic channel processes), nonfluvial disturbances (fire, insects, wind), variable light environment, variable soils, variable topography, and other upland processes (e.g. Gregory *et al.* 1991; Naiman *et al.* 1993; Sagers and Lyons 1997).

The following sections review the current literature on terrestrial biotic groups including forest structure, vegetation, birds, mammals, and other animals. Examples are used to describe the diversity and use of riparian areas by each group, and to review the use of riparian buffers on the their management. The focus of the literature review was on the Boreal and Rocky Mountain regions, but for some taxa there is a shortage of information. For these taxa, we have included studies from other forested systems within North America and Boreal Europe.

2.2.1 Forest Structure and Vegetation

The riparian forest community is unique from its upland counterpart.

The riparian forest community is generally considered to be distinct in both composition and structure compared to adjacent upland forest communities. In part, this is attributed to higher soil moisture, nutrient availability, and light in riparian areas. Greater plant diversity in near-shore areas compared to interior areas of intact forests was described around mid-order in Vermont (Spackman and Hughes 1995). The abundance and diversity of trees, shrubs, flowering herbs, ferns and fern allies, annuals, perennials, non-native and ruderal plants was higher within 10 m of the high water mark than areas further upland. Data from Korean deciduous - pine forests indicated that measures of species richness, Shannon-Weiner index, and species evenness were greater in riparian forests than in upland forests (Wang *et al*. 2002).

It is often characterized by a greater diversity and higher shrub biomass than upland areas.

In addition to differences in diversity of species, riparian and upland forests also differ in terms of structure. For example, Kinley and Newhouse (1997) examined vegetation structure in 14, 37, and 70 m wide river buffers in hybrid white x Engelman spruce - lodgepole pine forests of southeastern British Columbia. They found that riparian forests had a taller shrub layer and greater canopy cover but fewer live trees compared to upland forests. However, no differences were found with respect to deadwood resources or cover of low shrubs. Similarly, Holt *et al.* (1995) demonstrated that shrubs were the dominant biomass in lakeshore systems in Nova Scotia, particularly along high shoreline positions. They, like other researchers, demonstrated that the highest diversity of shrub species occurred along shorelines (Keddy 1989; Wilson and Keddy 1986; Wisheu and Keddy 1989). The increased dominance of shrubs in riparian areas has led to suggestions that they competitively inhibit herbs, grasses, and the long-term succession of the community to conifers. Keddy (1989) demonstrated that lakeshore shrubs were competitive dominants, displacing small herbs. The dominance of shrubs and lack of tree recruitment has been documented in harvested and unharvested riparian areas in Oregon's Coast Range (Hibbs and Giordano 1996) and Wood Buffalo National Park, Alberta (Timoney and Peterson 1996). Failure of trees to regenerate under these circumstances has been attributed to competition from shrubs (Beach and Halpern 2001) and inadequate seed and substrate availability (Hibbs and Giordano 1996; Timoney and Peterson 1996).

Both fluvial and nonfluvial disturbances play a major role in influencing the diversity of riparian communities. Fluvial disturbances include flooding, erosion, and sedimentation; nonfluvial disturbances include fire, wind, insects and pathogens, and anthropogenic disturbances associated with forest harvest. The presence of water, topographical features, and riparian vegetation may influence upland disturbance patterns. Flooding is solely in the domain of riparian areas and is generally considered the dominant disturbance in Boreal riparian ecosystems (Timoney and Robinson 1996). The temporary inundation by water creates a significantly more diverse plant community than would otherwise occur (e.g. Hupp 1982).

Increased water stage including that rising from beaver activity, wildfire and harvest play significant roles in establishing plant communities in riparian areas.

Flooding, deposition of sediments, and cutting of stream channels can produce non-vegetated areas along the streambank and river channel; these areas are subsequently available for primary succession. In comparison to upland habitats where primary succession is generally limited to areas of glacial retreat and volcanic flows, primary succession in riparian areas occurs more frequently. Marks (1983) argued that floodplains are thought to be one of the original habitats for weedy species evolution, largely due to the high disturbance rates and relatively constant production of habitats available for primary succession.

Annual or short-cycle flooding (<10 years) is responsible for the compositional and age gradient of nearshore terrestrial communities. In forested ecosystems, the general pattern from the shoreline is emergent vegetation, followed by grass/forb, then shrub, and finally tree communities. The width of this gradient is highly variable with strata either missing or combined. Farjon and Bogaers (1985) recognized four distinct floodplain seres along the Porcupine River in Alaska's boreal forests. Two of the seres were climaxed by white spruce, while others which experienced frequent floods were dominated by willows and willow/calamagrostis/horsetails. They concluded that flooding frequency was the primary variable that determined the composition of the sere. In a thorough examination of plant-environment interactions in the Ozark National Scenic Riverways, Missouri, Lyon and Sagers (1997) found a high degree of substrate heterogeneity across the riparian landscape. They concluded that most important variable associated with species composition in this are was the elevational gradient from streamside to upland. Low elevation, flood-prone zones had greater rates of species replacement than higher elevation zones. At higher elevations, plant communities exhibited a gradual change, although the strength of discontinuities in the communities increased when communities neared the boundary of trees.

Another source of disturbance to riparian areas in the Boreal forest is beaver activity. Beavers affect riparian vegetation by building impoundments and through flooding adjacent forests, regulating and limiting downstream flow, and direct utilization of riparian vegetation as food sources and building materials. In Boreal forests of northern Ontario, Barnes and Mallik (2001) found that beaver selected species of trees and shrubs based on different purposes. For example, alders and conifers were used in dam construction while trembling aspen was the primary food source with paper birch and willows as secondary food sources. Beavers significantly altered forest structure and plant communities in an area of at least 20 m around impoundments. After 12 years there was modest recovery of trembling aspen, however shrubs and conifers had not recovered. Beavers may have long-term affects on both terrestrial and riparian vegetation, however, their temporal and spatial impacts as agents of disturbance have not been well characterized along whole river networks.

Nonfluvial disturbance regimes from adjacent uplands also influence the composition, structure, and diversity of riparian vegetation. Disturbances such as wildfire and harvest occur in riparian areas. However, they may initiate different successional trajectories of plant communities. In the Peace River lowlands of Wood Buffalo National Park, the type of disturbance: flooding, fire, or logging produced divergent forest types with little evidence of gradual convergence towards a common forest type (Timoney *et al.* 1997). Timoney *et al.* (1997) postulated that pioneering willow communities on alluvial landforms would eventually succeed to white spruce dominated, mixedwood (white spruce-balsam poplar), or balsam poplar dominated forest. These community types would be relatively stable until disturbance by either wildfire or logging. Wildfire-origin stands would eventually progress to white spruce, mixedwood, or balsam poplar forest types; whereas harvest-origin (natural regeneration and scarification) stands, would succeed to balsam poplar or Alaska birch/aspen/balsam poplar forest types.

Riparian areas serve as dispersal corridors for plants.

In addition to disturbance, other factors such as the migration capacity of plants along riparian corridors may explain high species diversity (Naiman *et al.* 1993). Hanson *et al.* (1990) modeled the outcome of seed dispersal within fragmented and connected riparian corridors. For the fragmented landscape, the SEEDFLO model projected a lower overall diversity of plants but a higher abundance of species with greater dispersal probabilities, e.g. bird-dispersed species. However, within connected riparian areas seeds may be transported by water or by birds and mammals that use riparian corridors for travel. Johansson *et al.* (1996) examined the use of rivers and streams as dispersal corridors for plants with floating seeds. They found a correlation between the floating capacity of seeds and the extent of dispersal along riparian corridors. From these results, they argued that dispersal by water played a significant role in structuring riparian flora.

The effects of a riparian edge can penetrate from a few to tens of metres into the adjacent forest. The distance depends on the forest types, plant species, and forest structure measured as an indicator of riparian extent.

A number of studies have attempted to determine extent of the riparian plant community into the upland. This is then used as a basis for determining the width of riparian buffers. Harper and MacDonald (2001) demonstrated that a distinct lakeshore forest edge community extends for about 40 m around boreal lakes in central Alberta. The lakeshore forest community was characterized by greater structural diversity, larger amounts of coarse woody material, and a greater density of regenerating trees than interior forest. A study of mid-order streams in Vermont (Spackman and Hughes 1995) suggested that a zone between the stream edge and a few meters above the high water mark would be adequate to conserve annual, biennial and ruderal plant species. These studies, along with others, recognize that the boundary between riparian and upland plant communities varies with the strata, taxa, or species measured. However, the transition from riparian to upland is not abrupt but occurs along a gradient or continuum (Sagers and Lyon 1997) and by solely using plant communities to delineate riparian boundaries we may fail to recognize the wide array of ecological processes and communities associated in this transition (Gregory *et al.* 1991).

Although riparian buffer strips are intended to represent a strip of intact forest, when used as a management tool in harvest areas they can potentially undergo a number of changes. These include: blowdown of trees and snags due to increased wind stress, increase growth and abundance of understory forbs and shrubs, release of suppressed trees, and greater input of coarse woody debris. For example, Johnson and Brown (1990) compared the forest composition of buffers strips (~80 m wide) left after timber harvest to the composition of undisturbed lakeshore forests in Maine. They reported that shrub densities were greater in the buffer strips but tree and snag densities were greater in undisturbed lakeshore forests. A comparison of different widths and disturbance levels of riparian buffers in mature balsam fir forests in Quebec, indicated that narrower buffers (20 m intact and 20 m thinned) exhibited a greater density of conifer and deciduous shrubs than wider buffers (>40 m) or uncut controls (Darveau *et al.* 1998). In an effort to determine the relative difference between edge types, Whitaker and Montevecchi (1997) measured coarse vegetation attributes from riparian edge, nonriparian edge (clearcut or access road), and interior habitat in balsam fir forests in Newfoundland. They found the basal area of standing deadwood to be greater in interior forest compared to riparian or nonriparian edges, whereas shrub densities were greater in riparian and nonriparian edges compared to interior forest areas. Otherwise there was very little difference in structure between the edge types.

As indicated by the literature, edge effects need to be considered when developing buffer widths for riparian areas. Riparian buffers have two edges; one abuts the water while the other abuts the harvest area. The edge-effects on the buffer created by these interfaces are asymmetrical (Harper 1999). Variables which exhibited the greatest edge asymmetry included suppressed tree density, aspen sapling density, and total herb cover. Harper recommended that lakeshore buffers in aspen-dominated mixedwoods in central Alberta would have to be at least 200 m wide in order to maintain non-edge influenced corridor of interior forest.

Of special interest to timber managers is the ability of riparian buffers to withstand windthrow. Applied to Alberta, there are two important points. Firstly, windthrow is a prevalent feature for much of the residual material (Hanus and Crites 1999). The loss of riparian to windthrow is not an overall loss of habitat. As previously discussed, post-wildfire riparian systems are inundated with windthrow of fire-killed trees shortly after disturbance. It is likely that even the loss of a 60 buffer is well within the range variation tolerated by riparian biota. Despite some being inevitable, managers should attempt to retain as much vertical structure as possible into the next rotation. Fortunately, there is an extensive literature on preventing windthrow in residual stands (see Lee *et al.* 2002b). As an example, an exhaustive set of wind tunnel and field measurements were performed in the north central Quebec (Ruel *et al.* 1998; 2001). The authors examined buffer widths varying from 20 m thinned to 60 m. They found that over these widths, windthrow was not related to width rather wind direction and presence of topographic features were the most important variables. For the first five years after cutting, the presence and velocity of perpendicular winds were a major agent of windthrow. After this time, the impact of these winds was not correlated to subsequent windthrow. Buffers exposed to unobstructed upland areas such as those along wide valley floors were more prone to windthow.

2.2.2 Birds

Unique assemblages of birds have been associated with riparian areas.

For some bird species the riparian community represents a specialized habitat. Often characterized by a unique species assemblage. A number of features may explain the diversity of birds in riparian habitats. These include: diversity of vegetation structure, greater productivity of riparian areas, biological interactions facilitated by the ecotonal gradient, horizontal diversity of water, and resource inputs from the aquatic production (reviewed by Murakami 2001). LaRue *et al.* (1995) compared upland and riparian areas in balsam fir communities in Quebec and found that bird abundance, richness, and diversity were significantly higher in riparian areas. They attributed the increases to the greater structural complexity along a horizontal gradient created by the juxtaposition of the aquatic ecosystem, narrow grass wetland, shrub zone, and treed community. Similarly, Whitaker and Montevecchi (1997) surveyed bird assemblages from riparian edge, nonriparian edge (clearcut or access road), and interior habitat in balsam fir forests in Newfoundland and found distinct species assemblages associated with each of these habitat types. They reported that total abundance and richness were not different between riparian strips and interior forest controls but were significantly greater in nonriparian edges compared to riparian edges. In northern New England, forest songbird communities differed in riparian buffers left after harvest and uncut reference areas, and between main stems and tributaries (Meiklejohn and Hughes 1999). Common species were detected >20 times more along main stem streams than headwater streams. Riparian buffers had greater numbers of edge species and less common species while reference areas had more interior species and common species such as bay-breasted, blackburnian, black-throated, and Cape May warblers. Also, due to their narrower width, tributary buffers had more edge species.

The unique horizontal structure in riparian areas is a key feature for bird assemblages.

Structural and compositional features associated with riparian zones are recognized as key habitat components for a number of species. For example, aspen was considered the prime cavity tree for wood ducks in north-central Minnesota, with most of the cavity nest trees located within 100 m of the water (Gilmer *et al.* 1978). In a review of osprey populations in forested areas of North America, Ewins (1997) reported that most populations breed close to water. Furthermore, Becker and Hull (1987) reported that all breeding male merlin home ranges in their southeastern Montana study area contained significant portions of riparian habitats. The use of riparian habitats by merlins was about 3 times greater than the amount expected through random encounter. For some species, riparian areas are primary feeding sites. Whitaker *et al.* (2000) recorded relatively high numbers of insectivorous birds in riparian areas presumably resulting from higher densities of insects along lakeshores in boreal Newfoundland. Murakami and Nakano (2001) observed differences in the foraging of insectivorous birds on aquatic and terrestrial prey species. Foraging guilds of leaf-gleaners, air foragers, and leaf and branch gleaners were enhanced by the input from aquatic insects. The authors argued the increased productivity reinforced the presence of foraging specialists in riparian areas. A comparison of buffer strips (~80 m) left after timber harvest and undisturbed lakeshore areas in Maine indicated that density and species richness were lower in the buffer strip (Johnson and Brown 1990). Undisturbed lakeshore areas supported more cavity nesters, foliage nesters, ground foragers, and bark gleaners than buffer strips. However, buffer strips supported more leaf gleaners and aerial feeders.

Riparian areas provide a combination of core habitat and/or corridors for dispersal.

Based largely on arguments of maintaining pre-harvest assemblages, the literature contains a number of recommendations on buffer width. Whitaker and Montevecchi (1997) argued that creation of buffer strips would not increase overall biodiversity as represented by species richness but should be considered as a method of protecting a distinct riparian assemblage of birds. Spackman and Hughes (1995) argued that buffer widths of 150 and 175 m would be required to maintain 90% and 95% of preharvest species along mid-order streams in Vermont. Kinley and Newhouse (1997) compared bird communities in 14, 37, and 70 m river buffers in hybrid white x Engelmann spruce - lodgepole pine forests in southeastern British Columbia. Their results indicated that mean bird richness was similar between upland and riparian sites but that diversity, equitability, total densities, and abundance of most riparian species increased with riparian buffer width. They argued that the current buffer width regulation of 20 to 50 m in British Columbia would lead to reductions in bird density and changes in community structure. Lance and Phinney (2001) argued that riparian buffers made significant contributions to the forest songbird communities within partial retention cuts (15 to 20% retention) in sub-boreal conifer forests within central British Columbia. When individual species were considered, Lambert and Hannon (2000) suggested that buffer widths >100 m were required for the successful breeding of ovenbirds along lakes within the boreal mixedwood forests of Alberta. They also found no breeding birds within 20 m wide buffer strips. Pearson and Manuwal (2001) examined bird communities in managed Douglas-fir forests along 2nd and 3rd order streams in western Washington state. The authors compared 1) unharvested controls, 2) narrow buffers (~14 m each side), and 3) wide buffers (~31 m each side). Resident species, species associated with shrubs in forested habitats, and species associated with conifer trees declined while open shrub species increased in riparian buffers. Narrower buffers were characterized by a high species turnover indicating that they were unable to retain the pre-harvest community. In contrast, wide buffers had a very low species turnover suggesting their support of the pre-harvest community. In considering the life histories of all interior species, the authors recommended buffers of >45 m for 2nd and 3rd order streams.

Aside from the structure of the buffer itself, buffers may influence suitability of a landscape to support birds after harvest. Machtans *et al.* (1996) demonstrated that birds traveled through riparian buffers more than across boreal clearcuts. Buffers served as corridors particularly for juveniles, whereas adults were likely to be residents within wide (~100 m) buffers. They also suggested that the use of riparian buffers could offset some of the impacts of fragmentation caused by harvest. Similarly, Schmiegelow *et al.* (1997) found that forest songbirds used riparian corridors in boreal landscapes fragmented by timber harvest. In a study along lakes in the boreal mixedwood forest of Alberta, Lambert and Hannon (2000) noted that even individual trees within harvested areas could be used as "stepping stones" to cross open, cutover areas. Though the contribution of buffers to landscape structure has been demonstrated to be important to birds, specific recommendations are still to be formulated.

After harvesting upland areas, bird assemblages in riparian buffers often exhibit "crowding" of species and mixing of guilds such as edge species, forest generalists, forest interior species, and riparian species.

Although creation of riparian buffer strips provides benefits for some bird species, there are some potentially less desirable impacts associated with buffer strips. These include edge effects, crowding, and increased predation. Due to the juxtaposition of habitat types (water-forested-cutover), assemblages of birds may be a "forced" mix of species each preferring different habitat elements. Meiklejohn and Hughes (1999) found edge species more common in buffers and forest interior species more common in unharvested reference sites. Whitaker and Montevecchi (1999) found that buffer strips contained a higher total avian abundance than forested streamside control. This was attributed to the presence of edge and clearcut tolerant species. Narrow buffers were able to maintain many riparian and woodland species, however, interior forest species required wider buffers and some were not present in even the widest buffers (~60 m). Based on their data, they suggested buffers widths greater than >20 m for the retention of forest generalists.

Measurement of buffer strip effectiveness for bird communities is often influenced by "crowding" of individuals into buffer strips following harvesting, however, this effect may be temporary. Darveau *et al.* (1995) noted an increased bird density in streamside buffers the year after cutting of balsam fir forests of Quebec. These differences declined in following years. Density increases were greatest and subsequent declines were faster in narrower (20 m) buffers. However, thinning (33% tree removal) of 20 m strips did not appear to have as great an impact on bird species as reductions in overall buffer width. Results further indicated that buffer widths of 60 m could support forest dwelling species, whereas buffer widths of 20 m were more useful to ubiquitous species.

Increased predation of nests in riparian buffers after harvest of upland areas has been reported in some studies.

Increased predation is commonly found along the cutblock edge created by the implementation of riparian buffer strips. Monitoring artificial nests in sub-boreal Acadian forests in Maine, Van der Haegen and DeGraaf (1996) found that predation around harvested stands was greater in narrower buffers (20 to 40 m) compared to wider (40 to 60 m) buffers, although predation rates in buffers of either width were greater than unharvested riparian areas. Using remotely triggered cameras, they were able to attribute the increased predation in buffers to greater incursions by other vertebrates, primarily blue jays and red squirrels. They suggested that a buffer ≥150 m wide would remove the edge effect created by increased predation. In a comparison of different buffer types in balsam fir forests in Quebec, predation of artificial nests was highest in buffers along main roads, intermediate in lakeside buffers and buffers adjacent to forest roads, and lowest in the riparian buffers along rivers (Boulet and Darveau 2000). Predation within main road buffers was likely greater due to the presence of crows searching for vehicle-killed carrion.

2.2.3 Mammals

At regional scales, some researchers and managers expect the distribution of mammals may be tied to the distribution of water. However, there is relatively little empirical study to support this assertion.

At large regional scales, the distribution of large mammals may be tied to the distribution of water. Approximately, 70% of vertebrate species in the Pacific Northwest of the United States used riparian habitat in a significant way during their lifetime (Raedeke 1989). Quantitative data at these large scales is lacking for other ecosystems but dependency is thought to occur based on known natural history and empirical studies at smaller spatial scales. Strong ties are likely to occur for animals dependent on the water-land interface. As an example, beavers have a direct impact on the succession of the vegetation communities along the boreal waterbodies. Barnes and Mallik (2001) found that the majority of beaver herbivory occurred within 20 m of the water edge. Trembling aspen was the preferred food tree, however, long periods without rejuvenation of aspen stocks by fire meant a slow replacement with the less preferred conifers. Even after cessation of herbivory for twelve years, aspen did not recover and continued to be replaced by conifers. They suggested that this change in available food trees leads to the dispersal of the local beaver population and their lower densities. Harvesting of riparian areas would in all likelihood rejuvenate aspen stocks but it is unclear whether it would lead to other deleterious impacts on beaver populations through loss of cover in the upland areas.

For other mammals, the connections to a treed riparian habitat are looser. For moose in the boreal forests along the Sustina River, Alaska, natural early seral stages of felt leaf willow and old seral stages of balsam poplar or paper birch/white spruce riparian areas were identified as important wintering sites (Collins and Helm 1997). Radio-collared grizzly bears in the Mission Mountains of Montana were more likely to utilize riparian areas in spring and fall (Servheen 1983). These results were similar to those found for grizzly bears in northwestern Alberta (G. Stenhouse pers. comm). In west-central Idaho, black bears used aspen and riparian areas as bedding areas in summer and fall, although throughout the remainder of the year, they preferred other forested areas for bedding (Unsworth *et al.* 1989). Unsworth *et al.* (1989) recommended that forested buffers along streams, roads, and dense stands on north-facing slopes be retained for bear cover and bedding. However, Clevenger *et al.* (2002) noted that habitat models developed for black bears from expert opinion tended to overestimate the importance of riparian areas relative to empirical or literature-based models. This trend reinforces the widely held view on the importance of riparian areas to wildlife amongst professionals but that the literature may not support the strength of this view.

Riparian buffers left after harvest probably do not provide sufficient areas for core habitat for many mammal species but may be important for cover and feeding.

For most large mammals, buffers left after harvest are not wide enough to provide core reproductive habitat, however, they may provide sufficient cover to feed and travel. Van der Haegen and DeGraaf (1996) found that black bears used riparian buffers as travel corridors through harvested areas in Maine. Brusnyk and Gilbert (1983) found that moose densities were greater in riparian buffer strips left after harvesting than in blocks that did not retain buffers. Unlogged buffers zones were recommended for the management of white-tailed deer in lodgepole pine forests in Montana (Leach and Edge 1994). The width of corridors required for travel by most mammals may be relatively narrow. Spackman and Hughes (1995) found that the movement of white-tailed deer, coyotes, raccoons, foxes, snowshoe hares, and voles occurred primarily within 10 m of the high water mark for mid-order streams in Vermont. Forsey and Baggs (2001) found that track counts were greater in interior, uncut forests whereas track counts were greater in riparian strips (20 m) after cutting for three of the five focal species. These included Newfoundland marten, snowshoe hare, and red squirrel. These authors concluded that buffer strips left after harvest were of value to these species. No differences were found for short-tailed weasel and red fox. Darveau *et al.* (2001) studied small mammals in balsam fir forests along streams in Quebec and found no difference in the abundance of the two most common small mammal species among buffers of varying width (20, 40 and 60 m). They also reported that meadow vole, which was absent prior to harvest, invaded clearcuts and was a limiting factor to the occurrence of red back vole and deer mouse in buffer strips. They suggested that 20 m buffer strips may work as refuges for small mammals, however, wider strips would provide a more natural habitat for edge-avoiding species.

Impacts of riparian buffers left after harvest appear to have little benefit for small- and medium-sized mammals.

In contrast to these results, a number of studies find little bias in the distribution of mammals in buffers left after harvest. In comparing red squirrel, northern flying squirrel, and eastern chipmunk population parameters within upland and riparian strips, and forested blocks, Cote and Ferron (2001) found no differences among treatments and controls. The only difference was that red squirrel middens were more abundant in controls than either upland or riparian strips. The authors concluded that all species were able to tolerate the presence of timber harvest and riparian strips were no different than upland strips. De Groot (2002) found similar results for small mammals within mixedwood boreal forests in north-central Alberta. Abundances of red-backed voles, deer mice, and meadow voles estimated through trapping, did not differ in riparian forest strips (20 to 200 m) and controls adjacent to small lakes for four years after their creation. He also found no impact on density or demographics of these species with distance from lakeside in intact forest suggesting that there was no increase in habitat quality in riparian areas. Furthermore, winter tracking data for larger mammals from intact riparian areas and after the creation of 20 to 200 m wide buffers indicated little difference in movement patterns. In balsam fir forests of Quebec, Darveau *et al.* (1998) reported that snowshoe hares make only minimal use of riparian buffer strips regardless of buffer width (buffer widths of 20, 40 and 60 m were tested).

2.2.4 Other Terrestrial Animals

Relatively little data exists on the impacts of riparian buffers on other terrestrial taxa particularly for Alberta's forested ecoregions.

Though the data for vegetation, birds, and mammals dominates the research literature, the use of riparian areas and the impact of buffers has been assessed on other taxonomic groups. Whitaker *et al.* (2000) captured 125 to 200% more insects along buffer strips than in uncut shorelines; mostly Dipteran and Hymenopteran adults. They argued that buffers acted as windbreaks and that trapped insects were blown off the water or from the adjacent clearcuts. Timber harvest operations in eucalyptus forests in Tasmania caused a decrease in the abundance of adult stoneflies and leptophlebiid mayflies (Davies and Nelson 1994). The declines were primarily dependent upon the width of the buffer but were also affected by slope, soil erodability, and time since logging (one to five years). Impacts of logging on mayflies were not significant when buffers were at least 30 m wide.

Amphibians have life history strategies directly associated with the land-water interface. There are ten amphibians in Alberta with five[3] of them residing in forested habitats and potentially affected by present and future harvest operations. Long-toed salamanders and western toad are listed as sensitive, i.e. naturally rare and patchy in distribution, while the Canadian toad is listed as maybe at risk (Alberta Sustainable Resource Development 2002). The Boreal Chorus frog and Wood frog are listed as secure. The impact of riparian logging has only been evaluated for long-toed salamander. The current distribution of long-toed salamanders in Alberta is restricted to Cordilleran and Boreal ecoregions (Graham and Powell 1999), and their habitat suitability index is based on distance to water and amount of terrestrial cover (shrubs, herbs, horsetails, grass, moss, downed woody debris) (Graham *et al.* 1999). Logging would not directly impact this species through removal of riparian trees. Impacts, if present, would have to come through loss of understory layer or downed woody debris.

2.3 Discussion

2.3.1 Aquatic Components

Forested riparian buffers provide primarily for water temperature regulation in streams and provision of organic debris. These are important features of streams and small waterbodies.

The current research suggests that forested riparian buffers function primarily to regulate stream temperatures and light levels, and provide inputs of organic debris into streams. In some cases, they are also valuable in controlling sedimentation and providing streambank stability. The widths required to provide these functions are relatively narrow (< 35 m). A number of factors such as water yields, peak flows, and nutrient transport are driven largely by the amount and type of disturbance in the entire catchment. Management of total harvest within a catchment would seem to be more appropriate for regulating these factors. In general, lakes are less affected than streams. The larger volume and greater depth were at better buffering of disturbance effects.

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[3] Long-toed salamanders (Sensitive), western toad (Sensitive), Canadian toad (Maybe at risk), boreal chorus frog (Secure), wood frog (Secure).

A key question in determining any riparian management is the allowable amount of variation in conditions caused by either natural or anthropogenic disturbances. Furthermore, there needs to be a greater understanding of the factors that make systems resilient to and allow recovery from disturbances.

At the heart of managing aquatic ecosystems lies a question about permissible amounts of disturbance and variation. Yount and Niemi (1990) reviewed the recovery of lotic communities from disturbance and noted that most systems recovered in a relatively short period of time from most disturbances. Three factors that facilitated recover were commonly cited in the research literature: 1) life history patterns adapted to the recolonization and repopulation of affected areas, 2) the availability and accessibility to unaffected upstream, downstream, and internal refugia for recolonization, and 3) systems with high flushing rates tended to recover faster because of the removal of affected water and sediments. Longer recovery periods were noted in systems where permanent changes had been made to the physical conditions within or surrounding the stream. Also, recovery was not possible without cessation of the disturbance. Based on these results, it is clear that the amount of permissible natural or man-made disturbance must be assessed on a cumulative basis. Additional disturbances must be evaluated relative to the baseline natural and man-made disturbance already present on the system. Furthermore, chronic and periodic disturbances must be integrated to determine the long-term amplitude of disturbance and recovery cycles.

Data suggests that aquatic systems are capable of recovering from a wide range of periodic disturbances.

The underlying management paradigm for research has focused on the protection of riparian areas and aquatic systems from dynamic change either through natural or anthropogenic disturbances. An alternative management paradigm partially based on the ranges set by natural disturbance and recovery cycles would propose a greater amount of variation at the point of disturbance, however, it would also have to allow for sufficient recovery time to avoid long-term chronic changes to the ecosystem. This latter point is critical. Under this paradigm, research would focus on; 1) the effect of having short-term changes to biotic communities 2) the return time to reference conditions, and 3) the impact of chronic activities throughout the watershed. Wildfires maybe a template for estimating the range of variation permitted under timber harvest in riparian areas. Rieman and Clayton (1997) agree that wildfires and harvest can precipitate some similar changes in ecosystems including regeneration of stands around aquatic systems; however, they warn that there are significant differences, e.g. inputs of large woody debris. Restoration or maintenance of ecological structure, composition, and processes through manipulations such as streamside harvest and prescribed burning are largely experimental and potentially risky. The use of wildfire as a template for estimating and implementing the range of natural disturbance-succession paradigm will be explored in more detail in the Section 5.0.

2.3.2 Terrestrial Components

Management objectives in riparian buffers can vary according to their degree of utilization by terrestrial organisms. As the degree of use increases, the area of the riparian buffer needs to be increased.

The arguments invoked for maintenance of riparian buffers for terrestrial species can be summarized by four objectives:

- preventing habitat avoidance,
- maintaining special use habitat,
- providing temporary refugia habitat,
- and core habitat.

It is important to have clear management objectives within riparian habitat. These will differ between species. For example, objectives such as providing cover for movement and core habitat are likely to result in different riparian buffer designs.

These objectives can be ordered relative to the magnitude of their conservation goals. The minimal goal for riparian management is to prevent avoidance by terrestrial organisms. These areas may not be preferred but should at least be neutral to the presence and movement of plants and animals. A step up is the use of riparian areas as special use habitats. This type of use can vary from movement and dispersal, to feeding or breeding areas. This differs from refugia or core habitats in that special use implies that riparian areas only fulfills a portion of the life history requirements. Refugia refers to the use of riparian areas as a temporary habitat that facilitates the survival of a species until post-disturbance recovery provides more favourable habitat. Refugia are used for temporal dispersal. Core habitats represent areas capable of supporting long-term sustaining populations of species. Like refugia, core habitats are sources for dispersal into the surrounding post-disturbance habitats. These categories represent points along a continuum of dependence upon riparian areas. For different organisms, the properties of riparian areas that support these species maybe quite different. Managers must clearly spell out the objectives for species within riparian areas prior to development and application of different management actions.

It is important to distinguish between species who require trees close to water from species which are found close to water but are unaffected by harvesting of trees near water.

In developing objectives for riparian management in forestry, it is critical to distinguish species whose life history is significantly impacted by the loss of trees around the water's edge. That is, species that require a combination of treed and riparian habitats are more likely to be affected. Many studies on the use of undisturbed or harvested riparian areas by animals have not explicitly looked at the dependency of the species on specific features within riparian areas. Most of the empirical data have been measures of presence/absence or relative abundance. At best most researchers argue that this data only provides information on habitat avoidance. Few studies have explicitly looked at reproduction or survival. Results from territorial behaviour and nest predation studies of birds represent some of the few empirical datasets that can be interpreted as defining core or refugia habitat (e.g. Lambert and Hannon 2000).

One commonly argued feature of riparian areas is their use as a corridor for travel. The empirical data for the use of riparian corridors are not strong. There is a lack of data in Alberta's forested ecoregions to evaluate the use of treed buffers left after harvest as dispersal corridors.

One proposed function of riparian areas is to facilitate the movement or dispersal of organisms throughout a broader landscape. Streams and lakes form a nearly continuous network of corridors throughout the landscape, and are one of the few naturally linear features on the landscape (Simberloff *et al.* 1992). Disturbances that remove streamside vegetation are postulated to interrupt these corridors. Within harvested landscapes, the effect of corridor discontinuity may be compounded by fragmentation of adjacent forest cover. In part, the rationale for corridors comes from analysis of small isolated populations located in a large landscape. Small populations are more prone to extirpation (reviewed in Simberloff 1988; Caughley 1994) and the effects of corridors on these populations at a landscape scale are unclear.

Data from agricultural settings is more compelling than forested settings (e.g. Wegner and Merriam 1975; Henderson *et al*. 1985; Bennett *et al*. 1994). There is little quantitative evidence on the use of riparian corridors for the movement of mammals in forested systems (Simberloff *et al.* 1992; but see Beier and Noss 1998, Lomolino and Perault 2001). For forest- or shrub-dwelling animals in an agricultural area, the intervening matrix is relatively inhospitable and forms a relatively permanent loss of habitat. Hence, wooded corridors are very important for forest-dwelling biota in these landscapes. In contrast, timber harvest represents a conversion to a forb-grass community for the first years after harvest but this regenerates to a shrub/tree community shortly thereafter. Under timber harvest, the landbase tends to be a shifting patchwork of relatively young treed areas with all stands placed on a similar successional trajectory whereas agriculture tends to be a fixed patchwork of highly contrasting vegetation types. More work needs to be done to evaluate the role riparian corridors in a forested and timber harvest context.

Lastly, the spatial scale of corridors maybe important to their use in management. A number of advocates for corridors (Noss 1983; Noss and Harris 1986) focus on extensive corridors covering relatively large distances with re-colonization of species in parts of its former home range as a goal. An example of a large connectivity issue is the Yukon to Yellowstone Initiative that hopes to connect the Rocky Mountains in a north-south corridor (Yukon to Yellowstone Conservation Initiative 2002). At large scales, decreased local extirpation, reduction of demographic stochasticity, and prevention of inbreeding (reviewed in Simberloff *et al.* 1992) may be important. However, at smaller spatial and temporal scales, as those involved in timber harvest, the degree of local isolation may simply be not great enough to invoke these costs (reviewed by Bunnell *et al.* 1998). There exists no direct empirical data to suggest or refute costs associated with the temporary loss and revegetation of riparian corridors at smaller spatial scales.

In general, the ability to use existing research to directly guide the management of aquatic and terrestrial ecosystems in Alberta is hampered by the lack of studies in western Boreal and northern Rocky Mountain forests. Research from other regions can serve as a guide to identify potentially important factors and perhaps suggest the direction of outcomes. However, current database in Alberta cannot give reliable predictions on the outcomes of different management actions. There is a need for both baseline research and assessment of responses of aquatic and terrestrial ecosystems to specific management actions in Alberta.

2.4 Summary of Key Findings

- Definitions for riparian areas depend on their intended use and mandate of the agency or institution proposing the definition. Most current definitions include: water and a geophysical feature that holds or transports water for at least part of the year, emphasis on the aquatic-terrestrial interface, and variable widths and boundaries sometimes defined on a probabilistic basis.
- Water yields, peak flows, and nutrient yields increase immediately after disturbance (harvesting and wildfire) and decline thereafter. Buffers are relatively ineffective at mitigating increases in these factors. These are catchment-level effects and are a function of the overall proportion of disturbance in the catchment.

- Buffers are effective for the reduction of sediment flows into waterbodies from overland flows created by disturbance of the forest floor. They are less effective for sediment transport through channelized flow. The need for buffers for sediment control greatly declines when harvesting can be accomplished with a minimal soil disturbance. Skid trails, landing areas, and roads are the dominant sources for sediment from harvesting.
- Canopy cover from treed buffers provides significant modulation of stream temperatures. The importance of this on stream biota in Boreal and Rocky Mountain streams in Alberta remains unexplored.
- Treed riparian buffers provide a significant amount of allochthonous organic debris into streams. The loss of buffers will create a loss of habitat and food source in stream systems. The loss of treed buffers around medium to large lakes is less likely to impact organic inputs compared to smaller lakes.
- The response of aquatic invertebrates and fish communities to riparian disturbance has been mixed. Some studies indicated increases while others have indicated declines. In part, responses were dependent on the factors limiting growth and reproduction of organisms. If temperature was limiting, increases were often noted. However, if cover or water column heterogeneity was limiting then declines were recorded. Fish communities in Boreal and Rocky Mountain ecosystems in Alberta lack sufficient short- and long-term data to evaluate the impacts of changes to buffer guidelines on invertebrates and fish communities.
- Riparian areas have a relatively high diversity of non-vascular and vascular plants because of the relatively close spatial proximity of varying abiotic conditions and the complex interaction of disturbance and succession. Significant changes in the succession of riparian areas can occur after harvesting. The impact of tree removal through harvesting and disturbance is unclear for plant communities in and around the Boreal and Rocky Mountain waterbodies.
- Bird communities generally respond to creation of riparian buffers by an increase in open habitat and edge species. Preservation of interior species depends upon the width of the buffer. Some studies have noted a "packing" of individuals in buffers immediately after harvest, however, this condition declines in subsequent years. Little research has been done on bird species, whose life histories are dependent on the presence of trees close to water, e.g. boreal ducks. These species are likely to be impacted by removal of riparian trees.
- Evidence suggests that the distribution of mammals is positively affected by the presence of water. However, it is unclear whether disturbance of the treed riparian zones alters this distribution. For some species such as moose, there is a preference for treed riparian while for many small- and medium-sized the treed riparian was not important.
- Little data is available on the impact of riparian harvesting on terrestrial invertebrates, amphibians, and reptiles in the Boreal and Rocky Mountain ecoregions. In other forest types, riparian harvesting has had major impacts on amphibians.

- Table 5 summarizes the probable impacts of treed riparian buffers on riparian functions, structure, and biota in Alberta's Boreal and Rocky Mountain ecoregions. The table restates some of this section's findings in the context of Alberta's current guidelines.

Table 5. Summary of the potential efficacy of buffers in mitigating the impact of timber harvest on riparian function and biota for Alberta's current waterbody classifications types.

Function/Biota	**Ephemeral**	**Intermittent**	**Small Permanent Stream**	**Large Permanent Stream**	**Small Lakes >4 ha with fish**	**Large Lakes >16 ha**	**Water Source**
Water Yield/Peak Flow	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.	Not effective, catchment size affects yield.
Erosion and Sedimentation	Not effective provided slopes are not excessive, soils are not highly erodable, and minimal surface disturbance.	Not effective provided slopes are not excessive, soils are not highly erodable, and minimal surface disturbance.	Minor benefits provided slopes are not excessive, soils are not highly erodable, minimal surface disturbance, and increased peak yields do not create sheet or bank erosion.	Minor benefits provided slopes are not excessive, soils are not highly erodable, minimal surface disturbance, and increased peak yields do not create sheet or bank erosion.	Minor benefits provided slopes are not excessive, soils are not highly erodable, minimal surface disturbance, and increased peak yields do not create sheet or bank erosion.	Minor benefits provided slopes are not excessive, soils are not highly erodable, minimal surface disturbance, and increased peak yields do not create sheet or bank erosion.	Minor benefits provided slopes are not excessive, soils are not highly erodable, minimal surface disturbance, and increased peak yields do not create sheet or bank erosion.
Nutrient Transport	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.	Effective only for sediment bound nutrients. See above.
Temperature Control	Not effective.	Efficacy depends on cumulative increases to downstream reaches during flow periods.	Effective.	Less effective, due to larger surface area to buffer edge ratio.	Less effective, due to larger surface area to buffer edge ratio.	Less effective, due to larger surface area to buffer edge ratio.	Efficacy depends on surface area of water source.
Organic Matter Input	Effective.	Effective.	Effective.	Effective.	Some empirical evidence of importance.	Not effective.	Not effective.
Aquatic Invertebrates	Efficacy depends whether organic inputs or temperature are limiting.	Efficacy depends on whether organic inputs or temperature are limiting.	Efficacy depends whether organic inputs or temperature are limiting.	Effective for very large streams.	Efficacy depends whether organic inputs or temperature are limiting.	Not effective.	Efficacy increases if organic inputs or temperature are limiting.

Function/Biota	Ephemeral	Intermittent	Small Permanent Stream	Large Permanent Stream	Small Lakes >4 ha with fish	Large Lakes >16 ha	Water Source
Fish Communities	Not directly applicable, but may impact through cumulative downstream effects.	Not directly applicable, but may impact through cumulative downstream effects.	Efficacy depends on whether temperature, organic inputs, or food sources are limiting factors.	Efficacy depends on whether temperature, organic inputs, or food sources are limiting factors.	Efficacy depends on whether temperature, organic inputs, or food sources are limiting factors.	Less dependent than smaller lakes on shoreline tree vegetation.	Not applicable.
Terrestrial Vegetation and Forest Structure*	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Greater soil moisture, productivity, shrub density, deadwood resources, and vertical diversity.	Greater soil moisture, productivity, shrub density, deadwood resources, and vertical diversity.	Greater soil moisture, productivity, shrub density, deadwood resources, and vertical diversity.	Greater soil moisture, productivity, shrub density, deadwood resources, and vertical diversity.	Insufficient data for evaluation of buffers in boreal and eastern slopes.
Bird Communities	Buffer use for edge and interior species depends on width. Potentially similar patterns to upland residuals.	Buffer use for edge and interior species depends on width. Potentially similar patterns to upland residuals.	Relative use by edge, interior, and riparian species dependent on width.	Relative use by edge, interior, and riparian species dependent on width.	Relative use by edge, interior, and riparian species dependent on width.	Relative use by edge, interior, and riparian species dependent on width.	Relative use by edge, interior, and riparian species dependent on width.
Mammal Communities	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Preference data (presence/absence) for moose, black bears, and grizzly bears indicate buffers may be effective in fulfilling some habitat requirements.	Preference data (presence/absence) for moose, black bears, and grizzly bears indicate buffers may be effective in fulfilling some habitat requirements.	Small mammals unlikely to utilize as specialized habitat in boreal. Insufficient data on other taxa. Insufficient data to evaluate east slopes.	Small mammals unlikely to utilize as specialized habitat in boreal. Insufficient data on other taxa. Insufficient data to evaluate east slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.
Other Terrestrial Animals (insects, amphibians, reptiles)**	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.	Insufficient data for evaluation of buffers in boreal and eastern slopes.

*By definition riparian buffers are a reserve of the current shoreline plant community.

** Current recommendations for long-toed salamander include the provision of forbs, shrubs, and downed woody materials cover along waterbodies in the east slopes.

3.0 QUANTITATIVE REVIEW OF RIPARIAN FUNCTIONS, STRUCTURES, AND BIOTA

3.1 Introduction

The physical delineation of the riparian area has been of long standing interest to aquatic and terrestrial ecologists and foresters. Understanding the spatial extent over which the riparian properties extends is a critical component for management. The spatial boundary of the riparian area can be examined along three axes. These include: longitudinal, vertical, and transverse (after Malanson 1993; United States Fish and Wildlife Service 1997). Most of the past and current research focuses on the transverse properties of riparian areas. That is, the area bordered by the shoreline to the edge of the uplands. In part, this emphasis has been pushed by a long history of riparian management focused on best management practices at the spatial scale of harvest stands.

This section reviewed and summarized ecological studies that recommend buffer widths for a selection of aquatic and terrestrial habitats (function and structure) and biota associated with riparian ecotones.

In order to determine the width of buffers, researchers have attempted to measure the impact of harvesting by surveying structure, biota, and processes within riparian areas. This is done by 1) extrapolating the impacts of harvest from measurement on undisturbed riparian areas, or 2) experimentally altering buffer widths and measuring the resultant changes. In turn, these distances from shorelines are used as a basis for developing buffer width guidelines. This section examined the width of riparian buffers recommended from the ecological literature. We reviewed and analyzed studies throughout Canada and the United States with a special focus on Boreal and Rocky Mountain ecoregions.

3.2 Data and Analytical Methods

A large body of literature was reviewed to assess the transverse widths of riparian areas. The primary bibliographic search engines were Agricola (1970 to 2002) and Cambridge Science Abstract's Ecological Abstracts (1980 to 2002). Other articles were derived from the reference sections of papers. The focus was on peer-reviewed journals and government reports. Although the search included review articles, these were primarily employed as a source for original articles. More than 250 articles were examined. From each study, we recorded the riparian function, structure, or biota measured, buffer width recommendations (median, minimum, and maximum), forest type, and geographic location[4]. We relied on the interpretation by the author(s) for the recommended buffer widths. No attempt was made to re-evaluate the quality of the papers or reports. The analysis was restricted to studies from forested ecosystems in Canada and the United States. The dataset was further stratified and analyzed according to ecoregions, biota, and riparian habitat features found in Alberta.

[4] References available from the authors.

3.3 Results

3.3.1 General Recommendations from Canada and the United States

Median buffer width recommendations of 30 m, 23 m, 42 m, and 90 m were estimated for aquatic biota, aquatic habitat, terrestrial habitat, and terrestrial biota, respectively. All distributions, except aquatic biota were right skewed (Figure 4). We found the fewest studies for aquatic biota and terrestrial habitat. More studies were available for aquatic habitat features than aquatic biota. The reverse was true on the terrestrial side where there were more studies on biota and fewer studies on habitat features.

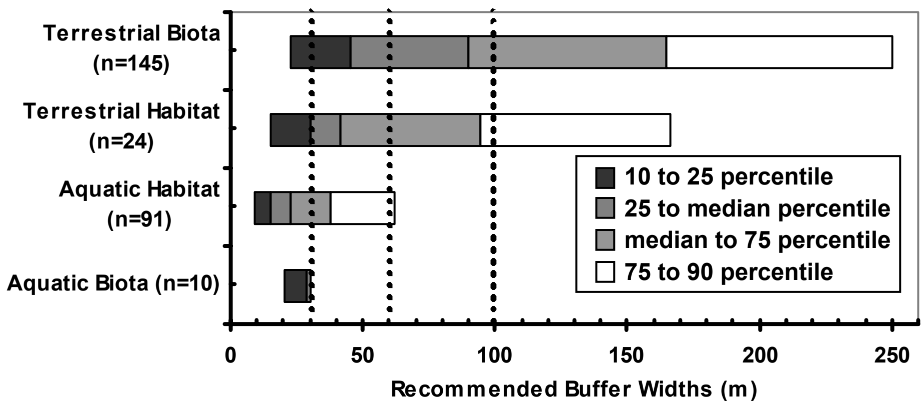

Figure 4. Distribution of buffer widths recommended in the scientific literature for broad habitat and biota groups. Different shades of gray represent the percentile of values for each buffer width. Dotted lines represent the buffer widths recommended for small permanent (30 m), large permanent (60 m), and lakes (100 m) in Alberta.

Recommended buffer widths for aquatic biota and habitats, and terrestrial habitats were less than for terrestrial biota. Water temperature, filtering of pollution and nutrients, and sediment and erosion control were associated with relatively narrow buffers. In contrast, birds and large mammals had the widest recommended buffers.

Stratification by specific features indicated that processes relating to water temperature, filtering of pollution and nutrients, and sediment and erosion control had the narrowest recommended buffers (Figure 5). Median values ranged from 18 to 30 m. There were relatively few studies on fish and aquatic invertebrates that provided recommended buffer widths. Our search only yielded a total of ten studies for both groups. For both groups, the median buffer width was 30 m (Figure 5).

Recommended buffers for plant communities associated with terrestrial riparian areas and organic debris input for both the terrestrial and aquatic components had larger median values of 38 m and 40 m, respectively (Figure 5). Buffers recommended for the protection of terrestrial vertebrates associated with riparian areas were the widest (Figure 5). Of this group, buffers for birds

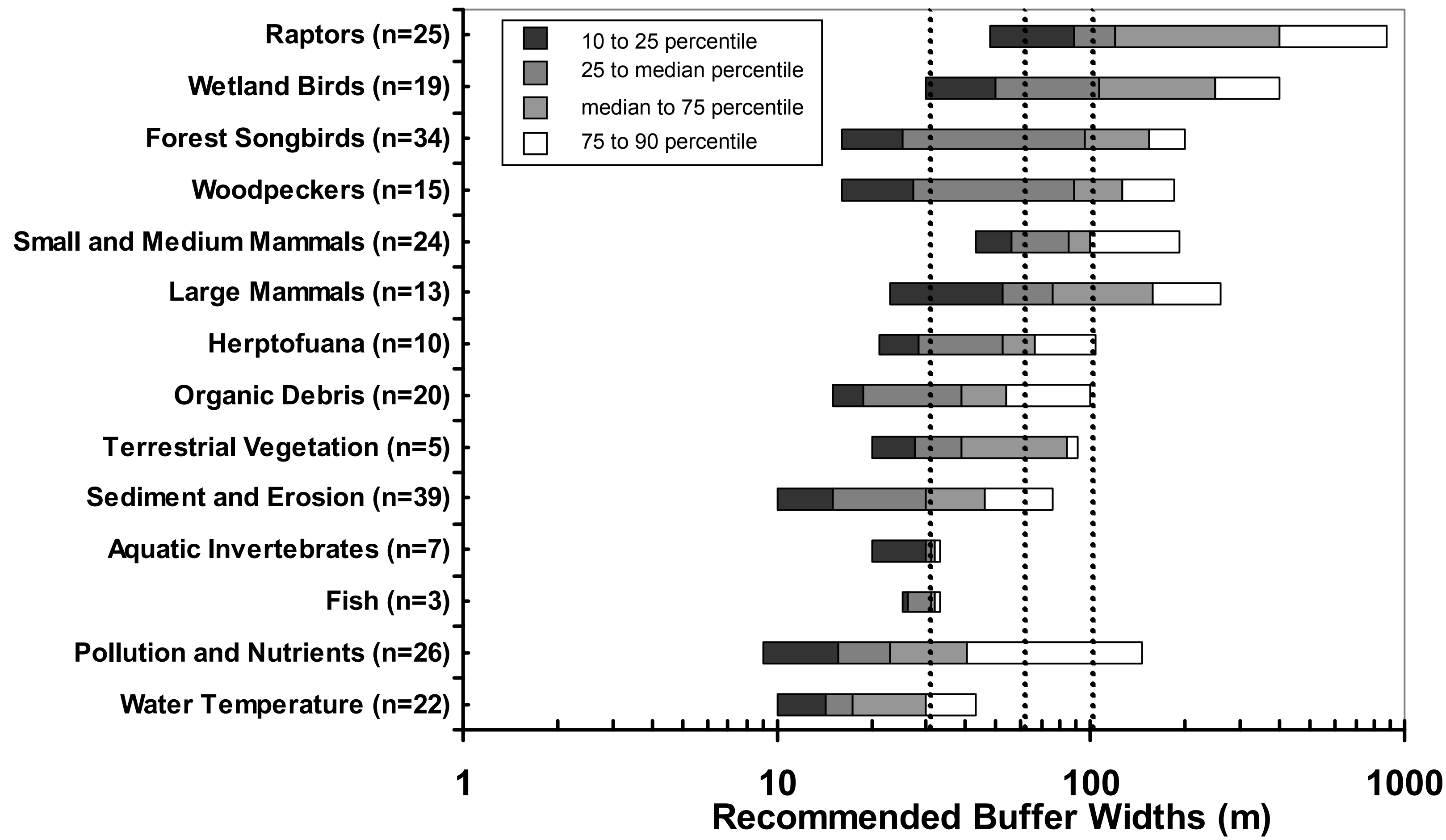

Figure 5. Distribution of buffer widths recommended by literature for specific ecological riparian functions, habitat, and biota. Different shades of gray represent the percentile of values for each buffer width. Dotted lines represent the buffer widths recommended for small permanent (30 m), large permanent (60 m), and lakes (100 m) in Alberta.

were greater than either herptofauna or mammals. The median values of 53 m, 76 m, and 86 m were recorded for herptofauna, small and medium mammals, and large mammals, respectively, while recommended buffers for forest songbirds, woodpeckers, wetland birds, and raptors were 96 m, 107 m, 88 m, and 119 m, respectively.

3.3.2 Recommendations for Boreal and Rocky Mountain Ecoregions

Most of the patterns applicable to Alberta were similar to those in other ecoregions in North America. The exceptions were recommended buffer widths for filtering of pollution and nutrients, and sediment and erosion control which were narrower.

Buffer widths from the subset of species or ecological functions that occur within ecoregions in Alberta were generally comparable to those from Canada and United States (Table 6). Median recommended buffer widths were 30 m, 40 m, and 100 m for aquatic biota, terrestrial habitat, and terrestrial biota. The major exception were the much narrower buffer widths (median 10 m) associated with aquatic habitats. Recommended widths for pollution and nutrients, sediment and erosion, and water temperature were 0 m, 9 m, and 15 m, respectively (Table 6). These results were from studies in Boreal regions. Unfortunately, there was a lack of data for aquatic habitats in the Rocky Mountains. Furthermore, little (n=1) or no data was available for the terrestrial vegetation for either the Boreal or Rocky Mountains.

There were fewer studies for Boreal and Rocky Mountain ecoregions than other ecoregions, particularly for aquatic invertebrates, fish, sediment and erosion, pollution and nutrients, terrestrial vegetation, herptofauna, woodpeckers, and wetland birds.

This leads to a second important pattern. There was an overall lack of applicable Boreal or Rocky Mountain data for most of the categories, particularly aquatic and terrestrial habitat. No data or fewer than five recommendations were found for eight subcategories of riparian habitat or biota in Alberta. Eleven subcategories when the Boreal ecoregion was considered alone, and 13 subcategories when the Rocky Mountains was considered alone (Table 6). The subcategories with few studies included: aquatic invertebrates, fish, sediment and erosion, pollution and nutrients, terrestrial vegetation, water temperature, herptofauna, woodpeckers, and wetland birds.

Alberta's 1994 guidelines cover the values recommended for aquatic biota and habitat and most biota and terrestrial habitat.

The widths in Alberta's 1994 buffer guidelines captured more of aquatic biota and habitat recommendations than terrestrial biota or habitat (Table 6). The current guideline width for small permanent streams, i.e. 30 m, was sufficient for median values for most aquatic biota and habitat. Six and four aquatic categories were less or equal than 30 m for Boreal and Rocky Mountain specific recommendations, respectively, while the remainder had no data. Current guidelines for large permanent streams, i.e. 60 m, extended the coverage to include terrestrial habitats. Guidelines for lakeshores, i.e. 100 m, further extended the coverage to include median recommendations for small and medium mammals and forest songbirds. Groups whose median buffer widths were greater than 100 m included raptors, woodpeckers, and large mammals. However, the median buffers widths for these groups were less than 30 m wider than the lakeshore guidelines for Alberta.

Results from this section should be interpreted with caution because of the lack of Alberta applicable studies for many riparian function, structures, and biota.

3.4 Discussion

Given the lack of data for most riparian functions, habitat, and biota specific to Alberta, we should be cautious about interpretation and extrapolation of information to management actions. For the Boreal and Rocky Mountain

Table 6. Medians and quartile ranges (in brackets), i.e. 25 and 75 percentiles, of recommended buffer widths for major and subcategories of riparian habitat and biota for ecoregions within Canada and United States. Recommendations were further subdivided by those studies that were applicable to Alberta and those that were applicable to other ecoregions outside of Alberta. n represents the number of data points. Degree of shading in cells indicated recommended medium buffer widths relative to Alberta's riparian buffer guidelines. No shading represented categories whose recommended median buffer width was equal or less than 1994 guidelines for small permanent streams (30 m), light grey cells were equal or less than 1994 guidelines for large permanent streams (60 m), medium grey cells were equal or less than 1994 guidelines for lakeshore buffers (100 m), and dark grey cells were greater than 1994 guidelines for lakeshore buffers. Black cells represented no available recommendation on buffer widths.

Major Categories	Subcategories	Canada and United States	Canada and United States (excluding Alberta	Alberta Only	Alberta – Boreal	Alberta – Rocky Mountain
Aquatic Biota		30 (29 - 30) n=10	30 (30 - 30) n=8	30 (20 - 30) n=3	20 (20 - 20) n=1	30 (30 - 30) n=2
	Invertebrates	30 (30 - 30) n=7	30 (30 - 30) n=6	20 (20 – 20) n=1	20 (20 – 20) n=1	No data
	Fish	30 (25 - 30) n=3	28 (25 - 30) n=2	30 (30 - 30) n=2	No data	30 (30 - 30) n=2
Aquatic Habitat		23 (15 - 37) n=91	25 (15 - 40) n=80	10 (0 - 20) n=11	13 (0 - 24) n=10	9 (9 - 9) n=1
	Pollution and Nutrients	23 (16 - 46) n=26	23 (15 - 45) n=24	0 (0 – 28.5) n=5	0 (0 – 28.5) n=5	No data
	Sediment and Erosion	30 (15 - 46) n=39	30 (16 - 47) n=37	9 (0 - 15) n=3	8 (0 - 15) n=2	9 (9 - 9) n=1
	Water Temperature	18 (14 - 30) n=22	18 (15 - 30) n=19	15 (10 - 37) n=3	15 (10 - 37) n=3	No data
Terrestrial Habitat		42 (30 - 94) n=24	44 (30 - 94) n=20	40 (28 - 100) n=5	40 (21 - 85) n=4	100 (100 - 100) n=1
	Organic Debris	39 (19 - 54) n=20	38 (23 - 53) n=17	40 (28 - 100) n=5	40 (21 - 85) n=4	100 (100 - 100) n=1
	Terrestrial Vegetation	38 (28 - 83) n=5	38 (28 - 83) n=5	40 (40 - 40) n=1	40 (40 - 40) n=1	No data
Terrestrial Biota		90 (45 - 165) n=145	80 (45 - 165) n=125	100 (61 - 196) n=57	100 (61 - 200) n=54	100 (60 - 200) n=48
	Herptofauna	53 (28 - 66) n=10	53 (28 - 66) n=10	No data	No data	No data
	Small and Medium Mammals	86 (56 - 100) n=24	76 (55 - 100) n=19	86 (45 – 121) n=10	86 (45 – 121) n=10	91 (56 – 142) n=5
	Large Mammals	76 (53 - 157) n=13	61 (40 - 125) n=10	117 (60 - 188) n=8	100 (60 - 180) n=7	133 (60 - 191) n=7
	Forest Songbirds	96 (25 - 154) n=34	83 (24 - 127) n=28	90 (25 - 138) n=15	95 (25 - 154) n=14	70 (24 - 135) n=12
	Woodpeckers	88 (27 - 126) n=12	61 (22 - 133) n=10	112 (100 - 124) n=2	112 (100 - 124) n=2	112 (100 - 124) n=2
	Wetland Birds	107 (50 - 250) n=19	107 (50 - 250) n=19	50 (30 - 200) n=3	125 (50 - 200) n=2	50 (30 - 200) n=3
	Raptors	119 (88 - 400) n=25	113 (88 - 369) n=24	120 (91 - 400) n=19	120 (91 - 400) n=19	120 (91 - 400) n=19

regions, the majority of categories (67%, 24/36) had less than five studies associated with recommended buffer widths. Eight (22%) categories had no recommendations based on research in these regions at the time of this report. Problems with interpretation caused by the lack of data may be further compounded because much of the current data on the Boreal and Rocky Mountains are from eastern Canada or the United States. In the case of the western boreal forest, the underlying geology and hydrology are different, hence, the results may not be appropriate (Tóth 1970; Devito *et al.* 2000). Much of work done in the eastern Canada should be re-designed and executed in the Western Boreal Plains (sic. Environment Canada 2002).

For the Rocky Mountains, there was a lack of riparian studies across all groups. Thirteen of eighteen groups (72%) had fewer than five studies with recommendations. Even for groups such as raptors many of the recommendations were species-based with much of the data originating from the Pacific Northwest. It is unclear to what extent these would apply to the same species residing in different ecoregions.

In general, the lack of studies was more notable with aquatic biota and habitats, and terrestrial habitat. Much of the previous effort has focused on finding terrestrial indicators (mainly biotic) for estimating the extent of the undisturbed riparian zone rather than measuring the degree of impact on aquatic biota after riparian disturbance. Many studies on terrestrial organisms were at the species level there-by increasing the number of studies. This was particularly true with raptors, ungulates, and large carnivores. With the exception of game fish, most aquatic studies focused on collectively reporting on large groups of organisms such as macroinvertebrates. Generally, this led to fewer reports.

Part of the attractiveness in focusing on terrestrial measurements is the relative simplicity of most experimental designs. Many studies focus on treatments of different buffer widths within a similar forest type. Aquatic organisms require a longitudinal, i.e. downstream and upstream, approach adding to the inherent complexity and logistic difficulties in experimentation. In both cases, treatments may take years to manifest themselves on habitat and biota features. The logistics of designing and replicating field experiments over the large spatial and temporal scales leads to fewer studies. However, the cost of these large and long-term experiments may be justified, in that, most of the knowledge-base on forestry and aquatic riparian effects comes from this type of experimentation (e.g. Coldwater Lakes Experimental Watersheds, Ontario[5], Experimental Lakes Area, Manitoba[6], Hubbard Brook Experimental Forest, New Hampshire[7]).

Despite the paucity of information on some groups, a couple of trends were evident. Generally, the recommended buffer widths for aquatic components

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[5] http://www.aquatic.uoguelph.ca/Human/Research/stu/steedmanr.htm
[6] http://www.umanitoba.ca/institutes/fisheries/
[7] http://www.hubbardbrook.org/

were narrower than terrestrial components. This is not surprising since disturbance of riparian areas by harvest directly affects terrestrial components while only indirectly affecting aquatic components. In general, terrestrial riparian studies followed three patterns. The first type of study analyzed the presence/absence or habitat use (e.g. track counts) of near-shore to upland gradients in undisturbed, intact riparian areas. The riparian zone was identified by changes in species densities or/and community composition. A second type of study measured changes after harvesting with varying widths of buffers. The riparian zone was identified by thresholds of species density or community composition changes that occur with changes in buffer width. These two types of experiments were the most common types for identification of riparian zones. Rarer were experiments that examined the potential changes to life history or dispersal characteristics of species when they are restricted to riparian buffer strips after harvest (for an exception see Lambert and Hannon 2000). Either by implicit or explicit design, these methodologies focus on the independence of near-shore areas from upland harvest as a major criteria for defining the riparian zone.

Even the large number of studies on terrestrial biota can be somewhat misleading. Most studies record presence/absence or density, hence it is difficult to extrapolate these metrics to potential functional uses of buffers such as corridors, temporary refugia, or core habitat. Furthermore, it is unclear which species require the proximity of trees to water as a vital component of their life history.

As previously discussed, the objectives for the management of habitat or biota need to be articulated prior to experimentation (Section 3.0). If the management objective is to retain core, i.e. breeding, habitat, then relatively detailed information on population dynamics is required to assess habitat parameters including the width of buffers. In contrast, management objectives may focus on the retention of sufficient riparian components to allow for the full recovery populations and communities within a specified time after disturbance of the upland and perhaps riparian. In this case, field measurements should concentrate on use of remaining riparian habitat as temporary refugia. The term “temporary” would have to be defined with appropriate spatial and temporal parameters. In all likelihood, this objective will require less stringent riparian buffer widths than the previous objective. Most studies did not articulate the conservation objectives prior to recommending buffer widths.

The role of riparian buffers has shifted from controlling sedimentation and erosion to provision of aquatic and terrestrial habitat for biota. Many of these features require the integration of management activities at larger spatial and temporal scales.

Current riparian guidelines in Alberta cover most aquatic features and terrestrial habitat on larger streams and lakeshores. For a number of components such as filtering of pollution and nutrients, and sediments and erosion, recommended buffer widths maybe considerably less than elsewhere in Canada and the United States (for discussion see Section 4.0). The original intent of the 1994 guidelines was to prevent sedimentation and erosion (Robert Anderson, personal communication). From this point of view, studies would suggest that narrower guidelines than the current ones may be warranted. However, changes would have to be balanced with other ecological functions that suggest the retention relatively wide buffers such as those for terrestrial habitat and biota.

Buffers are only a component of riparian management. Other elements include; roads and skid trails, stream crossings, landing placement, total watershed disturbance, and silvicultural activities. In Alberta, activities in the energy and transportation sectors also impact riparian areas. These activities act cumulatively to produce a total impact on the aquatic and terrestrial components of the riparian area. In this regard, the role of riparian buffers may be changing over time. Originally, they were applied for local controls on sedimentation and erosion. Improved best practices such as winter logging, low ground disturbance, and road and cutblock design may negate any current use of buffers as barriers for sediment movement. Instead, their value as temporary or long-term refugia for terrestrial organisms may become increasingly important given the amplified levels of disturbance in Alberta's north. Managers need to develop an overall plan for the maintenance of species and riparian functions and articulate the role of riparian buffers. In this regard, Alberta has fallen behind other jurisdictions in developing large-scale plans that integrate a number of different activities within watersheds.

3.5 Summary of Key Findings

- In general, aquatic biota and habitats, and terrestrial habitats have narrower buffer width recommendations than terrestrial biota (But see Discussion).
- Based on available data, Alberta's current guidelines are within the range of recommended buffer widths for protection of aquatic biota and habitat for small permanent streams, large permanent streams, water sources, and lakeshores. Current buffer width guidelines are narrower than those recommended for terrestrial biota.
- There are a lack of studies with buffer width recommendations for Boreal and Rocky Mountain ecoregions in Alberta, particularly for aquatic biota and habitats, and terrestrial habitats.

4.0 QUANTITATIVE REVIEW OF BUFFER WIDTH GUIDELINES

4.1 Introduction

Riparian zones have a long history of special management in forestry (Porter 1887). Implementation of treed corridors along streams dates to the 1700's in European forest management. The practice of leaving buffers was first implemented in United States in late 1960's (Calhoun 1988 reference in Broskfske *et al.* 1997). The primary reasons for their use today are similar to their use in the past. These include: flood control, reduction of sedimentation, stabilization of navigation corridors, maintenance of wildlife habitat, and protection of fishing stocks (See Section 2.0). The underlying philosophy in this type of management is the isolation of upland activities from aquatic zones. The continued prevalence of this paradigm is demonstrated by its adoption in nations attempting to modernize forestry practices such as China (Deng *et al.* 2001).

Riparian buffers left after timber harvest, have a long history of use in Europe and North America. Their primary objective is the separation of upland activities from riparian areas.

The application of no harvest or restricted harvest buffer strips along waterbodies has been the traditional response of managers for the maintenance and protection of riparian values. A variety of names are used to describe this area including; special management zones, riparian management zone, streamside management zones, and riparian management areas. A fundamental problem in the application of buffers is the underlying variation in the ecological, social, and economic value of riparian areas. Waterbodies vary in their topographical context, water quality parameters, sizes, aquatic and terrestrial biota, human use, and aesthetic value. Buffer width guidelines reflect trade-offs between ecological, social and economic factors, and historical perspectives riparian areas.

Despite similar objectives across different jurisdictions guidelines vary a great deal.

This section reviewed and quantified some of the differences and similarities among guidelines from Canada and the United States.

Guidelines for operating beside watercourses in Alberta are outlined in the *Alberta Timber Harvest Planning and Operating Ground Rules* (Alberta Sustainable Resource Development 2002) ("Ground Rules"). The Ground Rules describe the use of buffers under two scenarios: where logging and harvest related activities are prohibited within the buffer or, if approved, where modified logging operations/harvest related activities within the buffer region. Exceptions in the guidelines permit more extensive or permissive activities based on a wide variety of circumstance including local topography, wildlife use, and salvage logging. Our primary objective in this section was to review and analyze the structure and underlying riparian values embodied in riparian guidelines throughout Canada and the United States. Guidelines from other jurisdictions particularly Boreal and Rocky Mountain jurisdictions were compared with those for Alberta. The specific objectives included: 1) Comparison of national and regional differences 2) Examine differences in the structure of guidelines through the use of modifying factors, 3) Examine the application of selective harvest to riparian management areas.

4.2 Data and Analytical Methods

A database of riparian management guidelines and regulations were obtained by contacting provincial and state jurisdictions in Canada and the United States. Washington State utilizes different guidelines for the east and west sides of the state. These were included as separate jurisdictions. Nunavit, District of Columbia, Kansas New Mexico, and Arizona, did not have guidelines for forest harvesting at the time of this report. A total of 60 jurisdictions were analyzed (See 9.2 Appendix B).

A total of 60 jurisdictions were examined. In order to facilitate comparisons, jurisdictions were classified according to ecoregions and their guidelines were re-interpreted to be comparable to Alberta's 1994 waterbody classes.

To examine the effect of geography on riparian guidelines, we categorized jurisdictions into six regions (Table 7). Classification was broadly based on a combination of geographic location and ecological similarities and a general consensus of a number of general references (Bailey and Cushwa 1981, Environment Canada 2001, and United States Department of Agriculture, Forest Service 2002). Statistical analysis required at least three jurisdictions in each category. In this regard, the southwestern ecoregion comprising of New Mexico, Arizona, and Nevada had only data for Nevada, hence, there was no analysis of this ecoregion. A number of jurisdictions have guidelines that apply to different ecoregions. In these cases, the guidelines were applied in each region (Table 7). As an example, British Columbia's guidelines were examined in the Pacific as well as the Boreal ecoregions.

Factors that modify riparian areas such as slope, presence of fish-bearing streams, waterbody type, waterbody size, human water supply, and downstream effects were also examined.

In order to compare different waterbody classifications, we used the classifications from Alberta as a template and applied the guidelines from other jurisdictions to these criteria. This resulted in a standardized template by which the buffer widths assessed for large permanent stream, small permanent stream, intermittent stream, ephemeral stream, large lake, small lake, and water source (Table 8). Guidelines often had modifying factors which altered the baseline buffers prescribed to a waterbody. The diversity and relative frequencies of the commonly applied modifying factors were examined. The use of complementary, i.e. two or more, modifying factors was examined by recording the paired frequency of factors and comparing this with the expected frequency if each factor was selected independently of each other. For all these analyses, we focused on the most commonly available classification, i.e. large streams. Our purpose was to demonstrate the changes to buffer widths under different modifying factors rather than to exhaustively examine to all combinations of modifying factors and waterbody types.

Table 7. Classification of provinces, territories, and states from Canada and the United States into broad ecological regions.

Country	Regions	Jurisdictions
Canada	Boreal	Alberta, Manitoba, Newfoundland, Northwest Territories, Ontario, Quebec, Saskatchewan, Yukon, British Columbia
	Northeast	New Brunswick, Nova Scotia, Prince Edward Island
	Rocky Mountain*	Alberta, British Columbia
	Pacific	British Columbia
United States	Boreal	Michigan, Minnesota, Wisconsin, Alaska
	Rocky Mountain*	Colorado, Montana, Utah, Wyoming, Idaho, Nevada, Washington east
	Midwest	Illinois, Indiana, Iowa, Missouri, Nebraska, North Dakota, Oklahoma, South Dakota, Texas
	Northeast	Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, West Virginia
	Pacific	Alaska, California, Hawaii, Oregon, Washington west
	Southeast	Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Virginia

* The regions also include the Intermountain region.

Table 8. Definitions for model waterbody classes used to facilitate comparisons between different jurisdictions.

Waterbody Type	Basic Description	Size	Slope	Fish-Bearing	Human Intake	Streamside Forest Management
Large permanent streams	Permanent watercourse with defined bank, year-round flows	>5 m width	2.5%	Yes	Yes	Non-plantation
Small permanent streams	Permanent watercourse with defined bank, year-round flows	≤5 m width	2.5%	Yes	Yes	Non-plantation
Intermittent streams	Permanent watercourse with defined bank, no year-round flows	Any width	2.5%	No	Yes	Non-plantation
Ephemeral streams	Temporary water course	Any width	2.5%	No	No	Non-plantation
Small lakes	Standing waterbodies	>4 ha	2.5%	No	Yes	Non-plantation
Large lakes	Standing waterbodies	<4 ha	2.5%	Yes	Yes	Non-plantation
Water sources	Areas with permanent or temporary standing water or discharges of water. Includes springs, seepages, and wetlands without surface water, e.g. bogs, fens.	Any size	2.5%		Yes	Non-plantation

4.3 Results

4.3.1 Comparison of Canadian and American Jurisdictions

On average, guidelines from jurisdictions in the United States were narrower than those in Canada across all waterbody types except intermittent and intermittent streams.

In general, jurisdictions in the United States exhibited narrower buffer widths than in Canada for similarly classified waterbodies (Table 9). Large and small permanent streams, small and large lakes, and water sources all had significantly wider buffers in Canada than in the United States. Mean buffer widths from Canadian jurisdictions were wider by 33 to 58%. No significant differences were detected for either intermittent or ephemeral streams (Wilcoxon test, P=0.65). Mean widths of ephemeral buffers were 88% wider in Canada but the absolute difference was less than a metre and there were few jurisdictions (n=2).

Table 9. The mean (S.E.) and median widths (lower line) of riparian management areas for different waterbodies pooled by provinces and territories in Canada, states in the United States, and combined for both countries. % difference represented the degree to which Canadian jurisdictions were greater than American jurisdictions (based on a percentage of the mean width in Canada).

Stream Classes	Canada[b] (n=12)	United States (n=48)	Combined (n=60)	% Difference
Large permanent streams	44 (9)	24 (2)	28 (3)	+45[a]
	30	15	20	+50
Small permanent streams	30 (4)	20 (2)	22 (2)	+33[a]
	23	15	15	+33
Intermittent streams	14 (3)	16 (2)	15 (2)	-13
	20	15	15	+25
Ephemeral streams	0.8 (0.8)	0.1 (0.1)	0.2 (0.2)	+88
	0.0	0.0	0.0	N/A
Small lakes	47 (11)	23 (2)	28 (3)	+51[a]
	30	15	20	+50
Large lakes	55 (11)	23 (2)	29 (3)	+58[a]
	30	15	20	+50
Water source	40 (10)	22 (2)	25 (3)	+46[a]
	30	15	16	+50

[a] denotes statistically significant differences (P<0.05) between Canadian and American jurisdictions (Wilcoxon test, df=1, P range 0.012 to 0.0015).
[b] includes Alberta

Alberta's buffer guidelines were wider than most jurisdictions particularly for large permanent streams and lakes.

Alberta's guidelines for large and small permanent streams, and small and large lakes were wider than other jurisdictions in either Canada or the United States (Table10). In particular, mean percentages were greater by more than 70%. Water source guidelines in Alberta were narrower than mean and median values for the rest of Canada but were greater than the mean width of jurisdictions in the United States. In Alberta, intermittent streams are an onsite call with potentially no buffer whereas the mean widths were 14 and 16 m, in the remainder of Canada and United States, respectively. Only two jurisdictions in North America provide specific buffer guidelines for ephemeral streams.

Table 10. Comparison of Alberta 1994 guidelines with other jurisdictions in Canada and United States, and both countries combined. Percentage differences were based on mean and median buffer widths comparisons to Alberta's guidelines. Positive values indicate greater widths with Alberta guidelines. N/A (not applicable) was listed for intermittent and ephemeral streams because Alberta's guidelines do not specify a buffer width.

Waterbody Classes	Alberta 1994 Guidelines (m)	% Difference with other Canadian Jurisdictions	% Difference with American Jurisdictions	% Difference with all Jurisdictions
Large permanent streams	60	+30	+60	+53
	60	+50	+75	+67
Small permanent streams	30	+2	+34	+27
	30	+33	+50	+50
Intermittent stream	0	N/A	N/A	N/A
	0	N/A	N/A	N/A
Ephemeral streams	0	N/A	N/A	N/A
	0	N/A	N/A	N/A
Small lakes	100	+58	+77	+72
	100	+70	+85	+80
Large lakes	100	+50	+77	+71
	100	+70	+85	+80
Water source	20	-107	-8	-25
	20	-50	+25	+20

4.3.2 Regional Patterns

Guidelines from Boreal ecoregions were the widest amongst in Canada and the United States while those from Rocky Mountain jurisdictions were intermediate in width.

Of seven waterbody classes, the mean buffer widths for the Boreal jurisdictions were the widest (Table 11). Buffer guidelines for small permanent and intermittent streams were similar or less than other jurisdictions, respectively. In general, medium width buffers were recommended in Rocky Mountain, Midwest, Pacific, and Northeastern jurisdictions (Table 11). Southeastern jurisdictions had the narrowest riparian buffer widths. However, there were exceptions. As an example, buffers for small permanent and intermittent streams in the Midwest were narrower than other regions (Table 11).

Table 11. Comparison of regional differences in buffers widths for waterbody classes. Mean widths (S.D.) and median values (listed below) are presented. Superscripts represent significant differences (Wilcoxon test, $P<0.05$). No data was available for two of the three southwestern states i.e. New Mexico, and Arizona. These were not listed in this dataset.

Waterbody Classes	Boreal (n=13)	Rocky Mountain (n=9)	Midwest (n=9)	Northeast (n=16)	Pacific (n=6)	Southeast (n=11)
Large permanent streams	39 (6) ^{a}	24 (7) ^{ab}	26 (6) ^{ab}	30 (7) ^{ab}	24 (8) ^{ab}	19 (3) ^{b}
	30	15	18	18	18	15
Small permanent streams	26 (3) ^{ab}	24 (7) ^{a}	14 (1) ^{b}	24 (4) ^{ab}	23 (8) ^{ab}	18 (3) ^{b}
	30	15	15	15	17.5	15
Intermittent	14 (4) ^{b}	24 (7) ^{a}	12 (2) ^{b}	13 (3) ^{ab}	22 (8) ^{ab}	12.1 (3.4) ^{b}
	16	15	12	15	15	11
Ephemeral	0.8 (1) ^{a}	0 (0) ^{a}	0 (0) ^{c}	0 (0) ^{b}	0 (0) ^{a}	0.5 (0.5) ^{a}
	0	0	0	0	0	0
Small lakes	46 (9) ^{a}	23 (7) ^{ab}	22 (6) ^{b}	31 (7) ^{ab}	23 (4) ^{ab}	17 (3) ^{b}
	30	15	15	20	25	15
Large lakes	52 (9) ^{a}	23 (7) ^{ab}	22 (6) ^{b}	30 (7) ^{b}	23 (4) ^{b}	17 (3) ^{b}
	30	15	15	20	25	15
Water source	37 (7) ^{a}	23 (7) ^{ab}	18 (5) ^{b}	30 (7) ^{ab}	20 (3) ^{ab}	16 (3) ^{b}
	30	30	15	20	18	15

Alberta's riparian buffers were amongst the widest of Boreal and Rocky Mountain jurisdictions across all waterbody types except intermittent streams.

In comparison to other Boreal or Rocky Mountain jurisdictions, Alberta's 1994 Ground Rules recommended wider buffers for large and small permanent streams, and small and large lakes (Table 12). Percentages ranged from 17 to 319% greater across those waterbodies. The greatest disparities were for large and small lakes. Alberta's guidelines had no specific buffer width recommendations for intermittent streams whereas other Boreal and Rocky Mountain jurisdictions had mean (S.E.) widths of 16 and 24 m, respectively. Water sources in other Boreal jurisdictions had significantly wider buffers than those in Alberta while other Rocky Mountain jurisdictions had slightly wider buffers than Alberta (Table 12).

Table 12. Comparison of Alberta's 1994 buffer width guidelines with other Boreal and Rocky Mountain jurisdictions for waterbody classes. Positive values indicate wider buffers in Alberta. N/A was listed for intermittent and ephemeral streams because Alberta's guidelines do not specify a buffer width.

Waterbody Classes	% Difference Other Boreal	% Difference Other Rocky Mountain
Large permanent streams	68	139
Small permanent streams	17	26
Intermittent streams	N/A	N/A
Ephemeral streams	N/A	N/A
Small lakes	172	319
Large lakes	126	319
Water source	-44	-16

A number of modifiers were commonly applied to determine the width of riparian buffers. The five most common were ordered: waterbody type, slope, waterbody size, presence of fish, and human water supply/aesthetics.

4.3.3 Modifying Factors

Most jurisdictions used modifying factors in guideline formulation (Figure 6). Seventy-eight percent of jurisdictions used two or more modifiers while just under half (~44%) had three or more modifying factors in their guidelines (Figure 6). Waterbody type, shoreline slope, waterbody size, and presence of fish were the most common modifying factors (Table 13). Less common factors included: human water supply/aesthetics, forest management practices adjacent to waterbodies, presence of saltwater flow, types of shoreline vegetation, upstream of fishbearing waterbodies, downstream threat of sediment transport, and flow rates.

On average each jurisdiction in North America utilized 2.7 modifying factors with the Boreal and Rocky Mountain jurisdictions utilizing 3.8 and 2.4 factors, respectively. Boreal jurisdictions had a greater percentage of jurisdictions using waterbody type, waterbody size, presence of fish, drainage basin area, shoreline forest management, and shoreline vegetation as modifying factors than all other jurisdictions (Table 13).

For Boreal and Rocky Mountain jurisdictions, the types and order of modifiers were similar to those found across all jurisdictions. Boreal jurisdictions focused on modifiers such as waterbody type, waterbody size, presence of fish, drainage basin area, shoreline forest management, and shoreline vegetation while Rocky Mountain jurisdictions added slope to this list.

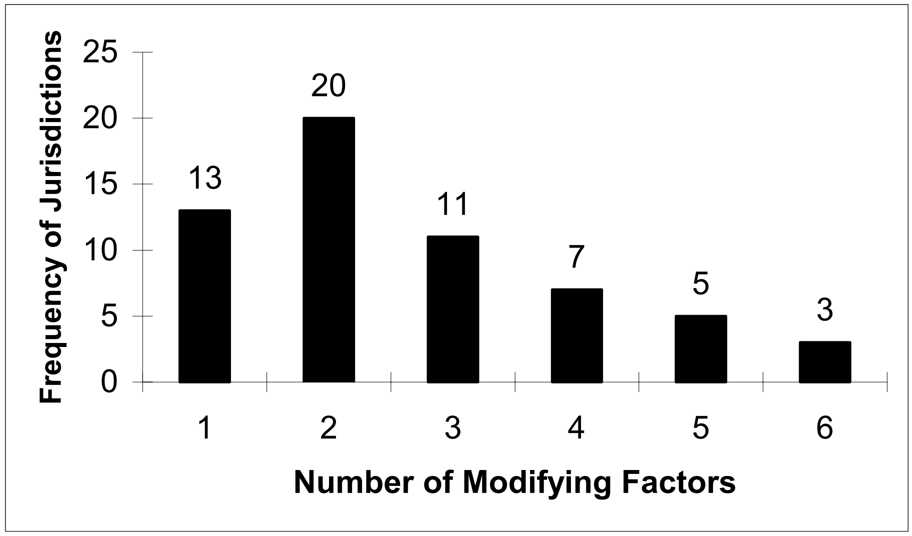

Figure 6. Distribution of jurisdictions using multiple factors in individual guidelines. Values are listed above bars.

Table 13. Percentage of jurisdictions using different types of modifiers in guideline formulation.

Madifying Factor	% All	% Boreal	% Rocky Mountain
Waterbody type	79	91	63.6
Slope	49	27	36.4
Waterbody size	33	55	27.3
Fishbearing	33	64	27.3
Human water supply/aesthetics	15	18	18.2
Drainage basin area	7	27	0.0
Shoreline forest management	8	36	9.1
Saltwater flow	10	9	0.0
Shoreline vegetation	5	27	9.1
Upstream of fishbearing	5	0.0	9.1
Downstream sediment threat	3	0.0	0.0
Flow rates	3	0.0	0.0

Modifiers were often applied in particular combinations. The most common combinations included waterbody type, slope, presence of fish, and waterbody size.

A lower percentage of Rocky Mountain jurisdictions used modifying factors than other jurisdictions in North America.

The most common combinations of two factors were waterbody type with either slope, waterbody size, or presence of fish (Table 14). Combinations of waterbody type and one of these other modifiers were found in ~30% of the guidelines. Less common pairs were frequently applied modifiers such as waterbody type, slope, waterbody size, or presence of fish with modifiers such as human water supply/aesthetics, adjacent forest management, and shoreline vegetation. Most pairs were represented more than would be expected by independent selection (Table 14). That is, managers appear to be selecting combinations of modifiers to complement each other. The strongest cases for preference were fishbearing streams in combination with human water supply/aesthetics and fishbearing streams with waterbody size.

Amongst Boreal jurisdictions, the most common pairs featured waterbody type, waterbody size, and fishbearing in the same guidelines (Table 15). Much less frequently, these modifiers were found in combination with modifiers such as shoreline forest management practices, shoreline vegetation, and drainage basin area. Rocky Mountain guidelines had relatively few combinations of modifiers (Table 15). Combinations with slope, presence fishbearing or upstream of fishbearing , waterbody size and type were the most common pairs.

Table 14. Most common (>10% of jurisdictions) combinations (pairs) of modifiers used by jurisdictions. N represents the number of jurisdictions with actual percentage based on the total number of jurisdictions sampled (N=60). Expected percentage is the number jurisdictions expected based solely on independently selecting combinations of modifiers. The difference between actual and expected values is expressed as a percent value in the last column.

Combination of Modifiers	N	Actual Percentage	Expected Percentage	Percent Difference
Waterbody type - Slope	23	39.0	38.7	0.7
Waterbody type – Waterbody size	18	30.5	25.8	18.2
Waterbody type - Fishbearing	18	30.5	25.8	18.2
Slope - Fishbearing	11	18.6	16.1	15.5
Waterbody size - Fishbearing	9	15.3	10.8	41.8
Fishbearing - Human water supply/aesthetics	7	11.9	4.9	144.4
Waterbody type - Human water supply/aesthetics	7	11.9	11.6	1.9

Table 15. Pairs of modifiers used by Boreal and Rocky Mountain jurisdictions. Percentages were based on the number of jurisdictions with combinations divided by the total number of jurisdictions (excluding Alberta).

Combination	% Boreal	% Rocky Mountain
Waterbody type - Fishbearing	63.6	12.5
Waterbody type – Waterbody size	54.5	12.5
Waterbody size - Fishbearing	45.5	0.0
Fishbearing – Shoreline forest mgmt.	36.4	0.0
Waterbody type – Shoreline forest mgmt.	36.4	0.0
Fishbearing – Shoreline vegetation	27.3	0.0
Shoreline vegetation - Waterbody type	27.3	0.0
Fishbearing – Drainage basin area	27.3	0.0
Waterbody type - Drainage basin area	27.3	0.0
Waterbody size - Shoreline forest mgmt.	27.3	0.0
Fishbearing – Human water supply/aesthetics	18.2	0.0
Slope - Fishbearing	18.2	0.0
Waterbody type - Slope	18.2	0.0
Slope – Upstream of fishbearing	0.0	12.5
Waterbody type - Upstream of fishbearing	0.0	12.5
Waterbody size - Slope	0.0	12.5

4.3.4 Waterbody Types

Streams were the most commonly recognized (71%) waterbody type in guidelines (Table 16). Forty two percent of jurisdictions recognized differences between intermittent and permanent flow streams. Lakes and wetlands were the next most common classes. A relatively large percentage of jurisdictions (21%) applied a single classification across all waterbody types. A number of jurisdictions utilized other classifications including; ponds, saltwater/brackish estuaries, coldwater/ warmwater bodies, and braided streams.

The most common separation for waterbody type was based on stream and lakes, and permanent and intermittent streams.

In comparison to other Boreal and Rocky Mountain jurisdictions, Alberta's guidelines use the major classifications, i.e. streams, and lakes (Table 16). However, Alberta's guidelines do not specifically cover smaller bodies of water, e.g. ponds, nor does it specifically recognize wetland complexes. Saltwater and brackish water, and cold/warmwater bodies were not applicable to Alberta's Boreal and Rocky Mountain jurisdictions.

Table 16. Percentages of waterbody types for all, Boreal, and Rocky Mountain jurisdictions (excluding Alberta).

Waterbody Types	% All	% Boreal	% Rocky Mountain
Streams	71	82	75
Permanent/intermittent streams	42	36	25
Lakes	37	82	50
General waterbodies*	21	18	25
Wetlands	21	27	25
Ponds	7	20	0.0
Saltwater/brackish estuaries	7	9	0.0
Cold/Warmwater bodies	3	9	0.0
Braided streams	2	0.0	0.0

* Some jurisdictions did not recognize differences in waterbody types and applied guidelines to a broad category including many types of waterbodies.

4.3.5 Slope

Across all jurisdictions that modified buffers by slope, 0.7 m was added to buffer widths for each percent increase in slope. For Boreal and Rocky Mountain jurisdictions, 0.5 m and 0.7 m were added for each percent increase in slope. Jurisdictions that did not have a modifier for slope had wider baseline buffers.

Jurisdictions that do not modify buffer widths in accordance with the slope of the near-steam area usually maintain a safety margin by implementing wider baseline buffers. The mean (S.E.) buffer width at 0% slope for jurisdictions with slope modifiers was 17 m (3) while the mean width for jurisdictions without slope modifiers was 33 m (3) (ANOVA, P>0.01). On average, jurisdictions with slope guidelines added a mean (S.E.) width of 0.7 m (0.08) for each 1% increase in slope. Based on this relationship, jurisdictions that did not have slope guidelines have an additional 16 m to their baseline buffer. This supplementary width would account for an additional 23% slope.

A similar trend was found when considering only Boreal jurisdictions (excluding Alberta). Mean (S.E.) baseline buffer widths for jurisdictions with slope modifiers were 30 m (5) while those with no slope modifiers had a mean base buffer width of 47 m (8). However, these differences were not significantly different (ANOVA, P=0.22). The rate of buffer width increase was 0.53 m per 1% increase in slope. This suggests that jurisdictions that had no slope modifiers were retaining a safety margin to account for approximately 26% of additional slope. Mean (S.E.) base buffers widths among Rocky Mountain jurisdictions (excluding Alberta) with slope modifiers was 18 m (3). Mean widths for those without slope modifiers was 40 m (11). This difference was significant (ANOVA, P<0.05). The rate of buffer width with slope was 0.7 m per 1% increase in slope. This suggests that jurisdictions that had no slope modifiers were retaining a safety margin to account for approximately 30% of additional slope.

If other Boreal and Rocky Mountain jurisdictions were used as a standard, Alberta's current large permanent stream guidelines could handle about a 25% slope.

Alberta's current guidelines do not explicitly include slope modifiers. Based on the results of this section, the current buffer guidelines for both Boreal and Rocky Mountains should be able to handle slopes up to about 25% particularly for large streams, and lakes in the Boreal and Rocky Mountain, and small permanent streams in the Rocky Mountains.

4.3.6 Waterbody Size

In general, guidelines distinguished three size for criteria for streams (mean widths 0 m, 5 m, and 9 m), two for lakes (median area 1 ha and 4 ha), and two for wetlands (0.3 ha and 2.3 ha).

Table 17 summarizes mean class boundaries used to delineate sizes of different waterbodies across North America. Jurisdictions varied in the types of waterbodies which were classified according to size. Most jurisdictions (n=54) classified streams fewer classified lakes (n=41), and fewer still classified wetlands (n=15). Of these only 15, 8, and 8 jurisdictions went on to establish a second size class for streams, lakes, and wetlands, respectively. For streams, the mean and median widths were both 5 m, while for lakes the mean and median widths were 4 and 3 ha, respectively. Wetlands were smaller than lakes, mean and median areas for categorizing larger wetlands were both 2 ha. Only guidelines for streams had a 3rd criterion for categorizing the largest channels. The mean and median for these criteria were 9 m and 12 m, respectively.

For Boreal jurisdictions, guidelines distinguished three size for criteria streams (mean widths 0 m, 2 m, and 4 m), two for lakes (median area 1 ha and 8 ha), and two for wetlands (0.3 ha and 5 ha). Rocky Mountain jurisdictions differentiated the sizes of wetlands (0.1 ha and 2 ha) but not other waterbody types.

In general, size criteria for streams in the Boreal and Rocky Mountains were smaller than other jurisdictions in Canada and the United States (Table 17). The mean lower threshold values for small, medium and large streams were 0.4, 2, and 4 m, respectively, for Boreal jurisdictions (excluding Alberta). For Rocky Mountain jurisdictions, the mean lower threshold values for small, medium and large streams were 0.2, 4, and 5 m, respectively. Classification of large Boreal lakes and wetlands was initiated at a larger size (mean 8 and 5 ha, respectively) relative to other jurisdictions in North America and Rocky Mountains (Table 17).

Table 17. Waterbody size criteria for streams, lakes, and wetlands summarized across all, Boreal, and Rocky Mountain jurisdictions (excluding Alberta). 2nd and 3rd criteria are breakpoints used to distinguish larger classes of waterbodies. Mean values are given on the top line while median values are given on the lower line.

Waterbody Type	All	Boreal	Rocky Mountain
Small streams (m)	0.1 to 5.0 0.0 to 5.1	0.4 to 2.0 0.0 to 3.0	0.2 to 3.0 0.0 to 3.0
Medium streams (m)	5.0 to 9.3 5.1 to 12.2	2.1 to 4.3 3.1 to 5.0	3.8 to 5.0 3.1 to 5.0
Large streams (m)	>9.3 >12.2	>4.3 >5.0	>5.0 >5.0
Small lakes (ha)	0.9 to 4.3 0.0 to 2.5	1.0 to 7.5 0.0 to 5.0	0.2 to 5.0 0.0 to 5.0
Large lakes (ha)	>4.3 >2.5	>7.5 >5.0	>5.0 >5.0
Small wetlands (ha)	0.3 to 2.3 0.2 to 1.5	0.3 to 5.0 0.0 to 5.0	0.4 to 3.5 0.2 to 3.5
Large wetlands (ha)	>2.3 >1.5	>5.0 >5.0	>3.5 >3.5

Other Boreal and Rocky Mountain jurisdictions featured: three size classes of streams, two size classes of lakes, and two size classes of wetlands.

Alberta's guidelines differed from those of other Boreal and Rocky Mountain jurisdictions. Alberta applied only two size classes to streams and the classes are generally larger than other jurisdictions with a boundary between size classes at 5 m. In comparison to other Boreal and Rocky Mountain jurisdictions, Alberta's guidelines lacked a specific delineation for smaller size classes for streams. For other Boreal jurisdictions, the mean size for small streams was 0.4 to 2.0 m while in the Rocky Mountains the mean size was 0.2 to 3.0 m. Similarly, the size criteria for lakes in Alberta were much larger than other Boreal or Rocky Mountain jurisdictions. Lakes were either 4 ha (fishbearing) or 16 ha (non-fishbearing). Other Boreal or Rocky Mountain jurisdictions set the lower size limit for defining lakes at ~1 ha. Wetland classifications are currently lacking in Alberta's guidelines. Other Boreal and Rocky Mountain jurisdictions recommended two size classes of each.

A number of jurisdictions based criteria on features other than waterbody size. These include navigability, stream order, and floodplain width.

A number of jurisdictions do not use channel width or surface area as a metric for size. In these jurisdictions, channel width or surface area is only indirectly part of classification. West Virginia and Saskatchewan base their classifications on stream order while Wisconsin bases classification on navigability. The Northwest Territories emphasizes the terrestrial and riparian interface as well as floodplain width to classify streams. A number of maritime jurisdictions, i.e. New Brunswick, Newfoundland, and Nova Scotia, base their classifications on mapping units such as width designation on 1:50,000 maps.

Like slope, jurisdictions added width (~142%) to baseline buffers when fishbearing bodies were encountered. Jurisdictions that do not have fishbearing modifiers compensated by having wider baseline buffers.

4.3.7 Fishbearing Waterbodies

Like the guidelines for slope, jurisdictions that do not have specific guidelines for fishbearing waterbodies utilize wider baseline buffers than those that recognize fishbearing waterbodies. In jurisdictions with fishbearing modifiers, the mean (S.E.) baseline widths for non-fishbearing streams was 19 m (5), however, the mean increased to 46 m (6) for streams with fish. The overall increase in buffer width was about 142%. For jurisdictions that did not explicitly consider fishbearing criteria in their guidelines, the mean (S.E.) baseline width was significantly wider at 29 m (3) (ANOVA, $P<0.05$). Thus, there appears to be some compensation in jurisdictions that do not explicitly consider fish in their guidelines. However, this "additional" width was lower than jurisdictions that explicitly accounted for fishbearing streams.

Alberta does not have explicit guidelines for fishbearing waterbodies (except small lakes) but its current guidelines covers a similar or greater amount of width as Boreal and Rocky Mountain jurisdictions that have fishbearing guidelines.

In Boreal jurisdictions without guidelines for fishbearing streams, the mean (S.E.) buffer width for large streams was 41 m (9) while for jurisdictions with fishbearing modifications the baseline buffer width was 26 m (4) (ANOVA, $P<0.05$). Buffer widths for this latter group increased to 69 m (11) when streams were classified as fishbearing, i.e. an increase of about 165%. Rocky Mountain jurisdictions with fishbearing guidelines recommended a mean (S.E.) buffer width of 35 m (8) when fish were present but a mean buffer of 16 m (4) when fish were not present. This was an increase of about 118%. Rocky Mountain jurisdictions without fishbearing guidelines had baseline buffer widths comparable to (30 m (15)) those with guidelines.

Alberta's current guidelines do not explicitly recognize fishbearing streams. However, the current set of buffer widths for large and small streams in Rocky Mountain and large streams in the Boreal are equivalent to jurisdictions that use specific guidelines for the presence of fish.

4.3.8 Human Water Supply and Aesthetics

Jurisdictions that did not have modifiers for human drinking water/aesthetic qualities maintained wider baseline buffers than jurisdictions that modified buffer widths on the basis of these factors.

Alberta's baseline guidelines are as wide or wider than those jurisdictions with drinking water guidelines.

Like slope and fishbearing streams, jurisdictions that do not have modifiers for drinking water recommended wider buffers than the baseline buffers of jurisdictions that have modifiers for human water supply, i.e. drinking water. Amongst the ten jurisdictions that recognized human consumption in their guidelines, the mean (S.E.) buffer width was 42 m (4) for waterbodies that served as drinking water but declined to 13 m (2) for comparable waterbodies not used for drinking water. Jurisdictions that have no explicit guidelines for drinking water utilized an intermediate baseline width of 33 m (13).

Only three Boreal (n=1) and Rocky Mountain (n=2) jurisdictions had guidelines for drinking water. Due to the small sample size, we pooled all these jurisdictions for analysis. In jurisdictions with consumption guidelines, the mean (S.E.) buffer width was 65 m (42) for waterbodies used for drinking water but was only 16 m (7) for waterbodies not used for drinking water. The mean (S.E.) buffer widths for Boreal and Rocky Mountain jurisdictions that did not have specific guidelines for waterbodies for drinking water was 44 m (8). Alberta's current guidelines for large permanent streams and lakes are equivalent to or wider than either those jurisdictions with or without human consumption guidelines.

4.3.9 Patterns of Selective Harvest

Most jurisdictions including Boreal and Rocky Mountain ecoregions permitted some harvest within designated riparian areas.

About 80% of all jurisdictions allowed some harvest within buffers. However, there was no difference in the mean buffer widths of jurisdictions permitting harvest (mean (S.E.) 27.4 m (2.9)) over those that did not allow harvesting (34 m (6)). Unlike the previous factors such as slope, fishbearing, or human consumption, no additional buffer width was added to areas that permitted some harvest within buffers.

This pattern continued when reviewing Boreal and Rocky Mountain jurisdictions (excluding Alberta). Amongst Boreal jurisdictions, 66% (8/12) allowed partial harvest of buffers while amongst Rocky Mountain jurisdictions, 75% (6/8) allowed partial harvest. Neither region exhibited any significant differences between harvested and non-harvested buffer widths. The mean (S.E.) widths for were 47 m (11) and 35 m (8), respectively, for harvested and non-harvest Boreal jurisdictions. Rocky Mountain jurisdictions had mean (S.E.) widths of 31 m (9) in harvested buffers and 38 m (23) buffers in non-harvested buffers.

Usually this harvest was restricted to less than 50% of trees and has some provisions to prevent "hi- grading" of larger trees.

Table 18 summarizes some of the guidelines from different jurisdictions. The general rate of removal in partial harvest systems was generally less than half of the merchantable trees in the buffers. Metrics used to measure level of

Table 18. Examples of partial harvest retention guidelines for riparian areas on medium and large channel streams from jurisdictions in Canada and the United States. RMA = riparian management area

Jurisdiction	Waterbody Type	RMA (m)	Minimum Retention
Alabama	Perennial stream (no wildlife objective)	11	50% cover
Arkansas	Permanent stream <7% slope	11	Minimum of 50 square feet basal area per acre. Harvest by individual tree selection
California	Class I Stream <30% slope	23	50% overstory and 50% understory retention
Connecticut	All waterbodies	30	Some harvesting activities are permitted. Final plans are made at the county level
Delaware	Open bodies of waters, streams (0-10% slope)	15	60 square feet per acre or 60% cover overstory; divided evenly amongst size classes
Florida	OFW, ONRW, and Class 1 waters (potable)	61	50% of canopy tree basal area
Georgia	Perennial streams, <20% slope	12	50% cover
Hawaii	Intermittent streams, permanent streams, and open bodies of water - slightly erodible soil, 0 - 5% slope	11	Partial harvesting acceptable to 50% of original crown cover; 50 square feet of basal area per acre, evenly distributed across stream management zone. No harvesting along stream banks
Idaho	Class I Stream (used for domestic water supply or by fish)	23	75% of shade
Illinois	Perennial streams/lakes (navigatible) 0% slope	15	Harvesting should be done so as to purposefully regenerate site while maintaining adequate vegetative cover to protect site
Indiana	Perennial streams 20 – 40 feet (>40% slope)	32 to 50	50% cover, no cutting along streambank
Iowa	Stream 20 - 40 feet	23	Minimize harvesting within stream management zone
Kansas	All waterbodies	8	75% of streamside forest.
Kentucky	Perennial cold-water aquatic habitat	18	75% of canopy tree
Louisiana	Perennial streams >20 feet wide	30	Extraction permitted provided other stream management zone goals are met
Maine	Tidal waters or flowing waters downstream from the point where such waters drain 50 square miles or more (P-SL1) and Resource Protection Districts adjacent to a great pond (>10 acres)	15	Forest harvest is permitted but no clearcutting permitted. Harvesting activities may not create single openings >14, 000 square feet in the forest canopy
Maryland	Permanent streams 0% slope	15	Partial harvest to 60 square feet per acre of trees >6 inches diameter at breast height
Massachusetts	All waterbodies and certified vernal pools	15	50% of overstory basal area

Jurisdiction	Waterbody Type	RMA (m)	Minimum Retention
Massachusetts	Streams >0.25 feet bank to bank; Ponds >10 ha; Designated scenic rivers; Outstanding Water Resources including tributaries; 0% slope	15	50% of overstory basal area
Minnesota	Designated trout streams and associated tributaries	46	60 square feet per acre
Mississippi	Perennial stream, 0 - 5% slope	9	50 square feet per acre, selective harvest only
Missouri	Streamside zone (land and vegetation areas adjacent to perennial and intermittent streams, caves, springs and lakes which require special consideration); 0% slope	8	Harvesting permitted throughout streamside zone but minimize soil damage and leave shade trees
Montana	Class 1 Stream <35% slope	15	50% of trees >8 inches diameter at breast height on either side of 10 trees per 100 feet which ever is greater
Nebraska	Perennial stream 20 - 40 feet wide	23	Forest harvest is permitted but single tree selection cutting is recommended
New Jersey	Permanent streams 0% slope	15	Partial harvest to 60 square feet per acre of trees >6 inches diameter at breast height.
New Hampshire	Protected waterbody	46	50% of the basal area; harvest must be distributed evenly amongst all tree sizes.
North Carolina	Perennial (<30 feet width) or intermittent stream, perennial waterbody,	15	75% of pre-harvest shade on stream channel
North Dakota	Stream (channel width 20 - 40 feet)	23	Limit harvesting to only problem trees.
Oklahoma	Waterbody	15	Retention of trees but no set guidelines. No harvesting along streambanks.
Oregon	Type F Streams	30	220-350 square feet per 1000 ft stream
Pennsylvania	High quality waters, perennial cold water streams, warm water streams, recreational and modified recreational rivers, and pastoral rivers	30	Area of special management, but harvesting permitted
South Carolina	Perennial and intermittent streams, lakes, and ponds, <5% slope	12 Primary, 0 Secondary	Primary zone - leave minimum of 50 square feet of overstory basal area per acre, Secondary zone - no log decks, avoid excessive rutting, do not expose more than 15% of mineral soil
South Dakota	Streams (intermittent and perennial)	15	Harvesting permitted within stream management zone, streamside trees left
Texas	Perennial streams, lakes, and intermittent streams	15	50% of original crown; 50 sq ft. per acre; harvesting evenly distributed over riparian management zone
Utah	Class I <35%	23	Selective tree removal; no clearcutting; leave hardwoods, unmerchantible conifers and shrubs.

Jurisdiction	Waterbody Type	RMA (m)	Minimum Retention
			No harvesting on streambank
Vermont	Stream slope 0 - 10%	15	Only light thinning or selective harvesting should occur
Virginia	Perennial stream	15	50% of crown cover, harvesting limited to cable and winch systems
Western Washington	Stream >10 feet Site 1	200	Core buffer no harvest no activity 15 meters. Inner Zone meeting basal area requirements or leaving trees close to water. Outer zone 20 trees per acre
Eastern Washington	Stream >15 feet Site 1	130	Core buffer no harvest no activity 10 meters. Inner Zone 21 meters some selective depending upon species and density; Outer zone = 9 meters 10 to 15 tree per hectare
West Virginia	Larger creeks and rivers	30	50% of trees in the stream management zone
Wisconsin	Lakes and Navigable Perennial Streams	30	60 square feet per acre of trees >5 inches diameter at breast height
Wyoming	Waterbody (0 - 20%)	0 to 30	Harvesting permitted; no specific targets
British Columbia	Stream S1(channel width >20 meters, fish stream)	70	Riparian Reserve Zone (no harvest) 20 meters; Riparian Management Zone (constraints to forest harvest applied) 50 meters
Manitoba	Class I Stream (A small stream with a gross drainage area of >50 square kilometers)	100	Selective harvest only where buffer management approved.
New Brunswick	Single Line Stream	30 to 100	70% of trees.
Ontario	Coldwater Streams (Slope is 0 - 15%)	30	Harvest Options include: no harvesting, selection cutting on restricted basis, avoid damaging banks, keep debris away, avoid damaging banks, keep debris away, avoid occurrence of erosion, maintain shade on both sides)
Quebec	Peat bog with a pond, a swamp, a marsh, a lake or a permanent watercourse	20	Notwithstanding the foregoing, when harvesting the trees, the permit holder shall not reduce the number of standing live trees per hectare to less than 500 trees of all species having a diameter of 10 cm or more. Cutting with regeneration and soil protection and strip cutting with regeneration and soil protection are nevertheless prohibited in the buffer strip
Yukon	Stream S1 (channel width >20 m, fish stream)	70	Riparian Reserve Zone (no harvest) 20 meters; Riparian Management Zone (constraints to forest harvest applied) 50 meters
Prince Edward Island	Watercourse or designated wetlands <9 % slope	20	Cut only 1/3 of trees in each of two size classes 10 to 30 centimeters and >30 centimeters; selection harvest. 2 hectares with. 1 hectare between cuts

retention included; % cover, % shade over the stream, and basal area. The minimum amount of basal area retained depended on forest type but values of 50 or 60 $ft.^2$ per acre were commonly applied particularly in eastern jurisdictions. In all jurisdictions, the area immediately adjacent to the streambank or trees with roots showing in the streambank could not be harvested. In some jurisdictions, this was extended to a wider no harvest zone around the streams. For example, Eastern Washington state had three zones of activity a core, inner, and outer zones. No harvest was allowed within the core zone. Harvest was permitted in the other two zones but the prescription options were more permissive in the outer zone. Further restrictions on the partial harvest of riparian areas included requirements to disperse the harvest over the full size range of trees, criteria for the maximum opening permitted, and distribution of trees along the streambank.

4.4 Discussion

Alberta has relatively wide baseline buffers compared to the rest of temperate North America.

Alberta's buffer widths were amongst the widest in Canada and the United States. With the exception of intermittent streams and wetlands, Alberta's classifications were about twice as wide as the average from other jurisdictions in Boreal and Rocky Mountain ecoregions. If the width of buffers from other jurisdictions were used as a standard for riparian protection, then the current recommended buffer widths on small and large permanent streams and small and large lakes are higher than the current standards from other jurisdictions. Whether they are adequately protecting riparian systems would require extensive field evaluation and testing against alternative protection strategies.

Alberta's guidelines can be classified as relatively simple. They contained relatively few modifiers, instead, relying on wide baseline buffers to cover most local conditions.

In general, guidelines follow two types of strategies. Some jurisdictions utilized relatively simple guidelines that did not include modifying factors in their formulation. Instead, these jurisdictions tended to recommend buffers that were wider than jurisdictions using modifiers. The primary advantage to this type of strategy lies in its relatively simplicity. The number of categories and the effort required in detailing the qualitative and quantitative nature of modifying factors is reduced. Broad, inclusive guidelines minimize the risks associated with a wrong classification. For example, if a stream is fishbearing but was not classified or misclassified, the intermediate buffer widths associated with broad guidelines would likely produce less of an impact. Hence, this strategy tends to minimize the risk across the landscape. However, the primary disadvantage is that intermediate width buffers may not be wide enough to fully protect fishbearing streams. Since specific guidelines on fish-bearing streams were generally wider, proper identification and application of the appropriate buffer widths would provide less risk through the application of wider buffers.

If Alberta considers implementing modifiers, the most commonly added would be fishbearing in the Boreal and fishbearing and slope in the Rocky Mountain region.

Amongst the most common modifying factors were waterbody type, slope, waterbody size, fishbearing, human water supply, drainage basin area, shoreline forest management, and shoreline vegetation. Modifiers were often formulated in specific combinations. Fishbearing was commonly added to other factors such as waterbody size, slope, or waterbody type. Other Boreal and Rocky Mountain jurisdictions commonly added presence of fish, shoreline vegetation, and slope as modifying factors. Currently, Alberta's guidelines are wide enough to cover many of the potential modifiers for large permanent streams and lakes. Application of fishbearing and slope buidelines similar to these in other jurisdictions would reduce the overall area of buffers.

Partial harvest allows for the introduction of some anthropogenic disturbance into riparian areas but is still done with the underlying paradigm of no impact on aquatic systems.

Most jurisdictions allowed for some harvesting within riparian zones. This followed the probabilistic model of the riparian ecotone (Ilhardt *et al.* 2000). In general, the total amount of harvest allowed did not exceed 50% of the basal area or canopy cover. In addition, the harvest is often dispersed amongst sizes of trees to prevent removal of high quality genetic stock, i.e. hi-grading, and large diameter trees within riparian areas after harvest. Types of harvest allowed within riparian areas included; single tree selection, group selection, and zoned harvest. Single tree selection was the most spatially dispersed of these options. It retains a relatively even canopy, shade. It also had the advantage of "hiding" the removal of trees more than the other forms of harvesting. However, disturbance to ground cover maybe more widespread because of the greater area accessed throughout the riparian area. In some jurisdictions, group selection was permitted but was limited to a maximum opening size. As an example, Maine allows openings by a maximum of 14,000 ft^2. In all cases, removal of trees whose roots are exposed in the shoreline or trees lining the shoreline is not permitted. Placement of two or more special management zones around waterbodies is the closest approximation to a probabilistic definition of a riparian ecotone. A special extension of this rule is the application of a wider no entry zone along the shoreline, which is then buffered by a zone for which harvesting in permitted.

In Alberta, two areas that require more clarification are intermittent streams and wetlands.

Given the emphasis in other jurisdictions, Alberta needs to clarify the delineation and riparian areas surrounding intermittent streams and wetlands. Further development of guidelines for intermittent streams may be warranted since these systems are the headwaters for larger, permanent downstream waterbodies. The recognition of this connectivity forms the basis for their protection in most other jurisdictions. In this regard, many jurisdictions did not distinguish the difference between permanent and intermittent streams. Alberta's current guidelines require an on-site call by personnel laying out cutblocks. The difficulty lies in the attempting to define the permanency of flow on the basis of channel characteristics. Snow melt puts flowing water in most low points in the boreal north. The retention of water by these systems is variable within and between years depending on snow pack and annual rainfall. There are a wide variety of channel types ranging from gravel bottoms and unvegetated channels to grassy or shrubby depressions. Also, connectivity is often not considered since the classification is done at a relatively small scale, i.e. harvest polygons. To address issues such as connectivity planning and stream designation should occur at larger spatial scales. Identification and classification using stream order as well as classification of local site factors such as relative channel permanency maybe a good first step. Large-scale planning requires the availability of accurate, universally accepted hydrological maps. The experience in this project has been that these maps are lacking for most of boreal Alberta.

Wetlands are a second waterbody that merits further consideration. This was a broad category that included bogs, marshes, sloughs, fens, and peatlands. These structures should not be considered as a single classification because their ecological structure, function, and associated biota are different (reviewed in Vitt *et al.* 1998). Hence, guidelines should reflect these differences. Most jurisdictions have not incorporated a great deal of ecological information in formulation of guidelines on wetlands. In Alberta, the boreal landscape is dominated by wetland complexes but our knowledge of impacts through forestry or other anthropogenic activities is relatively poor. Fortunately, there are number of studies underway in Alberta that will aid in evaluating the impact of harvesting on wetlands complexes (e.g. HEAD Project 2002).

4.5 Summary of Key Findings

- Canada has wider buffers for small and large permanent streams and small and large lakes, and water sources than the United States. No differences were found for intermittent and ephemeral streams.
- Recommended buffer widths in Boreal jurisdictions were generally wider than other jurisdictions. In contrast, Rocky Mountain, Pacific, Northeast, and Midwest jurisdictions were intermediate while the Southeast jurisdictions had the narrowest buffers.
- Alberta's riparian management areas are wider than most other jurisdictions for most waterbodies except for intermittent streams.
- Most jurisdictions (>78%) use at least one modifying factor to differentiate the riparian area associated with a waterbody. The most common modifying factors to riparian management areas were (ordered form highest to lowest) waterbody type, slope, waterbody size, fish bearing, human water supplies/aesthetics, drainage basin area, shoreline forest management, saltwater flow, shoreline vegetation, upstream of fishbearing waters, downstream sediment threat, and flow rates.
- Jurisdictions that do not have specific guidelines for modifiers (e.g. slope, presence of fish, and human drinking water) usually have baseline buffers that are wider than the baseline buffer widths of those jurisdictions with specific guidelines. However, the later group has wider buffers when the modifiers are present.
- Alberta's guidelines are relatively simple, generally focusing on waterbody type and size. Other Boreal and Rocky Mountain jurisdictions have presence of fish, shoreline vegetation, and slope as modifiers. However, the widths of buffers in Alberta are equivalent or greater than other jurisdictions (including Boreal and Rocky Mountains) that maintain specific guidelines for these other modifiers.
- A relatively high percentage of jurisdictions (>80%) permit selective harvest within riparian management areas. Usual rates of harvest were less than 50% of merchantable trees and usually with size restrictions to prevent "hi-grade" harvesting, i.e. selection of largest trees. Restrictions were also placed on shoreline harvesting and the permissible size of openings.
- Intermittent streams and wetlands require the development of protection strategies in Alberta.

5.0 NATURAL DISTURBANCE-SUCCESSION: AN ALTERNATIVE RIPARIAN MANAGEMENT PARADIGM

5.1 Introduction

Recently, forest management practices have been moving from a single or multiple-use management approach to an ecosystem management approach, in which ecological principles are used to integrate management practices for both timber production and maintenance of biodiversity (Hansen *et al.* 1991, Cissel *et al.* 1999, Fries *et al.* 1997, Armstrong 1999). The intent of this approach is use current ecological understanding and principles to sustain a healthy ecosystem that can provide a balance of commodity production and conservation of species diversity (Swanson and Franklin 1992, Hansen *et al.* 1991). One premise related to ecosystem management suggests that biodiversity will be used in a managed landscape if natural conditions are provided or the landscape is maintained within its natural range of variation (Swanson *et al.* 1994, Fries *et al.* 1998).

Natural disturbances can potentially serve as a model for managed landscapes.

One method of maintaining a landscape within its range of natural variability is to develop management practices based on the natural disturbance regime (Swanson *et al.* 1994, Attiwill 1994, Swanson and Franklin 1992). This method assumes that the native biota have evolved and adapted to natural disturbance regimes and if their environment is maintained within the range of these conditions then there is an increased potential to maintain biological diversity (Hunter 1993, Swanson *et al.* 1994). Aspects of natural disturbance that can be characterized and incorporated into harvest landscapes include patch size distribution, spatial distribution, shape, frequency or rate, and severity (Hunter 1993, Swanson *et al.* 1994, Fries *et al.* 1998, Armstrong 1999, Bergeron *et al.* 1999, Cissel *et al.* 1999). Some aspects of fire pattern such as burn rates (Armstrong 1999), live residuals (Eberhart and Woodend 1987, Delong and Tanner 1996, Ontario Ministry of Natural Resources 1997, 2001), deadwood (Lee *et al.* 1997), and size and distributions of disturbances (Cumming *et al.* 1996) have been described for boreal landscapes. Currently within Alberta, many forest companies incorporate some aspects of natural disturbance-succession (NDS) into their forest harvest practices (Alberta Forest Conservation Strategy Steering Committee 1997, Alberta Pacific Forest Industries Inc. 1996, Daishowa-Marubeni International Ltd. 2001). The current implementation of NDS is generally limited to small set of upland stand-level parameters including cutting to natural stand boundaries, leaving inblock residuals, one-pass rather than multiple-pass harvest systems, and larger cutblocks. The NDS model has not been extended to broader large, landscape-level planning (Lee *et al.* 2002b) or assessed with respect to riparian area management.

Natural disturbance patterns specific to riparian areas could be used to guide riparian management.

If riparian areas were managed under a NDS paradigm, guidelines for streamside vegetation would likely be radically different than the current protection/preservation paradigm (Alberta Sustainable Resource Development 2002; also see Section 2.0). This NDS management philosophy accepts that disturbance-succession is a part of the temporal and spatial dynamics associated with forested landscapes and that riparian zones are part of that dynamic landscape. Therefore, in terms of regulations for riparian zones, the NDS approach would differ from a protection/preservation philosophy primarily by allowing for variable buffer widths and streamside harvesting.

Some evidence suggests lower of wildfire disturbance in riparian areas.

One aspect of natural disturbance, which could apply to riparian buffer design in managed landscapes, is the distribution of live residual forest in riparian areas following wildfire. Under the NDS paradigm, the pattern of live residual forest around streams would serve as a template for treed riparian buffers left after harvest. The current literature arrives at no general consensus regarding the extent to which boreal riparian areas are affected by wildfire. One argument holds that riparian areas are unique environments in which fire is less frequent than surrounding upland areas (Rowe *et al.*1974, Timoney and Robinson 1996). The influence of topography on microclimate and fuel moisture may impede the movement of high-severity fires into valley bottoms (Gregory *et al.* 1991). On a landscape scale, water features are often considered to be effective fire breaks, limiting or stopping the spread of fire (Foster 1983, Dansereau and Bergeron 1993). For example, in northeastern Alberta, the distance to waterbody was related to the fire cycle length of a forest (Larsen 1997). Within the Peace River lowlands of northeastern Alberta, Lacate *et al.* (1965) observed a discrepancy in age structure of forests within riparian areas and attributed this to less frequent fires in alluvial lowlands adjacent to the river compared to landforms occurring further from the river. Further, the size of the water course may be an important factor in influencing the extent of disturbance by wildfire (Naiman *et al.* 1993). For example during the large Yellowstone fires of 1988, it was common for lower order watersheds to completely burn, whereas higher order watersheds only partially burned (Minshall *et al.* 1989). If the area surrounding streams is rarely disturbed and therefore not adapted to disturbance, then NDS patterns should be similar to the current protection/preservation approach; consequently, allowing anthropogenic disturbance within the riparian zone may be detrimental to the riparian ecosystem. In this case, a “no touch” buffer zone around streams may be an appropriate management practice under an NDS philosophy.

Other studies indicate riparian areas are frequently disturbed by wildfire.

However, evidence from broad-scale studies indicates that disturbance rates in riparian areas are similar to the surrounding upland. Based on available Alberta Vegetation Inventory (AVI) data, Burgess (1997) compared the proportions of old and young forests surrounding small and medium sized lakes to upland forest in northeastern Alberta. He demonstrated that there was no significant difference in the proportions of young and old forests for riparian and upland sites at increasing distances from lakeshores. In a large spatial study, Cumming and Pelletier (1995) described the relationship between fire frequency and the proportion of riparian area for a number of townships in northeastern Alberta. Results of this study confirmed the relationship between fire frequency and forest type, but the correlation between fire frequency and the amount of buffered riparian area was very low. Furthermore, calculation of fire intervals for a riparian area within the Peace River lowlands in northeastern Alberta, using 45 years of fire data found that 24.2% of the forested land burned (Timoney *et al.* 1997). This implies an annual burn rate of 0.54%, which is within the range (0.32% to

2%), albeit the low end of the range, of annual burn rates calculated for the boreal forest in a number of different studies (see Armstrong 1999).

Although results of these broad-scale studies suggest that riparian areas have little influence on natural disturbance patterns, the resolution of the data used may not have been adequate to detect the influence of waterbodies on the surrounding or adjacent landscape, particularly with respect to smaller waterbodies. Furthermore, when studies use timber harvest vegetation maps (i.e. AVI) to define polygons in the landscape, as the case with Burgess (1997), much of the small-scale spatial variation is "rolled-up" into the larger stand classification. Hence, the smaller residuals or narrower strips of residual riparian vegetation are often classified with upland vegetation polygons. Despite the research to date, the burn pattern in riparian areas after wildfires in the boreal forest remains unclear. A description of the extent that wildfire affects riparian areas and how the pattern varies with stream type and distance from stream is required before a NDS based model can be assessed for riparian area management.

5.2 Objectives

The objectives for this section were:

1. To describe the pattern of natural disturbance in riparian areas, specifically by examining the distribution of live residual forest remaining after wildfire,
2. To determine if stream, vegetation, and terrain variables were important in explaining the pattern of live residual forest, and
3. To translate the pattern of live forest residuals into a practical harvest plan for simulation and assessment in riparian and upland areas.

This study provides a description of the wildfire pattern for a watershed in northern Alberta and suggests how this information could be incorporated into riparian management.

The description and explanation of the pattern of live residuals, which constituted the first two objectives of this component, are summarized below. A more detailed description of the methods, results, and discussion are provided in Section 9.1 (Appendix A). Although not completed as part of the present component, the NDS model for a harvest plan for riparian management (Objective 3) was designed for simulation on an actual landbase; results of this simulation were to be compared with alternate riparian management scenarios in order to provide a relative comparison of ecological, economic, and social impacts of various management scenarios.

5.3 Description of Live Residual Patches Remaining after Wildfire

For the purpose of describing the residual pattern in riparian areas mapped live tree residuals (unburned patches of forest) within a large, single fire (DP2-9-80). The fire burned 36,000 ha in 1980 within the Notikewin watershed located (56° N, 118° W) approximately 130 km northwest of Peace River, Alberta. It extended over the Lower Foothills and Dry Mixedwood subregions (Strong and Leggat 1992). Due to the large area of the fire and the focus of this study on riparian areas, the study was limited to forested areas within 500 m of a watercourse. Digital AVI and watercourse coverage for the study area was provided by Daishowa-Marubeni International Ltd. (DMI). Streams were classified from aerial photographs as intermittent, small permanent, or large permanent; classification was based on stream size, apparent flow duration and channel development (Alberta Sustainable Resource Development 1994).

Live tree residuals (unburned islands) within the post fire study area were mapped.

Within the study area, live tree residuals were delineated and mapped from 1989 post fire aerial photographs at a scale of 1:20,000. Live tree residuals were identified as having canopy mortality less than 75%. Residuals were classified according to dominant tree species (aspen, white spruce, pine, black spruce). The pattern of residuals was described with respect to stream type (large permanent, small permanent, intermittent) and distance from stream. For each stream type, the percentage of area occupied by live residual forest was described in 20 metre intervals from 0 to 500 metres from streams.

More residuals occurred closer to larger streams.

Results indicate that for large permanent and small permanent streams the amount of residual was greatest streamside (30% and 16% residual area within the first 20m interval, respectively) and decreased with increasing distance from stream. For intermittent streams, however, the amount of residual was lowest streamside (12% residual area within the first 20m interval) and gradually increased with distance from stream. The area and number of residuals per kilometer of stream length (within 500m of stream) was compared among stream types. Large permanent streams had the highest amount of residual area per kilometer of stream length (22.4 ha/km), followed by intermittent (10.4 ha/km) and small permanent streams (5.5 ha/km). These results are relatively similar to the number of residuals per kilometer of stream length. Within 500 m of streams, large permanent streams had 13.0 residuals per kilometer of stream, intermittent had 4.6 residuals and small permanent had 3.6 residuals per kilometer of stream.

5.4 Explanation for the Pattern of Live Residuals Patches

Amount of residual was influenced by stream size, forest type and height, and distance from stream.

The purpose of this objective was to determine if there were landscape features or variables that could be used to explain the pattern of residuals. This information could be useful in placement of harvest retention over the landscape. The choice of landscape variables was made based on availability of digital datasets (i.e. stream data, digital elevation model, forest cover data) and potentially relevant variables based on other studies. The variables included stream (stream type, distance from stream), forest cover (pre-fire forest type, pre-fire canopy height) and terrain variables (elevation, slope, aspect, curvature). For this objective, the study area (as described in Objective 1), was divided into 20 m by 20 m grid cells. For each grid cell, the proportion of residual, distance from stream, type of nearest stream, pre-fire forest type, pre-fire canopy height, elevation, aspect, curvature, and slope was recorded. Grid cells were grouped based on the nearest stream type.

A generalized linear model was developed for each stream type (large permanent, small permanent, and intermittent streams) and the amount of variation in proportion of residual area (estimated by grid cells) explained by the landscape variables was reported. The total amount of variation in the proportion of residual area explained by the selected variables was 24.1% for large permanent streams, 10.9% for small permanent streams, and 9.0% for intermittent streams. The relatively low values for small permanent and intermittent streams indicated the fire or residual pattern was not clearly influenced by the landscape variables used in this study. However, of the variables considered in this study, those that explained the most variation, whether individually or interactively, were similar among stream classes. For all stream types these variables included canopy height, distance from stream, pre-fire forest type and for small permanent and intermittent streams north/south aspect. More details on the results of these analyses are provided in Appendix A.

5.5 Translation of Wildfire Residual Patterns into a Harvest Plan

A riparian harvest plan was developed based on the wildfire pattern.

The purpose of this objective was to use the information from the residual pattern (described above) to develop a NDS harvest plan that could be implemented by a harvest scheduling program, i.e. Woodstock/Stanley™. This harvest plan would constitute one of a number of harvesting scenarios that would be simulated on a digital landbase. Results of the simulations could then be compared and assessed to determine potential ecological, economic, and social implications of each harvesting scenario. In order for implementation into the harvest scheduling program the NDS model was required to be:

- as simple as possible, since complex rules would significantly increase computer processing time (i.e. division of the landbase into a minimum number of groups) and be difficult to formulate for the current suite of models,
- nonspatial, since the harvest scheduling program is essentially aspatial, prescriptions were limited to the total percentage of residual within a stand with no residual size criteria or positional information,
- practical from an operational perspective (i.e. limited to merchantable species of a harvestable size – aspen, white spruce, and pine generally ≥17m in height),
- based on relatively static landscape variables that could be easily developed or were readily available (at an appropriate scale) from a forest planning perspective (i.e. existing digital data or easily derived data), and
- descriptive enough to represent the range of natural variability of the natural disturbance pattern.

For each harvest stand, the percentage of residual to be retained was assigned.

Based on these criteria and the data obtained from Objectives 1 and 2. The harvest plan was designed to assign or specify a percentage of a potential harvest area that would remain unharvested (i.e. residual) within each stand. The percentage of residual area would vary based on landscape variables which were found to be important in explaining or influencing the pattern of residual after wildfire (Objective 2) and included pre-fire forest type, canopy height, nearest stream type, and distance from stream. The development of the NDS model harvest plan from the natural disturbance pattern information is described below.

The percentage of residual varied according to stream type, distance from stream, stand size, and forest type.

The pre-fire forested stands within the study area (described in Objective 1) and their associated percentage of residual area (based on post burn maps) were categorized according to stream class, canopy height, distance from stream, and pre-fire forest type. The percentage of residual varied with stand size, therefore pre-fire stand size was also included as a variable. The stands within each stream class (intermittent, small permanent, large permanent) were grouped and the stands were further subdivided based on size, pre-fire forest type, and distance from stream. Since the harvest plan was designed to be feasible from an operational perspective, only forested stands with a canopy height ≥17 metres were included (any forested stand below 17 m in height was considered to be unmerchantable). The subdivision of stands within each stream class based on distance from stream, pre-fire forest type, and stand size. Correlations between percentage of residual and each variable was used to determine which variables were associated with the percentage of residual and to indicate how the stands should be further subdivided. For example, if the percentage of residual was correlated to distance from stream, then the stands were divided into distance categories. If there was no correlation between percentage of residual and distance from stream, stands were not subdivided into distance from stream classes. The subdivision process was somewhat subjective in order to ensure an appropriate sample size for each subgroup (i.e. N>20). For stands in all stream classes the percentage of residual area was correlated with stand size, therefore stands were limited to the largest size class possible while maintaining an appropriate sample size. In most cases stands included were ≥5 ha. Due to the small number of stands around large permanent streams the percentage of residual was compared between aspen and white spruce and found to be not significantly different, therefore these two vegetation types were combined into a single subgroup.

The percentage of residual was drawn from a probability distribution.

For most subgroups, many of the stands had a low percentage of residual. Therefore, frequency distributions of percentage residual were non-normal and generally skewed to the left. Thus, parametric statistics such as mean and standard deviation would not provide an adequate representation of the variability of percentage of residual within stands. The use of probability distributions has been suggested as an appropriate method to characterize natural landscape variability (Swanson *et al.* 1994; Landres *et al.* 1999; Strauss *et al.* 1989). Probability distributions have been used to characterize pattern of natural disturbance such as annual area burned (Armstrong 1999) and fire size distribution (Cumming 2001). For this study, probability distributions were generated to approximate the distribution of percentage of residual for stands within each canopy type (Table 19).

Application of this look-up table would consist of determining a) what stream type is the stand/harvest area closest to, b) what is the dominant forest cover type is of the stand slated for harvest, and c) if the stand is dominated by white spruce and is closest to an intermittent stream, is the stand greater or less than 100 m from the stream. Based on the results of this analysis the probability distribution and associated parameters (Table 19) we can

Table 19. Natural disturbance-succession model scenario lookup table. AW = trembling aspen, SW = white spruce, and PJ = Jackpine.

				Percentage of Residual				
Stream	**Forest Cover**	**Distance from Stream**	**N**	**mean**	**median**	**min-max**	**Probability distribution**	**Parameters**
Large permanent	AW/SW	all	23	28.5	17.5	0.0-99.5	Beta	(0.261, 0.654)
Small permanent	AW	all	28	42.8	41.2	0.0-97.1	Beta	(0.481, 0.643)
Small permanent	PJ	all	32	20.2	11.5	0.0-99.8	Beta	(0.293, 1.155)
Small permanent	SW	all	49	15.0	6.7	0.0-87.9	Beta	(0.289, 1.646)
Intermittent	AW	all	27	50.3	50.5	0.0-100.0	Beta	(0.497, 0.492)
Intermittent	PJ	all	48	26.2	13.6	0.0-99.2	Beta	(0.295, 0.832)
Intermittent	SW	<100m	21	8.4	1.5	0.0-56.2	Beta	(0.221, 0.395)
Intermittent	SW	$\geq$100m	61	14.5	4.9	0.0-100.0	Beta	(0.286, 1.681)

Notes:

- All stands used were $\geq$17m in height when height was correlated to % residual
- Stands used in calculations were in the largest size class possible (if there was a significant correlation between percentage residual and stands size) while maintaining an appropriate sample size (all stands were $\geq$5ha except for LP: $\geq$2ha, SP-AW: $\geq$ 3ha, and INT–PJ: $\geq$15 ha)
- There were no pine stands in the vicinity of large permanent streams

determine the percentage of residual data for stands in that subgroup. A random draw from the appropriate distribution determines the percentage of the stand that should remain unharvested (i.e. residual).

The harvest plan needs to be tested and assessed through simulation.

The degree to which the above harvest plan creates a natural disturbance pattern within a harvest landscape will need to be tested through simulation. There are some potential limitations in the dataset that may cause the simulated landscape to differ from the naturally disturbed landscape. One limitation is the relatively low number of canopy subgroups. For example, the distance from stream categories were often amalgamated to ensure that there were a sufficient number of stands from which to derive a probability distribution. Therefore the harvest scenario does not always describe the difference in residual due to different distance from stream classes. Similarly, the low number of stands around large permanent streams resulted in the grouping of aspen and white spruce stands. In most cases percentage of residual was correlated to stream size, therefore a larger sample size would have allowed division of stands based on stand size class.

Other landscape variables may be important in explaining the amount of residual.

Based on the amount of variation explained by the generalized linear models in Objective 2, it is clear that there may be other variables that may be important in explaining the pattern of residuals. Other potential variables, based on results of similar studies, may include physiographic region (Kushla and Ripple 1997), aspect (Rowe and Scotter 1973, Hadley 1994), and weather (Bessie and Johnson 1995), although weather variables are spatially and temporally variable and could not be used as static landscape variables. Similarly, suppression information may help to explain the pattern of residuals for fires that have been affected by fire suppression efforts. For the present study, although there were fire suppression efforts for the fire, there are no detailed records that could be used to account for variation in residual pattern.

Although a larger sample size and more explanatory variables would increase the resolution of the natural disturbance pattern, it would also result in a more complex harvest plan, which in turn would increase the computational time of the simulations and difficulty in formulation for harvest planning. In that respect, there needs to balance between accuracy of information and effort required for simulations.

Another potential limitation is that the results for the lookup tables were based on a single fire. The effect to which a single large fire captures the variability of all fires is unknown. A large fire would span a large geographical area and burn over many days, therefore encountering many different burning conditions, which would increase the variability represented by a single fire. It has been documented that fire pattern varies among regions, therefore it is important to use fire information derived from a specific region (Kushla and Ripple 1997).

The landscape level implications of the NDS paradigm could also be developed.

Another point to consider regarding utilization of the NDS harvest plan described above is that it applies to stand level prescriptions and only implies larger landscape level patterns of natural disturbance. A separate type of modeling/simulations would be required to apply a landscape scale NDS scenario. The impacts of selecting only certain aspects of natural disturbance to apply to a forest harvest landscape is unclear.

How Does the Natural Disturbance Model for Harvesting Differ from Natural Disturbance?

From an operational perspective, limited aspects of natural disturbance could be implemented under a NDS scenario.

Differences between the NDS scenario and other riparian management scenarios can be compared and evaluated through simulation prior to field implementation.

Although the overall amount of live tree residual may be comparable between natural disturbance and a harvesting system based on the NDS scenario, there are a number of differences between the two types of disturbances. These differences primarily result from the need for the NDS scenario to be feasible from an operational perspective. For example harvesting is limited to areas with merchantable forest, whereas wildfire affects forests of all ages and levels of productivity, although to varying extents. Also in wildfire affected areas all dead trees remain on site whereas harvesting removes this organic matter from the site, the short- and long-term effects on the carbon balance and soil properties due to this removal of organic material is unknown. Specific to riparian areas, the removal of trees streamside would affect the input of coarse woody debris. Other features occurring in a harvest landscape include logging roads and skid trails. The effect to which these differences in harvest based on the NDS scenario and the actual pattern of wildfire will affect the biota, the key premise behind proposing a NDS model, is unclear.

Although the simulations of the natural disturbance-succession management scenario were not completed as part of this project and therefore results are not available, we can provide some discussion, based on the literature, as to what the effects of the NDS harvest strategy may be on terrestrial and aquatic components of the landscape in comparison to a preservation/protection riparian management strategy. Studies describing the effects of forest harvest on waterbodies are discussed in Sections 2.0 and 3.0 of this report.

Based on the NDS scenario, impacts to aquatic systems would result from removal of streamside forest.

The primary difference for NDS is that harvesting would occur streamside along all stream types. The removal of streamside forest would likely have a number of impacts. One of the impacts of vegetation removal would be increased solar radiation to stream and a resultant increase in stream temperature (Cissel *et al.* 1999). Increased light levels and stream temperature are likely to cause an increase in stream productivity, invertebrates and a shift in algal communities (Cissel *et al.* 1999, Carlson *et al.* 1990). A second impact from the removal of streamside forest would be reduced input of coarse woody debris to streams (Cissel *et al.* 1999). The effect of this would last until streamside vegetation was mature enough to contribute coarse woody debris to the stream. The reduction in coarse woody debris may affect channel stability, pool formation, fish habitat, and sediment retention (Bragg and Kershner 1999). Thirdly, the natural disturbance-succession scenario does not consider slope, therefore steep slopes or streambanks could be harvested which may result in increased erosion, channelization, slope failure, and subsequent sedimentation of streams (Bragg and Kershner 1999, O'Laughlin and Belt 1995).

Both the NDS scenario and the preservation/protection scenario retain live residual trees, however the spatial distribution of the residual material is quite different.

At the landscape level, the protection/preservation paradigm provides a forested corridor along watercourses, which may provide important habitat or movement corridors for some species (see Section 1.2). In the natural disturbance-succession scenario the amount and distribution of unharvested forest would differ from the preservation/protection scenario. Instead of continuous strings of 100% forest retention along larger streams, there would be patches of intact forest, of various sizes, which would cover an average of 12 to 30% of the streamside area, depending on stream type. If the natural disturbance-succession scenario was integrated into areas other than immediately streamside (i.e. upland areas), there would be a high retention of live residual trees (averaging 8% to 50% of stand area). This high amount of residual would not likely exist in the upland areas under a preservation/ protection scenario, since the upland is where the harvesting is currently concentrated. Therefore the total amount unharvested forest may be similar between the two scenarios although the distribution of unharvested forest would be very different. Based on simulated comparisons of a natural disturbance management plan and a more traditional harvest plan (riparian buffers and intensive upland harvest) in Douglas fir forests of the Pacific Northwest region, Cissel *et al.* (1999) reported that the distribution of stand ages differed between the two scenarios. Within the traditional plan, stand age was spatially segregated within the landscape; old stands occupied low slope positions (riparian reserves) and young stands occupied the higher slopes (intensive plantations), there were fewer mature stands resulting from this scenario. However in the natural disturbance scenario, the allowance for some harvesting along streams (although higher levels of live tree retention were specified closer to streams) tended to disperse stands of varying ages across the landscape. Further results from this study concluded that mean patch size was greater in the natural disturbance based landscape compared to the traditional harvest plan. In the traditional harvest landscape the harvestable area was constrained by the riparian reserve areas, resulting in smaller patch sizes, increased edge density, and lower forest interior area compared to the natural disturbance scenario. The edge abruptness was also lower in the natural disturbance scenario due to the higher levels of green tree retention prescribed in this scenario (15%, 30%, or 50% tree retention).

Economic and social implications of a NDS scenario need to be assessed.

With respect to economic impacts, harvest plans based on NDS may be more costly and include higher costs for planning and implementation due to retention of green trees on site and lower amounts of volume harvested per area accessed (Cissel et al 1999). In addition to potential ecological and economic impacts of NDS, there are potential social impacts that require consideration. These may include the aesthetic values offered by riparian buffers strips for recreational users of lakes and streams. For example riparian buffers can visually mask upslope landuse activities such as forest harvest (O'Laughlin and Belt 1995).

5.6 Summary of Key Findings

- The natural disturbance-succession pattern, as measured by the amount of residual or unburned forest, varied with stream type. Large permanent streams were associated with higher amounts of residual compared to small permanent and intermittent streams.
- The amount of residual varied with distance from stream. For large permanent and small permanent streams the amount of residual was greatest closest to the stream and decreased as the distance from stream increased. However, the amount of residual around intermittent streams increased with increasing distance from streams.
- Terrain, forest and stream variables associated with the amount of residual included distance from stream, dominant forest type, canopy height, and slope aspect. The amount of variation explained by stream, terrain, and forest variables was greatest around large permanent streams (24%) compared to small permanent streams (11%) and intermittent streams (9%).
- Patterns of live treed residuals after wildfire and variables associated with this pattern were complex and highly variable but were comparable to findings from similar published studies.
- If riparian management was based on a natural disturbance-succession model it would be quite different than current guidelines. An NDS would result in a) removal of streamside forest around large permanent and small permanent streams, b) retention of streamside forest around intermittent streams, and c) retention of forest at distance from stream (i.e. upland) exceeding current buffer distances.
- Further research and simulation modeling are required to assess the economic, social, and ecological impacts of using a natural disturbance-succession model for riparian management.

6.0 OVERALL MANAGEMENT RECOMMENDATIONS

For three of the four major components in this report, we can make a number of management recommendations.

6.1 Defining Riparian

- No specific recommendations.

6.2 Quantitative and Qualitative Review of Ecological Literature

- There is little urgency based on the current available data to shift Alberta's 1994 riparian buffer guidelines either to be narrower or wider for most stream and lake classifications. Current guidelines fell within recommended widths for aquatic biota and habitat and terrestrial habitat.
- Set clear objectives for the management of terrestrial biota within riparian areas and integrating riparian buffers with upland activities to achieve management objectives.
- Continue to focus on the reduction of sediment transport by created by roads, skidding trails, and landings.
- Evaluate of the total cumulative disturbance on watersheds (Rocky Mountain and Boreal) to access the impact additional impacts of harvest and other disturbances on water yield, peak flows, and nutrient transport.

6.3 Quantitative Review of Temperate North American Guidelines

- If Alberta were to consider moving away from the current "blanket approach", guidelines could reflect four general areas of revision:
  1) Development of separate guidelines for Boreal and Rocky Mountain ecoregions.
  2) Addition of fish-bearing modifier to Boreal and fish-bearing and slope modifiers to Rocky Mountain ecoregions.
  3) Further development of guidelines for intermittent streams and wetlands for all regions.
  4) Establish permissible levels of riparian harvest within buffers.
- Development of silvicultural classification systems and practices for riparian harvest (if this is planned).

6.4 Natural Disturbance-Succession

- Due to the large spatial scale and long-term "footprint" of disturbance-succession management regimes, computer simulation of alternate riparian management scenarios and active adaptive management should be undertaken as initial steps in identifying potential impacts and benefits. No large-scale implementation is recommended at this time.

7.0 KNOWLEDGE GAPS

- Due to the lack of ecological data on most aspects of riparian ecology in Alberta ecoregions, there should be an establishment of long-term experimental watersheds in Boreal, Northeast slopes, and Southeast slopes regions.
- Establish inputs and responses of aquatic and terrestrial biota to short- and long-term changes in organic debris and stream temperature from partial and fully harvested and wildfire disturbed riparian areas.
- Continued evaluation on the relative importance of treed riparian buffers to aquatic invertebrates, fish, amphibians, birds, and mammals including successional changes after wildfire and harvest. This should be done in the light of clear hypotheses on habitat use, e.g. riparian corridors.
- If partial harvesting of riparian areas is to be undertaken, then silvicultural protocols and successional trajectories need to be established for regeneration of riparian areas.
- Develop riparian habitat classification. Characterize and quantify vegetation communities in riparian zones, i.e. the spatial and temporal juxtaposition of plant communities. This will provide a common terminology when discussing riparian areas.
- Evaluation and comparison of the variability in fire pattern within and among large fires is required to determine the range of natural variability with respect to fire pattern. It is unclear whether management efforts should focus on maintaining local patterns or whether patterns should be derived from "averaging" a large set of wildfires.
- There is a general discrepancy between the scale at which forest harvest is undertaken (stand-level) and the scale at which natural disturbance occurs (landscape-level). Therefore stand level prescriptions based on natural disturbance patterns need to be assessed or applied at a landscape scale to determine if the information is transferable between scales.
- More research regarding the association between residuals and topographic features may explain the spatial distribution of residuals. The extent to which this can be achieved will be dependent upon the availability of reliable topographic information at an appropriate scale.
- Natural disturbance-succession and protection/preservation through buffers are two possible management scenarios. There are likely to be others. Research should focus on development of other possible management scenarios based on other paradigms of ecological maintenance. These should be compared using simulation models applied to large spatial scales. Favourable models can be assessed, and further refined for field-based trials.

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9.0 APPENDICES

9.1 Appendix A: Step by Step Procedures Used to Develop a Natural Disturbance-Succession Harvest Plan For the Notikewin Watershed

Step 1: Determine what variables were important in explaining the percentage of residual using the results of the GLM's for each stream class.

In most cases these included, either alone or in combination with other variables: height, distance from stream, vegetation type, and terrain variables. Terrain variables were not included in the division process for stands since appropriate scale terrain information is not available for P1 and P2. In addition stand size was used as a variable since this varied in the stand data was not include in the GLM analysis since a constant grid cell size was used.

Step 2: Test correlations between percentage of residual and stand area.

If stand size was correlated with percentage of residual attempt were made to use stands with the greatest area since this would coincide with the size of harvest areas. This inclusion of larger stands was weighed against maintaining a sufficient sample size.

Step 3: Test correlations with canopy height using the reduced sample size (i.e. larger size class of stands).

Test correlation with height to see if there is a difference between <17m and $\geq$17m tree height. The height class of 17m was selected as the division point as it is the low end of the height range but >17m would still maintain a sufficient sample size and that stands less than 15m are less likely to be harvested (Notes from meeting with DMI on Nov 2, 2001). Histograms showed that most stands were between 16m and 19m in height. Even if there was no significant difference between height classes, stands $\geq$17m were used in further analysis (unless N for the $\geq$17m height class was very small).

Step 4: Test correlations with distance from stream using the reduced sample size (i.e. larger height class of stands, >17m).

Distance from stream was tested to see if correlated with percentage of residual. Tests were then done to determine if there was a difference in percentage of residual between 100m interval distance classes i.e.: <100m vs >100m; <200m vs >200m; <300m vs >300m; and <400m vs >400m.

Step 5: Define probability distribution for the percentage of residual in each stand class.

Based on visual assessment, the distribution type that best fit with the percentage of residual data was selected. In all cases the distribution chosen was a Beta probability distribution. Shape and scale parameters for the Beta distribution for each group were calculated from the stand data.

Table 20. Process of division/aggregation of stands around large permanent streams.

Variable	Outcome	Comments
AW (N=37)		
Stand size	Significant correlation with stand size (Spearman correlation coefficient 0.542; p=0.001). Significant difference between stands <1ha (N=19) and ≥1ha (N=18) (p=0.006)	Limit stands to ≥1ha (N=18).
Height	No significant correlation with height (Spearman correlation coefficient 0.045; p=0.794). No significant correlation with height when stands limited to ≥1ha (N=18) (Spearman correlation coefficient 0.059; p=0.816). Only one stand <17m when stands limited to ≥1ha.	Limit stands to ≥17m in height (N=17).
Distance	No significant correlation with distance from stream (Spearman correlation coefficient 0.205; p=0.225). No significant correlation with distance from stream when stands limited to ≥1ha (N=18) (Spearman correlation coefficient -0.159; p=0.529)	No division based on distance from stream.
SW (N=36)		
Stand Size	Significant correlation with stand size (Spearman correlation coefficient 0.448, p=0.006). Significant difference between stands <1ha and ≥1ha (p=0.016).	Limit stands to >1ha (N=24).
Height	No significant correlation with height (Spearman correlation coefficient -0.303, p=0.073). No significant correlation with height when limited to stands ≥1ha (Spearman correlation coefficient -0.068, p=0.752). Stands <17m N=3; stands <18 N=8; no significant difference between stands <18m and ≥18m (p=0.926).	No significant difference between height classes and small number of stands <17m (N=3) and <18m(N=8). Limit to stands ≥17m (N=21).
Distance	No significant correlation with distance from stream (N=24) (Spearman correlation coefficient 0.174, p=0.309). No significant correlation with distance from stream when limited to stands >1ha (N=24)(Spearman correlation coefficient -0.111, p=0.606).	No division of stands based on distance from stream.
AWSW(N=38)		
VegType	No significant difference between AW (N=17) and SW (N=21) (Z=-0.015; p=0.988).	Combine AW and SW ≥1ha and ≥17m (N=38).
Distance	No significant correlation with distance from stream for combined AW/SW group (Spearman correlation coefficient –0.078; p=0.641).	No division of stands based on distance for combined AW/SW class.
Stand Size	Significant correlation with stand size for stands >17m in height (Spearman correlation coefficient 0.526; p<0.001) and significant difference between stands <2ha (N=45) and ≥2ha (N=23).	Limit to stands ≥2ha in area (N=23).

Table 21. Process of division/aggregation of stands around small permanent streams.

Variable	Outcome	Comments
AW (N=191)		
Stand size	Significant correlation with stand area (N=191) (Spearman correlation coefficient 0.288; p<0.001). There are significant differences between all stand size classes <1ha vs ≥1ha; <2ha vs ≥2ha; <3ha vs ≥3ha; <4ha vs ≥4ha; and <5ha vs ≥5ha.	Limit stands to ≥3ha (highest Z of pairs of comparisons; N=34).
Height	No significant correlation with height (N=191) (Spearman correlation coefficient 0.06; p=0.410). When limited to stands ≥3ha there was no significant correlation with height (Spearman correlation coefficient 0.014; p=0.939) nor were there significant differences between stands <17m and ≥17m (p=0.366). (Differences between classes were not significant when limited to ≥4ha either).	Limit to ≥17m although no significant correlation with height.
Distance	No significant correlation with distance from stream (N=191) (Spearman correlation coefficient -0.05; p=0.491). When limited to stands ≥3ha there was no significant correlation with distance from stream (Spearman correlation coefficient 0.238; p=0.175), there was no significant differences between distance classes: <100m vs >100m (p=0.192); <200m vs >200m (p=0.128); <300m vs >300m (p=0.551) or <400m vs >400m (p=0.913). The greatest differences in means were between <200m (N=20) and >200m (N=14) (mean % residual = 32.5; 50.4, respectively)(Differences between distance classes were not significant when limited to ≥4ha either).	No division based on distance from stream classes.
SW (N=354)		
Stand size	No sign. correlation (Spearman coefficient 0.088; p = 0.098). No significant correlation when limited to ≥17m (Spearman correlation coefficient = 0.049; p=0.429)	When limited to stands ≥17m, no significant differences between stand size classes: <1ha vs ≥1ha, <2ha vs ≥2ha, <3ha vs ≥3ha, <4ha vs ≥4ha, or <5ha vs ≥5ha
Height	No sign. correlation (Spearmen coefficient 0.101; p=0.057) Signficant difference between <17m and ≥17 m (Mann-Whitney test; p=0.03)	Limit stands ≥17m (N=261)
Distance	No sign. correlation (Spearman correlation coefficient 0.088; p=0.098) No sign. difference when limited to stands ≥17m: Spearman correlation 0.049; p=0.429). When limited to ≥17m no significant difference between distance classes: <100m vs ≥100m; <200m vs ≥200m; <300m vs ≥300m; or <400m vs ≥400m	No division based on distance from stream class.
PJ (N=200)		
Stand size	Significant correlation with stand size (Spearman correlation 0.343; p<0.001). Significant differences (at 0.05 level) between all pairs of size classes: <1ha vs ≥1ha, <2ha vs ≥2ha; <3ha vs ≥3ha; <4ha vs ≥4ha; or <5ha vs ≥5ha with largest Z at 5ha (N=55)	Limit stands to ≥5 ha (N=55).

Variable	Outcome	Comments
Height	No significant correlation with stand size (Spearman correlation 0.109; p=0.126). When limited to stands ≥5ha there was no significant correlation with height (Spearman correlation coefficient 0.125; p=0.365). When limited to stands ≥5ha there was no significant difference between stands <17m and ≥17m (Mann-Whitney test p=0.521) nor between stands <18m and ≥18m (p=0.584).	Limit stands to ≥17m although no significant correlation with height.
Distance	No significant correlation with stand size (Spearman correlation -0.105; p=0.141). When limited to stands ≥5ha there was no significant difference with stand size (Spearman correlation coefficient –0.119; p=0.386). For stands ≥17m there was a significant difference between <100m and >100m although N=8 for <100m. No significant differences between other distance class comparisons (i.e. <200m vs ≥200m; <300m vs ≥300m; <400m vs ≥400m.	No division based on distance from stream class.

Table 22. Process of division/aggregation of stands around intermittent streams.

Variable	Outcome	Comments
<u>AW (N=398)</u>		
Stand size	Significant correlation (N=398) with stand size (Spearman correlation coefficient 0.255; $p<0.001$). Significant differences between stand size classes tested <3ha vs ≥3ha ($p<0.001$); <4ha vs ≥4ha ($p<0.001$); and <5ha vs ≥5ha ($p<0.001$). Stands ≥3ha (N=65); ≥4ha (N=36); ≥5ha (N=27). But since there is no splitting of group further, since no significant correlation with distance or height, use stands ≥5ha (N=27) since this is a sufficient sample size and there is a significant difference in % residual area between stands <5ha and ≥5ha.	Limit to stands ≥5ha (N=27)
Height	Significant correlation (N=398) with height (Spearman correlation coefficient 0.144; $p=0.004$). For stands ≥3ha there was no significant difference between <17m and ≥17m ($p=0.324$) or <18m and ≥18m. No stands <17m when limited to ≥5ha (N=27).	Limit stands to ≥17m (N=61) although no significant difference with <17m.
Distance	Significant correlation (N=398) with distance from stream (Spearman correlation coefficient 0.150; $p=0.003$). No significant correlation for stands <3ha and >17m (Spearman correlation coefficient 0.166, $p=0.202$, N=61). Significant difference between stands (N=61) <100m and >100m from streams although N=6 <100m from streams which would be insufficient for group). No significant difference between groups <200m vs >200m, <300m vs >300m, and <400m vs >400m from streams.	No significant correlation with distance for stands ≥5ha.
<u>PJ (N=992)</u>		
Stand size	Significant correlation with stand size (Spearman correlation coefficient 0.233; $p<0.001$). Significant difference between size classes tested <3ha (N=524) vs ≥3ha (N=468) ($p<0.001$); <4ha (N=615) vs ≥4ha (N=377)($p<0.001$); <5ha (N=688) vs ≥5ha (N=304) ($p<0.001$); <10ha (N=854) vs ≥10ha (N=138) ($p<0.001$); <15ha (N=914) vs ≥15ha (N=78) ($p<0.001$);	Limit stands to ≥15ha (N=78).
Height	Significant correlation with height (N=992) (Spearman correlation coefficient 0.116; $p<0.001$). Significant correlation with height for stands limited to ≥15ha (N=78) (Spearman correlation coefficient 0.408, $p<0.001$). Significant difference between stands <17m (N=30) and stands ≥17m (N=48) ($Z=-2.631$; $p=0.009$).	Limit to stands >17m in height (N=48).
Distance	Significant correlation with distance from stream (N=992) (Spearman correlation coefficient -0.069; $p=0.029$). No significant correlation with distance from stream when limited to stands ≥15ha (N=78) (Spearman correlation coefficient 0.107; $p=0.350$) and no significant difference between distance	No division into distance classes.

Variable	Outcome	Comments
	classes: <100m vs ≥100m (p=0.373); <200m vs >200m (p=0.956); <300m vs >300m (p=0.319); <400m vs >400m (p=0.856). No significant correlation with distance from stream when limited to stands ≥15ha and >17m in height (N=48) (Spearman correlation coefficient 0.204; p=0.165) and no significant difference between distance classes: <100m vs ≥100m (p=0.173); <200m vs >200m (p=0.483); <300m vs >300m (p=0.123); <400m vs >400m (p=0.686).	
SW (N=544)		
Stand size	Significant correlation with stand area (Spearman correlation coefficient 0.163, p<0.001). Significant difference between stands <1ha vs ≥1ha (p=0.002); <2ha vs ≥ 2ha (P=0.001); <3ha vs ≥3ha (p=0.008); and <5ha vs ≥5ha (p=0.03). No significant difference between stands <4ha and ≥4ha (p=0.068).	Limit stands to ≥5ha (N=119).
Height	No significant correlation with height (Spearman correlation coefficient 0.056, p=0.195). When limited to stands ≥5ha there was a significant difference between stands <17m and ≥17m (p=0.006). When divided into two distance classes <100m and >100m from streams there was no significant difference between <17m (N=9) and ≥17m (N=21) for stands <100m (p=0.272) but there was a significant difference between <17m (N=28) and ≥17m (N=61) for stands >100m from streams (p=0.009).	Limit to stands ≥17m (N=82)
Distance	Significant correlation with distance from stream (Spearman correlation coefficient 0.118, p=0.006). When divided into 100m distance classes and compared (N=544) there was significant differences between <100m vs >100m (P=0.005) and <200m vs >200m (p=0.006) but not significant between <300m vs >300m (p=0.081) or <400m vs >400m (p=0.333). When limited to ≥5ha (N=119) the correlation with distance was not significant (Spearman correlation coefficient 0.115; p=0.212) but there was a significant difference between <100m (N=30) and >100m (N=89) (p=0.046). There was no significant difference between other distance classes: <200m vs >200m (p=0.286); <300m vs >300m (p=0.720); or <400m vs >400m (p=0.778)	Split stands into two distance classes: <100m (N=21) and >100m (N=61).

9.2 Appendix B: Reference List of Guidelines from Different Jurisdictions in Canada and the United States

Jurisdiction	Reference
Alabama	Alabama Forestry Commission. Undated (accessed 2002). Alabama's Best Management Practices for Forestry. Alabama Forestry Commission, Montgomery, Alabama. 28 pp. http://www.forestry.state.al.us/publication/BMPs/BMPs.pdf
Alaska	Division of Forestry, Department of Natural Resources. 2000. Alaska Forest Resources and Practices. Division of Forestry, Department of Natural Resources, Anchorage, Alaska. http:/www.dnr.state.ak.us/forestry/pdfs/forprac.pdf
Arkansas	Arkansas Forestry Commission. Undated. (accessed 2001). 2.0 Streamside Management Zones. Arkansas Forestry Commission, Little Rock, Arkansas. 3 pp. http:/www.forestry.state.ar.us/bmp/smz.html
California	California Department of Forestry and Fire Protection. 2000. California Forest Practice Rules 2000. Resource Management, Forest Practice Program, California Department of Forestry and Fire Protection, Sacramento, Califorina. 230 pp. http://fire.ca.gov/forest_practice.html
Colorado	Colorado State Forest Service. 1998. Colorado Forest Stewardship Guidelines to Protect Water Quality: Best Management Practices (BMPs) for Colorado. 1998. Colorado State Forest Service, Colorado State University, Fort Collins, Colorado. 32 pp.
Connecticut	Connecticut Resource Conservation and Development Forestry Committee. 1998. A Practical Guide for Protecting Water Quality While Harvesting Forest Products. Connecticut Resource Conservation and Development Forestry Committee, Department of Environmental Protection, State of Connecticut, Hartford, Connecticut. 36 pp.
Delaware	Delaware Department of Agriculture, Forest Service. 1996. Delaware's Forestry Best Management Practices Field Manual. Delaware Department of Agriculture, Forestry Department, Dover, Delaware. 71 pp.
Florida	School of Forest Resources and Conservation. Undated (accessed 2001). Special Management Zones. Florida Forestry Information, School of Forest Resources and Conservation, University of Florida, Gainesville, Florida. http://www.sfrc.ufl.edu/Extension/ffws/smz.htm
Georgia	Georgia Forestry Commission. 1999. Georgia's Best Management Practices for Forestry. Georgia Forestry Commission, Dry Branch, Georgia. 71 pp.
Hawaii	Division Forestry and Wildlife. 2001. Water Protection and Management Program. Division Forestry and Wildlife, Department of Land and Natural Resources, State of Hawaii, Honolulu, Hawaii. 23 pp.
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The Northern Watershed Project Stakeholder Committee
Initiated in April 1999, the project represents a collaborative research venture between the Alberta Research Council and the Alberta Conservation Association.

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