Forest Land – Fish Conference II:
Ecosystem Stewardship Through Collaboration

Proceedings of the Forest Land – Fish Conference II

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Abstract

The *Forest Land–Fish Conference II* was held on 26-28 April 2004 in Edmonton, Alberta. A diverse group of individuals attended this meeting including: resource managers and practitioners from industry and government; researchers from universities and Federal and Provincial governments; conservation organizations and public interest groups. The objective of this conference, modeled after the one held in Calgary, Alberta in 1996, was to provide a forum to exchange ideas related to linkages between land management practices and aquatic ecosystems. This document contains 62 conference presentations delivered in both oral and poster formats. Contributions are divided into four conference themes: i) Cumulative Effects, ii) Criteria and Indicators, iii) Riparian Management and iv) Access Management and those presented as posters.

Proceedings Committee

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Foreword

Developing management strategies that minimize the combined effects of human-induced disturbances on ecological systems is arguably the single largest challenge to sustainable resource management. Changes in the structure and function of aquatic ecosystems typically result from the interactions of numerous stressors arising from a multitude of land use activities. For much of Alberta's forested landscape, activities related to the conversion of forested lands to agriculture, commercial forest harvesting, exploration and extraction of oil and gas resources, and angling are widely viewed as potentially altering the health and integrity of aquatic resources. Additionally, the extent that management practices emulate natural disturbances and the potential consequences of this management approach also poses many challenges. While resource managers are aware of the potential consequences of these actions, in many cases they lack specific information on consequences of these actions. This paucity of information, precludes: i) the rapid development of remedial measures, when required, ii) a detailed understanding of the consequences of current practices, iii) the prompt development of best management practices. Taken together, this approach fosters a conservative, and often a non-integrative approach, where information gaps are poorly identified and seldom prioritized. The Forest Land - Fish II Conference provides the opportunity for resource managers and practitioners to become more acquainted with results from a variety of programs that can compliment ongoing efforts to share information and to gain an improved understanding of where investments in research are most needed.

Results from at least two decades of research have shown important linkages between the lakes and streams and the basins in which they are located. For instance, watershed attributes can profoundly influence the physical and chemical composition of surface waters, the types of habitats that they provide, and the abundance and diversity of aquatic flora and fauna. Our ability to be successful ecosystem stewards is, in part, determined by the: i) degree that past and present land use practices alter these linkages?), ii) how ecosystems can recover from these perturbations, and iii) the extent that the ecological footprint of future practices can be minimized. The Forest Land - Fish II Conference provides a forum to disseminate and debate the ecological consequences of human practices on aquatic ecosystems, evaluate current practices, and identify the potential benefits of adopting alternative practices. We hope that these discussions will assist resource managers as they weigh cost and benefits of various management options.

Our conference philosophy of "ecosystem stewardship through collaboration" speaks to the benefits of partnerships as a means to more effectively address environmental challenges. Financial considerations aside, a collaborative approach to ecosystem stewardship will enhance our capacity to identify information gaps, ensure that the most salient environmental landscape questions are being addressed and that this information can be conveyed to operational resource managers in both government and industry. We endorse the view that the search for sustainable ecosystems is attainable by finding a balance between economic, ecological and social considerations. In the words of Ralph Waldo Emerson:

"That which we persist in doing becomes easier - not that the task has changed – but our ability to do has increased"

Proceedings Committee

Preface

The *Forest Land–Fish Conference II* was held on 26-28 April 2004 in Edmonton, Alberta. The meeting was attended by a diverse group of individuals including resource managers and practitioners from industry and government; researchers from universities, Federal and Provincial governments; conservation organizations and public interest groups.

The objective of this conference was to provide a forum for presentation and discussion of current work related to linkages between land management practices and aquatic ecosystems. The first conference with similar objectives was held in Calgary, Alberta in 1996. The conference and the related Proceedings represents an extraordinary opportunity to learn about current and ongoing research programs, research and management challenges, future information requirements, and the ways in which this information supports the continued development of a science-based management model.

This document provides an account of the 62 presentations delivered in both oral and poster formats at the conference. Contributions are arranged within each of the four conference themes of: i) Cumulative Effects, ii) Criteria and Indicators, iii) Riparian Management, and iv) Access Management, and are followed by Poster presentations. However, last minute changes in availability precluded the delivery of some presentations. Lastly, several poster presentations were delivered at the conference but due to time constraints are not identified in these proceedings.

As the Chair of the Conference Proceedings Committee I am greatly indebted to my fellow Proceedings team members: Greg Eisler, Bruce McCulloch and Uldis Silins. They provided insightful, detailed, and constructive reviews of the conference submissions. Thanks also to Michael Monita who completed the design and layout of this document and to Shelley Pruss for reviewing the introductory material of the Proceedings. I am appreciative of their efforts to ensure that this, the Proceedings document, would be available to participants at the conference.

Garry Scrimgeour, Ph.D
Proceedings Committee Chair

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Meeting the Conflicting Objectives of Stream Conservation and Land Use Through Riparian Management: Another Balancing Act

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Abstract

Conflicts between objectives of resource management are most evident at the riparian edge of aquatic systems, whether the land use is forestry, agriculture, or urban settlement. Riparian management and reserve strips originated to safeguard fish habitat, but have since come to be a primary tool to serve other objectives, e.g., wildlife habitat and nutrient uptake. How are we doing with riparian protection? And have we got the right balance of protection? Repeatedly, studies have shown that riparian reserves contribute to woody debris supply, streambank stability, reduced thermal changes, maintenance of detrital inputs, etc., and occasionally that there are benefits to fish populations. In aggregate, these studies suggest the measures are working. However, we still need to determine if the amounts and configurations of riparian reserves are sufficient, and whether they will protect against extreme events. Few studies have tested whether a particular reserve size is the "right" width for meeting all our objectives and to deal with uncertainty. The need for protecting small streams remains a major issue in many places. The particular widths and degree of retention in the reserves also needs to be more clearly tied to the specifics of a particular landscape and land use, rather than assuming one size meets all needs in all places.

Richardson, J.S. 2004. Meeting the conflicting objectives of stream conservation and land use through riparian management: another balancing act. Pages 1-6 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

The expansion of industrialised forestry in the middle part of the 20th century made it clear that forest operations could affect fish habitat. However, in those days, in an illuminating example of how rapidly forest policies can change, wood was actively removed from salmon streams during operations. Greater understanding of the effects of clearcutting on the heating of streams (e.g., Hall et al. 1987) and the loss of woody debris (Bilby and Likens 1980) resulted in many jurisdictions moving towards using streamside reserves, or buffers, to reduce direct impacts of forestry on fish populations.

Since the initial use of riparian reserves (or "buffers") to protect streams and stream fishes, the use of reserves has expanded to serve forestry, agriculture, urban, and other land-use activities as one of the primary means of mitigating impacts on streams. The proposed dimensions of these reserves are most often based on expert opinion or limited empirical results, and rarely on detailed study. It is time to step back and ask several questions. First, what are the objectives we're trying to meet with these measures? Given the myriad of objectives it may be difficult to meet them all in any given location. Second, are these measures working? Despite world-wide use of riparian reserves, there has been a startling lack of tests of whether reserves of a given configuration are effective for particular objectives. Last, can we think more broadly about a landscape-scale approach (e.g., cumulative effects, refuges) and about the temporal dimension?

What are the objectives of riparian protection?

The list of objectives for streamside protection has continued to expand since the first measures were put in place to protect fish habitat in the mid 1970s (Table 1). Reserves initially served to provide shade to minimise the heating effects due to forest canopy removal and to supply LWD to maintain cover and structure for fish habitat in streams. It was only in the 1970s that the current view was articulated of the strong dependence of stream systems on riparian areas (Hynes 1975). Since then the strong interactions of streams and the riparian area have become clearer and clearer (Gregory et al. 1991, Naiman et al. 2002). Subsequently the importance of riparian forests to water quality became clear. Reserves alongside streams provide for filtering of sediments and uptake of excess nutrients in groundwater flowpaths into streams (e.g. Castelle et al. 1994), primarily in agricultural settings. In urban settings the realisation that the areas beyond the riparian area were often permanently converted to non-forested settings argued for a greater degree of riparian protection than might be necessary in a forested environment. Riparian reserves originally intended to protect fish habitat from forestry now serve to protect against multiple land use effects, and for multiple objectives, not just fish habitat.

In addition to the protection of fish habitat, primarily estimated as large wood debris, reserves are known to maintain the integrity of stream banks and reduce sediment transport. And the list of objectives continues to grow. Riparian areas reduce the transport of fine inorganic sediment (silt) into streams, another important objective. Forests provide organic matter to streams (leaf litter, arthropods from the forest canopy) that are critical to the productivity of stream food webs. All of these objectives can be met with relatively narrow bands of about 30 m of forest, so long as they don't blow down!

Riparian areas form critical habitat for a great number of species around the world that are riparian obligates, i.e., they absolutely require streams or riparian areas for some portion of their lives. Some of these species are piscivores and benthivores (consumption of benthic invertebrates) and so their dependence on streams is obvious. Studies of intact riparian areas around wetlands and streams has suggested that in order to protect most populations of amphibians and reptiles, some of them threatened, reserves of up to 290 m (with an additional 50 m buffer) away from the water margins would be needed (Semlitsch and Bodie 2003). This assessment was based on the habitat use of large number of amphibian and reptile (mostly turtles) species from many places in the world. In setting out particular guidelines for riparian reserves the additional value of habitat has been included (FEMAT 1993, BC FPC 1995) as a consideration. In FEMAT (1993) the reserve widths for U.S. federal lands were doubled from what they originally proposed in order to accommodate wildlife habitat. Some species that live in these areas include amphibians that spend their larval stages in streams and remain as riparian associates in their juvenile and adult stages. There are also vascular plants and bryophytes (Dynesius 2001), and other species, that are found nowhere else than within the riparian area, even right at the wetted edge.

In addition to critical habitat, there are many species that are considered riparian associated, but are not

Table 1. Some environmental protection objectives of riparian management strategies along streamsides.

Objective	Primary targets	Measures	Effectiveness and extent
Fish habitat	Shade and LWD supply for cover and for geomorphic controls	Fish population size, species diversity, physical habitat	Few examples at present based on fish populations – mostly based on association of fish with woody debris. Fish-bearing reaches.
Water quality	Filtering of fine sediments and uptake of excess nutrients from groundwater	Nutrient concentrations and turbidity	Many studies, most from agricultural and urban situations
Supplies of litter inputs	Fixed carbon sources for detrital-based food webs	Material export and concentrations	Most particulate inputs come from riparian areas, but much is transported
Streambank integrity	Maintain stability of streambank based on root structures	Shear stress; erosion	
Habitat - obligates	Critical habitats for many species of riparian obligates	Supply of wildlife trees, denning and nesting sites;	Many studies of vertebrates and vascular plants; few studies of invertebrates or bryophytes
Habitat – riparian associates	Important and productive habitats for riparian-associated species	Area with plants eaten by species such as grizzly bears, which can also feed elsewhere	Productive area retained near streams
Movement corridors	Dispersal and migration pathways		Very few studies, other than along very large floodplains
Aesthetics	Visual and recreational quality	Visual quality	Walking paths, canoe routes, etc., where the harvesting is "hidden" by riparian reserves

obligate users of these areas. For instance, grizzly bears make use of riparian areas for the berries they provide, and at some times of year capitalise on the returning salmon as a major dietary item. However, they are not dependent on these areas. The high productivity of riparian areas and the particular microclimate make these attractive to many species and they occur at higher densities there. In heavily managed areas, riparian reserves may be important.

Riparian areas also provide movement corridors for some species. Many species of birds evidently follow drainages during migration. Species such as marbled murrelets also apparently fly up rivers from the sea to the forest sites where they nest. It is not clear how retention of riparian reserves or no reserves might affect their use of these areas as corridors. It is also known that in large, alluvial systems that other species of wildlife use these areas as corridors for dispersal or for movements through their home ranges.

Can we really meet all of these objectives at every site? And is the same reserve width going to serve every objective to the same level of effectiveness?

Evaluation of riparian protection

We know that riparian areas provide a great many services for streams and riparian-dependent organisms, but how effective are our designated guidelines for reserves? It seems logical that leaving riparian reserves should meet most of our objectives better than clearcutting to the bank. However, how much better is leaving reserves and could we do it more effectively? What is the correct width or configuration? Most of the widths currently entrenched in guidelines were derived from studies showing that trees that become large woody debris in streams, largely originate from about one tree height or less from the streambank. Modifications of the mandated widths were made based on local environments or fears of litigation (FEMAT 1993).

As an example of how reserve widths may be modified, British Columbia has reserve widths that vary according to stream size, presence of fish, or the downstream use of water for domestic supply (BC FPC 1995). Streams lacking fish and not used for drinking water have no required reserve. Very small streams with fish (< 1.5 m bankfull width) do not get a reserve. Fish-bearing streams > 1.5 m get a reserve width that increases from 20 to 30 and then to 50 m based on increasing channel widths. A 50 m reserve is intended to serve more than the immediate objectives of LWD supply, and the large alluvial rivers where this reserve size would occur may have more riparian-dependent wildlife species than smaller streams. Other jurisdictions have other modifiers of reserve width depending upon local objectives, although it is not often stated explicitly what those objectives are.

Semlitsch and Bodie (2003) propose 290 m wide reserves for the protection of amphibians and reptiles, however, these may not be effective (even if they were possible) if edge effects penetrated too far into the reserves or if dispersal were eliminated. There have been many studies that have shown that clearcutting to the streambank results in dramatic changes in diversity of a variety of organisms, but relatively few where retention of reserves has been evaluated.

Reducing the band of forest around an aquatic system to a narrow riparian reserve faces the additional problems of enhanced rates of windthrow and potential edge effects. To reduce the threats from the two above processes, best management practices often advocate using an additional buffer around the reserve to reduce edge effects. Unfortunately these guidelines are not mandatory and the result usually retreats to the default enforced rules.

Guidelines in various states and provinces for the retention of riparian reserves vary widely, without any obvious reasons (Young 2000, Blinn and Kilgore 2001). Why are there such diverse guidelines? One has to do with the primary resources to protect (cool-water fish or water quality), and also with the natural disturbance regimes of the region. Although many countries use riparian reserves as a protection from forest harvesting and other land uses, there are startlingly few studies of how well these strategies work.

Examples

There have been many studies focused on the effects of forest harvesting on streams, and some testing the effectiveness of reserves to meet some of the objectives discussed above. Studies of nutrient and sediment flux across riparian areas have shown reserves to be effective at removing a large proportion of nutrients and fine sediments, at least in mostly agricultural settings (e.g., Castelle et al. 1994, Lee et al. 2003). In a study of streams in northern California, diversity of stream benthos reached an asymptote at about 30 m width and was depressed at narrower reserve widths (Newbold et al. 1980). A set of 13 streams in British Columbia were compared prior to and following forest harvesting with either no reserves (cut to the bank), 10 m reserves, 30 m reserves, or controls (Kiffney et al. 2003). They found that even with 30 m reserves there were still nearly 5-fold increases in algae in summer compared to the controls, although this was still better than clearcuts or 10 m reserves. Other changes in temperature and organic matter dynamics also occurred with clearcuts or 10 m reserves, but 30 m reserves maintained those attributes relative to controls (Kiffney et al. 2003). There are other ongoing studies of the effectiveness of riparian reserves for stream systems in North America (e.g., British Columbia, Washington, Alberta). Whether we judge particular strategies for protecting stream and riparian systems as successful appears to depend on the particular measures under consideration.

There have been a number of different kinds of studies of riparian reserves for riparian organisms. In Quebec, streamside riparian reserves of several treatments (20, 40, 60 m reserves versus controls) were replicated to assess the impacts on riparian wildlife (Darveau et al. 1995), with 20 m reserves not supporting most forest bird species. Small mammals in the same study showed little difference from controls even in reserves as narrow as 20 m (Darveau et al. 2001). The effectiveness of riparian reserves for protecting terrestrial habitat has been shown in other studies for small animals such as rodents and shrews (e.g., Cockle and Richardson 2003), birds (e.g., Pearson and Manuwal 2001), and amphibians (e.g., Willson and Dorcas 2003). Riparian reserves also protect more of the biodiversity of bryophytes (mosses and hepatics) and terrestrial gastropods than clearing of forest to the stream edge (Dynesius 2001).

In many jurisdictions reserves are partially harvested and often based around letting second-growth trees in riparian areas to reach a larger size through a rotation

Table 2. Some aspects of riparian – stream ecosystems that need to be further explored as we improve our management strategies

1. Rates of recovery relative to scale (magnitude and extent) of disturbance
2. A non-stationary matrix, so we need to consider the net changes in watersheds, not simply the local environment.
3. Tests of the effectiveness of guidelines take time and we should not relax guidelines prematurely
4. There is likely to be geographic variation (latitude and altitude) and site-specific effects in the rates of stream – riparian processes that need to be incorporated into future developments
5. Other environmental objectives in need of riparian reserves?

time on average double that of other trees in the managed area. Modelling by Meleason et al. (2003) suggests that even moderate-aged riparian reserves may not provide enough volume in each tree to have much impact on riverine fish habitat. This may argue against partial harvesting and simple longer rotations in riparian areas, but more field testing is needed.

There are many pressing questions currently facing forest management around streams. In most regions small streams are the least protected, but likely the most vulnerable to land-use. How can small streams be protected without unduly restricting forest harvest? This even becomes a legislative issue of what constitutes a stream for the purposes of stream protection laws. Another question is whether some other form of riparian reserves can provide the same protection as the commonly applied fixed-width reserves? Can any wood be withdrawn from the reserve without diminishing its effectiveness? Can we economically minimise the risk of windthrow, and what should we do with windthrown trees?

The future

It will be apparent that we can't easily meet every objective at every site. This will require a broader view than the block by block approach that is typical. An integrative view of how the fluvial network is geographically situated will be needed to ensure that habitats are not fragmented or that upstream effects do not accumulate downstream. We also need to consider the temporal view, as some impacts will be short-term in the scale of the forest and recover on time scales much shorter than the rotation time (Table 2).

We need to be cautious about extrapolation of results from forestry-fish studies. Most of these studies were initiated when more of their watersheds were still relatively pristine. As the proportion of the area of a watershed under management increases, the amount of protection needed may increase. For instance, forests now developing into mature and harvestable second-growth forests will provide the riparian reserves of the future, however, the relatively small trees (at least in coastal forests) may not provide the same structural contribution as large, old-growth trees (Meleason et al. 2003). The context is thus not constant and we should be mindful of non-stationarity in these systems and how it may affect future needs and effectiveness.

Management guidelines change frequently, and sometimes on a shorter time scale than the generation time of a fish, never mind the rotation length of a managed forest. We need to be willing to revise our guidelines for protection of aquatic resources. However, we should not do so prematurely, despite pressure to relax guidelines, until we are confident that the guidelines are either meeting or failing to meet our objectives and expectations. The effectiveness of guidelines need to be tested and it is essential that enough time elapses to avoid mere transient effects, but it is not an excuse for avoiding innovation.

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Cumulative Effects

The Role of Integrated Landscape Management to Assist With Exploring the Past, Present, and Future Effects of Landscape Activities on Alberta's Boreal Fish Communities

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Extended Abstract

In both pre-and post-industrial eras, Alberta's boreal fish communities have been exposed to highly dynamic landscapes. In the former, variances in climate, water flow, and fire were major drivers in altering physical and chemical properties of water and adjacent riparian habitat. The contemporary lentic and lotic landscape, however, is quite different from the one that preceded European landuse, as it is being reshaped by a multitude of disturbance regimes that include natural perturbation agents (fire, flooding), and, increasingly, landuses such as agriculture, forestry, energy, transportation, shoreline residential, and commercial and recreational fishing. As landuses have unfolded on the boreal landscape, resource managers have observed (and measured) significant changes (including abundance and distribution) in several species (bull trout, grayling) of Alberta's boreal fish fauna.

Changes to the health of boreal fish communities are related to three key and often sympatric drivers: habitat loss, habitat alteration, and excessive fish harvest. This presentation will focus on the first two, arguing that fish managers will not have the option of implementing sophisticated sustainable fish harvest strategies if appropriate amounts and quality of fish habitat are not maintained on the boreal landscape.

Intrinsic to landuse are numerous requisite industrial footprints that can compromise fish habitat. Some are physical features such as roads, lotic crossings, and the actual replacement of natural surficial water with industrial features (such as displacing a creek with a surface oilsand mine). Others relate to water flow, and include rates of water extraction from and discharge to surface and subsurface water pool reserves for the production and or maintenance of hydrocarbons, agricultural crops, livestock, wood pulp, and people. Of equal importance to water quantity are the effects of landuse on water quality, a deserving issue that should focus attention on numerous landuses including shoreline residential, livestock and crop production, and the processing technologies of the energy and forest sectors.

Stelfox, B.J. 2004. The role of integrated landscape management to assist with exploring the past, present, and future effects of landscape activities on Alberta's boreal fish communities. Pages 9-10 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

The importance of water managers in understanding and implementing integrated landscape management (ILM) can not be overstated. Using the ALCES© landscape model attributed for the Forest Management Agreement (FMA) area of Alberta-Pacific Forest Industries, back-cast simulations were performed to generate the industrial footprints emerging from historical landuses in the boreal forest of northeast Alberta during the past half century. Under the principles of ILM, ALCES was then used to project plausible future development trajectories (and associated water demand) for the energy, forestry, agricultural, and human population sectors for the upcoming five decades. These simulations are presented with the purpose of challenging fisheries managers to appreciate the challenges of managing sustainable fish populations in an increasingly industrial boreal forest landscape. The potential for "best practices" proposed by the forestry and energy sectors to mitigate adverse effects on the boreal landscape of will be illustrated; an approach that clearly demonstrates the sustainability rewards that await resource managers prepared to explore novel solutions to emerging problems.

Effects of Stream Temperature on Interspecific Competition Between Juvenile Brook and Bull Trout

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Extended Abstract

Bull trout (*Salvelinus confluentus*) populations are in decline across their native range, including Alberta. Competition with introduced fish species, particularly brook trout (*S. fontinalis*), is considered a contributing factor. The two species often maintain parapatric distributions, where bull trout are restricted to cool, high-elevation streams and appear resistant to invasion from downstream populations of brook trout. Bull trout have one of the lowest upper thermal limits and growth optima of the North American salmonids. To test the hypothesis that juvenile bull trout compete more successfully against juvenile brook trout in colder water, artificial stream and snorkel observations were performed in the Rocky Mountains of central Alberta during the summer and fall of 2002 and 2003. Relative competitive performance of both species was quantified by measuring rates of growth (i.e., change in weight and fork length), agonisim (i.e., frequency of nip, chase, and display behaviour) and foraging (i.e., frequency of forage attempts). In an artificial stream experiment, three replicates, each of single and mixed species treatments at high (i.e., 8 fish/m^2) and low (i.e., 4 fish/m^2) densities and cool (i.e., 15°C) and cold (i.e., 8°C) water temperatures, were conducted to determine the relative strength of interspecific and intraspecific competition for both species. Each replicate of the artificial stream experiment was two-weeks in duration. The majority of fish in all treatments lost mass. Per capita growth of bull trout within temperature treatment was similar at either density and lowest when sympatric with brook trout in the cool water treatment. Growth of allopatric brook trout was reduced in high-density treatments while sympatric growth rates were comparable to high density rates at the same temperature. Both species showed differential behavioural responses to treatments. Dominant individual brook

Rodtka, M.C.and Volpe, J.P. 2004. Effects of stream temperature on interspecific competition between juvenile brook and bull trout. Pages 11-12 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

trout could be unambiguously identified in most allopatric and mixed species groups while this was rarely possible for bull trout. Bull trout foraged approximately three times more often than sympatric brook trout in cold water, in cool water brook trout foraging rate was double that of bull trout. Overall, agonistic behaviour initiated by brook trout was positively correlated with foraging rate, no similar correlation between the aggressiveness and foraging rate of bull trout was observed. Intraspecific aggression rates among bull trout in allopatry and sympatry increased markedly with water temperature. No comparable increase was observed for brook trout. The per capita rate of intraspecific agonisim among bull trout sympatric with brook trout was approximately six times greater in the cool water treatment while brook trout rates were similar between temperature treatments. Both species altered the nature of agonistic and/or foraging behaviour in response to changes in density and water temperature. Similar trends were observed in the wild. These data suggest increasing water temperatures disproportionately affect bull trout and help explain observed intra-drainage habitat partitionment. Further, these results demonstrate the complex non-linear effects of water temperature change on species-specific interactions. The broader implications of watershed and global-scale processes that may influence water temperature need to be considered in management plans if conservation of native salmonids is a priority.

Large Woody Debris Replacement in Small Headwater Streams in Central British Columbia

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Extended Abstract

To identify the effect of current riparian management practices on long-term large woody debris (LWD) levels we are adapting a woody debris depletion model for use in north-central British Columbia. LWD is important in headwater streams, it influences channel morphology by creating pools and trapping sediment and organic material, improving in-stream habitat. LWD provides a substrate for invertebrates and algae that will drift downstream to provide fish food. The model is being applied to a series of experimental watersheds located in the vicinity of Prince George. The project is part of a cooperative small stream research project investigating the effects of riparian management strategies on the ecology of small streams.

We measured the quality and variability of LWD in a series of undisturbed small streams with bank-full widths between 80 and 160cm, and gradients less than 6%. The streams run through a variety of stand types and parent materials, and are located across a wide range of regional precipitation regimes (440-900mm year). Within each of the twelve streams, one or more 50m reaches were sampled. For each reach, all in-stream woody debris greater than 5cm in diameter was measured and described. An inventory of the riparian stand by species and size class was compiled for 10m on either side of the stream.

Current levels of LWD in all the streams were between 26 to 100 pieces per 50m of stream length. Over 30% of this woody debris provided hydraulic function at low stream flows. In all streams measured, over 40% of the LWD was in the size class "5 to 15 cm in diameter" and at least one piece (to 15 pieces) of LWD was in the size class "> 30 cm diameter". Lodgepole pine leading stands had fewer LWD pieces greater than 30cm diameter (1 to 2 per 50m). The proportion of large trees was 3 to 4% lower in the stream than in the undisturbed adjacent riparian stand. Small streams have large wood as natural instream components, which require replacement over time.

Two to 40% of LWD was classified as wood that is not yet in the stream but lying above the

Beaudry, L. 2004. Large woody debris replacement in small headwater streams in central British Columbia. Pages 13-14 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

stream. The LWD that currently provides function will initially be replaced by LWD that lies above the stream. As the legacy of LWD decays, future inputs will originate from the riparian retention along the stream. Current riparian practice is to retain 10-12 stems greater than 15 cm diameter per 100m on small streams. In the longer term, LWD will be provided by the second growth stand. However, the stand will provide few inputs until it reaches 20 years of age when only small pieces of woody debris will be available. It requires 60 years to grow trees greater than 30cm diameter on these sites.

We are adapting a LWD input and depletion model to assess long-term changes in LWD in small headwater streams with local data to determine if the current riparian retention practice will provide sufficient LWD replacement over a harvesting rotation in BC's Interior.

Troubled Waters: Cumulative Anthropogenic Activity and a Declining Bull Trout Population in the Elbow River Watershed

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Extended Abstract

Rapidly expanding urban centres, coupled with increased human exploitation of riparian areas, are generating a sense of urgency and concern for the diminishing Elbow River bull trout population. In addition to possessing a 'sensitive' designation in the province, this trout population resides directly adjacent to Calgary and is the closest of any in Alberta to a major urban centre. Currently, the Elbow River watershed is representative of several potential consequences of growing human interests in natural areas.

Reasons for declines in Alberta's bull trout populations are both biological and anthropogenic. Specific habitat requirements, slow growth rates, late maturity, and alternate-year spawning make bull trout particularly vulnerable. In addition, bull trout currently face numerous pressures including illegal harvest, habitat degradation, and competition and hybridization with introduced brook trout and Arctic char. The severity of these threats is growing in-step with expanding human activity in the watershed.

Because bull trout form aggregations during fall and winter, they are disproportionately susceptible to negative point-source effects such as late season angling and poaching. The goal of this study is to quantify and map Elbow River bull trout habitat use throughout the watershed, with specific emphasis on how land-use patterns affect bull trout movement and distribution.

Radio telemetry is being used in conjunction with snorkeling observations to measure fish community composition and abiotic factors associated with seasonal habitat selection. During fall 2003, 30 bull trout were implanted with radio transmitters. Radio-tagged fish are located using aerial and ground tracking, and resulting data are quantified by coupling UTM coordinates with GIS software.

Popowich, R.P. and Volpe, J.P. 2004. Troubled waters: cumulative anthropogenic activity and a declining bull trout population in the Elbow River watershed. Pages 15-16 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Preliminary transmitter data have led to several unanticipated findings. It is suspected that poachers have illegally harvested five of the 30 radio-tagged fish. Given current population estimates, the removal of these five fish represents a confirmed loss of 5-8% to the entire spawning population. Assuming comparable poaching levels exist for untagged adult bull trout, poaching rates may have approached 17% between September and October 2003. In all confirmed cases, fish were taken from areas easily accessed by Off Highway Vehicle (OHV) or pedestrian traffic.

In addition to poaching pressure, Arctic char and brook trout have been introduced into tributaries of the Elbow River. Suppressed native bull trout populations, combined with increased habitat disturbance, provide an environment conducive to the successful establishment of invasive species. Should either or both of these aggressive species become established, bull trout will face significant additional pressure through direct resource competition. Brook trout have been observed in direct competition with juvenile bull trout at numerous locations within the watershed, and suspected bull trout x brook trout hybrids have been collected. Hybridization alone represents a multi-faceted threat to bull trout through wasted reproductive effort, loss of genetic material, and the potential introduction of a highly competitive hybrid population.

The cumulative effects of extremely high levels of poaching, the introduction of non-native competitors, as well as increased sediment loading from OHV activity and a petroleum access road are likely related to the 25% reduction in redd counts from 2002 to 2003. Should current trends continue, the Elbow River bull trout population might soon face the possibility of extirpation.

Study Designs for Environmental Impact Assessment: An Example Using Bull Trout (*Salvelinus confluentus*) and the Kemess South Mine Project

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Extended Abstract

Testing hypotheses through the experimental approach remains a powerful technique for the advancement of science. Environmental impact assessments may draw on this power by formulating studies into before-after control-impact (BACI) experiments. The BACI method has only a single control and impact location and relies on time for replication. While the approach solves problems of psuedoreplication inherent with earlier methods, it fails when population trends between locations are not positively correlated with zero lag. Furthermore, as replication occurs through time, impact assessments using the BACI design are often doomed from their inception as statistical power is limited from inadequate replication during the initial (i.e., control) time period. For example, many impact assessment biologists would consider two years of pre-development data as a luxury. More recent studies have proposed a modified BACI design that incorporates spatial replication through multiple control locations. The modified design alleviates requirements of correlated trend data and increases statistical power through spatial replication.

We used the modified BACI design to assess the effects of the Kemess Mine on adult and juvenile bull trout (*Salvelinus confluentus*) populations within the Thutade Watershed of northwestern British Columbia. Bull trout were monitored from 1994-2002 (juveniles from 1995-2002) in seven tributaries to Thutade Lake. South Kemess Creek, one of these tributaries, was directly affected by mine construction that commenced in the late summer of 1996. Tailings dam construction resulted in the removal of 4.1 km (65%) of stream habitat utilized by bull trout in S. Kemess Creek. Total spawners (i.e., estimated from visual counts of spawning nests and unspawned females) in S. Kemess Creek decreased significantly following mine construction. The decrease may have been partially offset by displaced spawners using nearby tributaries, suggesting inherent compensation for lost spawning habitat. However, densities of

Paul, A.J. and Bustard, D. 2004. Study designs for environmental impact assessment: an example using bull trout (*Salvelinus confluentus*) and the Kemess South Mine Project.. Pages 17-18 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

age-0 and age-1 bull trout did not increase significantly in these tributaries. A negative relation between density and age-0 survival implies juvenile production is naturally limited in the tributaries. Finally, densities of age-1 bull trout in the remaining 2 km section of S. Kemess Creek have decreased following mine construction to 2002.

The modified BACI study design detected statistical differences in bull trout abundance or density, despite only two years of pre-construction data. In addition to limited temporal replication, variable trends between tributaries, including negative or lagged correlation, would have prohibited the use of a standard BACI design for this impact assessment. Therefore, this study supports the use of several control systems when designing an environmental impact assessment as it: a) increases statistical power and b) avoids underlying assumptions in trend data. By quantifying changes from environmental disturbance, the suitability of mitigation programs can be evaluated and future research planned. For example, in the Kemess Mine study, natural limits to juvenile production mean simple relocation of spawners will not compensate for lost habitat. Rather, habitat quantity must be maintained. To meet this requirement, approximately 9 km of new stream habitat has been made available to bull trout through removal of migration barriers; and, artificial spawning beds have been constructed below the tailings dam in S. Kemess Creek. Most recent results suggest the mitigation measures have been effective at creating spawning habitat and enhancing juvenile bull trout rearing. By continuing the modified BACI study design, these mitigation efforts can be tested rigorously through time.

WrnsAB2k for Hydrologic Simulation of the Long Term Effects of Forest Harvesting

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Extended Abstract

The U.S. EPA and the U.S. Forest Service developed WRENSS in 1980 to simulate the long-term hydrologic effects of forest harvesting. Computerized versions were later produced for Alberta and have been used for 5-10 years to assess the hydrologic effects of harvest plans for forest companies. WrnsAB2k simulates streamflow as a time series that includes the rate of hydrologic recovery (time for evapotranspiration and flows to return to pre-disturbance levels) under average flow and precipitation conditions. WrnsAB2k also provides estimates of increases in maximum daily and instantaneous flows. A version that uses randomly generated variable precipitation is currently under development.

Two WrnsAB2k simulations will be presented. The first is a medium sized conifer dominated watershed (99.7 km^2) with harvesting (22 % over 10 yrs) planned in 9 sub-basins (1.3 – 25 km^2) Simulated maximum increases in annual water yield in the sub-basins ranged from 6.2% to 14.7% (extra 10-25 mm). Increases showed a positive trend with the area harvested (15.5% to 34.5%). Maximum increase in annual yield for the total basin was 6.4% (an extra 11 mm). Hydrologic recovery was 20 - 50 years for yield and peaks to approach zero. Increases in maximum daily flows in the sub-basins ranged from 2.6% to 7.1%, 2.7% to 7.8% and 2.6% to 8.2% for the 2, 5 and 10-year recurrence intervals. The largest increase was in a sub-basin where 31% of forest cover was removed. Increased peak flows for the total basin were similar, averaging around 5%.

The second simulation was for a larger mixedwood watershed (1900 km^2) composed of 16 sub-basins (35 – 350 km^2). Total area harvested was 26%, with 6% between 1950-2000 and the

Rothwell, R.L., Swanson, R.H. and Spillios, L.C. 2004. WrnsAB2k for hydrologic simulation of the long term effects of forest harvesting. Pages 19-20 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

remaining 20% planned between 2000-2021. Maximum increases in annual water yield in the sub-basins ranged from 3% to 16%, (extra 10 -53 mm). Maximum increases are projected to coincide with the period when most harvesting will occur (2000-2021). Harvesting in the sub-basins ranged from 13.1% to 59.6%. Maximum increase in annual yield for the total basin was 9.3%, (extra 21 mm). Hydrologic recovery among the sub-basins and total basin ranged from 29 to 91 years. Increases in maximum daily flows in the sub-basins ranged from 4.6% to 30.7%, 2.1% to 15.8% and 1.2% to 8.7% for the 2, 5 and 10-year recurrence intervals. The largest increases were in two sub-basins with the most harvesting (59.6%, 53%). Increases in the remainder of the basins, where harvesting averaged 25%, were much smaller.

WrnsAB2k provides forest managers a relatively straightforward tool to assess the relative magnitude and duration of changes in annual and peak flows using available precipitation, streamflow and forest cover data. Interpretation of the results should be approached carefully as the use of constant precipitation makes simulated water yield less variable than actual values (precipitation-streamflow is highly variable from year to year). A common question from forest managers is, "What is an acceptable increase in water yield and peak flows". Each situation is unique with respect to instream and downstream water users, and their use must be part of any "acceptable increase" criterion.

Effects of Forest Harvesting and Fire on Fish Communities in Boreal Plains Lakes

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Extended Abstract

Natural disturbances, such as wildfire, have been promoted as models for forest management in boreal regions. Aquatic systems are linked to their surrounding drainage basins and therefore can also be sensitive to landscape disturbance. Because our understanding of if or how landscape disturbance (natural or otherwise) shapes populations and communities in lakes is poor, our capacity to formulate wise management strategies for fisheries resources in forested landscapes is also limited. To assess impacts of forest harvesting or fire on lentic fish communities, we applied a reference-condition approach to the fishes in 37 lakes with burned, logged, or undisturbed catchments in the mixed-wood boreal forest of north-central Alberta. In this approach, communities at disturbed sites are compared to those in reference lakes to determine if the former fall outside the range of expected conditions, indicating effects of the disturbance.

Fish communities in the reference lakes were classified into two types: those dominated by large-bodied piscivores and those dominated by small-bodied fishes. A discriminant function analysis with only two environmental descriptors (lake maximum depth and average slope of

Tonn, W.M., Scrimgeour, G.J., Paszkowski, C.A., Boss, S.M. and Aku, P.K.M. 2004. Effects of forest harvesting and fire on fish communities in Boreal Plains lakes. Pages 21-22 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

the catchment) could correctly predict community type in 84% of reference lakes. Depth likely reflects the influence that winter oxygen concentrations have on fish assemblage type, whereas catchment slope is correlated with a variety of landscape-level features. Although forest harvesting and fire might increase the susceptibility of lakes to winter hypoxia and alter connectivity to the regional drainage network, fish communities in 93% of the disturbed lakes did not deviate from the discriminant function predictions. This suggests little, if any, community-level effects of the disturbances over the 1–2-year time period of our study. Indeed, the level of disturbance in a catchment could explain less than 3% of the variation in community structure. A slight increase in the catches of white sucker and a greater proportion of small individuals in white sucker populations, however, may have reflected a modest enrichment effect in burned lakes.

We also conducted a parallel study on near-term effects of fire on small-lake fish assemblages in the sub-arctic Caribou Mountains. Ordination analysis revealed three simple but distinct fish communities: those dominated by northern pike, suckers, or Arctic grayling. Despite its extensiveness at burned lakes (84% of catchment area disturbed), fire did not appear to impact the fish communities. Disturbance accounted for < 10% of variation in fish community structure among lakes, and reference and burned lakes were represented among all three community types. Burned lakes had, however, fewer small northern pike than did reference lakes. Current landscape disturbances on the Boreal Plains appear to have minimal short-term effects on lake fish; in both study regions, however, population-level differences between disturbed and reference lakes suggest longer-term impacts are possible.

Historical Risk Analysis of Watershed Disturbance in the Southern East Slopes Region of Alberta, Canada, 1910-1996

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Abstract

Effects of ecological disturbances are likely to be more pronounced the longer the disturbance persists. The difficulty and costs of restoration work therefore are likely to be greater, while the probability of success in restoration is likely to decrease, as disturbance is prolonged. We examined the persistence of disturbance at a landscape scale in a region having among the most heavily-disturbed watersheds in western North America, the southern east slopes region of the Alberta Rocky Mountains. We used the British Columbia Level 1 Watershed Assessment Procedure (IWAP) to evaluate the risk of erosion and stream channel damage from logging and linear disturbances to 90 basins in the Crowsnest, Oldman-Livingstone, Highwood river and Willow Creek drainages for the decades 1910, 1930, 1950, 1970 and 1990 using GIS analysis of historical and current maps, air photographs and contemporary digital geographic data. Overall, 87 of 90 basins have been at moderate or high risk of damage for 20 to 100 years (mean 47.4 years). Presently 28 are at high risk, and have been so for 20 to 60 years (mean 30 years). An additional 59 are at moderate risk, and have been so for 20 to 80 years (mean 43.4 years). Land managers wishing to minimize the disturbance risk in these watersheds need to act immediately to decrease risk factors for the 28 basins at high risk, and to prevent the evaluations for the remaining 62 basins from attaining the high risk category. This can be done by removing roads, recontouring their beds and replanting their rights-of-way; and by replanting cutblocks more effectively to reduce equivalent clearcut area.

Mayhood, D.W., Sawyer, M.D. and Haskins, W. 2004. Historical risk analysis of watershed disturbance in the southern east slopes region of Alberta, Canada, 1910-1996. Pages 23-29 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

Watersheds are highly-integrated ecosystems. The nature of the geology, soils and biological communities in the uplands, and the physical, chemical and biological processes that occur there, ultimately influence the waterbodies, watercourses and channels in the valley bottoms (Rawson 1939, Hynes 1975, Lotspeich 1980). Resource development inevitably entails substantial disturbance of the land surface within watersheds. Road systems are built and maintained, exploration trails and transmission lines are cut, and forest is logged or cleared. In the case of surface mining, the land surface may be substantially re-sculpted or even physically removed. Inevitably these activities, if extensive enough, disrupt the ecological integrity of watersheds and the watercourses that drain them.

The types of disturbance caused by largescale development are often long-lasting. Once built, roads are seldom removed. Seismic and exploration trails are rarely reclaimed. Roads and trails beget more roads and trails as new developers and recreationists extend and adapt the network for their own uses. Cutblocks, wellsites and surface mines remain surface disturbances for at least several decades. Urbanized sites are effectively permanent and ever-growing. The kinds of changes caused by such longterm disturbances are often cumulative and synergistic. Increased surface erosion and increased, flashier runoff are two common, widespread effects of surface disturbance that are of the progressive, synergistic type. For these types of change, the duration of disturbance substantially determines the total damage that a watershed may sustain.

In this paper, we examine the duration of watershed disturbance in one of the most disturbed regions of the Rocky Mountains. We applied a measure of risk included in British Columbia's Level 1 Interior Watershed Assessment Procedure (IWAP) to historical data from 1910 to 1996 to study changes in, and duration of risk from, surface disturbances in 90 watersheds on the eastern slopes of southwestern Alberta. We then considered ways in which the degree and extent of risk to watersheds can be minimized.

Study Area

The 2455 km² study area lies in the Front Ranges of the southern Canadian Rocky Mountains on the eastern slopes of the Continental Divide approximately between latitudes 49°30' N and 50°20' N, and between longitudes 114°15' W and 114°50' W (Figure 1). It

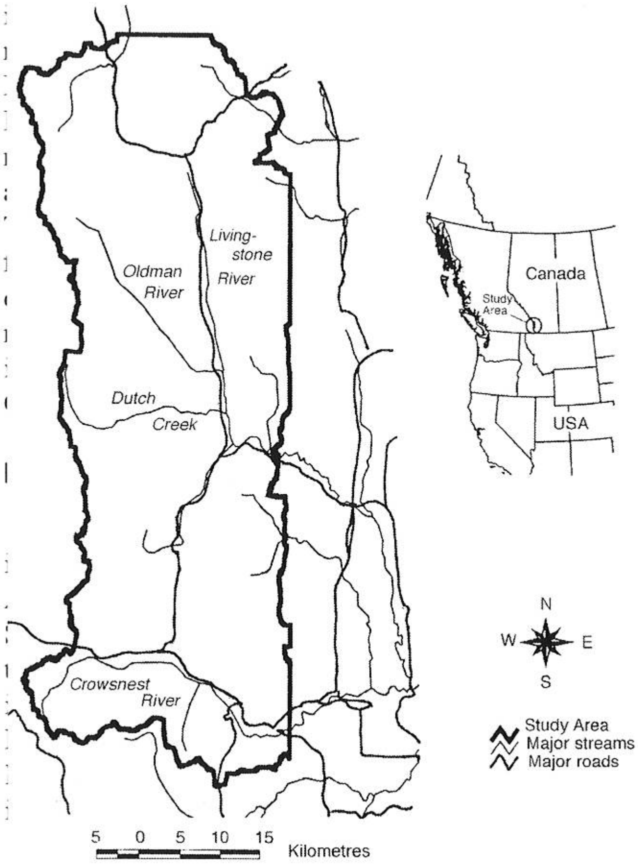

Figure 1. Location of the study area in southwestern Alberta, adapted from Mayhood et al. (1998).

a screening-level ranking of the potential for damage to watersheds from the combined effects of increased peak flows and increased surface erosion resulting from land disturbance. It is calculated from six measures of road development, plus estimates of equivalent clearcut area adjusted for the effects of hydrological recovery due to forest regeneration, and for the elevation of the clearcuts in the basin. Rationale, data requirements and details of the calculations are provided in the IWAP guidebook (British Columbia Forest Service 1995). Here we describe only those data and procedures specific to this study.

Recent linear data (roads, railways, transmission lines, vehicular trails), hydrography and digital elevation models (DEM) were obtained from Alberta Government 1:20,000 digital base map files. For road density calculations, we combined all linear disturbances that carried motorized traffic, but excluded foot and horse trails. Erodible terrain was mapped from surficial geology (Bayrock and Reimchen 1980) and slope data from the DEM using materials and slope criteria in Reimchen and Bayrock (1977).

Vegetation data, including species composition and stand disturbance modifiers, were derived from digital Alberta Vegetation Inventory (AVI) files. The linear and DEM data are current to 1994, and the vegetation data are current to mid-1995, supplemented with our ground observations up to 1996. Historical linear data were digitized from historical maps and aerial photography as available. The AVI did not provide basal area removal data to compute hydrologic recovery for regenerating clearcuts in the manner specified in the IWAP manual. We therefore estimated percent recovery from crown closure or crown removal, as available, in the AVI datasets (Sawyer et al. 1997).

Digital data were managed and analyzed with the ArcInfo geographic information system (ESRI 1995). Further details of our methods are provided elsewhere (Haskins and Mayhood 1997, Sawyer et al. 1997, Sawyer and Mayhood 1998b).

Results

The extent of landscape disturbance in the 90 study basins, and the associated level of risk, were mapped for alternate dacades beginning with the decade 1910-19 (Figure 2). From these maps, the number of basins at risk in each major drainage were tallied for each of the studied decades (Table 1).

Prior to the coming of the Canadian Pacific Railway in 1898, the study area was essentially undeveloped. The initial disturbances shown for the decade 1910-19 reflect the presence of the railway and associated developments in the Crowsnest Valley, and horse and wagon trails in the Oldman and Livingstone valleys. Risks to most watersheds from this level of development were negligible: only two (Oldman River between Hidden and Racehorse creeks, and lower Gold Creek) were at moderate risk.

By the 1930s the expansion of mining and urbanization in the Crowsnest Valley, and the introduction of automobiles to the region, had led to substantial expansion of the road network up several tributary valleys, some small surface mining pits, clearcuts in headwater creeks, and some cleared pasture or cropland at the lower end of the valley. Roads had replaced much of the old wagon trail in the Oldman and Livingstone valleys. This level of disturbance placed the upper Crowsnest valley and two additional tributary watersheds at moderate risk. Risk ratings for the remainder of the watersheds, including those in the Oldman-Livingstone basin, were unaffected by the additional development.

By the 1950s landscape disturbance in the Crowsnest drainage was extensive. Several urban centres now occupied substantial areas of the mainstem

Table 1. Numbers of basins at risk, by decade.

				Decade		
	Risk	1910	1930	1950	1970	1990
Total Study Area	low	88	85	61	8	3
	moderate	2	5	28	68	59
	high	0	0	1	14	28
Highwood	low	10	10	6	2	1
	moderate	0	0	4	8	9
	high	0	0	0	0	0
Willow	low	9	9	7	0	0
	moderate	0	0	2	8	6
	high	0	0	0	1	3
Oldman-Livingstone	low	44	41	40	4	1
	moderate	1	1	5	36	33
	high	0	0	0	5	11
Crowsnest	low	25	22	8	2	1
	moderate	1	4	17	16	11
	high	0	0	1	8	14

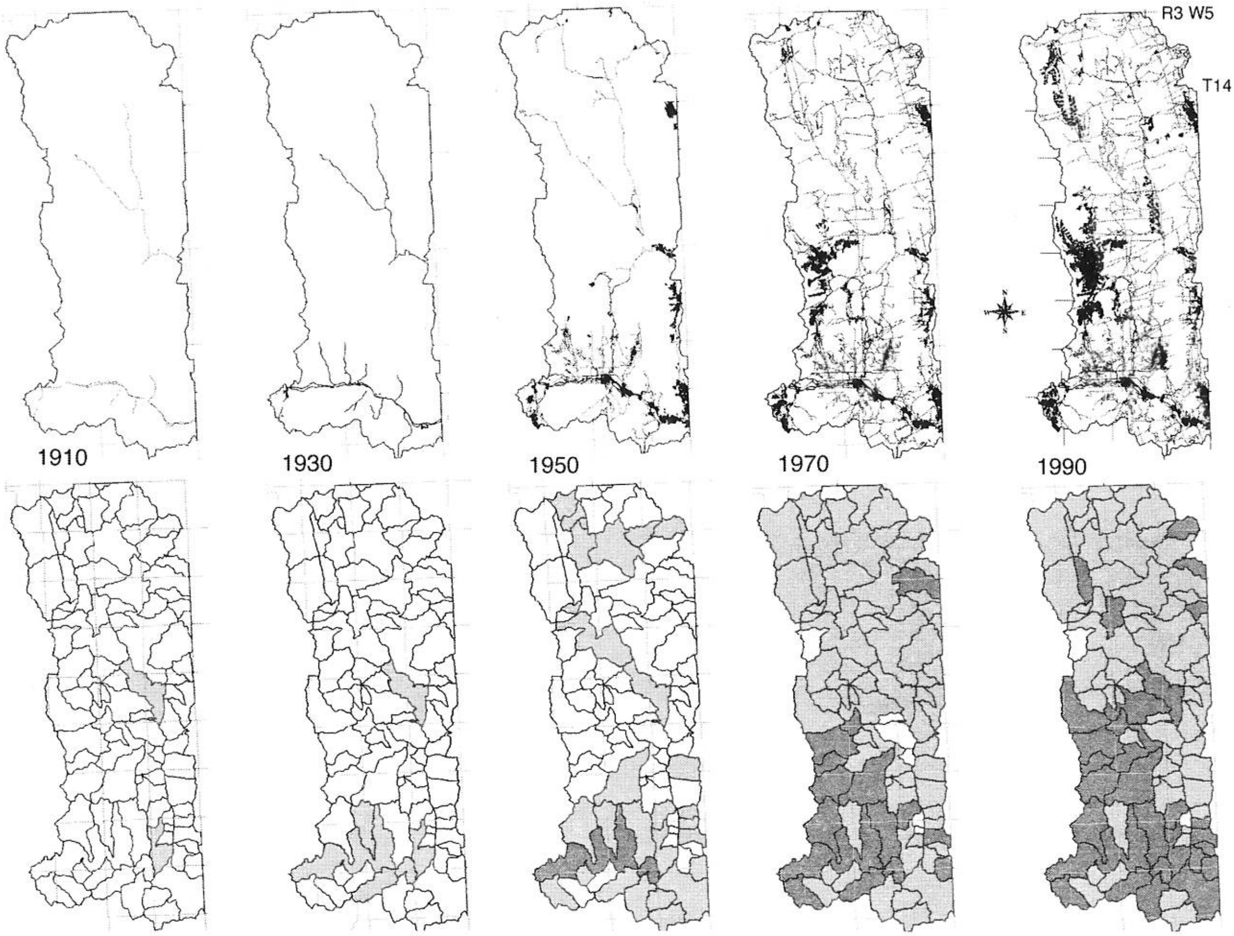

Figure 2. Disturbance history, alternate decades 1910-1990 (top row); watershed risk assessments for the corresponding decades (bottom row). White, low risk; light gray, moderate risk; dark gray, high risk. Township and range lines (9.6 kilometres, or 6 miles) are shown along map margins.

valley floor. An intricate network of roads penetrated up most tributary watersheds. Clearcuts and surface mines had expanded and new ones developed, while cultivated crop and pasture land was extensive in the lower Crowsnest valley and its northern tributaries. The Forestry Trunk Road, a major gravel highway, now traversed the entire length of the study area from north to south. From this road, other roads penetrated to the headwaters of the Oldman, Livingstone, and some headwater tributaries in the Highwood and Willow basins. A large forest block had been cut in the Westrup Creek basin (Willow drainage). Cultivated crop and pasture extended along the southern half of the eastern study area boundary. Risk assessments reflect the substantial increase in human development: by the 1950s, 28 of the 90 basins (approximately 31 percent) were now rated to be moderately at risk, more than half of them in the Crowsnest drainage (Table 1). One, the upper Crowsnest valley, was at high risk.

The effects of a major expansion of the logging industry were evident by the 1970s. Numerous large contiguous clearcuts existed in the upper basins of the Racehorse Creek (Oldman) drainage, in the Westrup Creek basin, and throughout Crowsnest Creek basin, with smaller but still extensive clearcuts in upper Cataract Creek (Highwood drainage), Dutch Creek (Oldman drainage), and various Livingstone drainage basins. Seismic trails for oil and gas exploration crosshatched most of the study area, while the road and trail network had expanded up most major, and a great many minor tributary valleys. Open-pit mines had expanded in the Vicary-Racehorse creek drainages. In response, there was a large increase in the number of basins at risk. Over 90 percent were classed as at risk. Over 15 percent were at high risk, most in the Crowsnest drainage (Table 1).

The remaining years into the 1990s saw continued expansion of commercial logging and the road and trail

Table 2. Numbers of basins at risk by duration of risk. Right column provides the number of basins with the indicated duration of total risk, both moderate and high, over the entire study period

Median Years at Risk	1990s Moderate Risk	1990s High Risk	1910s-1900s Moderate + High Risk
20	5	15	5
40	40	12	52
60	13	1	25
80	1		3
100			2
Total	59	28	87

networks. In the west, large contiguous cutblocks now extended over 20 km north to south and approximately 4 to 6 km east to west, covering the most southerly tributary basins of Racehorse Creek to as far north as Hidden Creek (Oldman drainage). The basin of Crowsnest Creek was now almost completely cut over, and a substantial new clearcut occupied most of the upper Gold Creek valley. An extensive pattern of linked cutblocks occupied most of the upper Oldman and Cataract Creek headwaters, and covered several basins near the Oldman-Livingstone river confluence. The seismic trail and road network extended into virtually every watershed, sparing only the summits of the Livingstone Range and the Continental Divide. As a result, nearly 97 percent of the 90 basins were at risk; more than 31 percent were at high risk (Table 1). Only three basins were considered to be at low risk, one of them — Cache Creek (Oldman drainage) — lying almost wholly within a protected natural area.

The historical analysis allowed us to estimate the period for which each watershed has been at risk. A summary of these estimates for the total study area is supplied in Table 2. These data were compiled by sequential inspection of the risk classification for each basin shown (Figure 2) over the 100-year study period. Basins at low risk (white) were excluded from the calculations. Most basins showed a progression from low to moderate, and often to high risk. Only one basin (Westrup Creek, north end of the east boundary of the study area) showed sufficient recovery during the study period to decrease from high risk (1970) to moderate risk (1990).

The data for each decade represented a composite of data from any time within in each decade of study. For this reason the possible duration of disturbance from one study decade to the next theoretically could have been as short as 11 years or as long as 29 years. We used the median, 20 years, as the best estimate of the duration between each decade studied. The 1890s, the next earlier comparable period in the time series, was taken as the zero-impact decade, although some impact must have occurred with the arrival of the railway in the last two years of that decade.

By the 1990s, the basins in the study area that were at moderate risk (approximately two-thirds of the total), had been at risk for a mean of 43.4 years. The basins at high risk (nearly one-third of the total), had been so for a mean of 30 years. In addition, many of the high-risk basins had previously been at moderate risk for various periods. When the total period of risk is taken into account (Table 2, rightmost column), 87 basins had been at some elevated level of risk for a mean of 47.4 years, ranging from 20 to 100 years.

Discussion

The Level 1 IWAP is a screening procedure designed to identify watersheds at risk as a result of surface disturbances, especially those associated with commercial logging. Watersheds identified at some level of risk typically would be further investigated through air photo interpretation, map work and field studies to determine the extent to which actual damage had occurred (British Columbia Forest Service 1995). This follow-up work remains to be done in our east slopes study area. Nevertheless, observable damage to stream channels and riparian zones should be expected, at least in the high-risk watersheds. In a similar study in the Carbondale River basin, a comparable 300-km^2 watershed abutting the upper Crowsnest basin on the south, we found that all eight sub-basins, and the Carbondale basin as a whole, ranked as high-risk, and all showed extensive channel and riparian damage (Sawyer and Mayhood 1998b).

Despite the very large clearcuts in some basins, the risk of watershed damage from peak-flow increases alone in the 1990s was rated as low for over 70 percent of the basins, moderate for more than 23 percent, and high for just five individual watersheds (Mayhood et al. 1998). In the same study, the risk of surface erosion was rated as high for over 90 percent of the basins, moderate for over 5 percent and low for only three basins. Other risks, including that to riparian zone integrity (87 percent of basins at low risk) and to watershed damage from mass-wasting (all basins at low risk), were also low. Accordingly, the risk of surface erosion from land disturbance is the principal risk to watershed integrity in most of these 90 basins. Here, as in many other forested regions, surface erosion from land disturbance is overwhelmingly attributable to the extent of road development (Waters 1995).

Roads and other linear disturbances constitute major geomorphic features in the landscape, extensions to the drainage network delivering water and sediments into the natural drainage system. In our study area the extent of roads and other linear disturbances approaches the total length of the natural drainage network (Mayhood et al. 1998). As long as they are allowed to persist, forest road networks fundamentally change the delivery of water and sediment to streams (Beschta 1978, Dietrich et al. 1982, King and Tennyson 1984, Reid and Dunne 1984, Chamberlin et al. 1991, Frissell 1997). Transport of water and sediment are the major factors shaping the mainly gravelbed channels in the study area, so if stream channels are to be restored to or maintained in something approaching their natural state, it is the extent of road development that must be controlled in most of our watersheds. We can control the extent of road development by minimizing their persistence, and the duration of the disturbance they create. We can, in other words, build them only when they are absolutely necessary, use them for as short a term as possible, and physically remove them once they have served their primary purpose.

A prudent land management policy in the study area would include plans to physically remove as many roads as necessary and restore their rights-of-way to a more natural condition, so as to reduce high IWAP risk scores at least into the moderate range. For the few basins in which increased peak flow risk is also high, additional efforts may be required to reduce equivalent clearcut area through enhanced reforestation. Alternative scenarios can be screened using the IWAP to find the most efficient ways of achieving these goals. Had it been possible to follow this policy in our study area over the last century, both the degree and the duration of risk to many watersheds could have been substantially reduced.

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Distribution and Abundance of Rio Grande Cutthroat Trout (*Oncorhynchus clarki virginalis*), Relative to an Introduced Salmonid, in Northern New Mexico

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Abstract

Thirty eight streams were surveyed for fish in the Carson and Santa Fe National Forests, of northern New Mexico, during the summers of 1994 and 1995. Seven new populations of Rio Grande cutthroat trout, (*Oncorhynchus clarki virginalis*), were documented, all of which are located in the Carson National Forest. Comparing field survey results with data from the literature and New Mexico Game and Fish records suggest that 86 Rio Grande cutthroat trout populations have become extant in the Carson and Santa Fe National Forests. Of these 86 populations, only 52 were determined, based on morphometric and genetic analyses, to be genetically pure populations. Only 14 of the 52 streams did not contain nonnative salmonids, such as the highly prevalent brown trout (*Salmo trutta*). Results from our study suggest that brown trout express greater density and biomass when in co-occurrence with native species. Management recommendations are suggested.

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Introduction

Rio Grande cutthroat trout, (*Oncorhynchus clarki virginalis*), are native to southern Colorado and New Mexico and are a member of the southern inland cutthroat trout complex (Sublette et al. 1990; Behnke 1992). The subspecies is believed to have evolved from the Colorado River cutthroat trout (*Oncorhynchus clarki pleuriticus*) and is closely related to the Greenback cutthroat trout (*Oncorhynchus clarki stomias*). Native populations of Rio Grande cutthroat trout occur in upper headwaters of the Rio Grande drainage of southwest Colorado and in headwaters of four drainages in New Mexico; the Rio Grande, the Pecos, and the Canadian, including the Mora. It is likely that the Rio Grande cutthroat trout occupied all waters presently capable of supporting trout in the Rio Grande drainage as well as the Canadian and Pecos River drainages of New Mexico and south-central Colorado (Sublette et al. 1990). Stumpff and Cooper (1996) speculate that this distribution may have covered approximately 40 hydrologic sub-basins in Colorado and New Mexico.

Once widely distributed throughout the Rio Grande, Pecos, and Canadian drainages the species now occupies a small proportion of its original range. In New Mexico, it is estimated that Rio Grande cutthroat trout occupy only about 5 - 7% of its original range (Sublette et al. 1990; Stumpff and Cooper 1996). Streams occupied by Rio Grande cutthroat trout are, for the large part, small headwater streams where productivity is low, stream gradients are high (> 4%), and connectivity to other tributaries is almost nonexistent.

Declines in both range and abundance of Rio Grande cutthroat trout are likely due to the widespread stocking of nonnative salmonids. Habitat loss and degradation, through water diversions, grazing, logging, and mining are also likely contributing factors to this decline, but are much less well documented and studied. Overexploitation by sport fishing has also been cited as a contributing factor to the decline of Rio Grande cutthroat trout (Behnke 1991).

Management programs developed to conserve this rare southwestern cutthroat trout subspecies have been implemented by the New Mexico Department of Game and Fish (NMDGF) and the United States Forest Service (USFS) and have been ongoing for several decades. Most of this effort has been allocated towards assessing the distribution and purity of stocks. Distributional lists and the genetic purity of Rio Grande cutthroat trout stocks, based upon point in-time genetic and morphometric studies, are available (Behnke 1980; Behnke 1982; Riddle and Yates 1990; Hartman et al. 1991; Davis and Yates 1992; Stumpff and Cooper 1996), however, these lists do not address presence or absence of non-native, introduced trouts, especially brown trout (*Salmo trutta*). Although brown trout are reproductively isolated from the cutthroat trout, its impact on the native cutthroat may be largely based on its competitive impact on the native Apache trout (*Oncorhynchus apache*) in Arizona (Rinne et al. 1981; Carmichael et al. 1995) and it may effectively replace the Rio Grande cutthroat trout.

The objectives of this paper are to delineate the historic and current distribution of the Rio Grande cutthroat trout, geographically define populations of pure cutthroat trout, document the location of new populations of putative Rio Grande cutthroat trout, report streams where brown trout are sympatric with pure populations of the cutthroat trout, and suggest through relative abundance data, the potential impact of brown trout on the Rio Grande cutthroat trout.

Study Area

The study area consisted of the Carson National Forest (CNF) and Santa Fe National Forest (SFNF), both of north-central New Mexico (Figure 1). Landscapes are generally mountainous with elevations ranging from 1,708 m in low elevation grasslands to at 4,011 m in alpine landscapes such as Wheeler Peak. Precipitation varies from 25.4 cm to 88.8 cm/year with the greater amounts falling at the higher elevations.

Methods and Materials

Smith-Root Model 12 backpack electrofishing gear was used to sample fish fauna of 38 streams in the Rio Grande drainage of New Mexico (i.e., 23 within CNF and 15 within SFNF) between June and August of 1994 and 1995. To determine the hypothesized negative impact of brown trout on Rio Grande cutthroat trout two dozen 50-meter study sections were established in reaches where brown trout were found in sympatry with Rio Grande cutthroat trout and in sections occupied only by Rio Grande cutthroat trout. Relative abundance estimates for each species were determined using Zippen (1958). Information on current distributions was gathered from the records of the New Mexico Department of Game and Fish (NMDGF) and United States Forest Service. Morphometric, electrophoretic, and mtDNA analyses were provided by the NMDGF. These determinations were made by NMDGF personnel and scientists contracted by the NMDGF during the 1970's, 1980's, and 1990's.

Results

Distribution

After review of agency records and the results of the 1994/1995 field-season, 86 populations of Rio Grande cutthroat trout were identified for the CNF and SFNF (Table 1 and 2). Of these, only 52 populations were recognized by morphology, meristics, and electrophoresis as genetically pure (i.e., A Grade, 95-100%) (Table 1 and 2). Introgression in the remaining populations varied from low (i.e., B grade, 75- 95%) to high (i.e., less than 25% pure) (Stumpff and Cooper 1996).

Of the 38 streams surveyed through field investigations, seven were identified within the CNF as new distributions for Rio Grande cutthroat trout: Osha Canyon (446050E,4002240N), Comales Creek (447750E,4001190N), Agua Piedras Creek (452640E,3998770N), Rio de las Trampas (429450E,4001150N), Rio San Leonardo (439360E,3988900N), Italianos Canyon (439360E,3988900N), and Yerba Canyon (453430E,4046970N). Definition of Rio Grande cutthroat distribution was expanded for two streams on the SFNF; Rito de Las Palomas and its tributary, American Creek. Records indicated presence of the cutthroat only on private holding in the lower reaches of Rito de las Palomas, although our 1994 and 1995 field studies revealed populations of Rio Grande cutthroat trout extant to the upper reaches of both streams. Records of Rio Grande cutthroat trout in Canjilon Creek, Tienditas Creek, Frijoles Creek, Rito de la Olla, Cabresto Creek (CNF) and Canoncito Creek (SFNF) were also substantiated.

Interactions with brown trout

Relative density and biomass estimates were made in June 1995 for allopatric populations of Rio Grande cutthroat trout and for populations in sympatry with brown trout. These populations were located in multiple sections of three of the 52 streams containing pure Rio Grande cutthroat. An inverse relationship in abundance and biomass between Rio Grande cutthroat trout and brown trout was noted in study sections where both co-existed. When Rio Grande cutthroat trout were not in co-occurrence with brown trout both abundance and biomass increased significantly (Table 3).

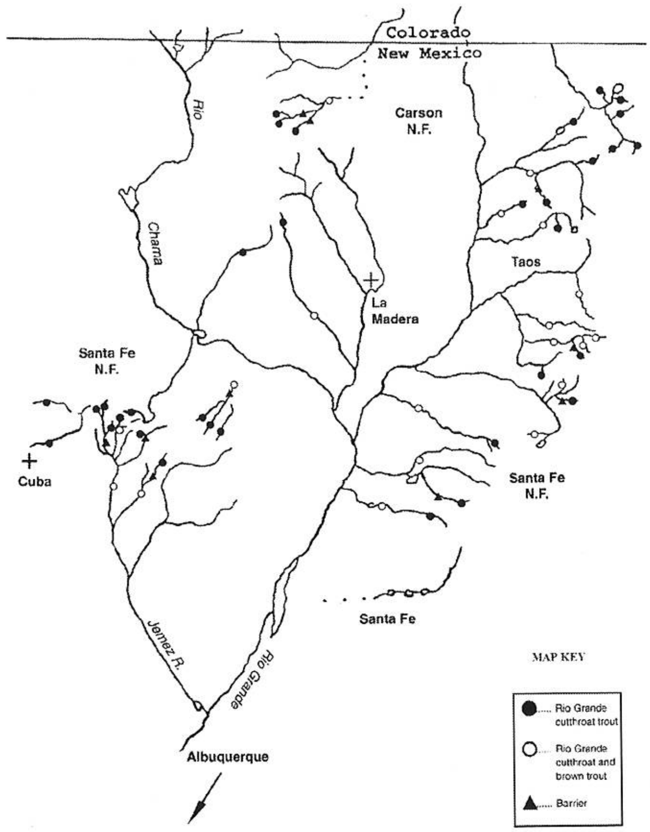

Figure 1. Study area including the distribution of Rio Grande cutthroat trout relative to brown trout in the Rio Grande Drainage of the Carson and Santa Fe National Forests, New Mexico, in 1998.

Discussion

Of the approximately 52 streams in New Mexico that contain pure populations, only 14 (27%) were occupied solely by the native cutthroat, conversely, 38 streams (73%) had brown trout co-occurring (i.e., usually in lower reaches) with Rio Grande cutthroat and of these only 20 (53%) had pure populations that were protected from upstream migration of non-native trout (Figure 1). Stumpff and Cooper (1996) suggested that only four populations of Rio Grande cutthroat trout were secure and stable, while three times that were either "at risk or unknown." These authors also listed 70 of 80 cutthroat populations in Colorado and New Mexico as co-existing with non-native salmonids. Most notably, brown trout were listed as the co-inhabitant in nearly half (i.e., n = 32) of these cases.

Introduced salmonids have been suggested to negatively impact native trout species in the West and

Table 1. The distribution and analysis of Rio Grande cutthroat trout distributions in Carson National Forest, New Mexico. Abbreviations: Ck = Creek, Morph. grade = Morphometric grade, Electro grade = Electrophorectic grade, UTM = Universal Transverse Mercator coordinates. Genetic purity determinations were determined by the New Mexico Department of Game and Fish and their contractors.

Waterbody name	UTM	Morph. grade	Year	Electr. grade	Year	mtDNA grade	Year
Agua Caliente	432630E,4012360N						
Agua Piedras	452640E,3998770N						
Alamitos Ck	456390E,3995770N	A	1982				
Bitter Ck	464030E,4061980N	B	1980	Pure	1992		
Cabresto Ck	459460E,4066150N	C	1975	Low level cross	1992		
Osha Canyon	446050E,4002240N						
Canjilon Ck	372600E,4079640N	A		Pure	1992		
Columbine Ck	454140E,4059410N	A	1980	Pure	1992		
Comales Ck	447750E,4001190N						
Comanche Ck	471700E,4076000N	A-	1982	Low level cross	1994	Pure	
El Rito (Lower)	387570E,4029110N	B	1980	Low level cross	1991		
El Rito (Upper)	-	B	1980	Pure	1991		
Italianos Ck	455620E,4048670N						
Jarosa Ck	454000E,4006100N	A	1982	Crossed	1997		
Jaroso Ck	454020E,4005800N	A	1982				
La Cueva Ck	453390E,3998410N	A-	1982				
Lake Fork of Cabresto Ck	454420E,4065370N	B	1980	Pure	1991		
Leanardo Ck	439360E,3988900N						
Little Costilla	473460E,4072060N	A-	1982				
Luna Ck	467880E,4007510N	B	1980	Low level cross	1990	Pure	1990
McCrystal Ck	489510E,4070220N	A-	1982	Unique allele	1990	Pure	1994
Palociento Ck	459950E,4012750N	A	1980				
Policarpio Ck	458880E,3999620N			Pure	1991		
Powderhouse Ck	474430E,4080460N	A-	1982				
Rio Chiquito	463950E,4018350N	A	1975				
Rio Costilla	466230E,4077770N			Low level cross	1994		
Rio de la Presa	459210E,4002330N	B	1980				
Rio de las Trampas	429450E,4001150N						
Rio de Truchas	433400E,3987980N			Pure	1994		
Rio Hondo (SF)	454620E,4047560N	A	1980				
Rio Nutrias	394520E,4078370N			Pure			
Rito Angostura	456390E,3995700N	B		Low level cross	1991		
Rio de la Olla	463000E,4014400N	A					
Rio Frijoles	462900E,4018350N	A					
Rito Primero	450920E,4071000N	C	1980				
Saloz Ck	452910E,4007390N	A	1980				
San Cristobal	444900E,4051720N	A	1980	Pure	1990	Pure	1990
Sardinas Ck	459210E,4002350N						
Sawmill Ck	465600E,4050430N	F	1980				
Tanques	391120E,4075350N			Low level cross	1990	Pure	1990
Tienditas Ck	466550E,4025050N	B-	1980	Pure	1991		
Tio Grande	393080E,4073860N	B		Low level cross	1990	Pure	1990
Agua Caliente	432630E,4012360N						
West Latir Ck	451200E,4075850N	F	1980				
Yerba Ck	453430E,4046970N						
Vidal Ck	475950E,4067800N			Pure			

Table 2. Distribution and analysis of Rio Grande cutthroat trout distributions, Santa Fe National Forest, New Mexico. Abbreviations: Ck = Creek, Morph. grade = Morphometric grade, Electro grade = Electrophorectic grade, UTM = Universal Transverse Mercator coordinates. Genetic purity determinations were determined by the New Mexico Department of Game and Fish and their contractors.

Waterbody name	UTM	Morp. grade	Year	Electro. grade	Year	mtDNA grade	Year
American Ck	338460E,3984710N	A		Pure	1990		
Apache Ck	369230E,4000100N						
Canoncito Ck	464300E,3985850N	C		Crossed			
Canones Ck	369970E,4001820N	A	1982	Pure	1990	Pure	1990
Cecilia Ck	338320E,4007360N	B	1977	Low level cross	1991		
Chihuahuenos Ck	368560E,3999540N	A	1975				
Clear Ck	338250E,3984360N	A	1975	Crossed	1990		
Dalton Ck	433750E,3947520N			Pure			
Doctor Ck	436970E,3958190N			Pure	1992		
Indian Ck	436950E,3952550N	A	1975	Low level cross	1993		
Jack's Ck	440880E,3964520N			Pure	1992		
La Jara Ck	327830E,3999340N	C	1977	Low level cross	1992		
Macho Ck	436010E,3948680N			Pure	1994		
Oso Creek	340990E,399820N	A	1976	Pure			
Peralta Ck	364720E,3951530N	B+	1982	Low level cross	1990		
Polvadera Ck	371980E,4000710N						
Polvadera (EF)	370060E,3994370N	A	1978	Pure	1992		
Quemado Ck	436520E,3984390N			Pure	1994		
Rio Capulin	335820E,4004460N	F	1977	Crossed	1991		
Rio Capulin	423520E,3966730N			Low level cross	1994		
Rio de la Cebolla	338490E,3965220N			Pure	1994		
Rio de las Vacas	337030E,3984370N	A		Pure			
Rio Frijoles	421310E,3977200N			Pure	1994		
Rio Nambe (Lower)	421050E,3967650N			Low level cross	1994		
Rio Nambe (NF)	429080E,3964020N	F	1975				
Rio Nambe (Upper)	428700E,3964050N			Pure	1994		

Southwest (Rinne 1985, 1988; Rinne and Minckley 1985, Behnke 1992). Preliminary information suggests this impact may be great (Table 3). In American Creek, Rito Cafe, and the Rio de las Vacas there appears to be an inverse trend of density and or biomass between the native cutthroat and the introduced brown trout. This impact may come through the mechanisms of direct predation and competition for food and space (Rinne 1994, 1995).

To conserve the Rio Grande cutthroat trout resource, managers of all agencies must have the latest information on this cutthroat subspecies's distribution and status. Distribution and population data for this rare southwestern trout are dynamic. The seven newly identified populations of Rio Grande cutthroat trout during the 1994 and 1995 field-seasons suggests that despite almost two decades of effort on this Southwestern cutthroat subspecies, much is yet unknown about its distribution and status. Continued effort toward defining the distribution of this subspecies in New Mexico and Colorado is needed. Accordingly, there is an urgent and continuing need to define the extent and degree of impact of introduced salmonids, especially brown trout. Definition of population dynamics (e.g. fecundity, recruitment, age class structure, and longevity) is needed to begin the

Table 3. Relative density (Number/hectare) and biomass (kilogram/hectare) of Rio Grande Cutthroat trout (RGCT) and brown trout (BT) in study reaches in American Creek and Rito Cafe, Rio de las Vacas, Santa Fe National Forest, 1995.

	Cutthroat trout		Brown trout	
Study section	**Density**	**Biomass**	**Density**	**Biomass**
American Creek: RGCT purity – pure				
1	324	9	279	22
2	702	32	0	0
3	90	6	90	13
4	488	13	195	9
5	300	9	486	12
6	89	5	714	58
7	56	5	617	46
Rito Café: Below Barrier; RGCT Purity - Pure				
1	676	14	2801	241
2	400	15	2300	18
3	0	0	3200	144
4	0	0	3875	96
Rito Café: Above Barrier; RGCT Purity – Grade A				
5	2545	30	91	30
6	2608	36	0	0
7	1080	22	83	5
Rio de las Vacas: Below Barrier; RGCT Purity – Grade A				
1	0	0	318	18
2	117	3	233	10
3	0	0	595	60
4	156	2	739	43
5	0	0	1468	84
6	61	1	1908	119
Rio de las Vacas: Above Barrier; RGCT Purity – Grade A				
1	3905	133	0	0
2	3386	121	0	0
3	2548	121	0	0

development of population viability analyses that will determine the potential long-term persistence of pure populations of Rio Grande cutthroat trout.

Management for the sustainability of this rare Southwestern salmonid should included the construction of migrational barriers in drainages containing currently unprotected populations of Rio Grande cutthroat trout. Reintroduction of this native fish into waters of its historic range should also be a priority. To facilitate gene flow and enhance connectivity among currently isolated cutthroat populations, the reintroduction effort should not be confined to small unproductive headwater streams but should be undertaken on a watershed scale.

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Effects of Industrial Activity on Bull Trout Populations in Alberta's Boreal Forest: An Evaluation of Current and Future Impacts

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Extended Abstract

Developing strategies that maintain the biological diversity amidst increasing levels of industrial development is arguably the single largest challenge to sustainable management of Boreal forest ecosystems. For bull trout (*Salvelinus confluentus*), these activities can profoundly impact the quantity and quality of habitat, in part, resulting from commercial forest harvesting and exploration and extraction of oil and gas resources that alter forest composition and age and produce road networks that can result in sediment inputs, create barriers to fish passage and provide anglers with increased access to remote areas. Along with similar studies, we use logistic regression to identify features that are important in the determination of presence and absence of bull trout in the Kakwa River watershed. Combined with Geographical Information Systems, we use these features to predict bull trout occurrence across the watershed under varying scenarios of land use practices.

The broader question on the strength of the relationship between bull trout occurrence and abundance is poorly understood. There is a moderately weak relationship between bull trout presence and their abundance. Unfortunately, the use of the logistic model to analyse abundance data can be problematic due to the preponderance of zero counts that result in over-dispersion of the data. Events with count data in nature tend not to conform to a Poisson distribution because of this over-dispersion. In an attempt to identify if the logistic model predicting presence is also useful to predict abundance, we use the Zero-Inflated Poisson (ZIP) model, which is useful to analyse the non-zero count data when compared with the logistic model describing occurrence.

Ripley, T.D., Boyce, M. and Scrimgeour, G. J. 2004. Effects of industrial activity on bull trout populations in Alberta's boreal forest: an evaluation of current and future impacts. Pages 39-40 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

The logistic model predicting bull trout occurrence was best described by a positive relationship with stream width and a strong negative association with the amount of timber harvest and deposition of fine sediment in stream channels. Timber harvest and road densities were negatively related to bull trout presence in many candidate models. We further extend the model to simulated increases in timber harvest across the watershed to evaluate the predicted response of bull trout occurrence.

Analysis of the abundance data resulting from the ZIP models indicates that variables important for predicting presence of bull trout are not necessarily the same variables associated with abundance. When compared with the best logistic model, we found that variables associated with stream width, stream substrate (fine sediment and cobble), distance fished and elevation to be important in predicting abundance. It is important to note that no industrial type disturbance variables were strongly associated with bull trout abundance. This is a likely result of these variables having a strong negative relationship with bull trout presence (i.e. industrial disturbance variables linked to sites with zero captures). Based on these results, it appears clear that abundance of bull trout is better predicted from habitat type variables, rather than industrial footprints on the landscape. A strong negative relationship between bull trout presence and these variables suggests further investigation of a direct linkage to bull trout habitat is important. Land managers should take a cautious approach to industrial development in the Boreal forest, with an understanding that the quality of habitat and quantity of bull trout presence may decline.

Influences of Basin, Stream Reach and Land-Use Characteristics on the Distribution of Bull Trout, Rainbow Trout and Brook Trout in Selected Foothills Model Forest Watersheds

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Extended Abstract

We used resource selection function (RSF) models and an information-theoretic approach to identify best models for explaining presence/absence of three fish species within headwater streams in the foothills of Alberta. Our objectives were to: i) quantify the relative importance of a number of ecological factors on fish species distributions; and ii) determine if specific land-use activities were associated with a reduced probability of occurrence. Our focal species were bull trout (*Salvelinus confluentus*), rainbow trout (*Oncorynchus mykiss*) and brook trout (*Salvelinus fontinalis*). Presence/absence information was collected using backpack electrofishing at candidate reaches representing the range of physiographic conditions across the study area. We identified four ecological factors and three land-use variables of potential importance. Using GIS, we determined the values for these potential explanatory variables at each sample location. The ecological variables included basin area, basin slope, reach slope and reach elevation. The land-use variables included percent of basin harvested, basin road density and the presence/absence of a downstream barrier. Based on knowledge and experience, we identified a number of candidate models from which the best model was identified using a standard Akaike Information Criteria (AIC) approach.

The best bull trout model had an ROC score of 0.857, representing intermediate accuracy. In order of relative importance, the variables in the model were basin slope, reach elevation, basin area, downstream barrier present, reach slope and percent of basin harvested. Both of the land use variables had a negative influence on probability of bull trout capture, however the presence of a downstream barrier had much greater effect than percent of basin harvested. The best rainbow trout model had an ROC score of 0.806, indicating intermediate model accuracy. In order of relative importance, the variables were basin slope, basin area, reach elevation, percent of basin harvested and basin road density. Both of the land use variables had

McCleary, R.J. 2004. Influences of basin, stream reach and land-use characteristics on the distribution of bull trout, rainbow trout and brook trout in selected Foothills Model Forest watersheds. Pages 41-42 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

a positive influence on probability of rainbow trout capture, however basin road density had a much greater effect than percent harvested. The best brook trout model had an ROC score of 0.910, indicating high model accuracy. In order of relative importance, the variables in the model were basin area, percent of basin harvested, reach slope, reach elevation, basin slope and basin road density. The land-use variables had differing influences on probability of brook trout occurrence, as harvest was associated with an increase in probability of occurrence, while roads were associated with a decrease in occurrence.

Results indicate that ecological variables had high relative importance in all models. These findings have two implications. First, they indicate that reasonable models of fish occurrence may be developed from GIS-generated ecological information, which in turn may be used to map predicted species distributions. Second, selection of sample locations for future modeling exercises could consider a range of ecological factors. Results also suggest that removal of fish migration barriers can help meet the goal of long-term conservation of bull trout.

Evaluating the Cumulative Effects of Industrial Activities on Stream Fish Communities in two Boreal Forest Watersheds of Alberta, Canada: A Score Card Approach

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Extended Abstract

We evaluated the cumulative effects of industrial activities on stream fish communities in two northern watersheds in Alberta, Canada. We quantified effects of industrial activities by determining whether current levels of industrial activity explain statistically significant variance in: i) fish presence, ii) fish density and biomass and iii) fish community structure.

Our analyses showed that the presence of fish, individual fish species and fish species groups were predictable based on watershed and stream reach attributes including watershed area, stream size (measured as stream bankfull width), and the elevation and slope of stream reaches. With two exceptions, metrics that defined industrial activities (e.g., cumulative percent of the watershed as roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines) were neither statistically significant nor powerful predictors of fish presence. The two notable exceptions were the statistically significant and negative relations between the presence of bull trout and: i) the cumulative percent of the watershed disturbed in one study watershed and ii) the cumulative density of stream crossings in the second study watershed. Variation in total fish biomass and density and the biomass and density of individual species were also typically related to stream size, reach elevation, stream reach slope, and the size of materials on the river bed. Metrics related to industrial activities seldom explained significant or substantial variance in total fish density and biomass and the biomass and density of individuals species and species groups. Analyses of the effects of industrial activities on the structure of stream fish communities were evaluated using a Reference-Condition Approach. Using this approach, we developed empirical models

Scrimgeour, G.J., Hvenegaard, P.J. and Tchir, J. 2004. Evaluating the cumulative effects of industrial activities on stream fish communities in two boreal forest watersheds of Alberta, Canada: a score card approach. Pages 43-44 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

predicting fish community structure from reference, that is, least impacted stream reaches (i.e., where levels of industrial activities accounted for < 10% of the total area of individual sub-watersheds) and asked to what extent do these empirical models predict fish community structure at potentially impacted sites (i.e., where levels of industrial activity accounted for >10% of sub-watersheds). These community-level analyses identified three and five fish community types in the reference stream reaches in the two study watersheds and that these community types were predictable using the re-occurring suite of watershed and stream reach variables. However, these empirical models were poor predictors of fish community structure at the potentially impacted sites (i.e., where levels of industrial activities accounted for < 10% of the total area of individual sub-watersheds). This finding suggests that current levels of industrial activity result in detectable impacts on the structure of stream fish communities in both study watersheds. This result arises because while the effects of industrial activity are associated primarily with bull trout, this species is a numerically important species in both study basins. Taken together, our data suggest that current levels of industrial activity alter stream fish communities and that alternative management practices need to be evaluated.

Access Management

Effects of Road Crossings on Beaver Pond Dynamics in the Boreal Mixedwood

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Extended Abstract

Riparian habitats are some of the most critical wildlife production areas in the Boreal Plains ecozone. They are the interface between terrestrial and aquatic ecosystems and are influenced by both anthropic and natural landscape alteration. Road development, an alteration of the terrestrial landscape, is directly linked to increased surface-water runoff, changes in subsurface flow and increased sediment release into streams. At stream crossings, culvert structures can also change surface and subsurface flows and sometimes create impoundments. The beaver (Castor canadensis) creates and maintains riparian vegetation and wetlands by altering: i) the aquatic landscape, through damming activity and causing the impoundment of water, and ii) the terrestrial landscape, though foraging for food and construction material. When beaver dams or culverts impound water: i) wetland/riparian plant species replace upland species, ii) flooded trees die, hence snag numbers increase, and iii) the area of open surface-water increases.

Based on field observations, it is evident that beavers tend to build dams in culverted low-order streams as well as in non-culverted stream networks. It was hypothesized that watersheds with culverted crossings would show an effect on beaver dam abundance and areal extent of open surface water, as compared to watersheds without road-crossing development. The mechanism for these phenomena could be an alteration of beaver damming behaviour due to the presence of road crossings.

In north central Alberta, a total of 21 third-order watersheds were selected as the unit of

Flynn, N., Foote, A.L. and Cumming, S. 2004. Effects of road crossings on beaver pond dynamics in the boreal mixedwood. Pages 47-48 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

analysis, with 11 of the watersheds having high road-crossing densities (per kilometer of stream) and 10 watersheds with low to no road crossings. Aerial photography, at a scale of 1:15,000, was collected and georectified for all selected watersheds over two time frames (i.e., years 1976-78 and 1998-2000). Respectively, these intervals represented pre- and post-road development periods. Ground-truthed sites enabled the validation of photo-interpreted riparian wetland characteristics. Beaver dam characteristics and parameters related to areal extent of open surface water were measured as change in: i) breached dam/intact dam ratio, ii) total open surface water area (m^2/km of stream) of beaver ponds, iii) number of intact beaver dams/km of stream and iv) number of breached beaver dams/km of stream._

Preliminary results show that both road crossings and beaver damming activity affect riparian and upland vegetation communities throughout low-order drainage networks. Impounded water increased in low-order streams associated with road crossings, possibly because beaver concentrated damming activity near road crossings; culverted crossings function as partially constructed dams and created more suitable damming sites. Another possibility is that subsurface water flow through the roadbed was impeded by soil compaction, redirecting subsurface water to the surface.

Road density is expected to increase in northcentral Alberta, and arguably so will stream crossings. Based on results from this study, an increase in road crossing density could have an impact on beaver abundance and distribution in low-order drainage networks, thus changing the distribution and characteristics of associated wetlands and riparian communities. Landscape simulations, under current management policies in the study area, predict that riparian areas will hold the majority of old growth forests in the future. Future landscape simulation models will incorporate the results of this study to examine what effect road crossing density and beaver activity may have on riparian vegetation under different management practices. This will then, in turn, influence riparian forest management and infrastructure planning decisions in low-order stream networks.

Development, Installation and Testing of the Enviro-Span Non-Toxic Archway Culvert for Fish Stream Crossings

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Extended Abstract

For many decades, fish bearing and/or fish habitable streams have been crossed using closed bottom circular culverts of metal fabrication, known as "corrugated steel pipe" (CSP). While these have been economical, practical and reliable from the point of view of the road, they present many inherent problems with respect to the sensitivities of fish and other aquatic life, or to the quality of local habitat. With ever increasing developmental pressures on our natural heritage, the importance of sustaining viable fish habitat, whether fish are present or not, has become paramount.

In more recent years, open bottom CSP culverts have been used with some frequency. These do much to retain a near natural stream bed. However, there are a number of issues pertaining to the installation phase such as short term disturbance of the channel substrate, shoreline and riparian areas. Furthermore, steel (CSP) is highly susceptible to material degradation in the natural environment, and is therefore commonly treated with Zinc or Aluminized coatings to lengthen the service life. Natural processes eventually cause the potentially harmful leaching of cations (Zn, Al, Fe) into the water. A different approach to stream crossings therefore seems appropriate.

The Enviro-Span archway is a non-corrosive, non-metallic and non-toxic modular crossing system that is designed for use over fish bearing or other environmentally sensitive streams. Enviro-Span archways are an open bottomed arch style crossing that sustains natural open channel flow with no special sub-grade preparation or foundation elements necessary. The system is held together with floating caps… no bolts or clamps. The modules are identical and interchangeable. Their flexible nature allows them to follow significant vertical irregularities without the need for special levelling and/or grade beams. Enviro-Span Technology Inc. developed the design for the archway and received a Canadian patent in January 2002 and USA patent in November 2002. Operational installation and testing was then carried out in the

Hammerstedt, R.W. 2004. Development, installation & testing of the Enviro-Span non-toxic archway culvert for fish stream crossings. Pages 49-50 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

winter of 2001/2002. An 1800 mm diameter modular installation was made for an operational resource road on a stream with fish bearing potential in the Slim Creek area of the central Rocky Mountain Trench. This installation was fitted with seven, 120 Ω strain gauges in critical load bearing areas. There was a minimum fill of one metre over the crossing, and testing was carried out with an L-75 load. The load was carried on a self-loading log truck. Test results indicated that strain fell very significantly under the allowable design deformation level. These results were corroborated by laboratory testing and thus indicate a good margin of safety for the arch culvert. Further to this, the arch has been in place for the complete development and harvesting program for a high volume harvest block. The operational performance has more than met expectations. Details of the results will be presented.

Restoration of Fish Passage at Elevated Culverts: Examples From CN's Mainline Operations in Western Canada

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Extended Abstract

The Canadian National Railway Company (CN) which operates across North America has an ongoing program to identify and prioritize environmental issues along their lines. As with roads, fish passage at railway stream crossings is sometimes restricted, especially if over time culverts become elevated above the stream channel bed. Two recent projects by CN illustrate options, other than full replacement, for restoring fish passage at such culverts.

In both cases the culverts could not be modified without compromising their flow capacity beyond reasonable limits. The first example, Paul Creek, near Kamloops British Columbia crosses the CN track through two 1300 mm CSP culverts that had become elevated by about 2 m above the channel bed. The target fish species for upstream passage were coho salmon during the fall (low-flow period) and rainbow and steelhead trout during the spring (high-flow period). CN's design team created fish passage for the target species by modifying an existing bypass channel, including relocating a portion of the channel, installing riffles in the channel and baffles in a culvert, and constructing a passive flow control structure at the bypass channel inlet. The project was completed in 2001 and coho have been observed spawning upstream of the crossing each year since then.

The second example, Hardisty Creek, in Hinton Alberta involved a more typical solution to a fish passage concern. Hardisty Creek passes through two 2 m concrete box culverts. Since their installation in 1927 the culverts had become elevated by nearly 2 m above the channel bed, blocking upstream migration by the target fish species, rainbow trout and bull trout. In order to create fish passage a 40 m long riffle was constructed in 2003 to raise the channel bed gradually (15:1 slope) to the culvert invert. The riffle was constructed in layers, with compacted fill on the bottom covered by geotextile and finally a cobble/ boulder layer.

Phillips, B. and Patterson, L. 2004. Restoration of fish passage at elevated culverts: examples from CN's mainline operations in western Canada. Page 51 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Stream Crossing Inventories in the Swan and Notikewin River Basins of Northwest Alberta: Resolution at the Watershed Scale

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Abstract

Aquatic habitat fragmentation, degradation and encroachment resulting from industrial activities can alter the distribution, abundance and subsequent viability of stream fish populations. Using GIS tools and field assessments we documented crossings of streams by road networks and assessed their potential to fragment stream habitats and to act as sources of sediment intrusions into stream channels in two boreal watersheds in Alberta, Canada. Our data showed that the density of stream crossings in the Swan River watershed (0.24 crossings / linear stream km) was about three times higher than that in the Notikewin River watershed (0.068 crossings / linear stream km). Culverts were the predominant crossing structure in both basins and occurred predominantly on first to third order streams. Our field assessments suggest that the majority of culvert crossings in the Notikewin (61% of all crossings assessed) and Swan basins (74%) have the potential to fragment stream habitats. Culverts can obstruct fish passage through various processes. Undersized or improperly graded culverts can result in velocity barriers and cause downstream scouring, resulting in a perched culvert. Assuming that these structures impede fish passage, culvert barriers could result in limited access to about 20% and 9.5% of the total length of stream habitats in the Swan and Notikewin watersheds, respectively. Assessments also identified many culverts less than bankfull width that reduce the surface area of benthic habitats. From our qualitative assessments relatively few culverts in the Notikewin (17%) and Swan River watershed (18%) potentially contributed moderate levels of silt to stream channels. The propensity of culverts and bridges to potentially contribute high amounts of sediment to stream channels in the Swan (19% of all culverts; 36% of all

Tchir, J. P., Hvenegaard, P. J. and Scrimgeour, G. J. 2004. Stream crossing inventories in the Swan and Notikewin river basins of northwest Alberta: resolution at the watershed scale. Pages 53-62 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

bridges) was higher than that in the Notikewin River watershed (3% of all culverts; 8% of all bridges). Taken together, our preliminary assessments suggest that fragmentation of stream habitats and sedimentation related to development of road networks is an important management issue and that additional efforts are required to better understand the effects of stream crossings of stream fish communities in northern boreal systems.

Introduction

The development of road networks is an inevitable outcome of any land-based development activity and has been shown to impact watershed health (e.g., Haskins and Mayhood 1997). Stream crossings can cause both immediate and longer-term effects on fish populations primarily by modifying water quality, substratum composition and fragmenting stream channels (Adams and Whyte 1990; Toepfer et al. 1998; Eaglin and Hubert 1993). Connectivity of fish habitats is considered to be critical in conserving the distribution and abundance of stream fish assemblages (Rieman and McIntyre 1993), while reductions in connectivity impede fish movements and alter fish community structure and likely threaten population viability (e.g., Morita and Yokota 2002).

Expansion of industrial activities in Alberta's boreal forest and the concomitant development of road networks have raised questions about the sustainability of stream fish communities. Like elsewhere, the extent that road networks impact stream fish communities in boreal ecosystems of Canada are not well understood. This information is required to evaluate the need to develop: i) remedial actions and ii) alternative management practices.

The objectives of the present study were to describe the density, type and condition (i.e., fish passage and erosion potential) of stream crossings in two watersheds in the boreal forest of northwest Alberta. Specifically, we addressed the following questions: 1) What is the density of road crossings?, 2) What is the predominant road crossing structure?, 3) To what extent do road crossings affect stream habitat? Based on watershed morphometry, we predicted that culverts would be the dominant crossing type and that the likelihood of culverts to impede fish passage would be sufficiently high to have potentially detrimental effects on stream fish populations. We also expected that stream crossings would likely pose risks to stream fish mediated by sediment intrusions.

Methods

Study Area and fish communities

The study was conducted in the Notikewin and Swan River basins located in northwest Alberta, Canada. The Notikewin River Watershed is located northwest of Peace River and drains an area of 9,799 km^2. GIS layers derived from Indian Remote Sensing II Satellite imagery collected in 1998 showed that the watershed includes 1,665 km of permanent roads and a total of 544 road-stream intersections (i.e., stream crossings). The Swan River Watershed is located south of Lesser Slave Lake, drains an area 3,117 km^2 and from queries of GIS databases includes 3,931 km of permanent roads and 759 road-stream intersections.

Fish communities in the Swan River drainage are comprised of 11 species and are dominated by lake chub, longnose suckers and Arctic grayling (Unpublished data Fisheries Management Information System 2003) (Table 1). In contrast, fish communities in the Notikewin River drainage are comprised of seventeen species dominated by Arctic grayling, long nose sucker, lake chub, brook stickle back, trout-perch and northern pike (Scrimgeour et al. 2003b) (Table 1).

Identifying study sites

The location of potential stream crossing sites were derived from Geographic Information Systems (GIS) databases that were queried to identify intersections (i.e., stream crossings) between roads and the streams, based on a hydrologically-corrected stream layer. Site coordinates were further attributed with several unique identifiers to generate a list of stream crossing sites for field assessments. Additional sites we identified by field surveys and not GIS layers were included in the total number of crossings assessed. We attempted to survey all crossings identified by GIS in the Notikewin River Watershed. Using the Strahler stream ordering system, 406 of the 759 crossings identified in the Swan River Watershed, were located on first order streams. Occurrences of fish in first order streams in the Swan River Basin are extremely low (Authors unpublished data) and as a result were excluded from assessments.

Stream crossing assessments

Stream crossing assessments were conducted from

Table 1. Common and scientific names of fish recorded in the Notikewin and Swan River basins. The presence of spoonhead sculpin in the Notikewin Watershed may result from the misidentification of slimy sculpin. Fish data for the Notikewin River Basin were provided from Scrimgeour et al. (2003b).

Family and common name	Species	Watersheds Notikewin	Swan
Cyprinidae			
Lake chub	*Couesius plumbeus* (Agassiz)	+	+
Flathead chub	*Platygobio gracilis* (Richardson)	+	-
Finescale dace	*Phoxinus neogaeus* Cope	+	-
Pearl dace	*Margariscus margarita* (Cope)	+	-
Longnose dace	*Rhinichthys cataractae* (Valienciennes)	+	-
Northern redbelly dace	*Phoxinus eos* (Cope)	+	-
Emerald shiner	*Notropis atherinoides* Rafinesque	+	+
Northern pikeminnow	*Ptychocheilus oregonensis* (Richardson)	-	-
Percopsidae			
Trout-perch	*Percopsis omiscomaycus* (Walbaum)	+	-
Gasterosteidae			
Brook stickleback	*Culaea inconstans* (Kirtland)	+	+
Percidae			
Walleye	*Stizostedion vitreum vitreum* (Mitchill)	+	+
Salmonidae			
Arctic grayling	*Thymallus arcticus* (Pallas)	+	+
Mountain whitefish	*Prosopium williamsoni* (Girard)	+	+
Rainbow trout	*Oncorhynchus mykiss* (Walbaum)		+
Bull trout	*Salvelinus confluentus* (Suckley)	-	-
Esocidae			
Northern pike	*Esox lucius Linnaeus*	+	+
Catostomidae			
Longnose sucker	*Catostomus catostomus* (Lacepède)	+	+
White sucker	*Catostomus commersoni* (Forster)	+	+
Cottidae			
Slimy sculpin	*Cottus cognatus* Richardson	+	+
Spoonhead sculpin	*Cottus ricei* (Nelson)	+	-

June through August in 2002. Assessments were designed to document crossing type and condition, and to provide a rapid assessment of the potential risk to impede fish passage and to contribute sediments to the stream channel in both watersheds.

Identification of crossing types. We identified the four crossing types of culverts, bridges, fords, and decommissioned crossings. Decommissioned crossings were defined as locations where a crossing structure had been installed and subsequently removed. These practices alter channel width and depth, substratum size composition and are often associated with high sediment inputs. Culverts (predominantly corrugated metal pipes) and bridges were further categorized by type and shape (e.g., Bridge/Bailey type). Cross-ditch drains and crossings on intermittent streams were not evaluated.

Stream and culvert properties - Replicate measures of rooted width (bank-full width ±1cm) were taken at least 25 m upstream of the crossing to avoid influence of the culvert on channel morphology. This measure was used to calculate encroachment ratios and habitat loss caused by encroachment defined by Harper and Quigley (2000). Culvert diameters (± 1cm) were measured on round (diameter) and elliptical, oval or box culverts as the widest distance across the culvert opening. Inadequately sized culverts encroach on fish habitat, constrict stream flow, increase velocities and may cause sedimentation and excessive scouring at the outlet (Parker 2000). Culvert outfall drop (±1 cm)

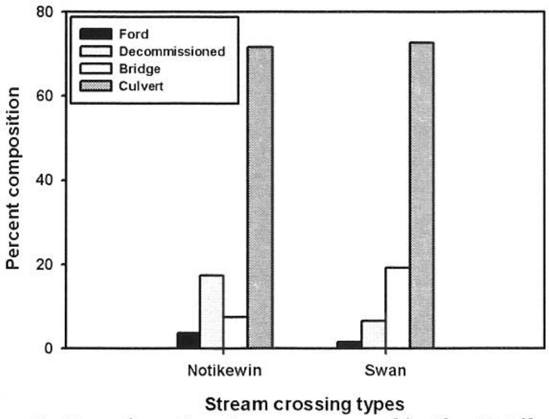

Figure 1. Crossing structures assessed in the Notikewin and Swan River basins in Northern Alberta, Canada. Ford = vehicular crossing bisecting the stream channel with no built up structure. Decommissioned = a crossing that was previously installed and has since been removed. Bridge = a structure placed above the stream channel typically spanning the bank-full width with or without in-stream abutments. Culvert = in-stream structure can be made of steel, concrete, plastic in round, half round, elliptical, or box shapes.

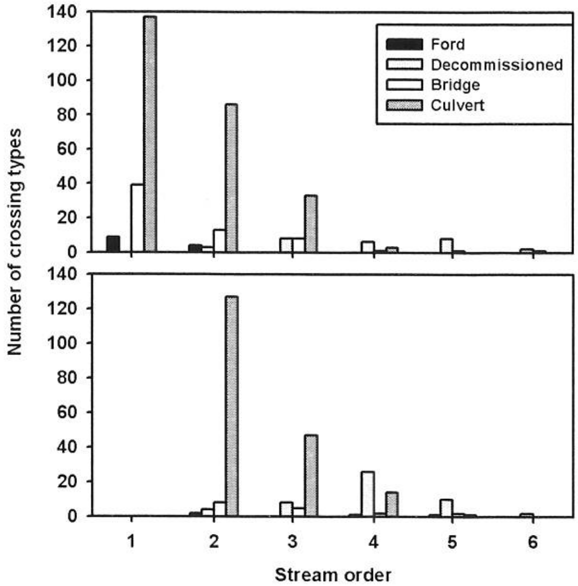

Figure 2. Number of crossing types as ford, bridge, decommissioned and culvert crossings in the Notikewin and Swan River watersheds, Alberta, Canada.

was measured as the distance from the bottom of the culvert outlet to stream water surface. Outfall heights exceeding 5 cm were considered to represent potential physical barriers to fish movements. Certain life stages of fish species may be unable to enter culverts that are set above the substrate [(e.g., slimy sculpin (*Cottus cognatus*)].

Sediment source assessments - The potential for sediment inputs to stream channels was evaluated visually by quantifying the potential for erosion of adjacent rights-of-way on the stream channel. Sites were assigned a rank of low, moderate or high potential for sedimentation based on evidence of: i) physically unstable river banks (i.e., evidence of slumping) at the site or immediately downstream of the site, ii) presence of non-vegetated soils or those with minimal vegetative cover, and iii) the absence or failure of sediment control structures to control sediment inputs. Our initial observations showed that bridges in both the Swan and Notikewin watersheds pose minimal risk of habitat fragmentation. As a result, we restricted our evaluation of the impacts of bridges on stream fish to those resulting from sediment inputs as previously described for culverts.

Photographic records - We obtained a photographic record of all crossings using a digital camera to document crossing condition and the condition of the adjacent rights-of-way (ROW). Digital photos were taken of the inlet, outlet and the two Rights of Way having the highest erosion potential. Lastly, our assessments also involved written documentation of other aspects of the crossing related to erosion potential and potential for fish passage.

Results

Data bases

Queries of GIS road and stream network layers identified a total of 544 intersections (i.e., potential stream crossings) in the Notikewin River Watershed and 759 intersections in the Swan River Watershed. Subsequent assessments indicated that 105 stream crossings (61 in Notikewin River Watershed and 44 in Swan River Watershed) were located in remote locations that precluded vehicular access and were not included in our assessments. Further assessments showed the presence of moderate numbers of stream crossings where water flow was either absent or insufficient to support fish communities. Using the Strahler stream ordering approach (Strahler 1964), the majority (406 of the 759) of stream crossings in the

Swan River Basin were located on first order streams. Because relatively few first order streams in the Swan Watershed support fish, data from first order systems were excluded from analyses (Tchir unpublished data). As a result, our assessments of crossing types and their potential to fragment stream habitats or result in sediment inputs were based on 413 and 352 road crossings in the Notikewin River and the Swan River watersheds, respectively.

Stream crossing density and structure types

Density of roads in the Notikewin Watershed (0.21 km of roads / watershed area km^2) was markedly lower and concentrated mainly in one sub-basin, compared to that in the Swan River Watershed (1.26 km roads / watershed area km^2). Similarly, in the Swan River Basin density of stream crossings (0.24 crossings / linear km) was more than three times higher than densities in the Notikewin River Watershed (0.068 crossings / linear km).

Culverts were the predominant stream crossing structure in both watersheds and were most prevalent on small streams (i.e., 1st, 2nd and 3rd order streams) (Figures 1 and 2). Bridges were prevalent on third order streams and larger and were the only structure present on sixth order streams in both watersheds. Fords were observed exclusively on 1st and 2nd order streams in the Notikewin River Watershed, where as, fords in the Swan River Watershed were found on 2nd, 4th and 5th order streams. In general, fords were located on reclaimed roads. While decommissioned crossings were observed on 1st to 6th order streams (Figures 1 and 2) in the Notikewin River Watershed, they were most prevalent on 1st order streams. In the Swan River Watershed, decommissioned crossings occurred most frequently on 2nd order streams and were located predominantly on deactivated and winter roads.

Multiple culverts were observed where previous culverts had apparently failed or where single culverts appeared to be insufficient to handle peak flows. Single and multiple steel pipe culverts (i.e., non-corrugated metal pipes) were rare and due to their condition were likely installed before the 1990s. The number of single and multiple culverts generally decreased with increasing stream order. Evidence of beaver activity was found at 35% and 16% of culvert crossings analyzed in the Notikewin River and Swan River watersheds, respectively.

Encroachment and habitat loss

The diameter of culverts was significantly less than bank-full measured at least 25 m upstream of crossings in the Notikewin (Paired t-tests: $t = 5.3$, df 111, $p < 0.005$) and Swan River Basin ($t = 4.4$, df 104, $p < 0.005$) and results in a reduction of the surface area of benthic habitats. The majority of culverts in both watersheds resulted in encroachment into stream channels (Notikewin River Watershed 69.9%, Swan River Watershed 83.7%). Crossing sites comprised of multiple culverts also resulted in habitat encroachment (Notikewin River Watershed 43.8%, Swan River Watershed 65.6%). There was a significant difference in the cumulative span of multiple culverts and bank-full width in the Swan River Watershed (Paired t-test: $t = 3.7$, df 68, $p < 0.005$) but not in the Notikewin River Watershed (Paired t-test, $t = 1.5$, df 15, $p > 0.05$).

Stream barriers and fragmentation

Based on our criteria, initial assessments showed that the majority of culvert crossings in the Notikewin (61% of all crossings assessed) and the Swan River watersheds (74% of all crossings assessed) likely impede fish movement and thus are potential barriers (Figure 3). Culverts with low water levels were typically, however, not considered to impede fish movement in either watershed. Barriers resulted from accumulations of organic debris at the inflow of culverts (i.e., debris blockages) and where flow from the bottom of the culvert is located above the water surface of the stream (i.e., a hanging culvert). Hanging culverts were the dominant cause of habitat fragmentation in both watersheds (Figure 3). Mean culvert outfall heights ranged from 26 –32 cm in the Notikewin and 19-67 cm in the Swan River Basin (Figure 4). However, inadequate culvert sizing and placement also caused debris blockages. Barriers caused by damaged inlets or outlets (i.e., damaged pipe) of culverts were uncommon in both watersheds.

If our assessment of what constitutes a barrier to fish movement is correct, our calculations suggest that fish populations may not be able to readily access about 20% of the headwater areas of the Swan River and 10% of headwater areas in the Notikewin Watershed. Because hanging culverts were identified to be the primary cause of potential impediments to fish movement, they also accounted for the majority of areas that are likely not freely accessible in both study watersheds.

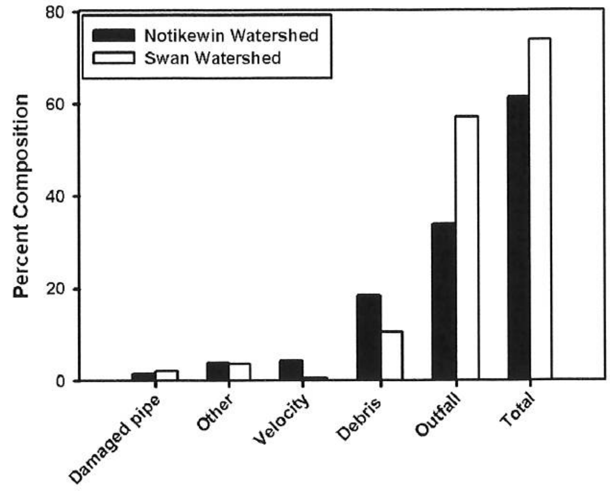

Figure 3. Percent composition of the dominant types of potential barriers to fish passage in the Notikewin and Swan River watersheds. Barriers due to damaged pipe occur as a result of physical damage of inflows and outflows of the culverts. The Other culvert barrier type includes beaver control structures (e.g., metal screens) at culvert inlets, hanging inlets, submerged culverts and agricultural modifications. Total represents the proportion of all culverts that were potential barriers to fish movement. Data are based on 362 and 260 road crossings in the Notikewin and Swan watersheds, respectively.

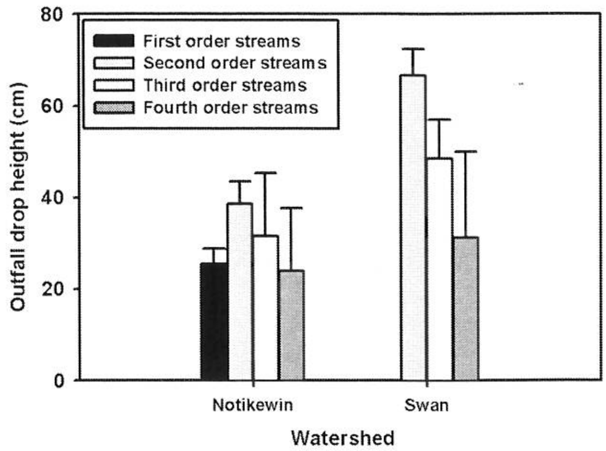

Figure 4. Mean (± SE) outfall drop height of culverts from first to fourth order streams in the Notikewin River Basin and second to fourth order streams in the Swan River Basin, Alberta, Canada.

Potential erosion

Our visual and qualitative assessments of erosion potential from crossing structures showed that 17% and 18%, respectively, of crossings in Notikewin and Swan River watersheds, likely contributed moderate levels of sediments to watercourses (Table 2). The propensity of culverts to contribute sediments was greatest in the Swan River Watershed (19%) compared to the Notikewin River Watershed (3%). In the Swan River Watershed, proportionately more bridges were rated as having high levels of silt deposition (Notikewin River Watershed 8%, Swan River Watershed 36%) (Table 2).

Discussion

We completed an environmental scoping exercise to evaluate the potential effects of road networks on stream fish communities inhabiting two boreal forest watersheds by quantifying their potential to impede fish movement and to reduce habitat quality by contributing sediments to stream channels. Our analyses identified culverts as the primary stream crossing structure in both watersheds and that the majority of culverts likely impede fish passage either because: i) water stage at culvert outflows was typically located above the water surface or ii) the presence of accumulations of debris at culvert inflows. In fact, mean culvert outfall heights ranged from 26 –32 cm in the Notikewin and 19-67 cm in the Swan River Basin. Our assessments also suggested that road networks and related stream crossings could impact fish assemblages by contributing sediment from road approaches ands rights-of-way that were physically unstable or poorly vegetated. Taken together, our preliminary assessment suggests that road networks in the two basins have the potential to impact stream fish communities. We suggest that additional efforts are required to more fully evaluate threats to stream fish communities in these basins and should include an evaluation of remedial actions to reduce crossings that likely severely restrict fish passage.

Our suggestion that road networks may potentially impact stream fish communities is not new. In fact, concerns about poorly designed roads and stream crossing structures have been expressed for at least two decades (Everest and Harr 1982; Adams et al. 1986; Furniss et al 1991; Weaver et al. 1987), while several recent studies have demonstrated negative effects of roads and associated stream crossings on stream fish populations (e.g., Beechie et al. 1994; Conroy 1997;

Table 2. Number of crossings rated as contributing moderate and high levels of sediments to streams in the Notikewin and Swan River watersheds. Decom. = Decommissioned stream crossings. Stream crossings on first order streams were not evaluated.

Notikewin	Bridge	Culvert	Ford	Decom.
Stream Order				
1	0	35	3	7
2	0	12	1	0
3	4	10	0	3
4	1	0	0	1
5	4	0	0	0
Unknown	0	8	0	0
Total	9	65	4	11
Swan	**Bridge**	**Culvert**	**Ford**	**Decom.**
Stream Order				
1	-	-	-	-
2	1	45	0	2
3	5	24	0	1
4	19	7	0	0
5	9	1	0	0
Unknown	1	0	0	0
Total	36	77	0	3

Pess et al. 1998). Additional concerns have also been raised about the ability of structures to provide fish passage under variable flow conditions (Ashton and Carlson 1984; Bates et al. 1999). Our observation that the majority of culverts in the Notikewin and Swan River basins likely impede fish movement is in part, a consequence of management practices in the 1970's and 1980's. During this period, culverts were designed and installed primarily to provide a stable and cost-effective water control structure capable of supporting vehicle use. Additional considerations of how different culvert designs and installation practices could fulfil these requirements, while not impeding fish passage, are a relatively recent advancement. In Alberta, this paradigm shift coincided with the rapid expansion of the forest sector in the early 1990's and the continued expansion of the oil and gas sector. While not quantified, our observations in both watersheds suggested that many, but not all, culverts that likely restrict fish passage appear to result from older structures.

Installation of under-sized culverts and those that are not embedded in the streambed are more likely to fragment stream fish habitat than alternate practices. Undersized culverts have been shown to increase the likelihood of habitat fragmentation resulting from water stage at outfalls being above the water surface and because of high water volumes and velocity exceeding swimming capabilities (i.e., velocity barriers). Non-embedded structures (i.e., culvert installed above the stream channel) can result in substrate scouring (i.e., removal of sub grade material) and in dewatering and erosion of stream habitat immediately downstream of the culvert outfalls (Harper and Quigley 2000). Our data suggest that culvert crossings were more severely undersized in the Swan River Watershed than that in the Notikewin River Watershed. As a result, we expect that the potential risk of habitat fragmentation in the Swan exceeds that in the Notikewin River Basin.

Efforts to minimize negative impacts of culverts on stream fish communities has focussed largely on understanding the ability of fish to: i) access culvert outfalls that are located above the water surface and ii) navigate high gradient and often hydraulically complex flow (e.g., Behlke 1991; Furniss et al. 1991). These sources of information have resulted in the development of guidelines related to preferred heights of culvert outfalls and design considerations related to culvert size, length, slope and in-culvert structures (e.g., baffles) that allow fish passage (refs here). While these studies have enhanced the ability of some species of fish to access and navigate culverts, effects of culverts on stream fish assemblages at the watershed-scale remain poorly understood. For instance, swimming abilities of many non-salmonids are poorly known; as are the cumulative energetic costs of navigating series of poorly designed or installed culverts. Despite these and other information deficiencies, our observations in the Notikewin and Swan River basins suggest that many of the concerns related to impeding fish passage may be alleviated by installing appropriately sized open or oval culvert designs that are set into, rather than above, the stream beds. In addition, increased focus on restoring vegetation cover to approaches to culverts and ROW may also dramatically reduce the number of stream crossings that potentially can contribute sediments to stream.

Our data suggested that the majority of concerns related to sediment inputs to stream originated from: i) physically unstable soils (i.e., where evidence of mass slumping was apparent), ii) poorly vegetated areas located immediately adjacent to stream crossing structures and within stream banks rights-of-ways and iii) bridge surfaces with large gaps between bridge materials that allow sediments on the bridges to fall into stream channels. Our observations also suggest that deactivated stream crossings in both watersheds surveyed were typically poorly vegetated and seldom associated with sediment control structures or if present were likely ineffective in minimizing sediment inputs to streams.

Although bridges were not analyzed for encroachment in this particular study, Harper and Quigley (2000) found bridges with shorter spans had a larger ecological footprint as a result of the adjacency of bridge abutments to the stream channel. These structures required extensive riprap to prevent substratum scour protection and resulted in increased habitat losses due to encroachment. They also suggested that bridges with longer spans were found to require less riprap and resulted in minimal disturbance of the stream banks (Harper and Quigley 2000). Eaglin and Hubert (1993) suggested that sediment delivery to streams might be positively related to the number of stream crossings. They found that the extent that large substrata were embedded in fine sediments increased with the number of stream crossings whereas the abundance of cobble substrates decreased with number of stream crossings. Cross-ditches with deeply rooted and abundant vegetation combined with sediment control structures (i.e., traps) can reduce the supply of sediments form road, ditches, and ROW to streams.

Restoring connectivity of watercourses has been contended to be an effective way to increase the availability of productive fish habitat. Roni et al. (2001) reported that the relative benefits of restoring fish passage can exceed those attained from development of off-channel habitats, instream structures and sediment reduction activities. Other studies have reported that removal of physical barriers can increase the abundance of parr, juvenile and adult salmonids (e.g., Beechie et al. 1998; Pess et al. 1998). Given that replacement and repair of culverts and other stream crossing structures can result in sediment input and alteration of instream fish habitat to some extent, and can be extremely costly, we urge resource managers to carefully consider where replacement and repair efforts are most appropriate. For example, we recommend that resource managers evaluate the quantity and quality of stream habitat upstream of crossings that impeded fish movements before they recommend any remedial activities.

Our data identified moderately high numbers of stream crossings in the Swan River Basin. In fact, our data showed that roads on average intersected streams at 4 km intervals and that the total number of stream crossings by seismic lines, power ROW, rail lines and small trails not identified in existing GIS layers are likely substantially higher. Our previous studies in two other boreal forest watersheds in northern Alberta have shown that stream crossings by roads comprise only 25-40% of all stream crossings (Scrimgeour et al. 2003a). The ecological consequences of high numbers of stream crossings on fish communities within Alberta's boreal forest and the mechanisms through which they are mediated are poorly understood.

We suggest that an improved understanding of the effects of road networks on stream fish within Alberta's boreal is required to ensure that industrial development activities can occur without compromising stream ecosystems. Broad research needs are required to provide an improved understanding of: i) landscape-level patterns in road networks and how they are progressing throughout the boreal forest, ii) current and forecasted effects of road networks on multiple life history stages (Toepfer et al. 1998) and migratory and non-migratory stream fish communities, iii) the extent that stream fish populations function as metapopulations (Riemen and McIntyre 1993; Morita and Yokota 2002) and iv) cost-effective remedial plans to reduce current impacts.

Acknowledgements

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The Hardisty Creek Restoration Project

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Extended Abstract

The Hardisty Creek Restoration Project (HCRP) is a multi-stakeholder initiative with the general mandate to foster increased public awareness of watershed concerns within Alberta and, specifically, to restore the fish, wildlife, vegetation and overall health of the Hardisty Creek watershed in Hinton, Alberta. The HCRP stakeholder group includes representatives from the Athabasca Bioregional Society, Foothills Model Forest, Town of Hinton, Alberta Sustainable Resource Development, Alberta Transportation, Fisheries and Oceans Canada, Hinton Fish and Game Association, Weldwood of Canada Ltd. and the general public. Other stakeholders, such as Canadian National Railroad, are also pursuing initiatives aimed at improving the quality of the Hardisty Creek watershed.

The overall objective of this project is to produce a detailed plan to guide the implementation of stream restoration activities at a number of sites within the Hardisty Creek watershed. These sites include the reach of Hardisty Creek within Kinsmen Park and in the vicinity of three selected road crossings. When completed, the restored sites will demonstrate practical and proven restoration techniques addressing a wide range of riparian and fish habitat impacts within the study area. The restored sites will also serve to educate interested parties from around the province.

A fish habitat assessment was conducted to identify opportunities for improving fish habitat and fish passage along Hardisty Creek. Habitat assessments focused on the target fish species of: rainbow trout, bull trout and mountain whitefish and the habitat assessment component of the study consisted of: i) a review of available information (base maps, current and historical aerial photography, aquatic habitat and fisheries inventory summaries, government and third party reports); ii) completion of sediment source surveys to identify major erosional features or unstable stream banks and iii) completion of fish habitat mapping at sites where potential restoration, rehabilitation or mitigation measures are possible. The fisheries assessment

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extended 50 m upstream and 100 m downstream of each culvert crossing and along the reach of Hardisty Creek within Kinsmen Park.

Following analysis of the fish habitat assessment, restoration options were developed to provide: i) fish passage during low flow conditions; ii) overwintering habitats for fish species; iii) effective rehabilitation techniques for riparian zones; iv) long term in-stream and floodplain stabilization designs; v) opportunities for public involvement in the implementation of restoration options.

Detailed fish passage assessments were conducted for selected culvert crossings to quantify the conditions preventing fish passage. Analysis determined water velocities at a variety of discharge regimes and assessed tailwater conditions downstream of the culverts. The fish passage assessment included: i) measurement of the contributing drainage area for each culvert; ii) calculation of appropriate conveyance and fish passage design discharges for each culvert, based on regional hydrologic analysis; iii) determination of appropriate fish passage criteria; iv) development of hydraulic models of each culvert configuration based on the survey data; v) evaluation of water velocities through each culvert, compared to fisheries criteria; and, vi) investigation of potential options for changing the culvert configuration to create acceptable hydraulic conditions for fish passage while maintaining flood conveyance capacity.

The Upper Bow River Watershed Off-Highway-Vehicle Stream Crossing Inventory and Assessment Program

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Extended Abstract

Recreational use of Alberta's forested areas has dramatically increased the number of open stream crossings. While off-highway-vehicle (OHV) use in the East Slopes of Alberta has dramatically increased over the last 10 years, there is a paucity of information describing the number, condition and potential impacts of these crossings on stream fish communities. Collection of information on the number and condition of stream crossings is an important resource management issue related to the No-Net-Loss provision of the Federal Fisheries Act and Provincial Government commitments to sustain stream fish communities.

In response to this issue, the Alberta Conservation Association initiated the Upper Bow River Watershed Off-highway-vehicle stream crossing inventory and assessment program in 2002. Completed in the East Slopes of Alberta, the objective of the program was to quantify the total number of crossings and to determine the condition of select OHV stream crossings in priority areas of the Ghost and McLean Creek drainages. The Program is a co-operatively funded, multi-agency initiative comprising financial contributions from the ACA, the Department of Fisheries and Oceans, Alberta Sustainable Resource Development, Municipal District of Bighorn and assistance form other organizations.

As part of initial efforts, we quantified the number and condition of select OHV stream crossings in each of the two study basins. These two geographic regions were chosen because: i) they are used extensively by OHV vehicles and ii) were identified by Government resource managers as areas that present current challenges in balancing increased recreational use while maintaining healthy streams. Our inventories, based on the interpretation of aerial photography and field-based ground-truthing exercises, identified a total of 178 OHV stream crossings in

Fitzsimmons, K., and Fontana, M. 2004. The Upper Bow River watershed off-highway-vehicle stream crossing inventory and assessment program. Pages 65-66 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

the two study areas. Our qualitative assessment of crossing health indicated that the majority (67% of all crossings evaluated) of crossings were categorised as being heavily (40 of the 178 crossings= 22%) or moderately impacted (80 of the 178 crossings=45%). The remaining 58 sites (33%) were categorised as being minimally impacted. Our preliminary analyses also suggest that some of the minimally and moderately impacted crossings are expected to deteriorate with increased vehicular use. As a result, the proportion of heavily and moderately impacted stream crossings is forecasted to increase in the absence of changes in management practices.

At this point the Program supporters need to prioritize a number of issues related to OHV use and stream fish communities to direct future initiatives. These include: i) quantifying the effects of OHV crossings on stream habitat and stream fish communities, ii) forecasting temporal changes in crossing density and condition, iii) evaluating a host of potential remedial measures and iv) launching a communication initiative to a wider groups of stakeholders to inform them of what has been learned from past activities to provide them with an opportunity to direct future efforts. The development of a longer–term plan based on input from a diversity of stakeholders will likely be an important component of addressing resource management issues related to recreational use and the maintenance of stream fish communities.

Alberta's Managed Access Program on Public Lands: A Collaborative Approach

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Extended Abstract

The Alberta Government is committed to wise management of natural resources and environment to support Alberta's economic, social, and environmental goals now and in the future. To realize this goal, Alberta Sustainable Resource Development (SRD) has developed the Access Management Program. This long term program is a collaborative strategy between the public, government and industry to deliver sustainable landscape management and use of public land while ensuring social, economic and environmental benefits to all Albertans.

The goals of this program are for all users of public land to: i) work together in a cooperative and collaborative fashion, ii) to possess a clear and substantial understanding of stewardship, and iii) to exhibit a high level of accountability. Stakeholders will examine the current situation and project future access use and requirements to ensure sustainability and contribute to a prosperous economy. Some key outcomes include certainty of access for industry and public; effective partnerships for consultation and integration of interests; efficient government approvals; a prosperous economy; opportunities for innovation; broad landscape-level perspectives; increased stewardship; sustainability of resource values through knowledge acquisition and sharing.

Included in the access program is an examination of the current situation and project future access use and requirements to ensure landscape sustainability and to contribute to a prosperous economy. Some key outcomes include certainty of access for industry and public; effective partnerships for consultation and integration of interests; efficient government approvals; a prosperous economy; opportunities for innovation; broad landscape-level perspectives; increased stewardship; sustainability of resource values through knowledge acquisition and sharing.

Anchor points for developing and sustaining an effective AMP are three major "project streams." The first of these streams addresses public access and recreational needs. This stream

Selland, G. 2004. Alberta's managed access program on public lands: a collaborative approach. Pages 67-68 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

is aimed at developing a comprehensive strategy and policy framework for the implementation of recreation and public use on public land with a focus on the backcountry. The second project stream focuses on industrial access. This element will develop a framework for collaborative industrial access on public land as a way to minimize the industrial footprint and enable industries to co-exist and prosper on a single landscape. The third project stream focuses on timely, planned reclamation of man-made disturbances to reduce the footprint on the landscape and return the productivity of the land.

It is critical that these three project streams are worked on simultaneously to effectively manage the wide range of uses (from industrial to recreational to general public) while at the same time recognizing and managing the significant contribution that progressive, ongoing reclamation can make to overall reduction in footprint and return to capability.

By working on these three project streams, the managed access program will go a long way to address important landscape challenges and to realize the Alberta government's to the sustainable development of natural resources.

Implementation of Watercourse Crossing Training for Harvesting Equipment Operators in Alberta by the Woodland Operations Learning Foundation (WOLF)

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Extended Abstract

The Woodland Operations Learning Foundation (WOLF) is a non-profit business established by the forest industry to develop and deliver training to individuals and companies involved in forest harvest operations in Western Canada. WOLF is administered by a Board of Directors made up of representatives from forest industry companies, harvesting and road construction contractors, forest consultants, harvesting equipment suppliers, Provincial Government, forestry associations, and Northern Lakes College.

The logging industry in Western Canada is evolving rapidly, being led by changes in harvesting practices and transportation systems, as well as environmental issues. More and more responsibility is being placed on the operators in the field. Logging equipment operators are our front line people in the forest industry, and it is necessary to provide them with proper skills in order to minimize the risk of costly mistakes and possible damage to the environment.

The Woodlands Operations Learning Foundation (WOLF) was established to meet the training needs of the forest industry and elevate the profile of harvesting equipment operators and is provides training in: i) Water course crossings, ii) Water quality.

The Watercourse crossing course was developed in conjunction with industry, government and subject matter experts to provide those working in the forest sector with an overview of considerations related to watercourse crossings. Increasing requirements have made it necessary for equipment operators who are installing stream crossings to have an increased awareness regarding the installation, maintenance and reclamation techniques for various crossing types. It is also important to understand the potential effects of improperly installed crossings on the environment and the legal implications.

The specific objectives of this course include: i) use of a stream crossing and potential downstream effects, ii) policies regarding stream crossings and how to follow the required guidelines, iii) alignments of approaches and methods of preventing run off into watercourses,

Eggleston, V.B. Implementation of watercourse crossing training for harvesting equipment operators in Alberta by the Woodland Operations Learning Foundation (WOLF). Pages 69-70 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

iv) correct location for a stream crossing based on alignment, gradient, and environmental factors, v) effects of various stream crossings on fish, vi) appropriate types of stream crossings to be used on a common cutblock in winter or summer applications, vii) culvert installation and understanding back-filling procedures and viii) removal of a stream crossing and reclamation of the site.

The Water quality module is intended to provide the industrial user with the foundation to make decisions regarding the protection of this valuable asset. In general, it provides: i) background on the importance of aquatic habitats and water source areas and their importance in ecosystem function, ii) an understanding of the effects operations on water quality and planning strategies to mitigate these effects, and iii) an understanding of the related acts and regulations that pertain to water quality.

Overall, the courses provide relevant and practical information related to forest management and module participants depart with newly acquired information and an improved understanding of: i) measures of water quality and pollution, ii) an understanding of watercourse ecosystems (aquatic life, stream temperature, the water cycle, water table, and natural disturbances), iii) stream classification (e.g., fish bearing and non-fish bearing, navigable, size), iv) effects of siltation, damming, diverting and blocking and v) an understanding of erosion and sedimentation in relation to forests.

The South Shore Watershed Project: Determining the Responses of Boreal Forest Stream Ecosystems to Harvesting

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Extended Abstract

Forest harvesting at the watershed scale has been shown to alter thermal regimes, dynamics of sediment deposition and transport, and water flow in streams. In boreal Alberta, a multiple-pass harvest system ensure that no more than 50% of a watershed could be harvested before sufficient regeneration occupied previous cutovers, limiting the potential for such stream-altering effects. This pattern of harvesting, however, differs markedly from a natural disturbance-based management model, where harvesting would produce a greater range and variation in forest opening sizes.

Research on the effects of large-scale harvesting within the boreal forest is limited; as a result, research results from coastal areas or the southern United States are often applied to operations in Alberta. In those regions, large-scale harvesting can alter stream geomorphology and the abundance and composition of aquatic organisms. The Boreal Plains differ from those regions, however, in site characteristics such as topography, vegetation, and soil types. As a result, the ecological effects of forest harvesting in other regions may not be a good predictor of the ecological responses of boreal forest streams.

As part of ongoing commitments to developing best management practices, Alberta Plywood Ltd. initiated a collaborative, multi-year study to evaluate the ecological effects of forest harvesting on stream ecosystems in central Alberta. The study area is located

Duffy, G., Tonn, W.M., Scrimgeour, G.J. and Proctor, H.C. 2004. The south shore watershed project: determining the responsiveness of aquatic indicators to forest harvesting. Pages 71-72 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

within a deciduous-dominated boreal forest consisting of patches of white spruce and mixed wood forest. The study will compare the effects of moderate (30% of watershed harvested) and high (70% of watershed harvested) harvesting levels with no-harvest references, with each of the three treatments replicated in three sets of three adjacent watersheds. Study systems were identified using a combination of watershed and stream characteristics, combined with operational considerations. These included the 1-10 yr and 11-20 yr forest harvest plans, watershed delineations, and stream hydrography, and resulted in the identification of nine similar watersheds. Within each watershed, study sites will be located in both the upper and lower reaches of the streams.

While in the final stages of planning, we expect to apply a ***BACI*** (***B***efore-***A***fter-***C***ontrol- ***I***mpact) design to quantify the responses of streams to the forest harvesting. These responses include a diversity of physico-chemical (e.g., pH, water temperature, suspended sediment concentrations, nutrients) and biotic (e.g., primary and secondary production [e.g., benthic invertebrates]) indicators.

Rigorous scientific data and other information is required to: i) support ongoing evaluations of current forest management practices and ii) ensure an alignment between economic and biological criteria, especially those related to evaluations of the natural disturbance-based management model.

Managing Access Impacts on Forested Watersheds – Timber Company Case Study

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Extended Abstract

Watersheds within Alberta's Eastern Slopes are the birthplace of some of the largest and most important rivers in Western Canada. Many different users, such as oil & gas, mining, timber harvest and recreation, operate within these watersheds. It is in everybody's interest that the watersheds forming these important rivers are managed to maintain water quality and fish and wildlife resources.

Today's consumers of forest products increasingly demand assurance that the products they purchase come from sustainably-managed forests. To meet these demands many companies are seeking third-party environmental certification through the Canadian Standards Association CAN/CSA Z809 standard. This standard requires public consultation to define values and goals, and guides the development of indicators, objectives and management strategies. The standard also requires development of an environmental management system, which identifies environmental risks and mitigation techniques. Rather than certifying a company, the CSA standard is applied to a Defined Forest Area (DFA). The Defined Forest Area in which Sunpine Forest Products operates is located in the southern east slopes of Alberta and considers all forest resource uses occurring on the DFA, within the limitations and responsibilities of licensing arrangements with the Crown.

Public input into the development of Sunpine's Sustainable Forest Management Plan, as required by the CSA standard, identified watershed protection as a primary concern. As a result, Sunpine has developed several Criteria and Indicators to address water quality. They include: i) pre-construction site plans on 100% of fish bearing streams, ii) design and installation of road crossings that provide fish passage, iii) practices that protect water quality, and the development of action plans to minimize impacts of point source occurrences of sedimentation and pollution. The first step in meeting these criterion indicators is to perform Pre-construction Stream Crossing Assessments. These assessments are designed to critique the

Kure, K.D. 2004. Managing access impacts on forested watersheds-timber company case study. Pages 73-74 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land- Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

crossings chosen by road layout personnel and to document the pre-disturbance attributes of the crossing as well provide construction specifications. A hydrology tool is used to calculate peak flows in individual drainages. This information is used to design bridges, culverts and temporary roads to meet a minimum of a 1:25 flood interval whereas long term tenure roads crossings are built to a 1:100 year flood interval. This data will be used in the approval process implemented under the new Water Act. All stream crossings are recorded spatially in a GIS system as well in a complete database. This database sets the foundation of the historical tracking of each crossing through to reclaimed status. Annual Stream Crossing Inspections, which includes a field visit, determines reclamation status that translates into environmental risk and sets remediation plans. This added information is useful in preventative planning, construction and monitoring programs. Monitoring is achieved through regular inspection, maintenance and annual audits.

By identifying issues regarding stream-crossing development, and through planned monitoring, forest products companies can effectively manage the risks associated with the impact of road construction on water quality.

Access Management in Calgary's Playground: Husky's Operations in Kananaskis Country

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Extended Abstract

Husky has operated in Kananaskis County since before Kananaskis was developed in 1979. Husky began its operations at Moose Mountain when the population of Calgary was about 700,000 today it is close to 1 million and showing no signs of slowing. From 1992-1998 Husky drilled 6 wells and constructed a 25km pipeline on the north side of Moose Mountain. Following this, in 2000 Husky drilled a sour gas well in McLean Creek Off-Highway Vehicle Zone located west of Calgary. Presently, Husky has four wellsites (i.e., pads) and about 30 km of pipelines in the Moose/Mclean area. Husky has also developed a 71,000 ha study area in which a wide variety of environmental information has been gathered.

Exploring and developing oil and gas reserves in the Kananaskis Country poses many challenges, several of which include: i) the majority of Calgary's drinking water comes from the Elbow River, which flows through the area; ii) the Elbow River Valley is the most highly used part of the Kananaskis Country and on a busy weekend in July 20,000 people can be in the area recreating; iii) the area is home to Canada's only legislated Off-highway vehicle zone, iv) the area has valuable wildlife habitat and v) other industries, including forestry, also operate in the area.

Husky, its partners and Shell Canada have developed several principles that they use to guide their operations in the Kananaskis region. The main principle is to minimize the industrial footprint, which in turn, will benefit both the wildlife and the people. This is achieved by co-ordinating our operations with other area users including the recreating public, the logging company (Spray Lakes Sawmills) and other oil and gas companies. Other principles include reducing the visual impact of our operations, minimal flaring and open and early public consultation.

Husky's future challenges include constructing an Interconnect Pipeline in partnership with Shell Canada (Shell Canada is the operator) and a wellsite close to Shell's infrastructure on the

Engstrom, C. 2004. Access management in Calgary's Playground: Husky's operations in Kananaskis Country. Pages 75-76 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

north side of Highway 66. In addition, there have been initial discussions between government, industry and local residents about the idea of habitat compensation.

This paper will focus on how Husky and its partners have tried to integrate its operations with the forest and water ecosystems as well as manage the recreational and timber users who also exist on the landscape. Specific items that will be addressed are environmental and development planning, public consultation activities, the development of a long-term environmental monitoring program with local residents, a trail-stream enhancement program and a proposed habitat compensation program for new wellsites.

Riparian Management

Effects of Managed Buffer Zones on Sedimentation and Invertebrates Associated With a Headwater Stream in the Indian Bay Watershed in Northeast Newfoundland

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Abstract

The effectiveness of a managed buffer zone on sedimentation and macroinvertebrate communities during forest harvesting was studied over a two year period in a small headwater stream in northeastern Newfoundland, Canada. Using a pre and post harvest design, we evaluated the effectiveness of the following four riparian management schemes: i) 20 m no harvest buffer; ii) 20 m buffer with 30% of the basal area harvested; iii) 30-50 m buffer with 30% of the basal area harvested; and iv) a no harvest control site to mitigate the effects of harvesting.

Sedimentation significantly increased in the 20 m buffer combined with 30% selective harvesting. Observations during this study suggest the sedimentation originated primarily from heavily used forwarding trails that resulted in exposure of mineral soils. The effects of selective harvesting on aquatic macroinvertebrates varied depending on the index and taxon. The number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) divided by the number of Diptera index was not significantly affected by buffer treatment and year. However, the number of total EPT, Ephemeroptera, Plecoptera, Trichoptera, Diptera (excluding Chironomidae) and Chironomidae were all significantly affected by buffer treatment and year. The most notable difference between pre- and post-harvest occurred within the 20 m buffer, where a large increase in *Oxyethira* sp., an algal consumer, was observed. The number of species observed for each of these buffer treatments was slightly greater post-harvest, however the differences were not statistically significant.

Overall, the reach with the 30-50 m buffer combined with selective harvesting appeared to

Wells, J.M., Scruton, D.A. and Clarke, K.D. 2004. Effects of managed buffer zones on sedimentation and invertebrates associated with a headwater stream in the Indian Bay watershed in northeast Newfoundland. Pages 79-86 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

be the least impacted, in terms of changes in sedimentation and invertebrate community composition. These results suggest that managed buffers of 30-50 m are capable of mitigating increases in sedimentation and changes in invertebrate communities. However, the results of this study should be cautiously interpreted, owing to the short post-harvest assessment, and that additional monitoring is recommended to assess the longer-term impacts of harvesting with managed buffers.

Introduction

Of the pressing concerns affecting Canada's boreal forest, logging is at the forefront due to its accelerating pace and because of its rapid advancements into fragile and slow growing northern regions. According to the Canadian Senate Subcommittee on the Boreal Forest (1999), about 90 percent of logging is clearcutting, which is the legislated method of forest harvesting in Newfoundland and Labrador (Government of Newfoundland and Labrador 2002).

Throughout the 1990s relatively few studies within Eastern Canada have evaluated the effects of selective harvesting within buffers on stream water chemistry and benthic invertebrates. Managed buffer zones differ from the conventional buffer zones in that they allow harvesting within the buffer. For this study, the effectiveness of selective harvesting within a buffer zone was examined to determine whether significant changes occurred to sedimentation and invertebrates within the stream. The objective of this study was to assess whether managed buffers are a superior method of riparian zone management, protecting the ecological integrity of an area, while maintaining the total wood production for the forestry industry within these areas. We also compare our findings to the Copper Lake study completed in western Newfoundland (Clarke et al. 1998).

Study Area

The study was completed in Hungry Brook in the Indian Bay watershed located in northeast insular region of Newfoundland. The Hungry Brook Watershed is approximately 750 km^2 in total area and consists mainly of moderately old (approximately 82 years) black spruce (Picea mariana) stands (average height = 12 m) (Dr. Gary Warren, Canadian Forest Service, pers. comm.). The site has a slightly sloping terrain (gradient 1% on average) with a north slope aspect. Hungry Brook has a mean wetted width of 4.8 m and has an average channel depth of 20.9 cm during the summer data collection. The substrate consists mainly of cobble, rubble and gravel.

Study Sites

Data were collected in 2000 to represent baseline pre-harvest conditions. Three experimental buffers were established: i) 20 m buffer, the current provincially legislated requirement for a brook of this size and location; ii) 20 m buffer with 30% of the basal area harvested; and iii) 30-50 m varying width buffer with 30% of the basal area harvested and iv) a no harvest control. Removal of the 30% of the basal area of forest within riparian of two of the four treatment zones was accomplished by initially measuring tree diameter at breast height and then selectively removing trees until 30% of the buffer area at each site had been removed. All of the sites were 500 m in length and harvesting was conducted with a mechanical harvester along one side of the brook during November 2000. Data collected in 2001 after the establishment of these buffers represented the experimental post-harvest data. All three buffer treatments were located along the same reach of Hungry Brook with no replication.

Material and Methods

Sedimentation

Whitlock-Vibert boxes were used to measure fine sediment accumulation (Wesche et al. 1989). Boxes were filled with cleaned gravel and duct tape was placed across the bottom of the box to prevent loss of fine particulates. Two Whitlock-Vibert sediment boxes were deployed at 100 m intervals, with placement beginning downstream at 0 m, within each 500 m experimental and the no harvest site. The boxes were changed again after the spring run-off in June 2000, and again late October 2000 prior to the harvesting commencing. For the post-harvest year, 2001, the sediment boxes were changed after the spring-run off in June and retrieved again in November. The contents of the sediment boxes were wet sieved and dried at 70°C for 30 h, and then weighed. The sediment was divided into two size classes: i) > 1.4 mm and ii) < 1.4 mm.

The data were analyzed using the G statistic from a chi-square distribution with one degree of freedom ($\alpha = 0.05$). This statistic measures goodness of fit of chi-square to the data (Devore 1995). The quantity of sediment accumulated for each size fraction and the combined classes within each site collected during 2000 (pre-harvest) and 2001 (post-harvest) were compared with the no harvest site to determine whether there was a significant difference between pre- and post-harvest.

Invertebrates

Six artificial substrates consisting of approximately 7.2 kg of 3.5 to 5.0 cm washed cobble encased in plastic Vexar mesh (1.5 cm, stretch measure) (Rosenberg and Resh 1982; Merritt and Cummins 1996) were deployed 11 May, 2000 and 2001 in the three experimental buffers and the no harvest site and retrieved three weeks after deployment. When lifting the rock bags from the stream, special care was taken to capture invertebrates that may swim or drift away by making a sweep of the immediate area with an aquatic sampling net. Each rock bag was shaken vigorously in a five gallon bucket of water for 60 seconds to remove organisms, and then visually inspected to ensure complete removal of colonized invertebrates. The water was then filtered through a 500 μm Nitex screen and all retained organisms preserved in 95% ethanol. Three of the six rock bags from each of the four sites were randomly selected due to the amount of time and work required for each sample, and invertebrates belonging to the Orders of Trichoptera, Ephemeroptera, and Plecoptera identified to the species or genus using taxomonic keys provided by Merritt and Cummins (1996), Larson (1997a, b), and Larson et al. (2000). Specimens belonging to the Family Chironomidae were identified to Family whereas all other Diptera were identified to Order.

For the insect data, the generalized linear model, with poisson errors, and log link was used (SAS 1988; McCullagh and Nelder 1989) with an alpha of 0.05. The assumptions of this model are that the residuals are homogeneous, normal, and independent. All of the assumptions were met. The number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) were totaled and divided by the number of Diptera (including Chironomidae) to create an EPT/D metric (Resh and Jackson 1992) that were observed for each sample using the explanatory variable of buffer treatment and year as an interaction term. This model was also used to determine whether there was a statistical difference between pre- and post-harvest between the four sites using the explanatory variable of site and year as an interaction term for the following response variables: the EPT/D, number of EPT, number of Ephemeroptera, number of Plecoptera, number of Trichoptera, number of Diptera (excluding Chironomidae), and the number of Chironomidae.

Results
Sedimentation

The experimental treatment that differed significantly between pre- and post-harvest conditions, compared to the no harvest site was the 20 m buffer with selective harvesting where significantly greater amount of sediment accrued for both sediment size fractions (Figure 1). The increase in sedimentation for the 20 m buffer with harvesting increased by about 290 g and was substantially greater than that recorded in the provincially regulated standard 20 m buffer (increase of 40 g), 30-50 m buffer with selective harvesting (decrease of 3.6 g) and the no-harvest control (increase of 19.6 g). Both 20 m buffers had greater sediment accumulated than the no harvest control treatment, with two times greater sediment in the 20 m buffer, and 14.5 times greater in the 20 m buffer with selective harvesting.

Invertebrates

We identified differences among the three harvest treatments and the no harvest control using general linear models and comparing the differences with the three harvest treatments in 2000 with that in 2001. These analyses were performed on the predominant invertebrate groups in 2000 (pre-harvest) with that in 2001 (post-harvest). These analyses showed differences between years varied with the taxonomic group and the harvest treatment (Figure 2). These analyses showed significant ($p < 0.01$) interactions between time and treatment for densities of EPT, Plecoptera, Trichoptera, Ephemeroptera, Chironomidae, and Diptera (excluding Chironomidae).

The EPT/D index was not significantly affected by the interaction term of buffer treatment and year ($p = 0.50$, Table 1). In contrast, the number of EPT was significantly affected by the interaction term of site and year ($p < 0.01$) and all three experimental buffer treatments were significantly different from the no harvest treatment (Table 1). The percent change for the number of EPT increased for the 20 m buffer and the no harvest control but decreased under the 20 m buffer with harvesting, and the 30-50 m buffer with harvesting (Figure 2).

The number of Plecoptera in both 20 m buffers (with and without selective harvesting) also differed

Table 1. Summary of p-values from the generalized linear model for comparisons of invertebrate metrics and densities of select taxonomic groups from rock bag substrata for the experimental treatments that differed from the no harvest control treatment.

Response variable	P-value	Sites statistically different from the no harvest site for the interaction term of treatment and year
Number of EPT/D	0.50	NA
Number of EPT	< 0.01	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Ephemeroptera	< 0.01	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Plecoptera	0.01	20 m; 20 m with harvesting
Number of Trichoptera	< 0.01	30-50 m with harvesting
Number of Diptera	< 0.01	20 m; 20 m with harvesting; 30-50 m with harvesting
Number of Chironomidae	< 0.01	20 m; 30-50 m with harvesting

significantly ($p = 0.01$, Table 1) from the no harvest site for the interaction term site and year with the percent change of Plecoptera decreasing for both 20 m buffers (with and without selective harvesting; Figure 2). The 30-50 m with harvesting site was not significantly different from the no harvest site.

The number of Trichoptera was significantly related to the interaction term site and year ($p < 0.01$, Table 1) with the number of Trichoptera increasing for all sites including the control. The only site that differed significantly from the no harvest site was the 30-50 m buffer with harvesting ($p < 0.01$, Table 1). The largest percent increase was observed for the 20 m buffer (Figure 2). Upon examination of the raw data, a single genus, *Oxyethira* sp. (Trichoptera) was responsible for this large increase in the 20 m buffer site, pre- and post-harvest, increasing 52 times ($n = 3$ pre-harvest; $n = 155$ post-harvest).

The number of Ephemeroptera also differed significantly in all three of the experimental compared to the no harvest site (treatment * year interaction, $p < 0.01$) (Table 1). Numbers of Ephemeroptera decreased in all of the experimental treatments in 2001 compared with 2000, with the greatest decrease observed in the 30-50 m buffer with harvesting treatment (Figure 2).

The number of Chironomidae was also significantly related to the interaction term treatment and year ($p < 0.01$) with the 20 m buffer and the 30-50 m buffer with harvesting both significantly different from the no harvest site. The 30-50 m buffer with harvesting demonstrated the only positive percent change, while the 20 m buffer demonstrated the greatest negative percent change in the number of Chironomidae (Figure 2).

The number of Diptera (excluding Chironomidae) was also significantly related to the interaction term site and year ($p < 0.01$) with all three experimental buffers differing significantly from the no harvest site (Table 1). All sites, including the control, demonstrated increased numbers of Diptera with the largest percent increase observed with the 20 m buffer with harvesting; an 18.9% increase (Figure 2).

Discussion

Sedimentation

The effectiveness of buffer strips of various widths to filter sediments and reduce inputs to streams has been investigated by various researchers. For this study, the only buffer that did not appear to be affected by overland flow of sediment entering the stream was the 30-50 m buffer. Evidence from existing studies suggests that logging roads can be an important source of sedimentation. Observations during this study suggest the sedimentation problem originated from heavily used forwarding trails that resulted in the exposure of mineral soils which may have been mobilized and entered the stream. There were a few locations along this brook where point sources were obvious and originated approximately 200 m from the stream, and sediment laden runoff was not completely filtered over a distance of 180 m that was clearcut and had slash remaining.

Salo and Cundy (1987) state the impact of yarding operations on ground disturbance can be extensive when tractors are used, but with minimal impacts when cable systems, that fully or partially suspend the logs, are implemented. In this study, logs and pulpwood were transported using a forwarder with a self contained loading rear rack. This forwarder was a six wheel drive unit with the two rear wheels on each side contained within a metal track, supposedly to reduce soil disturbance. Conversely, the effects of yarding using this forwarder were extensive with large

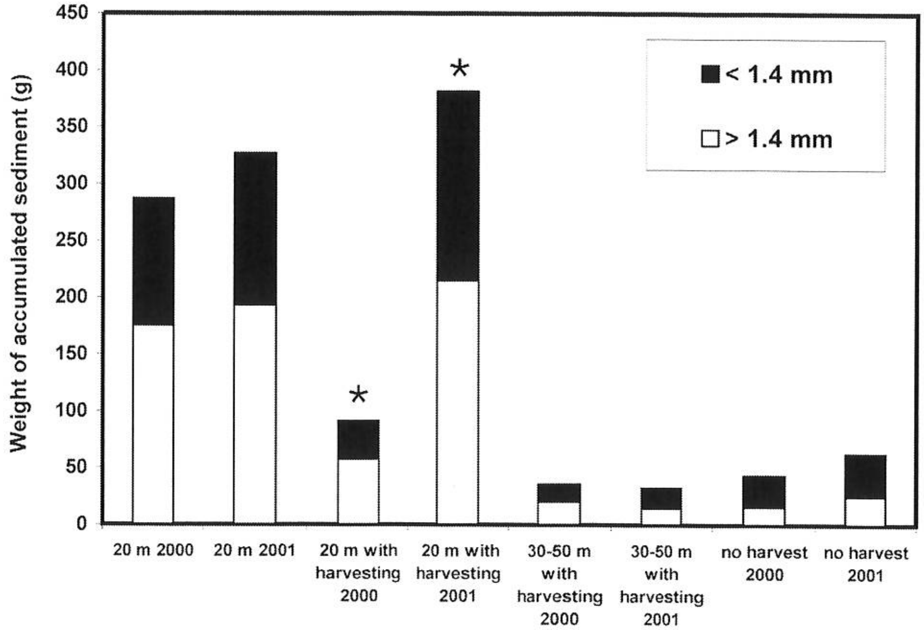

Figure 1. Comparison of total sediment accumulation (g) between the three experimental sites and the no harvest site separated into two size classes: (i) sediment greater than 1.4 mm and (ii) fine sediment (less than 1.4 mm).

vertical ruts in excess of 0.5 m in the forwarding trails that became a major source of sedimentation overland, and which eventually entered the stream.

The Copper Lake study conducted in western Newfoundland found that increased sedimentation largely resulted from road construction, specifically the installation of culverts (Clarke et al. 1998). A limited clear cut also resulted in increased sedimentation in addition to the accumulation attributed to culvert installation (Clarke et al. 1998). While the results of our study identify increased sedimentation in two of the harvest treatments, we cannot attribute the result to road crossings as they were absent from the study area.

Invertebrates

Results from our study identified taxon-specific changes in invertebrate abundance to the three harvest treatments, albeit effects were relatively variable. In contrast, the Copper Lake study conducted in western Newfoundland did not show any clear trend related to macroinvertebrates (Clarke et al. 1998). It has also been shown that macroinvertebrate biomass may increase with sediment addition resulting from a proliferation of sediment tolerant taxa (Blosser 1984). However, in this study, there did not appear to be any increase in macroinvertebrate abundance in relation to increased sedimentation which in part may result because many of the classically sediment tolerant taxa (e.g. oligochaetes) were not highly abundant at any of the study sites.

Plecoptera are typically considered to be intolerant of pollution and their presence often indicates good water quality (e.g. Lydy et al. 2000). Our data showed that the density of Plecoptera decreased for both 20 m buffer treatments, whereas their abundance increased in the remaining two treatments. This decrease corresponded with an increase in sedimentation for the 20 m buffers, specifically the 20 m buffer with selective harvesting. Similarly, Carlson et al. (1990) reported similar reductions in Plecoptera with logging.

Members of the orders EPT are known to be sensitive to pollution, and we expected the abundance of this group to decrease with a decrease in water quality (Norris and Georges 1993). However, Murphy and Hall (1981) and Murphy et al. (1981) found that the density of invertebrates is higher in clearcut sites. This is similar to the findings of this study where the percentage change in the number of EPT increased only in the 20 m buffer and this response was driven by the large percentage increase in Oxyethira sp. Erman et al. (1977) suspected the changes in the invertebrate communities within logged streams resulted from increased nutrients and light. These results suggest a possible increase in primary production possibly from increased nutrient loadings within Hungry Brook.

Individuals belonging to the genus *Oxyethira* sp. was responsible for the single largest percent increase in invertebrate densities between pre- and post-harvest

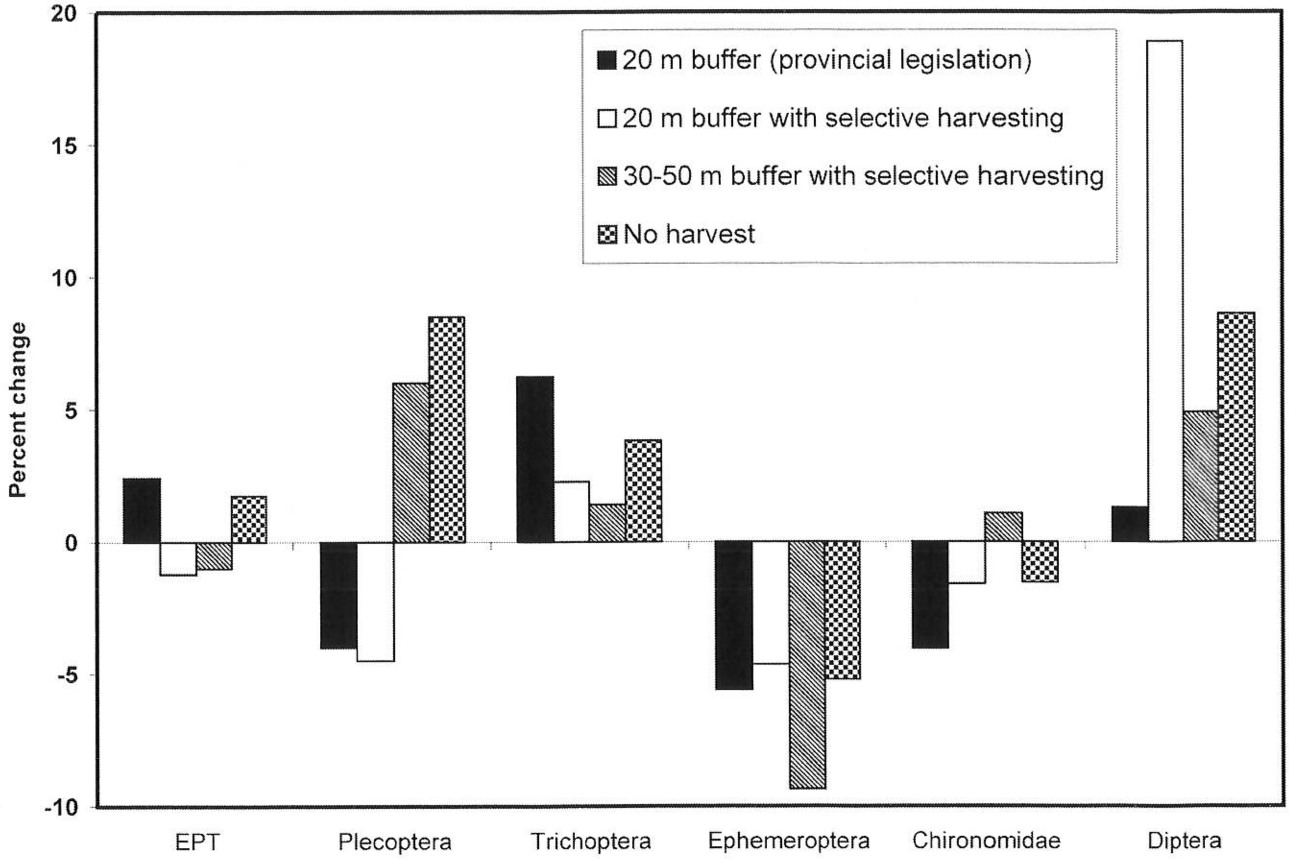

Figure 2. Percent change in various taxa between pre- and post-harvest observations for three experimental buffers and a no harvest (control) site. (N.B. Number of Diptera excludes Chironomidae).

in the 20 m buffer. This species is often associated with filamentous algae which it consumes (Winterbourn and Gregson 1989). Several authors have reported increased primary production after clearcuting compared to forested areas (e.g. Bormann, 1968; Gregory 1976; Johnson et al. 1986). While we did not quantify the efforts of harvesting on algal production, we expect that the increase in density of *Oxyethira* may have resulted from increased nutrient loadings and stimulation of the algal community that *Oxyethira* utilizes as its primary food source. The findings of increased density Trichoptera following harvesting is consistent with that reported by Carlson et al. (1990) who observed caddisflies to be more numerous at logged sites. Trichoptera increased in our study in the two selective harvesting experimental sites, but the increase was less than that observed for the no harvest site. The 20 m buffer observed the greatest increase and was greater than the no harvest site.

The number of Diptera (excluding Chironomidae) displayed the largest percent increase of all invertebrate orders. Carlson et al. (1990) also found that true flies were significantly more numerous at logged sites. Our results showed that Diptera increased at all sites, including the no harvest control and only the 20 m with harvesting exhibited increases greater than the no harvest site. The vast majority of Diptera were members of the family Simuliidae and this taxa was responsible for the large increase observed for the 20 m buffer with selective harvesting perhaps due to a mobilization of dissolved organic carbon.

In contrast, the number of Chironomidae did not display any pattern with respect to pre- and post-harvesting. Both 20 m buffers and the no harvest control displayed a decrease in percentage change, whereas the 30-50 m buffer with harvesting exhibited a slight increase. On the contrary, Erman et al. (1977) found increased numbers of Chironomidae in logged streams within their study.

The results of our study indicate that the buffer with the least amount of change overall was the 30-50 m buffer with selective harvesting. This result also suggests that increased sediment accumulation may not necessarily be accompanied with a decrease in numbers of macroinvertebrates, however, a change in the taxonomic composition of EPT was found. As well, the potential for increased primary production, as suggested by the increase in algal consumers for the 20 m buffer in our study needs more investigation in future studies with similar environmental characteristics as those in Newfoundland. Furthermore, the results of this study demonstrate the importance of repeating studies in areas where environmental, ecological, and soil characteristics differ, even within a small geographical area so that regulations can become site specific not general regulations for large areas.

Acknowledgments

I acknowledge the Department of Forest Resources and Agrifoods, the Indian Bay Ecosystem Corporation and the Western Newfoundland Model Forest for financial support. I also acknowledge the support, assistance and thoughtful suggestions from Murray Wells over the past two years during field work and for countless hours of assistance during field data collection.

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Assessing the Impacts of Forest Harvesting Within a Small Newfoundland Headwater System: With an Evaluation of a 20 Metre 'No Cut' Buffer as Means to Mitigating Harmful Effects

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Extended Abstract

The Copper Lake Buffer Zone Study was a multi-year research project initiated to evaluate the effectiveness of a 20 meter 'no-cut' buffer zone in mitigating the detrimental effects of forest harvesting on small headwater streams. The study was initiated in the summer of 1993 with pre-harvest conditions being monitored from 1993 to 1995. A limited clear cut was conducted during the winter of 1995 with harvesting being completed during the summer of 1996. Streams within the watershed were harvested with one of three 'buffer' treatments. Streams were either clear cut to the waters edge, i.e. no buffer, with a 20 meter buffer or with a 100 meter buffer which served as a control. The subsequent two years of the study (1997 and 1998) were intensively monitored for post harvest conditions.

This paper gives an overview of observed changes in physical habitat parameters such as large woody debris dynamics, sediment accumulation, water temperature, and water quality within the Copper Lake watershed over the course of the study. These physical changes are discussed with respect to their potential effect on brook trout (Salvelinus fontinalis) population characteristics (density, biomass, age-class structure, production), incubation habitats and migration patterns.

In general, the streams harvested to investigate historical conditions, i.e. no buffer, had expected physical changes. These streams could not replenish LWD, had higher summer

Clarke, K.D., Scruton, D.A., Curry, R.A. and McCarthy, J.H. 2004. Assessing the impacts of forest harvesting within a small Newfoundland headwater system: with an evaluation of a 20 metre 'no cut' buffer as means to mitigating harmful effects. Pages 87-88 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

and incubation temperatures and had increases in sedimentation. These physical changes resulted in localized reduction in habitat quality which in turn reduced brook trout recruitment and utilization of affected areas. The 20 meter buffers, for the most part, provided adequate protection from the physical habitat changes induced by forest harvesting. One of the 20 meter buffered streams however could not fully mitigate the sediment pulse during a major storm event in June 1995 and this stream did show increased diel summer water temperature post harvesting.

It is important to note that sediment accumulation appeared to be the most detrimental of the physical changes observed in the Copper Lake example. Most of this increased sediment appears to be derived from road crossings within the Copper Lake watershed; a problem that may not be mitigated by any size buffer strip. This increased sediment persisted through time and continued to be a problem up to the end of the study. The ability to remove accumulated sediment from small streams will depend on the particle size of the sediment, presence of 'refugia' for sediment to accumulate, and the hydraulic power of the streams during peak flow conditions.

Managing Disturbance in Riparian Zones I - Historical Patterns of Terrestrial Disturbance in Riparian Zones of West-Central Alberta

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Extended Abstract

From 1999 to 2003, an integrated natural disturbance pattern research program at the Foothills Model Forest, Alberta studied fire patterns within riparian zones at four different scales. Specifically, we looked at whether, or to what degree, historical forest fire burning patterns in riparian zones differ from that of the landscape as a whole. We hypothesized that differences in species composition, relative tree density, topographic position, and eco-site type create unique disturbance regimes in those parts of the landscape. In fact, we found no evidence of either the age-class distribution, or percentage of old forest of riparian zones differing from the rest of the landscape at very coarse scales. However, at finer-scales, patterns began to emerge. For example, we found evidence that small, partially burnt residual islands tend to form at or near riparian zones more often than expected. Such islands tend to survive the fires relatively intact, form at wide streams and on wetter sites, and chances are good that the surviving trees will be white spruce. We also found evidence that fires tend to stop at riparian zones more than expected, and particularly so on large streams with steep slopes. However, in all cases, the relationships were weak, and highly variable. Overall, fire burnt through the vast majority of the riparian zones we studied, and the majority of island remnants occur nowhere near riparian zones. Furthermore, the high variation in the results suggests that the most likely source of variation in fire behaviour is local fire weather. We also found evidence to suggest that fire in riparian zones is at least partially responsible for the unique habitat characteristics of riparian zones. Field data demonstrated that some riparian zones experience steady ingress of tree species for many decades after a fire. Presumably, fire maintains these sites in grasses and forbs when it burns through. Disturbance in riparian zone is also the only means of creating "young" riparian habitat, and reducing the landscape hazard of other natural disturbances.

Andison, D.W. and McCleary, K. 2004. Managing disturbance in riparian zones I - historical patterns of terrestrial disturbance in riparian zones of west-central Alberta. Pages 89-90 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Thus, although we found evidence to suggest that riparian zones burn somewhat differently than upland parts of the landscape, the fact that fire has been an integral process within riparian ecosystems is inescapable. The removal or prevention of disturbances from these habitats would thus presumably have significant ecological consequences. On the other hand, the re-introduction of cultural disturbance techniques comes with many aquatic and terrestrial ecological pitfalls. This dilemma was the catalyst for forming a multi-agency group of regulators and managers, whose shared goal was to identify and explore key disturbance-related riparian functions and impacts, and ultimately explore more sustainable management solutions for riparian zones. We specifically wanted to make the link between patterns of terrestrial disturbance and their impacts on both terrestrial and aquatic habitat, stream morphology, and ultimately, fish population levels. The three talks to follow are a summary of the direction and efforts of this group to date towards that goal.

Managing Disturbance in Riparian Zones II: Source, Quantity and Function of Large Woody Debris in Small Foothills Streams Following Fire

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Extended Abstract

In 2002, a group of regulators and managers with shared goals was formed to identify the effects of wildfire on the production and retention of large woody (LWD) debris and its ecological function in small streams of Alberta. These efforts are part of a larger initiative to identify management practices to support the sustainable management of riparian zones. One of the more obvious links between disturbance and riparian function is the generation of LWD. While much is known about the importance of LWD in fish-bearing streams from studies in the coastal environments of North America, considerably less is known about the importance of this structural element in headwater streams in continental regions.

In October of 2001, the Dogrib fire started in the front ranges of the Rocky Mountains west of Sundre, Alberta and quickly spread into the adjoining foothills. This event provided an opportunity to evaluate the effects of wildfire on the source, quantity and function of LWD in headwater foothills streams within a managed forest ecosystem. In the two-year period following the fire, I evaluated these elements using remote sensing and field-based inventories. Initial research included the mapping and division of streams within the study area into reaches with similar gradients and upstream drainage area classes. Candidate reaches included those within mature forest greater than 15 m in height that had not been locally influenced by: i) road building, ii) pre-fire harvest or iii) post-fire salvage harvesting. Sample sites were randomly selected among all of the candidate reach slope and watershed area class combinations. We interpreted low-level aerial photography to estimate the size of all visible LWD present at 19 sample reaches and completed field inventories at eight of these reaches.

We found that individual tree fall events rather than debris flows were responsible for delivering LWD to stream channels. In fact, initial analyses suggest that on average, 90 percent of the recently recruited LWD originated from trees that grew within 7.3 m of the stream. Data

McCleary, R.J. 2004. Managing disturbance in riparian zones II: source, quantity and function of large woody debris in small foothills streams following fire. Pages 91-92 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

from our field surveys at 8 sites showed that instream volumes of LWD averaged 2.9 m^3 / 100 m^2 (range = 0.1 to 9.1 m^3 / 100 m^2 bankfull channel surface). The input mechanism for instream pieces was unknown for 55 percent of the pieces and 4 percent of the pieces were confirmed as fire-generated. Of these fire-generated pieces, 50 percent formed full bridges across the channel, 25 percent formed partial bridges across the channel and none had any interaction with the stream channel. Of all instream pieces, 48 percent performed a function such as pool formation, sediment storage or bank protection. Of the functioning pieces, 76 percent were well incorporated into the channel bed or banks and 14 percent were partial or full bridges.

Managing Disturbance in Riparian Zones III: A Dendroecological Analysis of Large Woody Debris in Riparian Zones of the Foothills Model Forest, Alberta

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Extended Abstract

Two years ago, a multi-stakeholder group of regulators and managers with shared goals was formed to identify and explore key disturbance-related riparian functions and impacts, and to explore sustainable management solutions for riparian zones. One of the more obvious links between disturbance and riparian function is the generation of large woody debris (LWD). We formulated a study to describe the amounts, distribution, and types of LWD following a recent fire in west-central Alberta. We combined this "snap-shot" approach with the use of existing tree-ring techniques in an attempt to "date" LWD in and near headwater streams with known disturbance histories. Tree rings are often used as environmental proxies because variation in tree ring widths reflects environmental factors that limit annual growth. In ecologically good years, ring widths are wider than normal and in adverse years they are narrower. Matching the narrow and wide patterns in ring-width series of trees of the same species growing at similar sites is called crossdating. This method can be used to ensure that tree-ring data are accurate at an annual resolution and to assign calendar years to the tree-rings of snags and logs to determine the year that trees died. In our study, we used crossdating to: i) determine the year of death of trees that have fallen into streams, and ii) reconstruct the disturbance history of the surrounding riparian zones. More specifically, our goals were to identify historic trends in the death and recruitment of LWD, determine if recruitment of LWD is chronic or episodic and related to discrete disturbances and to quantify LWD decomposition and determine if

Powell, S.R., Daniels, L.D., Andison, D.W. and McCleary, R.J. 2004. Managing disturbance in riparian zones III: A dendroecological analysis of large woody debris in riparian zones of the Foothills Model Forest, Alberta. Pages 93-94 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

statistically discrete decay classes exist.

Although in the initial stages, our study has determined year of death of logs from two of eight sites by statistically crossdating ring-width series from logs to a master chronology generated from the living canopy dominant trees. At one white spruce site, the oldest coarse woody debris sample had an outermost ring date of 1855, although tree death was continuous during the 1900s in response to fine-scale disturbances. At another site, a lodgepole pine stand established after a fire ca. 1900. Here, tree death occurred in pulses from 1944 to 1955 and from 1964 to 1970 in response to canopy development and competition. The stages of decay of LWD varied with time since death and reflect the position and function of the wood in the stream. LWD in anaerobic environments, such as logs submerged in water and those incorporated into the stream bank, decays slowly. Our preliminary results provide evidence that LWD may persist in the riparian environment for more than a hundred years. Thus, in order to be ecologically sustainable, forest management must account for impacts on the amount and type of woody debris in riparian forests on the order of a century. Future work entails the analysis of data from the remaining six sites.

Managing Disturbance in Riparian Zones IV – Can Harvesting be used as a Surrogate for Natural Disturbance? Testing the Waters in the Weldwood Forest Management Area

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Extended Abstract

Integrated research at the Foothills Model Forest on natural disturbance patterns and key disturbance-related riparian functions shows the importance of fire as a disturbance agent in riparian zones. In contrast, Alberta riparian zones are managed by protecting buffer strips on both sides of stream channels. Over the long term, this traditional approach develops forested ribbons along streams, where fire events and timber harvesting are prevented. This alters the natural disturbance regime of riparian areas and, in particular, reduces the overall rate of riparian zone disturbance.

If disturbance in riparian zones is necessary for maintaining ecological functions over a long period, a new approach to riparian management is needed. One option is to use harvesting effort, in place of natural fire events, as a disturbance agent and as a method to increase the disturbance rate and approximate natural disturbance patterns. This strategy, however, must include the maintenance of important ecological functions such as the interaction of large woody debris (LWD) with stream channels. Identifying how many and which trees should be retained, for the maintenance of these critical functions, is also a priority. Similar questions arise for other riparian functions, as trees and other riparian vegetation also maintain channel stability, provide wildlife habitat, influence water quality, and provide nutrient flows to aquatic ecosystems.

Weldwood is currently developing and testing a riparian management strategy with two main components. The strategic component looks at disturbance and function as important elements of natural riparian variation and plans the general directions needed to maintain variability that is consistent with fire events and applicable to various watershed scales. The site-specific component looks at individual stands and develops disturbance plans based on riparian values and functions for each site, while considering related topographic, ecological, and other factors. For example, LWD conservation on a watershed basis would be a function

Bonar, R.L. 2004. Managing disturbance in riparian zones IV – can harvesting be used as a surrogate for natural disturbance? Testing the waters in the Weldwood Forest Management Area. Pages 95-96 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

of the rate and location of disturbance in the watershed. LWD conservation at a particular site would depend on channel size, tree proximity to the channel, and the role LWD plays in maintaining ecological function within the channel. LWD conservation away from the channel would depend on other values such as wildlife habitat.

This approach is illustrated with an example watershed planned to approximate a fire event within the watershed, and experimental harvesting at sites within the riparian zone. The disturbance-based approach is compared to the traditional approach at multiple scales and the differences and perceived costs and benefits of the new approach are discussed.

Replacing natural fire events with harvesting programs is an attractive option for disturbance in some riparian zones, but is not suitable in all situations. Some sites are too environmentally sensitive for machinery, while harvesting in others would not be economically feasible or socially acceptable. Other disturbance agents such as prescribed fire may be more appropriate in these areas, while some stands should be left undisturbed to undergo age-related stand breakup. Together with a multi-stakeholder group formed over two years ago, we will continue to explore linkages between disturbance regimes, investigate ecological functions, and apply adaptive management tests such as this one to help develop sustainable management solutions for riparian areas.

The Impact of Riparian Management on Old Growth: Simulation and Analysis of Four Buffer Guidelines in Northwestern Alberta

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Abstract

As timber harvest expands over previously unharvested landscapes, riparian buffers become an increasingly important component of old growth. This paper examined the impact of different riparian buffer guidelines on the amount and type of old growth in northwestern Alberta. Four different sets of riparian guidelines were applied through computer simulation over two forest management units (>312,000 ha timber merchantable) over a 200 year time horizon. Scenarios included; 1) Alberta's 1994 guidelines, 2) "Business-as-usual", reflecting a field interpretation of the 1994 guidelines, 3) Greater protection for fishbearing waterbodies, and 4) Two-zone buffers with partial harvest. As a prerequisite, all scenarios met wood supply and other economic targets. Across all scenarios, the amount of old growth initially accrues for 20 to 40 years but greatly decreases thereafter. In terms of location, it shifts from the harvested landbase to being concentrated in riparian buffers. Also, there was a virtual disappearance of old growth, mixedwood types from the landbase. Of the different buffer criteria, fish-bearing guidelines retained more buffer area particularly conifer. Few-sizes-fit-all retained the most deciduous and mixedwood old growth. Two-zone buffers exhibited less variance in either peaks or declines of old growth. Operational and field interpretations of the Alberta's 1994 guidelines resulted in the retention of 3 times more riparian area and 7 times more old growth than just the planning phase alone. This last result indicates the application of a precautionary principle by local timber managers at the field and operational phases overrode many of the specifics of different criteria for riparian buffers.

Lee, P. 2004. The impact of riparian management on old growth: simulation and analysis of four buffer guidelines in northwestern Alberta. Pages 97-103 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

The use of riparian buffers has been an important riparian management tool since its implementation in European forestry in the 1700's (Porter 1887). In North America, the practice of leaving buffers was first utilized in the late 1960's (Calhoun 1988 referenced in Broskfske *et al.* 1997) and today is almost universally applied (Lee et al. 2004a). Riparian buffers play a role in delineating and protecting aquatic biota and habitats, and some terrestrial biota and habitats (reviewed in Lee and Smyth 2003). Depending on the network of streams and waterbodies, and buffer width guidelines, the area retained within buffers can be significant. Buffers can represent significant repositories of old growth. As harvest proceeds over the landbase, much of the old growth within the boreal forest will be converted to younger seral stages. The caveat to this is that old growth found within riparian areas is anticipated to play an increasingly important role in the determining the old growth structure of the landscape.

This paper examined the impact of three different criteria for establishing riparian buffers on projections of old growth for different canopy types on both the operational (harvestable) and riparian landbases over a large area in the western foothills and mixedwood areas of Alberta. It also examined the disparity between estimations of riparian buffers based strictly on the planning phase and operation/field implementation phases of timber harvest.

Methods

The study area was the P1 and P2 Forest Management Units in northwestern Alberta (475,871 ha). It is bounded from 57.4 deg N. lat. to 56.4 deg. N. lat. and 117.0 deg. W. long. to 119.4 W. long. The area supports three ecological subregions. Areas of higher elevations were Upper Foothills, mid-elevation areas were Lower Foothills, and lower elevations including major river valleys were Dry and Central Mixedwoods (Alberta Environment 1994). The major watersheds in this area included the southern Notikewin, Whitemud, Hotchkiss, and Doig river basins. Watershed areas ranged from a mean of 26 ha for 1st order streams to 326,315 ha for 6th order streams. The total lengths (km) of streams declined exponentially for 1st through to 6th order; 4,737 km, 1,954 km, 938 km, 716 km, 655 km, and 384 km, respectively.

Scenarios

The commonly used planning model, Woodstock/Stanley 4, was parameterised to run four scenarios for riparian buffer management (Remsoft® Inc. 2003). The model split the landbase into merchantable timber and non-merchantable, i.e. non-operable areas, non-timber canopy types, water, and infrastructure. All scenarios were forced to meet current economic and timber supply targets (allowed 10% variance). Woodstock/Stanley 4 made small adjustments in the defining these landbase base types in order to economically optimise timber supply in each scenario. The area of timber merchantable land ranged from 312,752 ha (scenario 4) to 314,156 ha (scenario 2) while the non-merchantable area ranged from 161,715 ha (scenario 2) to 163,118 ha (scenario 4). These resulted in relatively minor differences (less than a 0.1% variance) amongst scenarios for seral stage and canopy type. Across all scenarios, the initial timber merchantable landbase was dominated by pure deciduous (~49% of timber landbase) and conifer (~34%) with less prominent areas of conifer-deciduous (~8%) and deciduous-conifer (~9%) mixedwoods. Initial age class distributions were dominated by older groups. Approximately, 62% of the timber landbase was mature (61 to 110 yrs), ~23% was old growth (>110 yrs) while ~15% was in younger seral stages. Further details are available in Lee et al. (2004b). The use of 110 years as a delineating age for old growth follows an extensive review of these forest types by Song (2002). Buffers widths used in scenarios are summarized in Table 1.

Scenario 1: Current Alberta Environment (1994) Riparian Buffer Guidelines

This scenario applied the current riparian guidelines in Alberta (Alberta Environment 1994). These are relatively simple, "few-sizes-fits-all" guidelines. Buffers are applied on the basis of waterbody types and waterbody size. Their original intent was for controlling sediment and shoreline erosion.

Scenario 2: Business-as-Usual

This scenario was based on an operational interpretation of the 1994 guidelines (Alberta Environment 1994). Variation from the above scenario was based on discussions with personnel at Peace River Pulp (Daishowa-Marubeni International), Peace River, Alberta. Quantitative correction factors were estimated for: 1) inappropriately labelled or classified waterbodies

Table 1: Riparian buffer guidelines used in this study. Bars represent waterbody classes that did not apply to the scenario.

Waterbody Class	Scenario 1	Scenario 2	Scenario 3	Scenario 4	
				No harvest	50% Volume
Large Permanent Stream (>5 m channel)	60	60	-	30	30
Large Permanent Stream (with fish)	-	-	55	30	30
Large Permanent Stream (no fish)	-	-	26	30	30
Small Permanent Stream (<5 m channel)	30	30	-	15	10
Small Permanent Stream (with fish)	-	-	43	20	10
Small Permanent Stream (no fish)	-	-	17	15	10
Intermittent Stream (defined bank, no permanent flow)	0	0	-	0	10
Intermittent (buffered)	-	30	25	0	10
Intermittent (buffered or no fish)	-	-	12	0	10
Intermittent Stream (no fish)	-	-			
Ephemeral (no defined bank, no permanent flow)	0	0	0	0	5
Small Lake (with fishing or recreational potential; >4 ha)	100	100	-	70	30
Small Lake (with fish)	-	-	55	-	-
Small Lake (no fish)	-	-	28	-	-
Large Lake (with little or no fishing or recreational potential; >16 ha)	100	100	-	-	-
Large Lake (no fish)			55	70	30
Large Lake (no fish)	-	-	28	70	30
Water Source	20	20	55	15	5
Water Sources (no fish)	-	-	28	-	-

on the map layers, 2) the presence of streams in the field not marked on maps, and 3) application of wider buffers due to local topography.

Scenario 3: Fishbearing Streams

This scenario used fish habitat and the presence of fish as the primary drivers for the application of buffers. Based on a review of Boreal and Rocky Mountain jurisdictions (Lee and Smyth 2003), mean buffer widths were estimated and applied in this study. Data on the distribution of fish was available for some waterbodies on the study site (Scrimgeour et al. 2003). Waterbodies with no empirical data were assigned fish presence on the basis of a relationships developed from empirical data (Scrimgeour et al. 2003).

Scenario 4: Zoned Riparian Buffers

This scenario was based on jurisdictions that utilize a two zone harvest systems in riparian areas (reviewed in Lee and Smyth 2003). The buffer widths applied in this scenario were derived from those found in other Boreal and Rocky Mountain jurisdictions. The outer zone included partial harvest (50% volume removal) while the inner zone closest to the shoreline was no harvest.

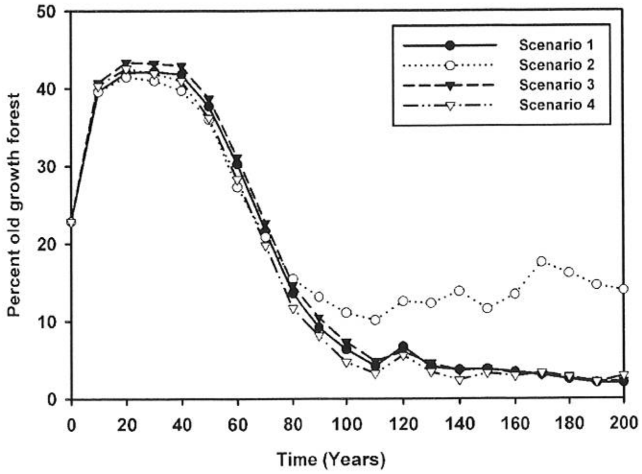

Figure 1: Changes in the percentage in old growth (>110 years) for a timber merchantable landbase in northwestern Alberta. Each line represents changes created a single scenario (see text for explanation).

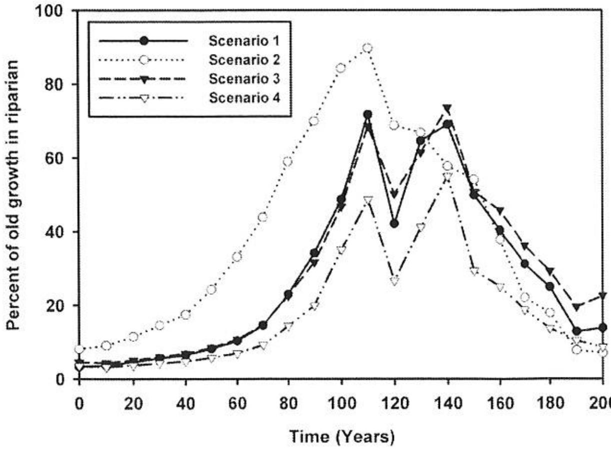

Figure 2: Percent of all old growth (>110 years) found in the riparian buffer area for a timber merchantable landbase in northwestern Alberta. Each line represents changes created a single scenario (see text for explanation).

Results

All scenarios projected an initial increase but later an eventual loss of old growth forest resulting from its conversion into younger seral stages (Figure 1). The amount of old growth was expected to increase for the next 20 to 40 years and peak to about 40 to 45% of the timber merchantable landbase. After this period, old growth declined rapidly for the next 90 years. For scenarios 1, 3, and 4, less than 3% of the timber merchantable landbase was old growth at the end of the 200 years. Scenario 2 did not significantly differ from others for the first 80 years. However, in contrast to other scenarios, it retained 10 to 17% of the timber merchantable landbase in the final 50 years of the simulation.

The location of old growth was also anticipated to change over time (Figure 2). From 0 to 90 years, most of the old growth was found on the harvest landbase whereas from 90 to 140 years, most of the old growth shifted to riparian buffers. Scenario 2 projected an earlier shift to the riparian landbase (~80 years) than other scenarios (Figure 2). This was due to a more aggressive cutting from the harvested landbase. The switch was less dramatic for scenario 4 with only a maximum of 56% of old growth being found in riparian areas. After 140 years, the old growth was expected to transfer back to the harvest landbase.

Lastly, the presence of mixedwood old growth (conifer-deciduous and deciduous-conifer) was forecast to greatly decline by end of the simulation runs (Figure 3). Initially, mixedwoods decreased to ~10% of all old growth in 40 years. After, models suggested an increase to ~ 40%. This is followed by a second projected decrease to 3 to 10% in 80 to 100 years and a second peak to about ~27% for scenarios 1 and 4 at 120 years. By the end of the simulation (>150 years), mixedwood old growth accounted for less than 5% of all old growth with a very low chance of recovery. The importance of riparian buffers as the repository mixedwood old growth increased with time (Figure 4). After 80 years, all of the mixedwood old growth was found in riparian buffers for all scenarios. Afterwards, the sudden shifts in location old growth mixedwood were due to its the relatively small area (<4,000 ha maximum; mean ~1,300 ha across all scenarios).

Discussion

The results of this study support the widely held view that forest harvesting will result in an overall decline of old growth through time. This change comes about as the age or seral class distribution, in this case, over 200 years becomes more regulated. That is, stands are harvested prior to the natural onset of old growth characteristics. Due to the configuration of the study landbase and the dominance of older ages, mature stands only are primarily harvested over the first 40 years. This allows a pulse of old growth to reach a peak at 20 to 40 years. These patterns may lead to a view that loss of old growth is not an issue. However, by about 60 years, the amount is less than what was present at the beginning of the simulation. Furthermore, all scenarios suggest that

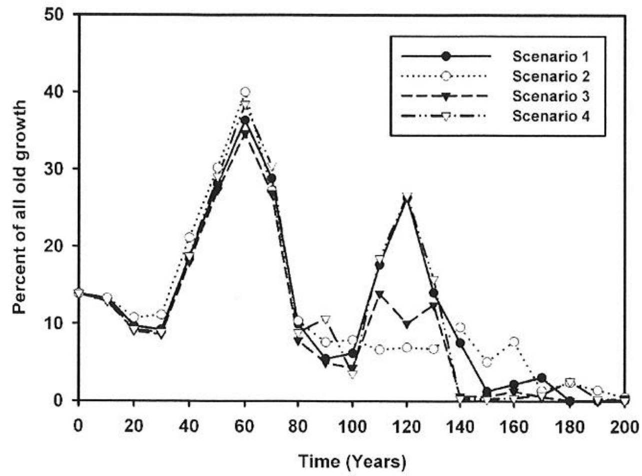

Figure 3: Percent of all old growth that is mixedwood (conifer-deciduous or deciduous-conifer) for a timber merchantable landbase in northwestern Alberta. Each line represents changes created a single scenario (see text for explanation).

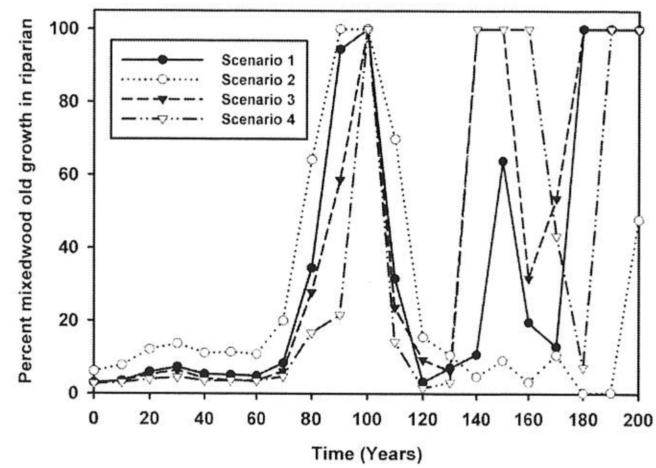

Figure 4: Percent of mixedwood old growth in riparian buffers for a timber merchantable landbase in northwestern Alberta. Each line represents changes created a single scenario (see text for explanation).

these levels were unlikely to ever occur again.

The projected shift of old growth from harvest areas to riparian buffers occurs for all canopy types (Lee et al. 2004b). The return of old growth to harvest areas is largely due to the deciduous forests. Conifer and mixedwood old growth forests are projected to remain in the riparian landbase after 80 years. In particular, mixedwood old growth forests are found at lower elevations within buffers associated with Central and Dry mixedwood subregions (Alberta Environment 1994). These subregions maintain a greater mixedwood component. Hence, the buffers of higher order streams with their wider channels and buffers are projected to become important refugia for this mixedwood old growth.

Comparison of scenarios suggested that for the business-as-usual rules as utilized by Peace River Pulp would provide for more riparian areas and, hence, more old growth. In this regard, a corporate culture applying a precautionary approach was critical and resulted in three important amendments to the assignment of buffers to waterbodies. The majority of hydrological map layers contain errors relating to misclassifying stream size and stream presence. The map layers were conservative in their classification of waterbodies. That is, field assessment of waterbodies particularly intermittent stream types re-classified (about 50% of the time) these to larger classes, often to small permanent streams. The delineation harvesting areas in the field often yielded the presence of waterbodies not marked on maps. These would be field classified and appropriate buffers attached. Field personnel suggested that unmapped waterbodies would be found for 2 to 5% of the harvest blocks. Lastly, the lack of an accurate digital elevation layer (2-5 m resolution) meant that buffers attached to many waterbodies were often too narrow for local topographic conditions. Greater slope, erodable soils, or channel configuration often led field personnel to attach wider buffers than would be attached through a strict interpretation of available maps. In these cases, the field personnel would place buffer widths that were at least the width of the 1994 guidelines and applied an additional width to account for local topography.

Of the three guideline formulations examined, fish-bearing guidelines (scenario 3) retained the most riparian area with "few-sizes-fits-all" (scenario 1) and two-zone buffers (scenario 2) retaining about the same amounts. Scenario 3 retained more conifer while scenario 1 retained more deciduous and mixedwood forests in buffers. Scenario 4 tended to have much less variability than the other scenarios. Peaks and troughs in the amount of riparian area were not as dramatic as other scenarios. The retention of some buffer through partial harvest reduced the overall variability.

Throughout this paper, I have cast riparian buffers as serving dual ecological roles of riparian protection and old growth. There is a relatively large amount of empirical evidence reviewing the ecological function of riparian zones (reviewed in Wenger 1999; Lee and Smyth 2003). However, the question of whether riparian buffer strips contribute significantly the maintenance of old growth on the landbase is much less clear. Under a low disturbance regime, i.e. wildfires or harvest, they certainly will become older but it is unclear whether they will support old growth habitat and biota. Their unusual configuration; relatively narrow, linear network, and topographic bias in location leaves open this question. For now, most managers are content with the use of buffers for riparian protection. In order for managers to accrue riparian buffers as old growth, research will be required to determine whether the buffers function as old growth.

Management Implications

This study has three important implications for managers. First, the degree to which field calls influence the amount of riparian buffer placed over the landscape. More detailed maps would be costly and the decision on whether their benefits warrant the large financial investment is not well understood. The primary benefit is more predictive planning and estimation of riparian areas. However, it could be argued that greater detail is not required provided estimations are prefaced by an understanding that they will underestimate the riparian area.

Second, the amount of riparian and old growth is dependent on a corporate culture that applies a precautionary principle when it comes to dealing with riparian areas. This was a quantifiably more important factor than the criteria-basis for the guideline. With additional field calls, the area of buffers was three times greater than strict adherence to current and other proposed guidelines. To achieve the type of corporate culture that applies this principle, managers must emphasize the flow of higher level, stewardship objectives to everyday planning and operations through education, training, and incentive programs. This would appear to be the case, in this study, with Peace River Pulp (Daishowa Marubeni Industries), Peace River, Alberta.

Third and arguably the most important, is that our modelling suggests that riparian buffers are projected to become an increasingly important landscape components of old growth. If buffers act as an ecologically significant landbase for old growth, then their planning and retention are critical. They will increase in ecological value with time. Managers should be willing to invest now, in high quality, riparian buffers not only for their use in riparian protection but as old growth. In particular, riparian areas associated with larger order streams are an important landbase for mixedwood old growth.

Acknowledgements

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Effectiveness of Variable-Retention Riparian Buffers for Maintaining Thermal Regimes, Water Chemistry, and Benthic Invertebrate Communities of Small Headwater Streams in Central British Columbia

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Abstract

Variable-retention riparian buffer strips are used extensively in the forest harvesting industry; however, their efficacy for maintaining natural stream and riparian functions has not been well documented. We studied the effects of different variable-retention riparian buffer treatments applied to small sub-boreal headwater streams on stream temperature, water chemistry, and benthic invertebrates. Increases in stream temperatures were related to the amount of merchantable timber retained within the buffer strip. A patch treatment had the largest effect, followed by a low retention treatment and a high retention treatment. Seven years after harvesting completion, none of the treatments showed temporal recovery in stream temperatures. Stream water chemistry changed in all treatments. Significant increases in total dissolved phosphorous (TDP) and nitrate (NO_3^-) were observed, but similar increases in conductivity were not found. Lack of correlation with treatment type suggests that watershed scale processes, and not riparian processes, are largely responsible for water chemistry changes. Benthic invertebrate abundance and biomass changed in the high retention buffer only. A response in the low retention buffer may have been masked by high sedimentation in the first two post-harvest years. While offering some mitigation, variable-retention buffers do not appear to fully protect headwater streams from changes to thermal regimes, water chemistry, and invertebrate communities.

Herunter, H.E., Macdonald, J.S. and MacIsaac, E.A. 2004. Effectiveness of variable-retention riparian buffers for maintaining thermal regimes, water chemistry, and benthic invertebrate communities of small headwater streams in central British Columbia. Pages 105-113 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

Variable-retention riparian buffer strips are used for forest harvesting but their efficacy for maintaining natural stream and riparian functions has not been well documented. The concept of variable-retention buffers is to mitigate the impacts of clear cut harvesting, while allowing harvesting of merchantable timber within the riparian area for reasons of timber-value and windthrow management. Current forest practices within British Columbia (i.e., Forest Practices Code Act and Forest and Range Practices Act) permit harvesting of commercial timber within the riparian area of small fish bearing (< 1.5 m wide) and non-fish bearing (< 3 m wide) streams. These small tributary and headwater streams are ubiquitous throughout watersheds of the Pacific Northwest. While the effects of forest and riparian harvesting on larger coastal systems have been studied (Salo and Cundy 1987; Hartman and Scrivener 1990), there is little information on the applicability of these findings to small interior B.C. streams.

Riparian forest harvesting can impact stream and riparian functions such as thermal regimes, water chemistry, and invertebrate communities. Among the stream characteristics influenced by riparian harvesting, increased stream temperature is one of the most extensively studied (Salo and Cundy 1987; Johnson and Jones 2000; Macdonald 2003b). A variety of changes in stream water chemistry, in response to forest harvesting, have also been found (Bormann and Likens 1979; Salo and Cundy 1987; Hartman and Scrivener 1990). Benthic invertebrate abundance has been shown to change after riparian harvesting. It is believed these changes are in response to alterations in stream primary production and allochthonous inputs and changes in abiotic factors such as thermal regime, solar radiation, nutrients, substrates, and hydrology (Newbold et al. 1980; Kiffney et al. 2003).

As a component of the Stuart-Takla Fish/Forestry Interaction Study in the central interior of B.C. (Macdonald 1994), we tested the effects of different variable-retention riparian buffers on small headwater streams within the Baptiste watershed. This study included evaluations of the effects of variable-retention riparian buffers on stream temperature (Macdonald et al. 2003b), groundwater properties, freshet discharge and suspended sediment transport (Macdonald et al. 2003a), bedload movement, windthrow, water chemistry, ultraviolet radiation (Clare and Bothwell 2003), geochemistry, and benthic and drift invertebrate communities (MacIsaac 2003). This paper presents an update of temporal changes in stream temperature previously reported by Macdonald et al. (2003b), as well as a portion of the water chemistry and benthic invertebrate data collected.

Materials and Methods

Study Area

The study area is located in the Hogem Range of the Omineca Mountains, at the northern end of the Sub-boreal Spruce Biogeoclimatic Zone, in the northern most drainage of the Fraser River basin (Macdonald et al. 2003a, 2003b). Data from four small headwater streams with similar physical attributes were included in this analysis (Table 1). Peak discharge is largely controlled by spring snowmelt in late April to early June. During the period of lowest discharge, from November to April, ice and snow cover the streams.

Harvesting of two 60 ha cutblocks took place in the winter of 1996/1997 using feller-bunchers and skidders. Riparian prescriptions were: 1) low retention - removal of all merchantable timber [> 15 or > 20 cm diameter at breast height (dbh) for pine or spruce/balsam, respectively] within 20 m of the stream, 2) high retention - removal of large merchantable timber > 30 cm dbh within 20 m of the stream and, 3) "patch cut" - a high retention along the lower 60% of the stream and removal of all riparian vegetation in the upper 40%. A 5 m machine-exclusion zone along the streambanks, combined with efforts at falling and yarding trees away from the streams, maintained most riparian understory vegetation with the exception of the upper portion of the patch treatment.

Temperature

Stream temperatures were recorded at five stations, two controls (B5Hi and B4), and three treatments (B2-patch cut, B3Lo-high retention, and B5Lo-low retention) (Figure 1). Stream temperatures were recorded hourly throughout the year, using Vemco™ dataloggers, with precision to ±0.2°C. Recording began 18 months prior to harvesting and continues to the present day, providing seven years of post-harvest data.

Graphical data analysis was used to measure the temporal changes in average daily temperature between the control (B4) and the treatment streams (treatment temperature – control temperature = ΔT). Temperatures of less than zero occurred at some stations during the winter, when sensors were exposed to ice or air. These temperatures were assumed to be zero.

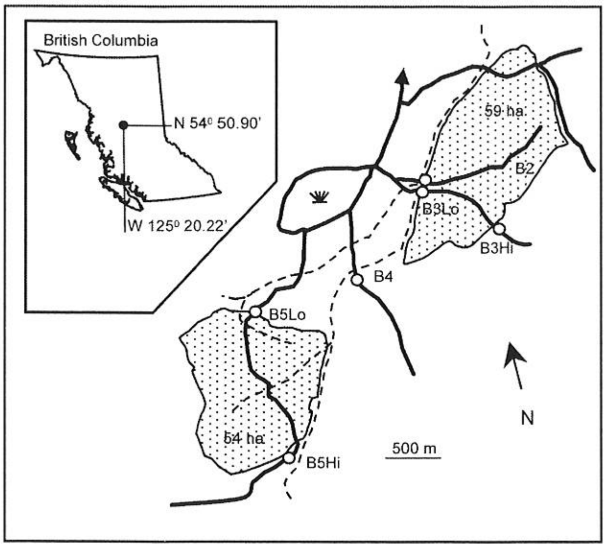

Figure 1. The location of small stream study sites in the Baptiste watershed. Thick lines denote watercourses, thin lines denote cutblocks, dashed lines denote roads, and circles denote sampling stations. Cutblock sizes in hectares are noted within the cutblocks.

Water Chemistry

Grab samples for water chemistry analysis were collected during summer months (May, June, July, August, and September) in 1997, 1998, and 1999. Samples were collected at two treatment stations located below both cutblocks (B3Lo and B5Lo) and at two control stations located above the cutblocks (B3Hi and B5Hi). Random duplicate samples were collected for quality control measures. In 1996, pre-harvest samples were collected in August and September only. Total dissolved phosphorus (TDP), nitrate (NO_3^-), and conductivity data are presented in this paper.

All water samples were filtered in the field. NO_3^- and conductivity samples were frozen in washed Nalgene™ bottles until analysis. TDP samples were filtered into and stored in digestion tubes. Nitrate plus nitrite was analysed as the azodye after cadmium reduction and is reported as NO_3^-. TDP was digested with persulphate and analysed as molybdate-blue after reduction with stannous chloride. Conductivity was measured with a WTW model LF330 conductivity meter.

Annual parameter estimates were calculated for each station by averaging the five monthly grab samples taken each year. A Two-Way Anova was performed for each parameter (TDP, NO_3^-, and conductivity) with station (four levels: B3Hi, B3Lo, B5Hi, and B5Lo) and year (three levels: 1997, 1998, and 1999) as factors (α = 0.05), using the monthly samples as response data. The 1996 pre-harvest data was excluded to keep the Anova design balanced (i.e., there were only two samples from each station for this year compared to five samples for the successive years).

Invertebrates

Benthic invertebrates were collected using colonization baskets at the four stations associated with water chemistry sampling (B5Lo, B5Hi, B3Lo, and B3Hi). Plastic colanders (11 cm high, 16 cm diameter) were filled with clean, 1-5 cm diameter gravel. Three colanders were embedded flush with the streambed at each station in riffle/glide habitat. Invertebrates were sampled three times during the summer, usually in June, July and September, after 4 to 6 weeks of sampler deployment. Each sample was washed with filtered

Table 1. Physical and treatment characteristics of the study streams in the Omineca Mountains, British Columbia, Canada. The patch treatment (B2) had 250 m of clear-cut and 560 m of high retention riparian treatment. RMA = riparian management area, elevation is in meters above sea level as measured with a GPS.

Stream	Riparian Treatment	Riparian Mgmt. Area Width (m)	Aspect	Bankfull Width (m)	Elevation (m)	Length through Cutblock (m)	Gradient (°)	Watershed Size (ha)	Percent watershed harvested
B2	Patch	20	NW	1.0	980	810 (250+560)	12	18	89%
B3	High Retention	20	NW	0.6	980	550	26	43	38%
B4	Control	-	NW	0.9	980	-	30	48	-
B5	Low Retention	20	N	1.4	980	800	7	150	40%

water three times, and the invertebrates were decanted off. The remaining coarse material was scanned for invertebrates, and all samples were preserved in 10% buffered formaldehyde. Invertebrates > 250 μm were enumerated and identified to family level (where possible) and functional group using Merritt and Cummins (1996). Biomass was obtained for each taxa using standard dry weight methods.

Annual estimates for the stations were obtained by first averaging the three samples from each station deployment, then by averaging the results from the three sampler deployments for each year. Two-Way Anova tests were performed using invertebrate biomass and abundance averages from each deployment as response variables. Station and year were used as factors in a similar manner as in the water chemistry analyses. Due to the inherent highly variable nature of invertebrate sampling, we chose to use an alpha value of 0.10. The patch treatment (B2) was not sampled for water chemistry or benthic invertebrates.

Results

Temperature

Temperatures for the control creek (B4) ranged from 0 to 10°C, with summer peaks usually occurring in early August. Post-harvest effects were evident and were most pronounced in the patch treatment (ΔT_{max} = 4.6), followed by the low retention (ΔT_{max} = 3.1) and high retention (ΔT_{max} = 2.5) treatments, respectively (Figure 2). Peaks in ΔT normally occurred in mid-June. No temporal recovery is evident after seven years of post-harvest monitoring in all treatments. Riparian windthrow events affecting the buffer strips in 1997, 1998, and 1999 may have contributed to the lack of recovery.

The relationship between the control stations (B4 and B5Hi) did not appear to dramatically change after logging, although there were small annual negative ΔT immediately followed by positive ΔT in early June (Figure 2). These small spring differences were due to earlier snowmelt in B4, which was located approximately 100 m lower in elevation than B5Hi. Positive deviations were generally less than 1°C, well below the magnitude of harvesting-related impacts recorded in the treatment streams.

Water Chemistry

The limited pre-harvest data collected in 1996 showed that although both TDP and NO_3^- levels had a tendency to increase downstream, their pre-treatment levels were relatively low (Figure 3). Both TDP and NO_3^- generally showed higher post-harvest levels at both the high and low retention treatment sites, relative to their respective controls. TDP was higher at both treatments (Tukey Simultaneous Test: $p < 0.01$ for B3 and $p = 0.01$ for B5), and the average increase was 2.2 fold for B3Lo and 1.6 fold for B5Lo. Higher NO_3^- levels, relative to the control stations, were detected only in B5 (Tukey Simultaneous Test: $p < 0.02$), representing an average increase of 3.7 times the control values. Nitrate levels in B3 were relatively high at the control (B3Hi) in 1998. No differences were noted in conductivity measurements between treatment and control stations, although there was a statistically significant year effect ($p < 0.01$).

Invertebrates

Average benthic invertebrate abundance and biomass (Figure 4) were elevated at the high retention treatment station (B3Lo), as compared to its control (B3Hi) (Tukey Simultaneous Test: $p < 0.01$and $p < 0.07$, respectively). This difference was primarily due to increases in collectors-gatherers of Diptera (Chironomidae) and Ephemeroptera (Baetidae). This was not the case for B5Lo, where there was no difference between the treatment site (B5Lo) and its control (B5Hi). The abundance and biomass values for B5Lo in 1997 and 1998 were equal or lower when compared to the values of the control, particularly with respect to the biomass estimates.

Discussion

Similar to other researchers (Brown and Krygier 1970; Holtby and Newcombe 1982; Johnson and Jones 2000), we show significant increases in stream water temperature after forest harvesting events. Our data suggested that the variable-retention buffers provided some mitigation from clear cutting (see Macdonald 2003b), but did not fully protect the streams from thermal impacts. The effectiveness of the variable-retention buffer depended on the amount of commercial vegetation left within the buffer strip, which is positively correlated to canopy density (Macdonald et al. 2003b). Riparian canopy closure largely determines the amount of direct solar radiation reaching the stream surface, a process frequently cited as being most important in controlling temperatures of small forested streams (Salo and Cundy 1987). Conversely, air temperature

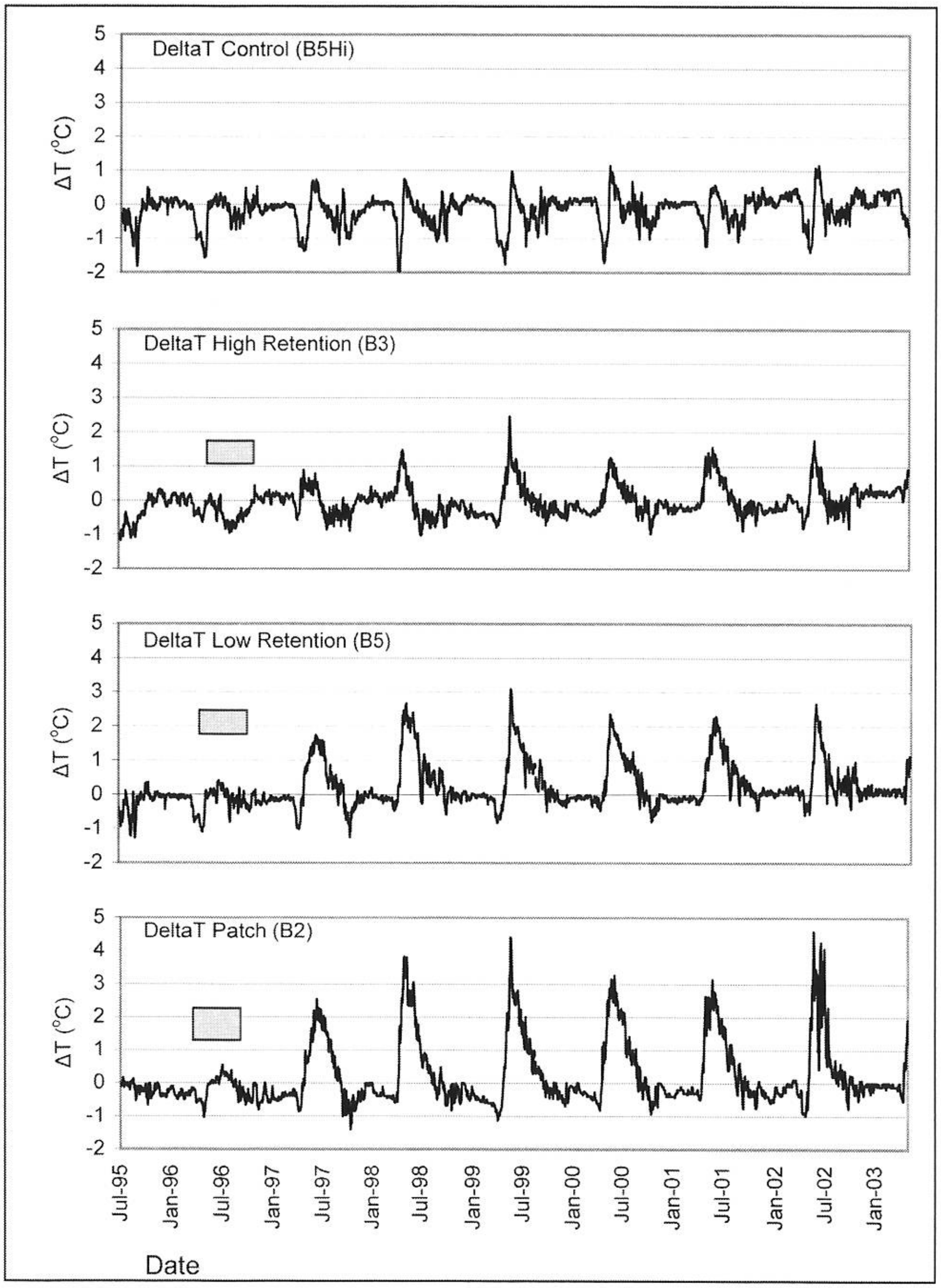

Figure 2. Thermal responses of streams with variable-retention buffer strips. Delta T is the average daily temperature of the control stream - treatment stream. Shaded box indicates when forest harvesting occurred. B5Hi served as a secondary upstream control.

and energy loss from the stream due to evaporation are thought to play minor roles (Brown and Krygier 1970). Other factors which control stream temperatures also bear consideration such as aspect, gradient, geomorphic features, elevation, and groundwater regimes.

Our analyses suggest that thermal regimes in the three riparian treatments did not recover to pre-harvest levels, even 7 years after harvesting. In contrast, most studies of stream thermal recovery predict substantial return to pre-harvest levels within five to seven years of harvest (e.g., Brown and Krygier 1970; Feller 1981; Harr and Fredriksen 1988). In the absence of comparable research from northern sub-boreal forests, we can speculate that stream temperatures in this type of ecozone may require greater recovery periods, due to slower growth of under- and over-story vegetation.

Several studies (e.g., Bormann and Likens 1979; Hartman and Scrivener 1990; Kiffney 2003) have

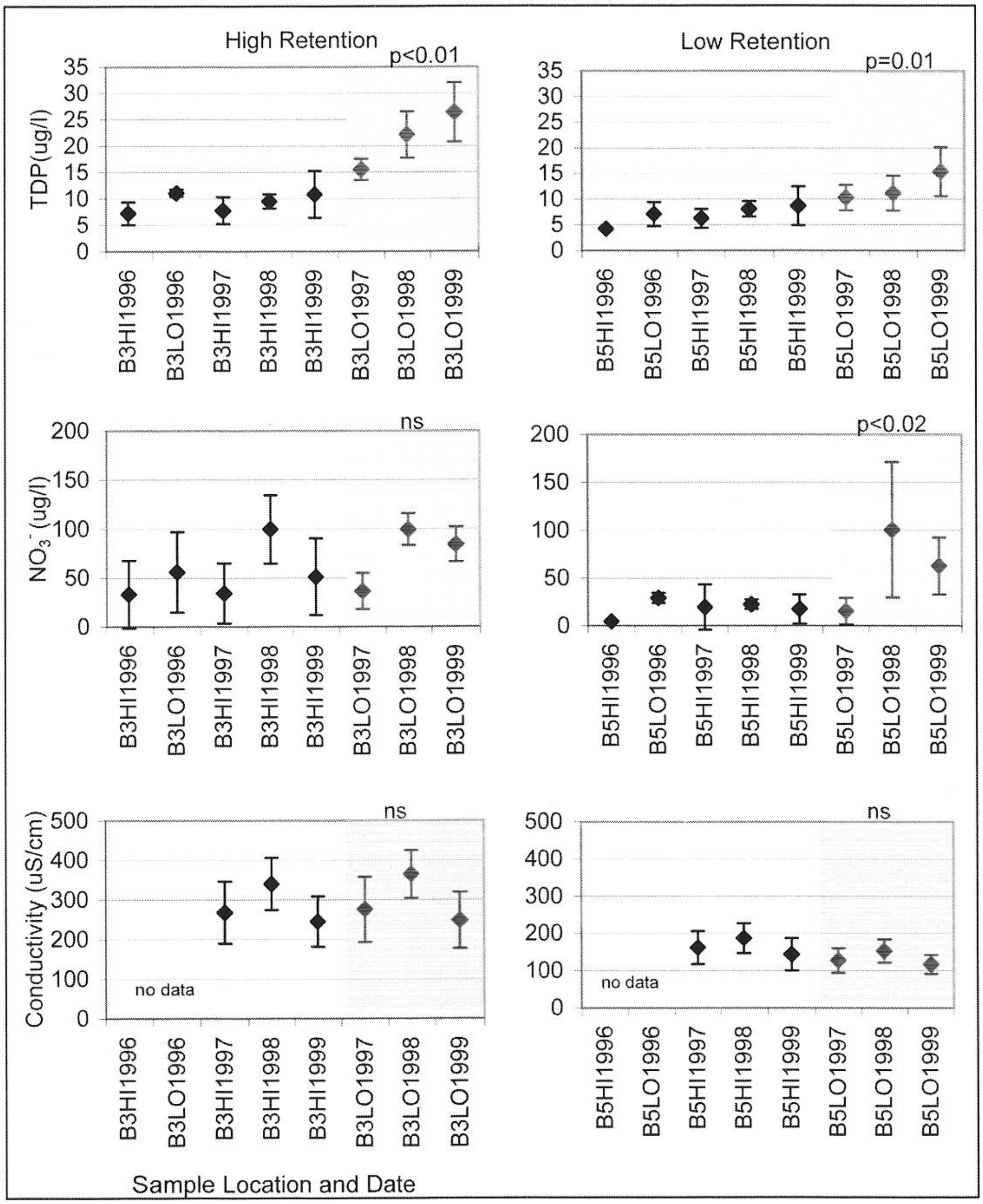

Figure 3. Mean annual (± 95% confidence intervals) water chemistry responses to variable-retention buffer strips. B3Lo had a high retention buffer while B5Lo had a low retention buffer. Post-harvest treatment sites are identified by shading. Significant differences (Two-Way Anova) between control and treatment sites are indicated above the shaded box (ns=not significant). Concentrations of TDP differed significantly for both treatments, where as concentrations of NO_3^- were significantly different for B5Lo only.

reported changes in stream water chemistry after forest harvesting. However, the effects often vary depending on site-specific processes such as geological weathering, atmospheric precipitation, soil chemistry, and terrestrial and aquatic biological processes. It is commonly reported that increases in nitrate levels and conductivity are observed, although little or no change in phosphorus levels is expected (e.g., Hartman and Scrivener 1990). In contrast, our results show a significant increase in TDP in the B3 and B5 streams and that nitrates exhibited higher trends in both treatments, but only in a statistically significant manner in B5. Conductivity did

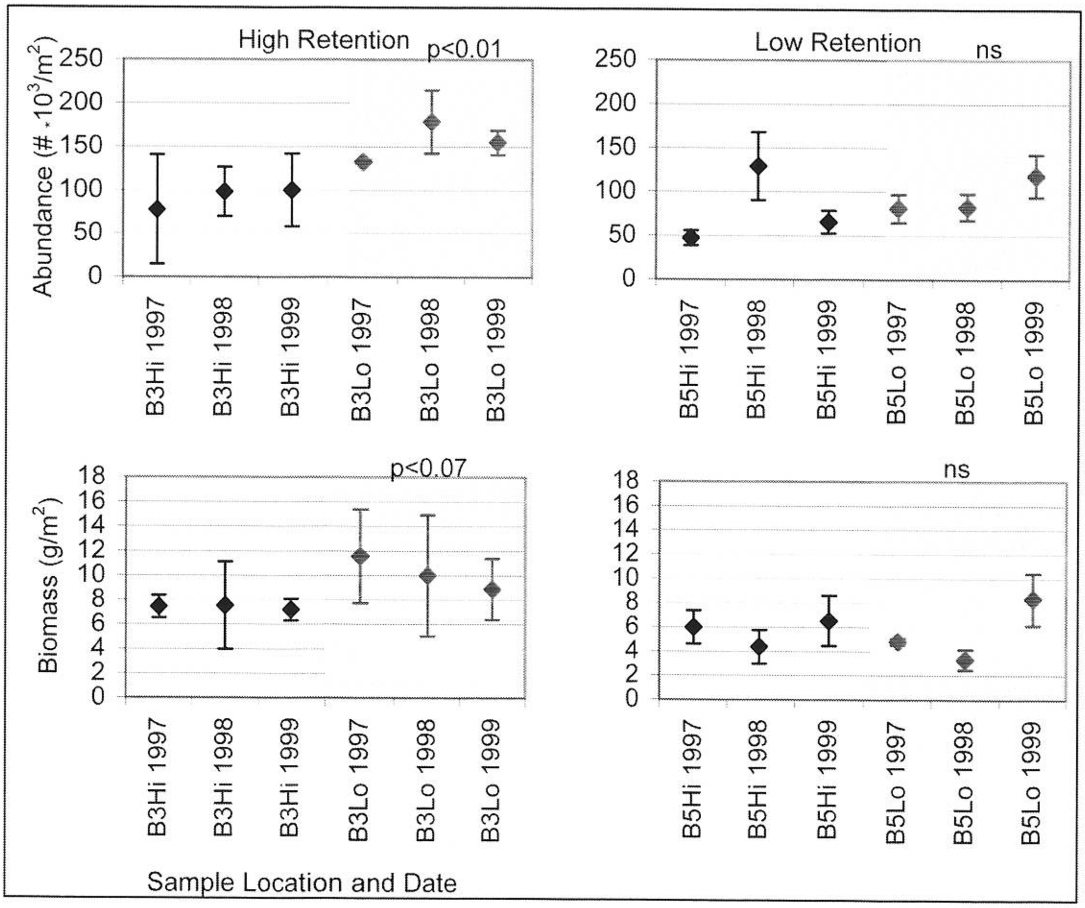

Figure 4. Mean annual (± 95% confidence intervals) variable-retention riparian buffer strip effects on relative invertebrate abundance (# * $10^3/m^2$) and biomass (g/m^2) at control (B3Hi and B5Hi) and treatment (B3Lo and B5Lo) sites. B3Lo had a high retention buffer while B5Lo had a low retention buffer. Post harvest treatment sites are identified by shading. Significant differences (Two-Way Anova) between control and treatment sites are indicated above the shaded box (ns=not significant). Both abundance and biomass were significantly different between B3Hi and B3Lo.

not differ between treatment sites and controls. Clare and Bothwell (2003), while studying the effects of UV on invertebrates in B5 during 1997, also found a similar increase in TDP after harvesting. In addition, they found a similar, significant increase in total dissolved nitrogen and unchanging levels of nitrates. However, their samples were taken over a shorter time frame and at the beginning of the summer season, when concentrations of NO_3^- are naturally low and when variations in water chemistry within streams may be limited.

There may have been a delayed response to forest harvesting in observed concentrations of NO_3^- and TDP. Both treatment sites showed relatively higher levels in 1998 and 1999 but not 1997, the first summer after logging (Figure 3). While conductivity was not significantly affected by our harvest treatment, it did differ among years. Conductivity increased at all stations in 1998, a year with little precipitation and low flow conditions. These conditions may have contributed to a lack of dilution, thereby increasing conductivity and resulting in high conductivity and concentrations of NO_3^- in B3Hi in 1998.

Riparian vegetation can significantly reduce nitrogen and phosphorus in overland and subsurface water flows (Naiman and Bilby 1998). Therefore, after forest harvesting, an increase in stream nutrient and ion levels would be expected. In addition, higher riparian retention might help buffer against water chemistry changes. Although this is consistent with the NO_3^- data in our study, it is not constant with our TDP data. Changes to TDP levels were highest in the high retention buffer (i.e., B3 showed a greater increase than B5).

Increased abundance of benthic invertebrates is a commonly reported response to forest harvesting (Newbold et al. 1980; Salo and Cundy 1987; Kiffney 2003). For instance, Newbold et al. (1980) found increases in both, Chironomidae, Ephemeroptera (Baetis) and Plecoptera (Nemoura) after forest harvesting. The increases we observed in abundance and biomass in B3 were driven largely by the grazing collector-gatherers Diptera (Chironomidae) and Ephemeroptera (Baetidae). Clare and Bothwell (2003), during their 1997 work in B5, found increased chironomid abundance in B5Lo. Our sampling occurred over a larger time span so some changes may have been masked or averaged out in our sampling.

As with water chemistry, there did not appear to be a variable-retention effect on invertebrates. Kiffney et al. (2003) found increased invertebrate abundance was correlated with narrower buffer widths and attributed this to increased primary production caused by increased photosynthetically active radiation. The lack of increase in B5Lo invertebrate abundance and biomass in 1997 and 1998 may have been due to increased sedimentation, associated with the removal of a stream crossing (Macdonald 2003a). High sedimentation has been shown to decrease invertebrate density (Salo and Cundy 1987). In 1999, B5Lo invertebrate density and biomass values were observed to be relatively high. It is, therefore, possible that riparian harvesting effects may have been masked by sedimentation during the early post-harvest years.

In summary, while offering some mitigation to the effects of clear-cutting, variable-retention buffers do not fully protect headwater streams from changes to thermal regimes, water chemistry, or invertebrate communities. Increases in stream temperatures were related to the amount of merchantable timber retained within the buffer strip. Lack of correlation with treatment type suggests that watershed level processes, and not riparian processes, are largely responsible for water chemistry changes. Benthic invertebrate abundance and biomass changes may have been due to changes in riparian vegetation as well as point source sedimentation. The study-streams continue to be monitored to assess whether temporal thermal recovery will occur. As there is more pressure on resource managers to move towards variable-retention buffers, further study is required to elucidate the specific mechanisms inflecting forestry induced change on sub-boreal streams.

Acknowledgements

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The Forest Watershed and Riparian Disturbance Study: Moving Forward from Buffer Strips to Integrated Watershed Management

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Extended Abstract

The goal of the Forest Watershed and Riparian Disturbance project is to model processes that link disturbance (i.e., fire, forest harvest) to the quantity and quality of water leaving small (1st order streams, up to 15 km^2) and ultimately larger (3rd to 4th order streams) watersheds in Canada's western boreal forest. Data are being collected on weather (e.g., precipitation, temperature, wind and solar radiation), soils (e.g., factors related to nutrient export, runoff, compaction and erosion), riparian vegetation (herb, shrub and tree community descriptors), stream discharge, stream water quality (in particular the nutrients nitrogen and phosphorus, and suspended sediments) and bioindicators. Impacts of fire and time-to-recovery are being evaluated in two large and two small watersheds, which were severely burned (84 to 89% of drainage basin area) in 1998. Two large and at least two small watersheds (see below) that were not burned serve as reference systems. Pre-fire data were collected in one large burned and one large reference watershed, and post-fire data were collected from all burned watersheds

Prepas, E.E., Russell, J.S., Smith, D.W., Putz, G., Meyer, W.L. and Burke, J.M. 2004. The forest watershed and riparian disturbance study: moving forward from buffer strips to integrated watershed management. Pages 115-116 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

beginning in 1999. The remaining ten watersheds in the study are small. Data were collected for three years prior to forest harvest (of more than 50% drainage basin area) during winter 2003/2004 and will be collected for at least two years after harvest. Five unharvested watersheds serve as reference systems for harvest and fire disturbance in small watersheds. The harvest design will evaluate the effectiveness of buffer strips in protecting surface water quality. Data will be used in two modeling approaches: 1) deterministic (SWAT) and 2) stochastic (artificial neural networks). Model output will have a spatial (scale up predictions to larger areas) and temporal (multi-decade horizon) component. Our industrial partners will incorporate these outputs in a spatially-explicit planning process to develop: 1) harvest constraint levels on a sub-watershed basis and 2) best management practices. This research also tests hypotheses related to effects of watershed disturbance on soils, hydrology, and water quality in the boreal forest of western Canada. Data generated can be linked with other small boreal watershed projects across the Canadian boreal forest (e.g., Turkey Lakes, ON).

Development of Alternative Streamflow and Water Quality Modelling Approaches for Simulation of Forest Disturbance Effects

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Extended Abstract

The Forest Watershed and Riparian Disturbance (FORWARD) project is investigating the impact of fire and harvesting disturbance on streamflow and water quality in several small (6 to 16 km^2) forested watersheds on the Boreal Plain in central Alberta. The project will also assess the importance of riparian buffers to moderate the effects of fire and harvesting on streamflow and water quality. These efforts will assist forest management planners to predict hydrological responses to harvesting activities. Two alternative modelling approaches are being investigated. The first is a deterministic approach that utilizes an adaptation of the Soil Water and Assessment Tool (SWAT) model developed by the United States Department of Agriculture. The second approach is the development of an artificial neural network model (ANN) for stochastic predictions of streamflow and water quality.

SWAT is a semi-distributed deterministic model that provides simulation of streamflow and water quality given precipitation input and environmental conditions such as topography, soils and vegetation within a watershed. The FORWARD project is attempting to use this model to predict long-term impacts of landscape changes due to fire and harvesting upon water quantity

Mckeown, R., Nour, M.H., Khan, A., Putz, G. and Smith, D.W. 2004. Development of alternative streamflow and water quality modelling approaches for simulation of forest disturbance effects. Pages 117-118 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

and quality. Recognizing that SWAT was developed for agriculture, efforts are being made to adapt it for better representation of forested ecosystems. Initial changes to the model are concentrating on the incorporation of a litter layer under a treed stand to better represent the upper soil layer and other hydrologic processes inherent to forested environments. Future improvements of the SWAT model for forested watersheds will address the interaction of solar radiation and snow cover. Finally, the vegetation growth simulator within SWAT must be modified to include additional tree species in the forested environment and expanded to include multiyear growth capability over decades.

Artificial neural network (ANN) is a computing system made up of a number of simple and highly interconnected processing elements (neurons) which process information by their dynamic state response to external inputs. ANNs have been successfully used to identify dynamic systems, which exhibit complicated and non-linear behavior. Considerable research efforts have targeted rainfall runoff modelling as well as water quality modelling using ANN. Results were comparable to highly sophisticated physically based data intensive models making this field very promising. However, none of the efforts reviewed managed to account explicitly for spatial discrepancy of landscape variables and their impact on the system response. With the advent of remote sensing (RS) satellite imagery, data availability has become economically feasible. Yet, incorporating spatial variations of soil and vegetation parameters using ANN and utilizing (RS) images, as inputs, are very challenging and need significant improvements. A conceptual design formulated to overcome these problems is described in this study as well as how RS time series images are processed and incorporated in the modelling phase.

The FORWARD project utilizes both modeling approaches to achieve the best possible results with reasonable model input requirements. The final delivery is expected to be a hybrid deterministic/stochastic modelling approach that capitalizes on the strength of the two methods.

Forest and Fishes: Effects of Flows and Foreigners on Southwestern Native Fishes

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Abstract

Habitat alteration in physical (stream channel characteristics), chemical (nutrients, temperature), or biological (introduced species) form can have dramatic effects on native southwestern USA fishes. Southwestern flow regimes, their alterations, and introduction of alien species have had a dramatic, negative impact on native southwestern fishes. The cumulative and interactive impacts may result in various responses by native fish assemblages. Managers should not expect the same result when one or more factors are in operation that may affect an aquatic ecosystem in the southwestern USA. Ultimately, consideration of temporal-spatial influences, natural factors, interactions of factors, and sound monitoring or research activities will determine which factors most influence southwestern fish assemblages in respective situations.

Rinne, J.N. 2004. Forest and fishes: effects of flows and foreigners on southwestern native fishes. Pages 119-124 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

Native fishes of the southwestern United States (Minckley 1973, Rinne 2003a) have declined dramatically in range and numbers in the last century (Rinne 1994, Mueller and Marsh 2002). Multiple, cumulative factors such as dams, diversions, introductions of non-indigenous species, and varying land uses have been implicated as factors causing their demise. The question can be asked, "What are the relative impacts of hydrology, introduced fishes and other organisms, and land uses such as timber harvest and livestock grazing on native fishes occupying southwestern riparian ecosystems?" The primary objective of this paper is to briefly introduce and delineate factors that have impacted historically and will potentially continue to negatively impact the native, largely threatened and endangered fish fauna of the American Southwest (Rinne and Minckley 1991, Rinne 2003a, 2003b). Each factor that individually, and ultimately, cumulatively impacts native fish assemblages will be introduced and evidence presented documenting the degree of impact on native fish assemblages in the southwestern U. S.

Cumulative, impacting factors

Hydrological and physical habitat alteration and introduction of nonnative fishes (Miller 1961, Rinne 1994, 2003b) are the two factors most commonly associated with the marked decline in range and numbers of most native fish species in the Southwest. As a result, the majority of the southwestern fish species have been officially listed as threatened or endangered (Rinne 2003a). Recently, land uses such as domestic livestock, grazing of forest landscapes and their riparian corridors (Rinne 2000) have been implicated as a negative impact on native fish assemblages. Studies of changes in fish assemblages on the upper Verde River, Arizona over the past decade (Figure 2) and literature on the topic over the past few decades will be used to demonstrate and document the relative impact of flows or stream hydrographs on fishes and the removal of livestock grazing and associated habitat changes.

Natural and human-altered hydrology

Where natural flow regimes persist, rivers change dramatically and abruptly temporally and spatially from flood to drought across the arid, more xeric regions of the interior West (Hubbs and Miller 1948). Similarly, the natural hydrology of southwestern desert rivers and streams is highly variable and episodic (Minckley and Meffe 1987, Rinne and Stefferud 1997) (Figure 1). In absence of any human-imposed factors, native fishes appear to be adapted to survive and sustain themselves under these conditions. Natural flow regimes have generally been considered optimum for sustaining native fishes (Poff et al. 1997).

The Southwest has sustained extensive and recently intensive human immigration. Accompanying this influx of Europeans was the ever-increasing demand for water that has resulted in dramatic alteration of the historic hydrology of the Southwest (Rinne 2002). The 1902 Bureau of Reclamation Act instituted these dramatic changes. The first Reclamation dam, Roosevelt, was completed on the Salt River in 1911 and the hydrology of the Salt River downstream was irreversibly changed. This dam and others retained peak flows that originated from upper elevation, forested areas in the Central Arizona Mountains.

In 1932, completion of Hoover Dam on the Colorado River and additional dams such as Glen Canyon Dam impounding Lake Powell imposed a dramatic and lasting change in the hydrologic regime of the Colorado River mainstem. Periodic natural and often quite dramatic flood flows (Mueller and Marsh 2002) were forever lost to the system. Rinne (1994) calculated that over 75% of the large mainstream river habitats in Arizona were either lost or altered between 1911 and 1970. Diversions such as Imperial Dam on the lower Colorado River and groundwater pumping imposed additional alterations to natural flow regimes. Coolidge Dam and the Ashurst-Hayden Diversion on the Gila River, both completed in 1928, effectively dried the Gila River to its confluence with the Salt River. Other major tributaries to the Gila from the south, the San Pedro, San Simon and Santa Cruz Rivers have been dried primarily as a result of groundwater pumping.

Fish response to altered hydrology

The Gila topminnow, *Poecilliopsis occidentalis*, was once (1940s) the commonest native species in the lower Colorado River (Minckley 1973, Hubbs and Miller 1941). It now persists naturally in fewer than a dozen, isolated diminutive spring heads or spring runs in southern Arizona (Meffe et al. 1983). The large Colorado pike minnow (*Ptyocheilus lucius*), historically referred to as the "Colorado salmon" by locals because of large spawning runs, is now extirpated in the lower Colorado River and might be only locally present as a result of restoration-repatriation programs. Similarly,

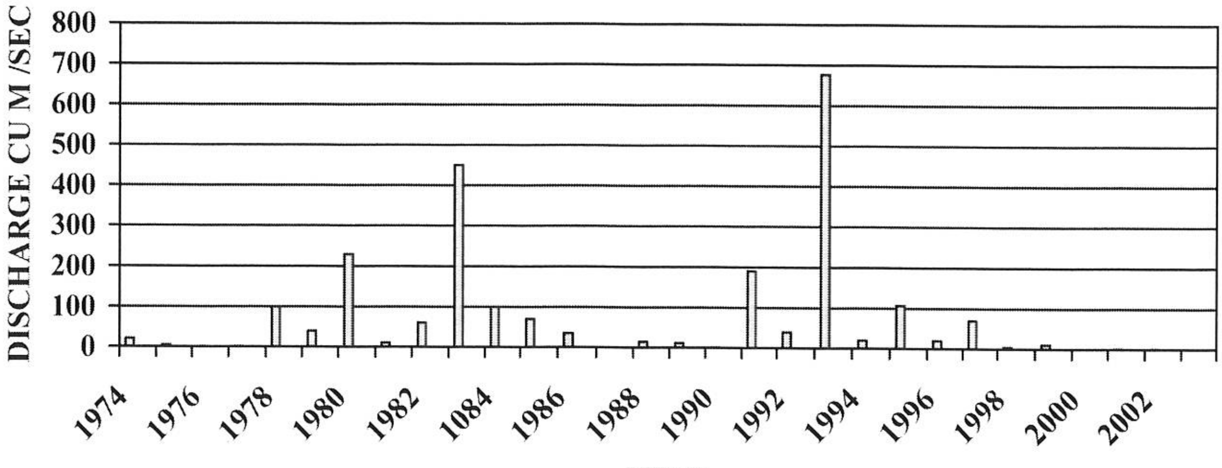

Figure 1. Instantaneous peak discharges in the Verde River, 1974-2002.

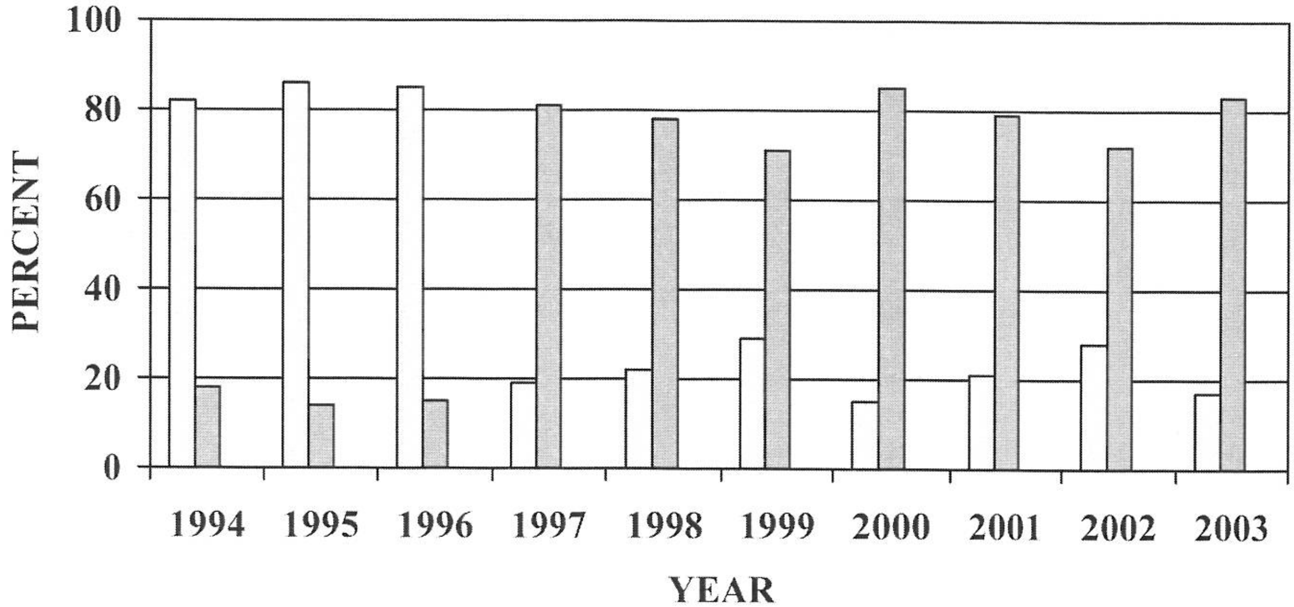

Figure 2. Change in relative proportions of natives (light bars) and nonnative (dark bars) in samples in the upper Verde River, 1994-2003, a period of low, drought flows (see Figure 1).

the razorback sucker (*Xyrauchen texanus)* was once so abundant in the river that it was pitch forked from canal systems in the Phoenix area and used as fertilizer. The bonytail chub (*Gila elegans)* along with razorback sucker occurs only in Colorado River reservoirs either as scenescent populations that correlate well with dam closures or as repatriated individuals. All these species are now officially listed as endangered. Others, such as spikedace (*Meda Fulgida)* and loach minnow (*Rhinichthys cobitis*) are threatened species. In the upper Verde River peak flows (Figure 1) have been demonstrated, in the short term, to be positively related to native fish populations (Figure 2; Rinne 2002).

Foreigners

Changes in fish assemblages

The native fish fauna of the Southwest is low in diversity and high in uniqueness and specialization (Miller 1961, Minckley 1973, Rinne and Minckley 1991). Fewer than 50 species of fishes naturally occurred in the waters of the Southwest and only two dozen were historic inhabitants in the waters of Arizona (Minckley 1973, Rinne and Minckley 1991). By comparison, over 100 species of fishes have been introduced into Arizona alone (Rinne 1994) and half have become established (Rinne 2003a) as self-sustaining populations. Most of the introductions were for sport fishing, which naturally followed the massive increase in reservoir surface water acres and habitat (Rinne 2003a). Rinne and Janisch (1995) reported the extensive coldwater introductions, and Rinne et al. (1998) the warmwater introductions in Arizona streams and lakes.

Nonnative, or non-indigenous fish introductions into foreign waters have generally been shown to have a negative, often dramatic impact (Courtenay and Stauffer 1984). In the Southwest, increased presence and abundance of these species is negatively correlated with native species. In the upper Verde River, in 1994, nonnative fishes comprised less than 10% of fishes captured (Figure 2). Only a decade later, in 2003, 90% of the fishes captured were nonnative species. In the Gila River, Colorado (Mueller and Marsh 2002) and Rio Grande rivers similar patterns of increase in non-native fishes is paralleled by an often, dramatic decrease in native species. Native trout species have declined dramatically with the introduction on nonnative trout. Rinne and Minckley (1985) documented the inverse distributions of the native Apache trout (*Oncorhynchus apache*) and introduced rainbow (*O. mykiss*) and brown (*Salmo trutta*) trout. Gila topminnow populations decrease in presence of the introduced mosquitofish (*Gambusia affinis)* (Meffe et al. 1983). Replacement can come by way of competition, hybridization or direct predation (Minckley 1983, Rinne 2003a). In summary, native southwestern fishes and non-native, predatory or competitory fishes generally cannot co-exist (Rinne et al. in press) in the same reaches of stream. Hydrological and geomorphological influences and interactions can alter this statement (Rinne 2002).

Other foreign species

In addition, other foreign aquatic species also have been introduced into the waters of the West and Southwest. Two principal species are a vertebrate, bullfrog (*Rana catesbiena*) and an invertebrate, crayfish (*Procambarus sp)*. Data, albeit mostly observational, indicate the dramatic impact of these two foreign aquatic species. White (1999) documented the impact of crayfish on the native Colorado spinedace (*Lepidomeda vittata)* through predation on eggs of this native, threatened fish species. However, in general data are lacking on the potential or real impact of these two species.

Domestic Livestock

Grazing of domestic livestock on upper elevation forested landscapes and riparian areas is generally thought to have an effect on fish habitats and fish species. However, most of the information pertain to salmonid species (Rinne 2000) and would apply only to the three native species of southwestern trouts (Gila (*Oncorhynchus gilae*), Apache (*O. apache*), and Rio Grande (*O. clarki virginalis*) cutthroat). Data on the upper Verde River, a warm water aquatic ecosystem in Arizona, do not corroborate the contention that livestock have a significant or even a demonstrable effect on native fishes (Figure 2). Removal of livestock on the upper Verde River in 1997 has resulted in markedly improved riparian conditions in form of increased vegetation and stream bank and channel alterations (Rinne and Miller in press). However, most native species, including the threatened spikedace, have declined in abundance and distribution in the upper Verde River. Most of the information addressing livestock grazing effects on fishes is 1) largely opinionated and conjecture, 2) based on qualitative, short term, non-replicated data, 3) primarily for salmonids, and 4) not based on sound science. Further, complicating and confounding factors make it difficult to produce definitive answers. The negative effect of grazing on native, cypriniform species for such variables as stream banks (Rinne and Neary 1997) and sediment levels (Rinne 2001) are not demonstrable. At present, there is no evidence, based on sound science, that grazing by domestic livestock has an obvious and well-documented negative effect on native fish species.

Cumulative, inter-active factors

The above factors that potentially negatively impact native southwestern fishes obviously do not act independently. That is, several factors operating simultaneously may produce a different result on fish assemblages in southwestern rivers. For example, flood flows on the upper Verde River in 1993 immediately favored the native fishes (Rinne and Stefferud 1997). Subsequently, low or drought flows (Figure 1) were paralleled by an increase in non-native species. Removal of livestock grazing on the river corridor was then superimposed. Although this management action improved riparian vegetation and is generally considered a favorable restorative action for "fish habitat," it has not resulted in an increase in native fishes (Figure 2). Indeed, the opposite appears to be true. The increase in cover and change in water depths have favored introduced, "cover seeking," more lentic species such as smallmouth bass (*Micropterus dolomieus*) and green sunfish (*Lepomis cyanellus*) (Pflieger 1975), yellow bullhead (*Ameiurus natalis*), mosquito fish (*Gambusia affinis*) and red shiner (*Cyprinella lutrensis*). The question becomes "Which of the two factors, flows (natural and altered) or foreigners in the form of nonnative fishes and domestic livestock has the greatest influence on native fishes?" Further, "Do livestock and non-native fishes have a greater influence on fish assemblages than does the hydrograph?"

In the upper Gila River, New Mexico, natural, historic flow regimes are extant in the Gila-Cliff Valley (Rinne 2002). Grazing occurs in most reaches of the river, however, livestock have been removed from the Gila Bird Area for a time period similar to that of the upper Verde River. The same native fish assemblage that occurs in the upper Verde consistently has comprised greater than 90% of the total numbers of fishes captured in these reaches over the past six years (Rinne and Miller in press). Native fishes also are predominant in contiguous grazed reaches. These data suggest that a natural, more variable hydrograph characterized by frequent flood events may override or more strongly influence fish assemblages than does domestic livestock grazing (Rinne 2002).

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An Evaluation of Large Woody Debris Restoration Efforts on the Manistee and Au Sable Rrivers

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Extended Abstract

The addition of large woody debris (LWD) may offer a practical management technique for managers enhancing habitat in midwestern forest streams. Although it is well established that LWD provides cover, structure and nutrients, which are critically important habitat values for salmonids in conifer-dominated watersheds, the relationship between LWD and fish habitat has not been previously investigated in northern hardwood forests.

In 2001, a helicopter was used to place whole trees in the Hodenpyl reach of the Manistee and Mio reach of the Au Sable Rivers, Michigan, to help restore structure and function to the aquatic ecosystem. This technique is used successfully in the Pacific Northwest. Helicopter placement has allowed for aquatic ecosystem restoration without compromising future LWD recruitment potential as trees were obtained from outside the riparian zone. A mix of large hardwood and coniferous trees were dropped in a semi-woven pattern to secure them to one another, minimize loss of deposited trees (i.e., runaways), and to create desired habitat conditions.

These rivers in Northern Michigan were sampled to determine the response of gamefish to whole tree treatments and to determine the mechanisms responsible for any observed responses. In particular, we determined if whole trees produce a response by altering stream habitat by increasing the food available via enhanced production of aquatic invertebrate drift, or increasing the survival or retention of trout near the structures. A split-plot design with paired treatment and reference reaches was implemented to determine the response of gamefish and aquatic invertebrates to whole tree implementations. Treatment and reference reaches 100 m long were selected in each stream and broken into three sections (left bank, middle, right bank). Preliminary results show little difference in the catch per effort between treatment and

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reference reaches, age frequency distributions in brown trout, our most abundant species, are very similar indicating similar survival rates and mark/recapture studies produced low recapture rates that may indicate low site retention. Also, preliminary drift sample results indicate higher invertebrate densities in areas influenced by LWD structures versus non-influence areas.

These data will provide a baseline for possible future studies on established LWD, and will aid in the design in future evaluations of LWD implementation projects. Numerous studies have emphasized the critical role that LWD plays in creating and maintaining fish habitat in streams. However the effects vary for several reasons, structural complexity, age of structure, diel fluctuations in fish distribution, and species present. Evaluation of fish habitat restoration projects, both in the shorter and longer term, are essential to improve biological, technical and cost effectiveness of projects, and to identify and incorporate innovations. Stream habitat is often manipulated to enhance gamefish under the assumption that habitat is somehow limiting. Managers are urged to identify the specific factor limiting the population of interest and manage for the population accordingly.

Decomposition and Longevity of In-Stream Woody Debris: A Review of Literature From North America

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Abstract

This paper presents a brief summary of a literature review completed to document the current state of knowledge regarding the decomposition and longevity of in-stream woody debris in western Canada. The summary provides brief overviews of riparian management implications, decomposition processes and rates, sensitivity of in-stream woody debris models to decomposition parameters and a summary of the longevity of in-stream woody debris.

Introduction

Over the last three decades, in-stream woody debris, more commonly termed small (SWD) and large woody debris (LWD), has received a great deal of research and management attention in forested regions of North America (Naiman et al. 2002). This focus stems from the recognition of the many important roles that in-stream woody debris plays in the ecology, geomorphology and biodiversity of streams and large rivers (Harmon et al. 1986; Bisson et al. 1987; Naiman et al. 2002). This focus stems from the many critical roles that in-stream woody debris plays in the maintenance of the ecology, geomorphology and biodiversity of streams and large rivers that either flow through or drain forested watersheds (Harmon et al. 1986; Bisson et al. 1987; Naiman et al. 2002). In-stream woody debris physically alters stream channel morphology creating areas of local channel scour and deposition (Beschta and Platts 1986; Fausch and Northcote 1992). Habitat for fish and aquatic organisms is created by woody debris altering channel morphology and through the dissipation of stream energy (Keller and Swanson 1979; Montgomery et al. 1995). In-stream woody debris also plays critical roles in creating cover for fish (Tschaplinski and Hartman 1983), providing long-term food for aquatic organisms (Dudley and Anderson 1982), retaining transported sediment and organic matter (Bilby and Ward 1989), cycling of nutrients (Bilby and Likens

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1980) and provide substrate for aquatic invertebrates (Anderson et al 1984; Sedell et al. 1988).

Even though a significant amount of research knowledge has been gained over the last three decades, several critical knowledge gaps still exist in relation to woody debris dynamics. Two of these knowledge gaps include the delivery rate and decomposition of in-stream woody debris (Naiman et al. 2002). Although not a comprehensive review, the focus of this paper is to provide a brief summary of the decomposition and subsequent longevity of in-stream woody debris. For purposes of this paper decomposition is defined as the decay, fragmentation and fluvial transport of woody debris within a stream channel network (i.e., small headwater streams downstream to large rivers).

Management Implications

An understanding of the decomposition processes and the longevity of woody debris have several important implications to the management of forested riparian environments. Current forest management has placed a significant amount of attention on the development of regulations and guidelines that protect proper riparian and stream functions. As a result, wood budget (e.g. Benda and Sias 2003) and riparian woody debris models (e.g. Welty et al. 2002) have been developed to help improve riparian management decisions, but more key information such as the decomposition of woody debris is required before a comprehensive riparian management decision-making tool can be created (Naiman et al. 2002). Critical in the development of such a tool is an improved understanding of the decomposition processes and longevity of woody debris in the various regions of western Canada (e.g. interior versus coast).

Further understanding of woody debris decomposition also has important implications in the design and longevity of stream restoration projects (Bilby et al. 1999). A common practice is to reintroduce wood into stream environments to restore or enhance fish habitat. Typically logs from coniferous species such as Douglas-fir or western red cedar are utilized in stream restoration projects since coniferous species decompose at a slower rate, are generally larger in diameter and are structurally stronger than deciduous species. The commercial value of these logs and the cost to transport the logs to a restoration site often makes the cost of restoration projects prohibitive. Recent research into the depletion of woody debris in western Washington has shown that deciduous species such as red alder and bigleaf maple may provide an alternative to the more commonly used coniferous logs when the logs remain submerged (Bilby et al. 1999). These deciduous species may provide cost savings since many deciduous species grow in riparian areas and are in close proximity to restoration sites, thus minimizing transportation costs.

An understanding of woody debris decomposition also has important implications in modeling CO_2 emissions and global carbon budgets associated with climate change. Issues related to climate change and the calculation of CO_2 emissions from land-use change and forest management has received much attention recently (Mackensen and Bauhus 1999). The decay of biomass associated with land-use change and forest management is a critical component in modeling CO_2 emissions. In a recent review, estimates of CO_2 emissions from decay of coarse woody debris in terrestrial environments were identified as being subject to large uncertainties (Mackensen and Bauhus 1999). This level of uncertainty is further complicated by limited information of decomposition rates and CO_2 emissions from aquatic or riparian environments (Guyette et al. 2002).

Decomposition Processes and Rates

The decomposition or depletion of woody debris in stream environments is complex and involves many biological and physical processes (Harmon et al. 1986; Golladay and Webster 1988; Maser and Trappe 1984). Harmon's (et al.1986) comprehensive synthesis paper outlined seven major categories of decay and material transfer processes. The processes include: fragmentation, leaching, collapse and settling, seasoning, transport, respiration and biological transformation. Additional controlling factors include climate, tree species (chemical content), piece size (diameter or length), decay class, position (suspended, on ground, buried, fully submerged), site conditions (temperature, moisture levels, oxygen and carbon dioxide levels), channel bed stability, channel morphology, flood intensity and riparian forest composition (Harmon et al. 1986; Naiman et al. 2002).

In determining the longevity of in-stream woody debris a decomposition rate (decay coefficient) is often calculated through long-term studies (e.g. time series), chronosequences (e.g. dendrochronolgy), input-biomass ratio (Harmon et al. 1986) or through a hybrid approach called the decomposition vector method that combines components of the long-tem study and chronosequences methods (Harmon et al. 2000). Decomposition of woody debris is most often expressed as a negative exponential decay rate function of:

$$X = X_0 e^{-kt}$$

where; X_0 is the initial mass, density or volume of woody debris, X is the quantity of material left at time t (years) and k is the decay rate constant. Although numerous studies have calculated woody debris decomposition rates in terrestrial ecosystems (refer to Caza 1993; Harmon et al. 1986 for summaries), few have documented decomposition rates in stream environments (Table 1). Decomposition rates summarized in Table 1 range from approximately 0.01 to 1.20 per year in stream environments. Variations in decomposition rates are highly dependant on wood species, wood chemistry, piece size and stream environment (e.g., submerged vs. partially submerged). Caution should be used in making comparisons between these studies since several different study methods were used in determination of the decomposition rates. Also, the reported decay constants often include different components of decomposition such as mineralization, fragmentation and transport.

As shown in Table 1 decay coefficients from stream research conducted in the Pacific Northwest were in general agreement with the combined range of values between 0.01 to 0.03 (Murphy and Koski 1989; Bilby et al. 1999; Hyatt and Naiman 2001). No studies were identified for the Intermountain region of the United States or interior regions of British Columbia and Alberta.

In-stream Woody Debris Models and Sensitivity to Decomposition Parameters

In the Pacific Northwest a few in-stream woody debris models or wood budgets have been developed to simulate the abundance and distribution of woody debris in forested streams (e.g., Kennard et al. 1998; Beechie et al. 2000; Bragg 2000; Welty et al 2002; Benda and Sias 2003). Only two of the models cited in the examples above provide a discussion of model sensitivity to decomposition parameters such as decay or fluvial transport.

Welty et al. (2002) identified their model to be highly sensitive to varying decomposition rates. In their sensitivity analysis example, a low and high decomposition rate of 0.6% to 5.4% generated a wide range in woody debris pieces per 100 meter of stream channel with the low rate generating an output of 180 pieces per 100 m of stream channel and the high rate generating an output of 4-8 pieces per 100 m of stream channel.

Benda and Sias (2003) highlighted the sensitivity of varying decay rates and provided a hypothesized algorithm for the persistence of in-stream woody associated with fluvial transport. For decay, Benda and Sias (2003) found that when a single exponential decay model is used 70% of the wood volume is lost within 40 years using a decay coefficient of 3%; whereas, if a decay coefficient of 1.5% is used the same loss of wood volume requires approximately 80 years. Alternatively, if a decay coefficient of 6% is used the wood volume loss requires approximately 20 years. In regards to fluvial transport of wood, Benda and Sias (2003) provided a detailed discussion of the complexity of modeling fluvial transport and stated that field data for the parameters and functions included in the hypothesized fluvial transport algorithm do not exist; therefore, no sensitivity analysis was provided regarding the fluvial transport component of the model. The parameters identified included the width of the stream channel in relation to a given wood piece length, variations in stream power associated with slope and cross-sectional area of the channel, log diameter, piece orientation, characteristics of woody debris jams and presence of root wads. To further highlight the complexity, Lienkaemper and Swanson (1987) point out that decomposition probably occurs disproportionately in years with relatively large flooding events. Hyatt and Naiman (2001) also identified reduced decomposition rates for a low proportion (< 20%) of the in-stream woody debris that was buried in floodplains and later "exhumed" by large flow events that mobilize the channel bed and cause channel migration.

The sensitivity of varying decay constants can further be highlighted through sensitivity analysis conducted on simulation models developed for coarse woody debris (CWD) in terrestrial environments. Wright et al. (2002) explored the sensitivity of a simulation model called FIRECWD developed to compare the dynamics of CWD from two different forest fire regimes, stand-replacing fire regime vs. mixed-severity fire regime with differing mean fire return intervals of 300 and 125 years, respectively. Wright et al. (2002) found that the mean mass of CWD modeled for decay constants ranging between 0.01 to 0.05 ranged between 387 Mg/ha to 111 Mg/ha for simulation of stand-replacing fire regimes and ranged between 319 Mg/ha to 96 Mg/ha for simulations of mixed-severity fire regimes.

Table 1. Reported decomposition rates from various in-stream woody debris studies.

Reference	Decay coefficient (k) in years^{-1}	Components included in decomposition rate	Species	Study location	Stream order and adjacent forest	Decay model used	Method used for determination of decomposition rate
Murphy and Koski, (1989)	0.01 to 0.03	Mineralization (decay), fragmentation, transport	Primarily western hemlock (*Tsuga heterophylla*) and sitka spruce (*Picea sitchensis*)	Southeast Alaska (7 streams)	2nd to 5th order, Old growth	Single Exponential Model	Input-biomass ratio method and dependant vegetation age determination
Bilby et al. (1999)	0.026	Fragmentation and mineralization	Douglas fir (*Pseudotsuga menziesii*)	Tributary to Deschutes River, western Washington	3rd order, (adjacent forest type not available)	Single exponential model	Time series method: initial and final volumes of submerged logs measured over 5 years
	0.026		Western red cedar (*Thuja plicata*)				
	0.031		Western hemlock (*Tsuga heterophylla*)				
	0.033		Red alder (*Alnus rubra*)				
	0.038		Big leaf maple (*Acer macrophyllum*)				
Hyatt and Naiman, (2001)	0.03	Mineralization (decay), transport and burial	Primarily Sitka spruce (*Picea sitchensis*) Red alder (*Alnus rubra*) Western red cedar (*Thuja plicata*)	Queets River watershed, old growth forests, Olympic Mountains, western Washington	≥ 5th order old growth	Single exponential model	Chronosequence method: cross-dating, radio carbon dating and dependant vegetation age determination
Golladay and Webster (1988)	0.107 to 0.281	Mineralization and fragmentation	Red oak (*Quercus rubra*), 1-3 cm diameter	Big Hurricane branch and Hugh White Creek, North Carolina	2nd order (undisturbed and harvested hardwood forest)	Single exponential model	Time Series method: initial and final weights of stick packs measured over 4 year period

Based upon Kennard's et al. (1998) description of model sensitivity and uncertainty, current comprehensive models that include output rates associated with decomposition introduce major uncertainty into the accuracy and precision of model analyse. This uncertainty is primarily evident due to: i) model sensitivity to varying decay coefficients, ii) the observed variation in the decay coefficients as summarized in the above sections, and iii) the lack of appropriate functions and parameters for fluvial transport (Benda and Sias 2003). These examples emphasize the importance of the role decomposition plays in determination of the abundance and distribution of woody debris. This further highlights the importance of an improved understanding of the decomposition and longevity of in-stream woody in development of riparian management decision-making tools.

Conclusions

A main purpose of this paper was to provide a brief summary of the literature describing the

Table 1 (continued). Reported decomposition rates from various in-stream woody debris studies.

Reference	Decay coefficient (k) in years^{-1}	Components included in decomposition rate	Species	Study location	Stream order and adjacent forest	Decay model used	Method used for determination of decomposition rate
Melillo et al. (1983)	1.20 (1st order stream)	Mineralization (assumed)	Alder chips (*Alnus rugosa*)	Four streams in eastern Quebec	1st order to 9th order (k values only shown for 1st order stream)	Single exponential model	Time series method: initial and final weights of mesh bags measured over 16 months
	0.95 (1st order stream)		Birch chips (*Betula papyrifera*)				
	0.40 (1st order stream)		Aspen chips (*Populus tremuloides*)				
	0.35 (1st order stream)		Spruce chips (*Picea mariana*)				
	0.25 (1st order stream)		Balsam-fir chips (*Abies balsamea*)				
Guyette et al. (2002)	0.0015	Mineralization and fragmentation	Non-species specific (67 species sample that included Querdus spp., Juglans nigra, Acer spp., Betula spp., Populus spp., Salix spp.)	Medicine Creek, Missouri	Stream order not specified, floodplain of forest and agricultural land	Single exponential model	Chronosequence method: dendrochronology and radiocarbon dating

decomposition and longevity of in-stream woody debris. The decomposition of woody debris in stream environments is complex and involves the interplay of many biological and physical processes (Harmon et al. 1986; Golladay and Webster 1988). The main factors that control woody debris decomposition include climate, tree species (e.g., chemical content, lignin content), piece size (e.g., diameter or length), decay class, position (e.g., suspended, on ground, buried, fully submerged), major decomposition process underway (e.g., respiration and leaching or fragmentation), site conditions (e.g., temperature, moisture levels, oxygen and carbon dioxide levels), bed stability, channel morphology, flood intensity and riparian forest composition (Harmon et al 1986; Naiman et al. 2002).

A simple answer regarding the longevity of in-stream woody debris is problematic based on the interplay of these factors and the fact that these factors can vary greatly across temporal and spatial scales. However, based upon the majority of in-stream woody debris studies that have been completed in the coastal regions of the Pacific Northwest, woody debris typically remains in a channel for 70-100 years with several pieces remaining for many centuries to millennia (Naiman et al. 2002). In other terms, rates of decomposition range between 1 to 3% per year in coastal stream ecosystems. (Benda and Sias 2003). No research was identified that directly addressed this question for in-stream woody debris situated in the interior regions of western Canada or the United States, and I consider this paucity of information to be a relevant research area that needs to be addressed.

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Criteria and Indicators

Riparian Health Evaluation - Tools for Assessing the Function of Riparian Areas

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Extended Abstract

Recognising the value placed on riparian areas in our landscape, we need to understand the current condition of riparian areas and how they are influenced by current management practices. Specifically, we need to be able to answer the question "How healthy are these areas?" Riparian health is determined by the ability of a riparian site to perform certain ecological functions. These functions include the ability to: i) trap and store sediment; ii) build and maintain banks and shores; iii) store water and energy; iv) recharge aquifers; v) filter and buffer water; vi) reduce energy; vii) maintain biodiversity; and viii) create primary productivity.

Understanding which components of riparian health are intact and provide these functions involves examining vegetative and physical characteristics of a riparian site. Riparian health assessment and inventory are key awareness and education tools that build a common language and ecological literacy between landowners, resource management professionals, and the public. Limited resources in government and non-government programs means there is a need to assist landowners and land managers in making management decisions with scientifically

Ambrose, N.E., Bogen, A.D., O'Shaughnessy, K. and Spicer-Rawe, K. 2004. Riparian health evaluation - tools for assessing the function of riparian areas. Pages 137-138 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

rigorous information. Riparian health evaluations provide ecological knowledge that supports the land management decision making process. At both community and individual levels, these methods allow landowners to make more informed decisions and to take a key role in driving the process of identifying and addressing issues related to riparian management. The methods are land use neutral - any riparian site with any land use can be examined (bogs and fens have not been extensively tested). In addition, riparian health evaluations are critical in establishing baseline information to: i) identify if issues exist, ii) help direct management actions needed, and iii) monitor the success of those management choices in the future.

Riparian health assessments examine a number of parameters, including vegetative cover, bare soil, structural alterations to the bank or shore, and changes in hydrologic regime or plant community that may impact the ability of the area to perform those ecological functions. Vegetative parameters also include regeneration and utilisation of the tree and shrub community. Consideration of site potential is also included in the method. For example, where there is no tree or shrub species potential (e.g., saline prairie wetland), these questions are excluded from the method.

Site evaluations are based on either representative or site-specific sampling. Thorough field examination of a site results in a riparian health score. Sites are rated based on the thresholds met or exceeded in terms of vegetative or physical parameters, and fall into one of three categories: i) healthy and functioning, ii) healthy, but with problems (some functions impaired), or iii) unhealthy and non-functioning (most or all functions missing or severely impaired). In Alberta, less than 20% of the sites we have examined are healthy._ A discussion on the health of Alberta's and other jurisdictions' riparian areas will be included.

Monitoring Aquatic Ecosystems as part of the Alberta Biodiversity Monitoring Program: Identifying Appropriate Indicators and Developing Sampling Protocols

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Extended Abstract

The rate and diversity of human development in Alberta will increase greatly in the future. Without careful management, the cumulative impacts of this human development will affect biodiversity. The Alberta Biodiversity Monitoring Program (ABMP) is a rigorous, science-based initiative designed to detect broad-scale changes in biodiversity and landscape patterns, thereby improving management of natural resources in Alberta. The ABMP relies on a core set of sampling protocols developed to survey a broad diversity of biota, habitat structures, vegetation communities, and landscape patterns. These protocols were developed in consultation with a range of scientists from across North America, and have been amalgamated into an integrated design. In 2002, the protocols were field-tested at a number of sites in Alberta to examine their effectiveness and efficiency. Although most of the terrestrial protocols worked well during the testing phase, there were numerous difficulties in finding appropriate sites to conduct the aquatic protocols.

Aquatic habitats support diverse plant and animal communities that are expected to be greatly affected by human development activities and climate change. Thus, the development of appropriate aquatic sampling protocols is a critical component of monitoring broad-scale change in Alberta's ecosystems. Initially, ABMP aquatic protocols were designed to sample basin characteristics, water physiochemistry, benthic macroinvertebrates, zooplankton, phytoplankton and benthic algae, amphibians, and fish for standing and flowing water. These protocols are under revision, as they were not appropriate for the aquatic sites actually encountered during initial field tests. My work focuses on identifying aquatic organisms that are good indicators of environmental quality, and that can be sampled efficiently and with statistical repeatability. I am presently revising protocols to sample these organisms and associated physical and chemical parameters of aquatic ecosystems. These aquatic sampling protocols will be appropriate for standing and flowing water systems in Alberta.

Eaton, B. R. 2004. Monitoring Aquatic Ecosystems as part of the Alberta Biodiversity Monitoring Program: Identifying Appropriate Indicators and Developing Sampling Protocols. Pages 139-140 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

The ABMP will allow land managers to meet stated commitments to conserve biodiversity, and help ensure that development and resource extraction activities are sustainable. Monitoring by the ABMP will provide measures of changes in biodiversity that are real and scientifically credible, and ensure that changes in biodiversity can be related to changes within the systems examined, especially changes associated with human activities. Aquatic monitoring is an important component of the ABMP, and development of appropriate protocols is critical towards ensuring that this monitoring produces appropriate datasets for tracking changes in biodiversity, temporally and spatially, within Alberta.

An Empirical Analysis of Flood Peak Changes Due to Forest Harvesting Along the Alberta Foothills

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Extended Abstract

Logging activities in forested watersheds often result in a variety of changes. The magnitude and extent of these changes, however, is highly dependent on the local climate, surficial geology, altitude, and other site-specific factors. The timing and magnitude of stream peak flows is one type of change that is often reported from scientific experiments involving paired-watershed studies. Along the foothills of the Rocky Mountains in Alberta Canada, the change in peak flow magnitude due to logging has also been empirically-derived from the analysis of gauging station data. The records indicate a direct correlation between peak flow magnitude and the percentage of equivalent clearcut area. The scope of the analysis involved the forest management agreement areas near Rocky Mountain House and Hinton Alberta, where detailed watershed and clearcut information was available. This paper presents a multivariate prediction model for design flood peaks, and an example application comparing previous peak flow predictions. The model provides predictions for the 2-year 25-year and 100-year return period peak instantaneous flows, based on catchment area, average catchment elevation, equivalent clearcut area, and road density. Other factors were also considered, but were statistically insignificant compared to the selected model parameters. The other factors included: maximum catchment elevation, average catchment slope, average aspect in terms of percent northern or eastern aspect. The example provided is for Hardisty Creek, at Hinton Alberta. Hardisty Creek is a small ungauged catchment, and a design discharge was needed for

Bender, M., Wu, S., Chan-Yan, D., Bonar, R. and Denney, P. 2004. An empirical analysis of flood peak changes due to forest harvesting along the Alberta foothills. Pages 141-142 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

several culvert rehabilitation projects. Three previous regional analyses provided estimates of the design discharge, but were determined to produce poor results based on historical anecdotal evidence and based on hydraulic model analysis at the culverts. The multivariate model provided a more realistic estimate of the design discharge, and it provided a means of describing the likely changes due to recent logging activities upstream of the culverts. The example illustrates that while the prediction model may not account for any mitigating site-specific conditions, it seems to be more accurate for small watersheds compared to traditional regional hydrologic analyses, and that the multivariate model may be practical for the sizing of culverts and bridges.

Laurance Lake Hydrodynamic and Temperature Modeling Study

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Extended Abstract

The Clear Branch of the Middle Fork of the Hood River has been listed as temperature impaired by Oregon Department of Environmental Quality under the Clean Water Act. The temperature impaired reach is below Laurance Lake (Clear Branch Reservoir) in Hood River County, Oregon. Laurance Lake was built in 1968 to provide irrigation for 2590 hectares of agriculture in the upper Hood River Valley, part of the second largest fruit growing district in Oregon. The 48.5 hectare reservoir stores 3565 acre feet of water and is fed by two streams, Pinnacle Creek and Clear Branch. Laurance Lake, Clear Branch above the reservoir and Pinnacle Creek hold the only known population of bull trout (*Salvelinus confluentus*) on the Mount Hood National Forest. The purpose of this study is to create a fact based Reservoir Management Plan for Laurance Lake and to determine if the temperature in Clear Branch below the reservoir can be reduced. The Study began in August 2002. A weather station was installed on Clear Branch Dam to measure wind speed, wind direction, air temperature, and solar insolation hourly. Stream gauges on Pinnacle Creek and Clear Branch record stream stage hourly. Stream flows have been measured weekly and stage discharge curves have been developed ($r^2 = 0.98$ after 40 stream measurements). Twenty-eight thermistors record temperature simultaneously at half hour intervals. A string of thermistors from the deepest point (36.5 m) in the reservoir to the surface record water temperatures at 3 meter intervals. There are three thermistors above the Lake in Clear Branch, one above the lake in Pinnacle Creek, five in drains and springs below the dam, and three in Clear Branch below the dam. A data sonde was used weekly to collect vertical profiles at 4 locations in the lake. A handheld GPS is used to find the same three locations and the 4th profile is chosen randomly. The vertical profile data includes temperature, pH, conductivity, and dissolved oxygen (DO) taken at one meter intervals. The data sonde is also deployed in Clear Branch and Pinnacle Creek above the reservoir. All of these data is being audited. Thermistors not in the reservoir are audited monthly

Connors, W. B. 2004. Laurance Lake hydrodynamic and temperature modeling study. Pages 143-144 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

with a traceable thermometer. Thermistors in the reservoir are checked against data sonde temperatures, DO values are checked with Winkler titrations, and pH and conductivity are checked with independent instruments. The Water Quality Research Group at Portland State University is developing a CE-QUAL-W2 Version 3 water quality model of the lake. CE-QUAL-W2 is a Corps of Engineers two-dimensional, laterally averaged, hydrodynamic water quality model. The model will be used to evaluate management options for Laurance Lake. Audit data indicates temperatures at the bottom of the reservoir and Clear Branch below the reservoir exceed Oregon's temperature standard in July, August and September. Management options that could lower temperatures below the reservoir include creating shade near the base of the dam, modifying instream flow requirement timing and/or modifying penstock intake at the reservoir bottom.

Utilizing Large Scale Photography (70 mm) for Stream and Riparian Monitoring

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Extended Abstract

Fixed-base, large-scale, stereo aerial photography has been utilized over the past four years for various stream and riparian monitoring studies throughout southern British Columbia and Alberta. Salmon bearing rivers, with widths ranging from 3 to 50 m, have been photographed for the purpose of recording attributes such as: 1) spawning Chinook salmon counts, 2) Chinook salmon redd identification and counts, 3) stream width measurement, and 4) dominant river substrate identification. Aerial photography also continues to assist with studying fire disturbances within riparian zones. The objectives of this type of assessment include tracking the recruitment of large woody debris within the stream channel and riparian area, collecting stream morphology data, and providing comparisons between stand structure plots within and outside the riparian zone for piece count and volume trends.

When acquired, each stereo pair of large-scale aerial photographs is immediately linked to a set of Global Positioning System coordinates. This provides a permanent historical reference that can be used for relocation (i.e., for future monitoring). The system is very flexible, and can adapt to individual project requirements and parameters such as the desired photo scale. For example, if the goal is to collect stream morphology of different streams, flying height adjustments may be required to achieve the best resolution of substrate, while ensuring the capture of the entire stream profile. Methods of data capture are also of consideration. When the goal is to collect data at a fixed radius plot, a single pair of stereo photos will be sufficient. If the intention is to capture information along an entire stream reach, an intervalometer can be used to capture 100% photo coverage. Photo mosaics can then be created to obtain a large-scale visual image of the entire reach.

Lalonde, D. and Sandvoss, M. 2004. Utilizing large scale photography (70 mm) for stream and riparian monitoring. Pages 145-146 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Originally designed to be a cost effective tool for forest inventory sampling and to provide quantitative data capture, the system offers a highly mobile remote sensing platform for extremely high resolution, low altitude, large-scale (i.e., 1:200 to 1:2000) stereo aerial photographs. The system uses two computer controlled Hasselblad MK 70 photogrammetric cameras. True colour, false colour infrared, and black and white film (i.e., various emulsions, film speeds, and in either negative or diapositive form) can be used to suit the mission parameters. Interpretation is completed using a Ross Stereocomparator, a photogrammetric instrument designed specifically to use large-scale aerial photographs during photogrammetric measurement. The Stereocomparator, when mated to a Photogrammetric Analysis Software System (PASS) compatible personal computer, is capable of measuring horizontal and vertical distances to an accuracy of 10 cm at photo scales of 1:1000 or larger.

Large Scale Photography (70 mm) has been used for a variety of sampling projects for over 20 years. Research and technological advancements within the past few years have helped to make this tool a viable option for stream and riparian monitoring.

An Approach for Screening Forest Harvest Plans Based on Predicted Changes to River Systems: Weldwood Experience

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Extended Abstract

A hydrologic assessment framework and application is presented for the purpose of screening forest management plans based on predicted watershed changes. The framework was developed for the foothills of the Rocky Mountains in Alberta Canada, and the application is used by Weldwood of Canada to provide a screening level assessment of potential forest management plans. The purpose of the assessment tool is to highlight watersheds or sub-basins that will be affected the most, and to determine whether the effect is likely to be significant. In this way, Weldwood has a tool for prioritizing follow-up detailed assessment work, and a basis for improving its forest management plan. Assessments are based on a number of measurement endpoints that are derived from compartment timber yield estimates, watershed boundaries, estimated equivalent clearcut areas, natural vegetation subregion classifications, stream morphology, infrastructure location, and social values. First, the relative change of several hydrologic variables are estimated: sediment yield, peak flows (and flood levels), low flows, water temperature, and water yield. The relative change is then interpreted based on threshold levels as 'high', 'moderate', or 'low'. Measurement endpoints are key end-use indicators that are then evaluated: stream morphology, fish habitat, and infrastructure. For example, the net effect on fish habitat is estimated based on a combination of predicted effects to low flows, sediment yield, water temperature, and stream morphology. The measurement endpoints are then evaluated in terms of spatial scale and relative sensitivity, to account for cumulative effects and social values. The end result is a predicted 'acceptable' or 'unacceptable' watershed impact. In this way, the screening method avoids many of the scientific complexities that

Bender, M., Bonar, R., Lougheed, H. and Sawatsky, L. 2004. An approach for screening forest harvest plans based on predicted changes to river systems: Weldwood experience. Pages 147-148 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

often prevent us from making decisions due to a lack of accuracy or certainty. The selected method accomplishes Weldwood's goal of quickly identifying possible environmental inequities across the land base, and the method is integrated within a larger planning process. The method is focused on providing a transparent and reproducible basis for decisions, and a clear basis for allocating the available human resources for environmental protection. Weldwood applied this methodology as part of their 10-year Forest Management Plan development process. Weldwood has since continued the development of screening tools for detailed harvest plans. This paper presents the overall methodology, initial application for forest management plans, and recent adaptations of the methodology for screening detailed harvest plans.

Application of the Ontario River/Stream Ecological Classification Techniques (ORSECT) Towards Development of Criteria and Indicators of Aquatic/Riparian Ecosystem Sustainability: A Conceptual Framework

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Extended Abstract

Model Forests are designed to be "models" for testing innovative approaches to sustainable forest management. This focus includes a commitment to develop indicators to address sustainability criteria for conservation of biological diversity and soil and water conservation - including indicators that will address aquatic and riparian area protection. The conceptual framework described herein is a first step towards implementing the aquatic and riparian ecosystem research strategy for the Lake Abitibi Model Forest.

The Lake Abitibi Model Forest (LAMF) is located in the boreal forest of northeastern Ontario. The objectives of the LAMF Aquatic and Riparian Ecosystem Research Strategy are to: i) conduct collaborative research related to aquatic and riparian ecosystems in support of ecosystem management; ii) develop a suite of tools to support on-going work on Local Level Indicators, support decision-making and monitor impacts related to aquatic and riparian ecosystems, and iii) contribute to a knowledge legacy and facilitate further research on aquatic and riparian ecosystems and related topics.

This conceptual framework constitutes the initial phase of strategy implementation of the aquatic and riparian strategy by exploring the use of: i) existing and emerging digital databases and ii) GIS software and technology in an integrated fashion to generate meaningful aquatic and riparian indicators of sustainability. Specifically, the Ontario Ministry of Natural Resources' (OMNR) Northeast Science and Information Aquatic Ecosystem Team is exploring the utility of the Ontario River/Stream Ecological Classification Techniques (ORSECT) to identify and classify cold-water streams within the Boreal Claybelt.

ORSECT is a GIS-based decision support tool that uses a variety of digital databases

McGovern, S.P. and Chang, C. 2004. Application of the Ontario River/Stream Ecological Classification Techniques (ORSECT) towards development of criteria and indicators of aquatic/riparian ecosystem sustainability: a conceptual framework. Pages 149-150 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

to segment stream networks based on determined landscape characteristics. ORSECT includes a user-friendly, interactive GIS-based software with accompanying databases to automatically determine various physical characteristics of a stream segment. Subsequently, ORSECT can assist in the development of ecological classification systems and ultimately the development of landscape–level indicators for aquatic and riparian protection.

While brook trout is the initial focus of the study as a candidate indicator for aquatic sustainability, it is expected that this study will reveal parameter associations beyond those solely linked to cold-water streams. These indicators, as well as products from developing the integrated tools, models and techniques will subsequently be applied in a forest management context to improve planning decisions (e.g. aquatic values identification, water crossings selection, effectiveness monitoring of water crossing and riparian protection prescriptions and risk assessment).

The study will provide a significant advance in the application of information management and GIS tools to generate aquatic resource values. These tools will be applied to an existing and developing science knowledge base in a manner that optimizes recently available digital data to fill information gaps otherwise only obtainable via expensive field inventory procedures. The end result will be value-added, cost effective tools and models that will assist in future planning and decision-making. The study will demonstrate efficient use of existing science and information toward better informed, and hence, improved decisions regarding aquatic/riparian ecosystem protection.

Forests, Fish and Fire: Relationships and Management Implications for Fishes in the Southwestern USA

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Abstract

Until recently, the effects of wildfire on aquatic ecosystems and fishes in forested landscapes in the southwestern USA have been given little attention. In 1989-90 the post-fire impacts of two wildfires on fishes and their habitats increased concern for this management issue. In summer 2002 alone, wildfires burned over 5 million acres in the western USA. Several large wildfires occurred in the Southwest in 2002 and 2003 and provided opportunity to delineate the effects on over a dozen native fishes--several federally listed species. Information was gathered on three fires in 2002 and two in 2003. In summary, all fishes were lost in fire-impacted reaches of one stream. In two streams, a 70% reduction in total fish numbers was recorded. In 2003, up to a 90% reduction of total fish numbers were recorded in four more streams. Based on data collected in summer 2002-2003 and that from two fires in 1989-90, immediate, post-fire stream water quantity and quality and both short and long-term alteration of habitat are primary determinants of impacts on fishes. Recent study and historic information indicate that all species of native fishes could be affected markedly by post wildfire impacts in southwestern stream ecosystems. The impact of wildfire on native fishes is rapidly emerging as a primary management concern in forested landscapes in the Southwest.

Rinne, J. N. 2004. Forests, Fish and Fire: Relationships and Management Implications for Fishes in the Southwestern USA. Pages 151-156 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

The topography and hydrology of forested ecosystems in the southwestern United States largely delimits native fish distributions, both historically and at present (Rinne and Minckley 1991, Rinne 1995, 2002). Because of widespread alteration of historic hydrology and introduction of non-native fish species, the fish fauna of the Southwest is largely comprised of rare and listed species of fishes (Minckley and Deacon 1991, Mueller and Marsh 2002, Rinne 2003a, 2003b). In addition to altered hydrology and introduction of invasive species, forest multiple uses such as grazing, mining, recreational activity and timber management potentially affect fishes. The activity of suppression of wildfires over the past century in southwestern forest landscapes has resulted in increased tree density and fuel for wildfires (Covington and Moore 1992, Rinne and Neary 1996). The change on the landscape combined with an imposing drought cycle (Gray et al. 2003)) may render wildfire as an emerging, limiting factor to native fish survival.

Wildfire is increasingly becoming a major threat to sustainability of southwestern native fishes (Rinne and Carter in press, Rinne 2003a) and in general across the West (Gresswell 1999, Rieman et al. 2003). Indeed, wildfire may be considered of equal merit as a threat or risk factor as habitat alteration and introduction of nonnative fish species to native fish fauna sustainability (Rinne in press). Brown et al. (2001) suggested wildfire is the most significant risk factor for the sustainability of the endangered Gila trout (*Oncorhynchus gilae*). Further, in headwater streams, over the past 15 years, extensive wildfires on landscapes encompassing this endangered trout corroborate Brown et. al.'s suggestion (Table 1). In summary, a fire has occurred on average every two years that either really or potentially affected this rare, endangered fish species.

In the southwestern United States, there is little information available on the effects of wildfire on fishes. Propst et al. (1992) first discussed the impacts of fire on a native endangered trout, the Gila trout, in headwater streams in southwestern New Mexico affected by the Divide Fire, 1989. More recently, Rinne (1996) reported on the effects of fire on rainbow (*O. mykiss*) and brook (*Salvelinus fontinalis)* trout in three streams affected by the Dude Fire in central Arizona in 1990. Rinne and Neary (1996) summarized and assessed the probable effects of wildfires on streams in the Southwest. They suggested the increase in crowning wildfires cold have a major impact on native fishes and their habitats.

Study of the effects of wildfire on fishes and their habitats in the Southwest by the USDA Forest Service, Rocky Mountain Research Station escalated in 2002--one of the worst years for wildfires on record. Rinne and Carter (in press) and Rinne (2003a) documented the historic, short- and long-term impacts of wildfires on southwestern aquatic habitats and native fishes in upper elevation, forested ecosystems and suggested the need to aggressively study and manage southwestern native fishes. Because the fish fauna of this region is 1) low in diversity (Rinne and Minckley 1991), 2) dispersed in isolated reaches of streams (Rinne 1995), 3) rapidly declining due to multiple effects, (Rinne 2002, 2003c), and 4) largely comprised of threatened and endangered species of fishes, (Rinne 2003b) forest managers and researchers must collaboratively study and manage this rapidly emerging forest issue.

Species affected, 2002-2003

In 2002, twelve new species of fishes were sampled in streams affected by the wildfires in Region three of the U. S. Forest Service (Table 2). In Ponil Creek, Carson National Forest (Picture Fire), information was obtained on densities and biomasses of three cypriniform species: creek chub, *Semotilus atromaculatus*, white sucker, *Catostomus commersoni,* and longnose dace, *Rhinichthys cataractae.* These three species comprised

Table 1. The sustained effects of wildfire on the endangered Gila trout, 1989-2003.

Fire	Year	Acres	Creek	Comments
Divide	1989	12,000	Main Diamond	type locality population
Spruce	1994	131,000	Spruce	unique Gila trout pop.
Bonner	1995	29,000	South Diamond	reduced pop
Sprite	1995	< 100,000	Mogollon	Gila trout recovery pop
Lookout	1996	15,000	West Fork Mogollon	Gila trout recovery pop
Cub Mountain	2002	13,000	West Fork Gila River	fish removed, 2002
Whiskey	2003	10,000	Whiskey Creek	fish removed 2003

the major portion (98%) of the fish assemblage in this creek. Rainbow trout *(O. mykiss)* also were present in low numbers.

Only brown trout, *Salmo trutta,* was present in Rio Medio (Santa Fe National Forest), however, this represented a new species of trout for which fire effects (Borrego Fire) was determined. In the West Fork of the Gila River, Gila National Forest (Cub Mountain Fire), data were collected on six native species of the Gila River basin: longfin dace, *Agosia chrysogaster;* speckled dace *Rhinichthys osculus;* Sonora, *Catostomus insignis,* and desert *(C. clarki)* sucker; roundtail chub, *Gila robusta;* and the threatened spikedace, *Meda fulgida.* This reach of river also contains the threatened loach minnow, *Rhinichthys cobitis,* however, this species has only been collected a kilometer downstream (Rinne et al. in press). Spikedace also have been collected at an established, long-term monitoring site (Rinne et al. in press) positioned between summer 2002 sample sites on the West Fork.

In 2003, post wildfire impacts were documented on four additional species, two native and listed (Table 2). The recently described headwater chub (*Gila nigra*) was dramatically (79-90%) reduced in numbers following a wildfire on the Tonto National Forest. In addition, the effects on two additional nonnative species, green sunfish (*Lepomis cyanellus)* and yellow bullhead (*Ameiurus natalis).* Total loss was documented for the Gila chub (*Gila intermedia*) in 2003 in a stream impacted by the largest wildfire in New Mexico history. In 2002-2003 information were collected on the post-fire effects on 17 mostly native and non-salmonid species.

Historic, long term effects

Mcknight Fire, 1950

The McKnight fire occurred in 1950 on the Gila National Forest on the headwaters of McKnight Creek, southwestern New Mexico. The fire burned 20,000 ha situated on the steep landscapes of the western flank of the Black Range Mountains. In the early 1970s the endangered Gila trout was introduced into the stream to meet goals and objectives of the recovery plan for the species. From the late 1970s to middle 1980s a series of floods impacted McKnight Creek (Rinne and Neary 1996). Gila trout in McKnight Creek have fluctuated markedly during that time and declined almost to extinction in 1988. Although cycles of drought and flooding are the norm in the southwestern US (Rinne 2003c, in press), these flood events may, in part, relate to the chronic effects of wildfire a half century ago.

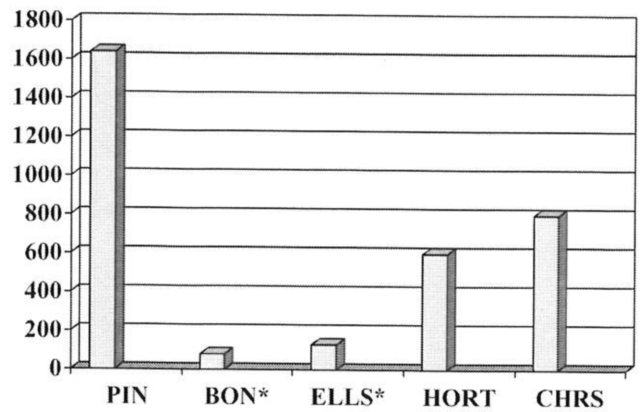

Figure 1. Longer-term (11 yrs) post-fire effects on two trout populations (numbers per kilometer) in two streams impacted by the Dude Fire (denoted with an asterisk), Tonto National Forest, Arizona, 1990, compared to three, contiguous, un-impacted streams. Streams are (et seq.) Pine, Bonita, Ellison, Horton, and Christopher.

Divide Fire, 1989

The Divide Fire occurred on the Gila National Forest landscape (a ridge) between Main and South Diamond creeks in 1989. Both contained populations of the endangered Gila trout, the former the type locality population (Miller 1950). Because of the emerging information of the post-fire effects of the Yellowstone fires, over 500 Gila trout were removed from the stream (Propst et al. 1992). The remaining individuals were all lost to post-fire ash and ultimately floods flows.

Dude Fire, 1990

The Dude fire was ignited on June 25, 1990 and quickly spread and affected the watersheds of Dude, Bonita, and Ellison Creeks. The fire burned ca. 12,000 ha, destroyed 55 homes, cost 6 million dollars to extinguish, and very unfortunately, 6 lives were lost (Rinne 1996). The post-fire ash and flood flows were fatal to all brook (Bonita Creek) and rainbow (Dude and Ellison creeks) trout. In September 2001, Bonita and Ellison creeks and three additional contiguous Mogollon Rim streams were sampled that were not affected by the Dude Fire and contained previous density and biomass estimates (Rinne and Medina 1988). These data (Figure 1) suggest that the trout populations in these two streams have yet to recover.

Table 2. List of species that have been affected by wildfire in the southwestern United States from 1989-2003. Species are listed by respective named fires and locations. Non- native species are denoted by an asterisk.

Fire	Common Name	Scientific Name	Stream
Divide (Arizona)	Gila trout	*Oncorhynchus gilae*	Main Diamond Cr.
Dude (Arizona)	*Rainbow trout	*Oncorhynchus mykiss*	Dude, Ellison Creeks
	*Brook trout	*Salvelinus fontinalis*	Bonita Creek
Ponil Complex (New Mexico)	*Rainbow trout	*Oncorhynchus mykiss*	Ponil Creek
	Longnose dace	*Rhinichthys cataractae*	Ponil Creek
	Creek chub	*Semotilus atromaculatus*	Ponil Creek
	White sucker	*Catostomus commersoni*	Ponil Creek
Borrego (New Mexico)	*Brown trout	*Salmo trutta*	Rio Medio
Cub Mountain (New Mexico)	Desert sucker	*Catostomus clarki*	West Fork Gila River
	Sonora sucker	*Catostomus insignis*	West Fork Gila River
	Longfin dace	*Agosia chrysogaster*	West Fork Gila River
	Speckled dace	*Rhinichthys osculus*	West Fork Gila River
	Spikedace	*Meda fulgida*	West Fork Gila River
	Loach Minnow	*Rhinichthys cobitis*	West Fork Gila River
	Roundtail chub	*Gila robusta*	West Fork Gila River
Picture (Arizona)	Headwater chub	*Gila nigra*	Turkey, Rock, Spring Crs
	Speckled dace	*Rhinichthys osculus*	Turkey, Rock, Spring Creeks
	Desert Sucker	*Catostomus clarki*	Turkey, Rock, Spring Creeks
	*Green sunfish	*Lepomis cyanellus*	Rock, Spring Creeks
	*Yellow bullhead	*Ameiurus natalis*	Rock Spring Creeks
	*Brown trout	*Salmo trutta*	Rock, Spring Creeks
Dry Lakes Complex (New Mexico)	Gila chub	*Gila intermedia*	Turkey Creek

Forests, fish and fire relationships

The change in forested landscapes resulting from fire suppression over the past century has imposed a potentially devastating impact on sustainability of native southwestern fishes. The combination of 1) markedly increased tree density (Covington and Moore 1992), 2) rarity and listed status of many southwestern native fish species (Rinne 2003b, 2003c), 3) their isolated dispersions in disjunct stream systems (Rinne 1995), 4) climate change and emerging drought (Gray et al. 2003), marked increase in bark beetle outbreak (Figure 2), and 6) probable historic recovery rates for aquatic ecosystems (in reality and potentially) will markedly affect the sustainability of the southwestern native fish fauna. Marked (70-90%) reduction to total loss of native fishes in upper elevation, montane aquatic ecosystems suggest the potential impact of wildfire on native fishes is certainly equal to or greater then that of past habitat alteration and introduction of non-native fishes. Indeed, post wildfire impacts and its negative impact on native southwestern fishes, if not aggressively and timely addressed, may be the ultimate limiting factor to many native fish species that persist in the more permanently-watered, upper elevation aquatic refugia on southwestern forested landscapes.

Management implications

Data collected on the short-term effects of five wildfires on twenty-one species of fishes in six streams on National Forest lands corroborate previous long-term findings by Propst et al. (1992) and hypotheses of Rinne and Neary (1996). Because the majority of southwestern native fishes are threatened, endangered, or Forest Service sensitive and state-listed species, managers must be vigilant of opportunities to remove fishes from streams whose watersheds are affected by wildfire. There are often very brief (2-3 weeks or less) windows of opportunity to salvage stocks before toxic ash or flood flows result. Efforts such as those conducted for Gila trout following the Divide Fire (Propst et al. 1992) and for Gila chub following the Aspen Fire (summer 2003) may be considered a fundamental management approach to address native fish sustainability in the Southwest following wildfires. Because most populations of rare, southwestern fishes are isolated and unique genetically they are evolutionary significant units. As such, they cannot be replaced once lost. Further, the climate and landscapes of the Southwest dictate fragmentation of aquatic habitats (Rinne 1995). Such fragmentation precludes natural repatriation that occurs more readily and frequently

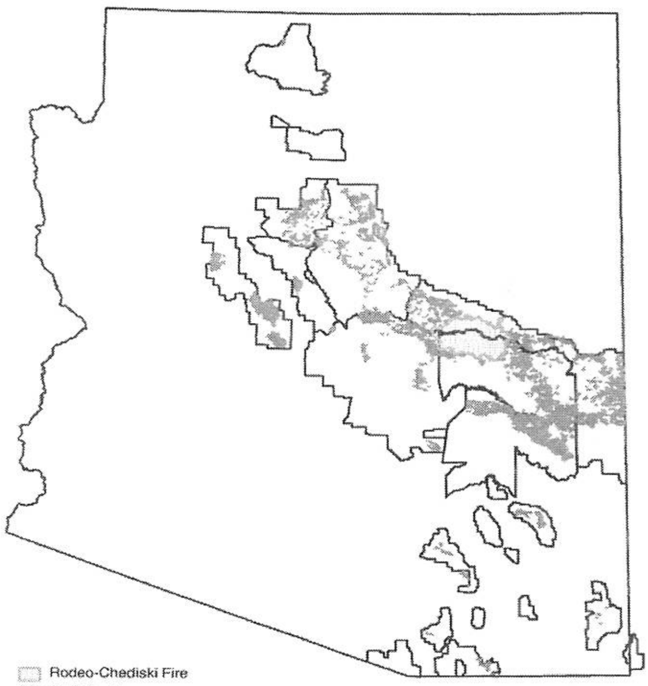

Figure 2. Extent of bark beetle outbreak in Arizona, 2002.

with primarily salmonid species in the more mesic, northern Rockies and Pacific Northwest (Rieman et al. 2003).

Finally, 2002 and 2003 were characterized again by extensive and extreme fire activity in Region 3 (Arizona and New Mexico) of the U. S. Forest Service. The Southwest is currently in a period of prolonged drought (Gray et al. 2003). Continued drought combined with the massive outbreak of bark beetles across forested landscapes has resulted in thousands of acres and millions of dead and dying trees in the national forests of the southwestern region (Fig. 2). The potential is high for even greater wildfire activity in the future. In parallel, the probability likewise increases that additional streams containing rare and endangered fishes will be impacted by the aftermath of these fires. Recent (2002-2003) data combined with long-term data on the McKnight, Divide and Dude fires suggest a call to action to address the issue of wildfire effects on rare, declining native fish fauna in the Region. Linkages of fire intensity on the landscape with fish mortality and habitat alteration need definition. Linkage of post-fire Burned Area Emergency and Restoration (BAER) activities and stream habitat and fish response must be delineated. Fishery professionals must be included on BAER teams. Timely removal of fishes and their respective gene pools are vital to conserve often isolated, rare listed populations. In summary, land and fishery resources managers in the Southwest must be prepared to take strategic, coordinated, and timely responses to these events as they potentially affect the invaluable, often locally irreplaceable resource, native southwestern fishes.

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A Water Quality Indicator for Sustainable Forest Management: The SCQI Experience

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Abstract

One of the goals of sustainable forest management is the maintenance of water quality. One of the biggest forestry related impacts to water quality is accelerated sediment delivery to streams at road crossings. Good road building and maintenance practices will minimize the erosion hazard and related negative impacts to water quality. Based on this assumption, several divisions of Canadian Forest Products Ltd. in British Columbia and Alberta have decided that a good water quality indicator should be based on a field-survey that evaluates effectiveness of controlling accelerated erosion and sediment delivery at stream crossings. This has led to the development of a sediment source hazard assessment procedure called the Stream Crossing Quality Index (SCQI). The procedure evaluates and scores the size and characteristics of road-related sediment sources at crossings and the potential for the eroded sediment to reach the stream environment. A high score infers that there is a significant erosion problem which may in turn cause sediment-related water quality problems. The sum of the individual crossing scores provides an indicator of the sediment hazard at the watershed scale. The SCQI is a good management tool because it identifies specific problems in the landscape and provides future direction to minimize them.

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Introduction

One of the goals of sustainable forest management (SFM) is to implement best management practices so that water quality is maintained within natural ranges of variability (CCFM 2000). Within an SFM framework there is a requirement for a set of clearly defined performance criteria and indicators to gauge progress towards the goal of maintaining water quality. Designing a meaningful indicator to address this goal is not an insignificant challenge. Forestry activities are an extensive type of disturbance that generally cover many hundreds of square kilometers and numerous watersheds. Forest harvesting activities can affect many water quality characteristics, but increased sediment loading has been identified as one of the most detrimental (MacDonald et al., 1991). Several forest harvesting activities can cause increased erosion rates and sediment delivery to aquatic environments. However, road building and maintenance, particularly at stream crossings, is the dominant point source for forestry-generated sediment in landscapes where landslides are not a dominant process (Beaudry 2001, Beschta 1978, Bilby et al. 1989, Cafferata and Spittler 1998).

Within any given watershed, there may be dozens or even hundreds of stream crossings, each being a potential source of sediment. Although the impacts of forestry disturbances on water quality can be relatively small and subtle at any given point within a watershed, the sum of the impacts may add up to significant downstream cumulative effects. If good road building and maintenance practices can minimize (or eliminate) accelerated erosion and sediment delivery to streams then negative impacts to water quality will be minimized. Based on this assumption, several B.C. and Alberta Divisions of Canadian Forest Products Ltd. (Canfor) have decided that a good water quality indicator should be based on a field survey that evaluates how well accelerated erosion and sediment delivery are being controlled in the vicinity of stream crossings. The stream crossing quality index (SCQI) was developed as an SFM indicator to provide a meaningful measure of the potential hazard that a stream crossing may present for water quality.

Development and Refinement of the SCQI

In 2000, the Prince George Division of Canfor considered a variety of SFM indicators for use in its forestry certification program. As an indicator of protection of water quality Canfor was considering the concept of the stream crossing density used in the BC Watershed Assessment Procedure (WAP), i.e. # of stream crossings counted on a map divided by the watershed area (BC Government 1995). We suggested that although the stream crossing density is very simple and inexpensive to measure, a better alternative would be to complete a field assessment of the crossing and score its real potential for accelerated erosion and sediment delivery to the stream. Such a procedure would provide accurate field based information and would be a large improvement on the stream crossing density concept that assumes that all crossings produce the same amount of sediment to the stream environment. Thus was born the concept of the SCQI, a field-based hazard assessment of the potential for accelerated erosion and sediment delivery at stream crossings.

The origins of the SCQI methodology were based on the concepts of the sediment source survey (SSS) presented in version 2.01 of the WAP (B.C. Government 1999). In the WAP, the road related SSS is used as an indicator of the level of hazard that forestry roads have of delivering sediment to the aquatic ecosystem and thus potentially reducing water quality. One of the major refinements provided by the SCQI methodology is the systematic description and evaluation of all individual sediment sources at a crossing that have the potential to deliver sediment to the stream network.

As an SFM indicator, the basic assumption that underlies the SCQI is that if erosion and sediment delivery in the vicinity of stream crossings is minimized, through proper road building and maintenance practices, then the potential impact to water quality from increased sediment delivery is also minimized. The SCQI is a useful management tool because it provides a clear incentive to improve erosion and sediment control (ESC) practices in the vicinity of stream crossings since it documents practices that create a water quality hazard and those that minimize it. Improvement of forest management practices over time is a clearly explicit goal of all forest certification schemes. The Canadian Council of Forest Ministers (CCFM, 2000) clearly recognizes the potential negative impacts to water quality associated with road crossings. In their sustained forest management program they have defined one of the aquatic indicators as being: "percentage of forest area having road construction and stream crossing guidelines in place" (Indicator 3.2.2). The SCQI approach takes that concept several steps further by actually completing a field-based evaluation on the quality of road construction and stream crossing maintenance relative to the production and delivery of to the aquatic environment.

Method

The execution of an SCQI survey begins with the mapping of current access within the watershed and planning an effective way of completing a 100% sampling of stream crossings with that watershed. In many situations 100% sampling is not possible but at least 90 to 95% sampling is usually achieved. Stream crossings are accessed using trucks, quads or by walking.

Once the surveyor has arrived at the stream crossing, the procedure begins by evaluating the size and characteristics of all sediment sources that can potentially contribute sediment to the aquatic environment. The area contributing to each stream crossing is divided into eight distinct and independent "elements" that could deliver sediment to the stream network. These include four road ditches that run into the stream, two road fill slopes and two road running surfaces, each of these potential sources being assessed independently. The sediment source hazard score for each individual element is a product of the erosion potential and the delivery potential of that source. The erosion potential is calculated as a function of several factors which include:

1. the size of the sediment source
2. the soil texture of the source
3. the slope gradient of the source
4. the age of the source, and
5. the level of road use

The cornerstone of the SCQI procedure is the measurement of the size of the sediment source (m^2). The other four variables act as modifiers to increase or decrease the hazard associated with the size of the sediment source. Each of the four modifiers is scaled from 0 to 1, where zero (0) represents a condition that would eliminate the hazard (e.g. coarse gravel, no slope or an abandoned fully revegetated road) and one (1) represents a condition that would maximize the hazard (e.g. silt, slope greater than 15% or active mainline). The size of the sediment source (m^2) is multiplied by the value of each modifier to generate an erosion potential score for the particular element being assessed. This is then multiplied by the delivery potential (scaled from 0 to 1) to obtain the element score. The delivery potential represents a qualitative assessment of the percentage of the eroded material that will likely reach the stream. A series of definitions are provided to assist in the determination of the delivery potential, i.e. 0 means that there is no connection between the erosion source and the stream and no delivery is possible, 0.5 means that the delivery is indirect and filtered through trees, grasses and/or sediment control structures, 0.8 is used when sediment is weakly filtered through a sparse grass cover and most of the material reaches the stream and 1.0 means that delivery is evident, direct and uninterrupted with no obvious depositional zones before reaching the stream. The total score for the crossing is simply the addition of the eight scores for each of the individual elements. The final SCQI crossing score generates four hazard classes as defined in Table 1.

Table 1. Correspondence between SCQI score and hazard level

Score	Hazard Level
0	no hazard
$0 < \text{score} < 0.4$	Low
$0.4 < \text{score} < 0.7$	Moderate
Greater than 0.7	High

The values for each of the modifiers are based on the concepts and values developed for the Revised Universal Soil Loss Equation (RUSLE) presented in Wall et. al. (2002). The universal soil loss equation was initially developed by Wischmeier and Smith (1965). The objective of the RUSLE was to provide a quantitative tool to assess the potential for soil erosion at a given site.

For the purposes of the SCQI objectives, the stream crossing scores are analysed both on an individual basis and collectively for a watershed. Individual stream crossing scores can be used to identify problem areas or unique problem sites. The collective scores can be used as an overall indicator of the potential for water quality problems within a given watershed, thus providing a landscape level perspective.

The SCQI procedure is a useful management tool because it identifies the specific location and magnitude of erosion problems. If scores are high, the crossing can be improved through remedial actions and current practices can be altered to avoid high scores in the future. If scores are low, then it shows that good erosion

Table 2. Levels of risk associated with increases in turbidity (adapted from Department of Fisheries and Oceans, 2000)

Induced Turbidity (NTU)	Risk to Fish Habitat	SCQI Hazard Level
0	None	None
1 to 8	Very Low	Low
8 - 35	Low	Medium
35 - 66	Moderate	
66 - 130	High	High
> 130	Unacceptable	

Table 3. Correspondence between SCQI scores, hazard levels and induced turbidity

Score	Hazard Level	Induced turbidity (NTU)
0	No hazard	0
0 < score < 0.4	Low	0.1 to 8
0.4 < score < 0.7	Moderate	8 to 70
Greater than 0.7	High	Greater than 70

and sediment control practices are being implemented and by extension water quality is being protected. The procedure has been presented to numerous field practitioners in a series of field workshops and received a favourable response because it clearly identifies the specific location of the problem and the practice that generates the problem.

It is important to note that the SCQI method was designed to be quick (about 15 minutes per crossing) so that a maximum number of crossings can be assessed, thus providing a better landscape level perspective. The SCQI has evolved over the last three years from its initial structure based mostly on subjective assessments. The procedure is now more objective, repeatable and transparent, using values based on the RUSLE.

It must be noted that the whole SCQI approach is a largely a conceptual model, based on the general concepts of the RUSLE, and was not developed based on an experimentally acquired set of empirical relationships. It provides a score in a consistent way that can be compared with other crossings in a given watershed and evaluated for how "good" or "bad" the crossings are. The SCQI does not provide a quantitative evaluation (e.g. kg/ha/yr) of exactly how much sediment is entering the stream or what the impact of that sediment is on the stream environment. The SCQI approach tells you where there are erosion and sediment control problems, how frequent in the landscape those types of problems appear and provides a basis of information to judge the magnitude of the problem and how to fix it so that impacts to water quality will be minimized. It is important to emphasize that the SCQI focuses exclusively on the evaluation of the sediment source and the potential of that sediment to reach a stream (i.e. the "hazard"). It does not in any way attempt to measure, evaluate or score the sensitivity of the stream or the impact of increased sediment delivery to the aquatic environment (i.e. it does not evaluate "consequence"). Work is currently underway to develop a methodology to evaluate the sensitivity of a stream to increased sediment loads. If this effort is successful, it could be combined with the SCQI approach to produce a true risk assessment procedure.

Evaluation of the SCQI Procedure

In 2001 an evaluation program was initiated by Canfor, Prince George Division, to test the validity of the SCQI procedure by monitoring stream turbidity levels at selected stream crossings. The validation program began by completing SCQI surveys over a variety of topographic and climatic conditions across the Prince George Timber Supply Area. Several hundred stream crossings were surveyed in the spring of 2002 to generate a population of possible sampling sites. These data were then sorted by SCQI scores and stream characteristics. From this database we eliminated all large streams (relatively rare occurrence in the landscape) and streams that were too small to be instrumented. Our objective was to focus the measurements on "small" streams with an average bankfull width of 1 to 3 metres since about 90% of stream crossings in the Prince George region occur on small streams (P. Beaudry and Associates Ltd. 2002). The crossing scores were then grouped into one of three hazard levels, i.e. low, moderate or high (see Table 1). We then completed a random selection of seven stream crossings per hazard level to serve as our experimental sample (i.e. total of 21 crossings).

Each crossing was instrumented with an electronic continuous turbidity sensor in an "upstream-downstream" experimental design. The assumption behind this approach is that the difference between the upstream and downstream measurements can be attributed to the erosion and sediment delivery at the stream crossing (i.e. induced turbidity). The objective was then to compare the measured induced turbidity with the hazard level to see if there was an acceptable correlation. To make this correlation, standards provided by public resource agencies were adapted.

The BC Ministry of Water Land and Air Protection (WLAP) has established water quality guidelines (Government of BC 2001). These suggest that land-use activities should not create "induced" turbidity levels greater than 8 nephaloturbidity units (NTU) for the maintenance of aquatic life. Based on these guidelines, we define a low hazard as a situation that will not cause induced turbidity levels to exceed 8 NTU. For the definition of the Medium and High hazard classes we use the guidelines suggested by the Department of Fisheries and Oceans (2000) (Table 2). Based on these provincial and federal guidelines we defined hazard levels relative to levels of induced turbidity (Table 3). As an example, an SCQI hazard level of high is defined as a site that generates enough sediment to the stream that it will consistently cause an increase in turbidity greater than 70 NTU when significant rainfall occurs.

Results from the 2002 turbidity measurements generally showed a good correspondence between assessed hazard level and induced turbidity measurements throughout the Prince George Timber Supply Area. The validation process also identified some specific problems with the procedure and these were addressed for the 2003 field season. The validation program was continued in 2003, but results have not yet been processed.

Conclusions

Canfor has completed SCQI surveys over a wide range of their operating areas as part of their forest certification programs (well over 3,000 crossings). These include areas within central and northern B.C. and eastern Alberta. Several independent certification audits have identified this approach as a meaningful and well structured process to objectively document the extent of effective erosion control practices in the landscape. Road construction and maintenance supervisors find this a useful tool because it locates and identifies specific problems and provides direction for remedial action with the built-in incentive of obtaining a better SCQI score in the future. The SCQI tool is also useful to show improvements in erosion control practices over time, a requirement of many forestry certification schemes.

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Spawning Gravel Quality and Salmon Production in British Columbia

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Extended Abstract

The manner in which salmon, trout and char excavate redds in the gravelly substrate of headwater streams has been extensively documented. Eggs are laid in pockets buried in the deepest portions of the redds. The selective transport of fine bed materials during redd excavation decreases the amount of matrix material in the gravel. The resulting higher permeability and changes to the stream bed surface increase aeration of developing embryos and hence assure better rates of egg-to-fry survival.

In northern British Columbia there are a select number of channel reaches where the volume of coarse (> 2 mm) bed-material mobilized by salmonine excavation is comparable to that mobilized by annual floods. From the perspective of habitat management and conservation policy this has a number of implications. It is notable that, in the Fraser and Skeena River watersheds of BC, a few such spawning sites produce most of the fish in the watershed. This paper discusses some of the important conditioning effects that large numbers of spawners have on their own habitat.

Over the past 12 years we have studied movement of bedload in spawning beds of the early Stuart sockeye of the Upper Fraser Watershed. In the bse streams, the topography of the stream bed is completely altered by sockeye bioturbation. The streambed topography switches annually from a streamlined flood morphology to a more complex bioturbated one. The salmon-created bedforms are present in these streams for about nine months of the year.

At high density sites, chinook salmon align their redds transverse to the stream flow and

Gottesfeld, A.S., Tunnicliffe, J.F. and Hassan, M. A. 2004. Spawning gravel quality and salmon production in British Columbia. Pages 163-164 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

create gravel dunes which survive for years and are used by subsequent generations of salmon. At these sites chinook clearly dominate the sediment movement regime.

Chinook salmon spawn in more than 57 streams in the Skeena, however 7 reaches account for 87% of the total Skeena production. Spawning dunes characterize at least 5 of these sites. These spawning reaches total less than 66 km and comprise less than 0.4% of the available higher order stream lengths. Sockeye spawn in about 90 streams in the Skeena, but 13 spawning stream reaches account for 90% of the Skeena productivity. In at least portions of each of these spawning reaches, much of the streambed area is reworked annually by sockeye redd excavation. Similar but less extreme concentrations of spawning habitat utilization are seen in steelhead and coho salmon.

Where salmon provide a significant contribution to the sedimentary regime, a positive feedback situation arises: increases in egg-to-fry survival increase recruitment, which provides the spawner density to greatly influence the sedimentary regime. The special characteristics of these spawning areas makes them keystone ecosystems of their respective watersheds. If we are to retain these critical areas, we need to identify them, and then to manage their upstream areas to retain the low fine-sediment transport levels that characterize them. We also need to manage fisheries so that large fish populations are maintained. Considering the very small portions of the stream network which serve this function, preservation of these areas is an appropriate and attainable management objective.

Stream Invertebrate Communities as Indicators of Logging Disturbance in Northern Hardwood Forests of Ontario

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Extended Abstract

Aquatic insect communities were assessed in three streams across a range of selection-based logging intensities in low-order catchments of a northern hardwood forest, and compared to a nearby reference stream. Logging was conducted in study catchments of the Turkey Lakes Watershed near Sault Ste. Marie, Ontario by a mechanical harvester and cable skidder at harvesting intensities of 29, 42, and 89% basal area removal. No riparian reserves were assigned to the streams. Temporal and spatial differences in aquatic insect community structure were examined over a 5-year study period (2 years before and 3 years after the logging) through an analytical pathway that included multivariate ordination, analysis of similarity, a bubble plot procedure to determine discriminatory taxa, and analyses of population trends among discriminatory taxa. No differences in community structure, community metrics, or relative abundance of discriminatory taxa were detected between the low-disturbance (29% removal) and reference sites. At the moderate-disturbance site (42% removal), small changes in community structure were detected and these were primarily the result of increases in detritivorous collector-gatherer taxa, especially the mayflies *Paraleptophlebia* and *Eurylophella* spp. Increases in collector-gatherer taxa were associated with a significant increase (about 2.5 times) in stream bed deposition of fine particulate organic matter. There were no measurable changes to the stream canopy cover at the moderate-disturbance site. Larger and more distinct changes in community structure were detected at the high-disturbance site (89% removal).

Kreutzweiser, D.P., Capell, S.S. and Good, K.P. 2004. Stream invertebrate communities as indicators of logging disturbance in northern hardwood forests of Ontario. Pages 165-166 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

These were largely driven by increased richness and abundance among grazer and grazer/gatherer taxa, particularly *Baetis* mayflies. In addition to significant increases in abundance of *Baetis,* four other grazer taxa (*Heptagenia, Epeorus, Rhithrogena, Diura*) appeared at the high-disturbance site after the logging that had not been previously detected at any site. The increases in abundance of grazer and grazer/gatherer taxa were accompanied by a small decline in the proportion of shredder taxa. Effects on insect communities at the high-disturbance site were in response to a 65% reduction in stream canopy cover after the logging.

Overall, effects of this selection-based logging on stream insect communities were generally small, even when statistically significant. Over the course of the 5-year study period, changes in insect community structure similar in magnitude to the post-disturbance responses were observed among pre-disturbance years and at the reference site. Although none of the logged sites had protective riparian reserves, the low- and moderate-disturbance sites were logged in accordance with a riparian code of practice that prohibited tree removal or equipment operation within 3 m of stream edges. It appears that compliance with this riparian code of practice and careful logging practices mitigated harmful effects on stream habitat and insect communities at a logging intensity of up to 42% basal area removal in these hardwood forests. The clear effects on aquatic insect communities at the high-disturbance site were associated with a harvesting intensity that was intentionally high for experimental purposes and well beyond normal harvesting prescriptions for hardwood forests in this region.

Stewardship Program Evaluation Tools: Logic Modeling a Common Vision

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Extended Abstract

Vision without action is merely a dream. Action without vision just passes the time. Vision with action can change the world.

Joel Barker

Stewardship has a long tradition in the Prairies though its First Nations, and many early farmers, foresters, and fishers never knew it by that name often understanding it only by practice. Today, stewardship is synonymous with an ethic of caring for the land to sustain its riches for future generations to enjoy. Like a chameleon, its form is a reflection of the unique characteristics of its surroundings - the community and landscape that nurtures it. Stewardship persists despite its arduous challenges because its rewards are both personally fulfilling and socially moral. But is it making a difference?

Government, industry, and communities recognize the value of stewardship and create programs and initiatives to further their objectives. All too often, however, institutions journey along a path driven by demands for immediate and quantifiable results that band-aid symptoms as opposed to treating source. Despite decades of organized conservation efforts we are still experiencing incremental habitat loss and its consequences. Project-based funding has evolved into a policy instrument of its own influencing the vision and activities of stewardship organizations that depend on and answer to it.

This introduction sets a stage to illustrate where Fisheries & Oceans Canada is taking a decidedly different approach to stewardship program delivery and evaluation in the Prairies. Its fledgling Stewardship-in-Action program is piloting a process-based support program intended to engage industry, NGOs, communities and employees and enable their capacity to influence change without influencing approach.

Objectives driven, the program strengthens stakeholder capacity to take ownership and

Ogilvie, K. 2004. Stewardship program evaluation tools: logic modeling a common vision. Pages 167-168 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

responsibility for environmental issues and craft solutions that are both practical and palatable among the people that ultimately influence and affect the environment. The challenge, is how to pair up resource and process measures as well as proximal and distal outcomes to evaluate the program's effectiveness. Logic modeling is helping the department see what is effective and where to apply adaptive management strategies.

Logic models are tools for evidence-based storytelling that map how a program or initiative is supposed to work. Effective logic models make an explicit, often visual, statement of the activities that will bring about change and the results expected. A logic model creates a common vision to ensure that actions move process in the same direction by providing a common language and point of reference.

Functional partnering, ongoing information exchange, and logic model evaluation are key strategies for responsive and adaptive management that have born successes, identified challenges, and strive to support the Prairie community's vision of stewardship along a continuum of awareness, understanding and action.

We can't all be charismatic leaders, but we can use practical tools
such as logic models to find the story in our own work to change the world

Community Tool Box
http://ctb.ku.edu

Poster Presentations

Behavioral and Population Responses of Ovenbirds (*Seiurus aurocapillus*) to Increasing Forest Dissection by Seismic Exploration

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Extended Abstract

While the physical area disturbed by linear features is relatively small in most forested systems, linear features have a disproportionate effect on forest fragmentation. In the boreal plains of northeastern Alberta, exploration by the energy sector has resulted in extensive dissection of pristine landscapes through the creation of 8 m wide seismic lines. Levels of fragmentation are highly variable due to the clustered nature of energy resources and the cumulative effects of multiple exploration and extraction events. Combined with the natural patchiness of the boreal forest, seismic lines can result in landscapes with highly variable fragmentation geometries (e.g., forest slivers).

A spatially explicit model was developed in GIS to test how the density of birds in areas of high forest dissection might change if individual birds respond to seismic lines under the passive-displacement (i.e. used the seismic line as a territory boundary) versus active avoidance hypotheses. Assuming birds had fixed territory shape and size, our model demonstrated that increasing seismic line density from 0 to 8 km per km^2 could result in a 38%, 62% and an 82% decline in bird populations if individuals use seismic lines as a territory boundary or avoided lines by 25 or 50 m, respectively. We empirically tested the model using the Ovenbird (Seiurus aurocapillus) as a model species. Based on radio-telemetry (n = 12), male Ovenbirds crossed

Bayne, E.M., Boutin, S., Van Wilgenburg, S.L. and Hobson, K.A. 2004. Behavioral and population responses of ovenbirds (*Seiurus aurocapillus*) to increasing forest dissection by seismic exploration. Pages 171-172 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

seismic lines as part of their daily activities. However, over 80% of their home range and almost 100% of singing locations were on one side of the line. This suggests that Ovenbirds used seismic lines as territory boundaries. Despite a behavior that our model predicted would lead to significant population declines, spot mapping of 24 plots with linear feature density ranging from 0 to 8.6 km per km^2 revealed no differences in Ovenbird density at the stand level. Similarly, in 4 km^2 landscapes (n = 62) sampled with a systematic series of nine point counts, we also detected no changes in Ovenbird numbers across the same range of seismic line densities. At the individual point count level, Ovenbirds declined once a threshold in seismic line density of 8.5 km per km^2 was reached. Above the threshold, Ovenbirds declined 19% per 1 km per km^2 increase in seismic line density. Below the threshold, Ovenbirds seem to have compensated for increased fragmentation, likely through changes in territory shape, size or overlap. The number of point locations where seismic line densities reach 8.5 km per km^2 is still relatively low in Alberta but definitely could become an issue if current energy exploration practices continue. Under the precautionary principle, we recommend energy sector companies consider adjusting their practices to utilize new seismic techniques that allow narrower lines (2 m) to be used. This should reduce potential impacts on birds by more closely mimicking natural gap patterns.

Juvenile Coho Off-Channel Pond Habitat Development Adjacent to an Interior British Columbia Glacier-Fed River

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Extended Abstract

The Telkwa coho ponds were constructed on the Telkwa River floodplain located approximately 18 km southwest of Smithers in north-central B.C. The initial ponds were built in 1993 by Fisheries and Oceans Canada as a pilot juvenile coho enhancement project. They were extended in 1995 and modified again in 1997 so that they now comprise 8700 m^2 of wetland habitat. The ponds consist of two pond complexes with an inter-connecting creek and side channel extensions. Inlet flow into the ponds are mainly derived from subsurface flows in the Telkwa floodplain, and the ponds are normally only accessible via a small outlet creek connecting to the mainstem Telkwa River from mid-May through late July. Telkwa River coho spawn 15 to 30 km upstream from the ponds, and newly-emerged fry as well as yearling coho enter the small connecting tributary during the high-flow snowmelt period from mid-May through July. Mark-recapture population estimates have been conducted for nine years in the pond complex during early May since pre-development (1993) through until 2001. As well, all fish movements in and out of these ponds have been monitored at traps for a six-year period up to 2001. Coho account for 99% of the fish movements into the off-channel ponds, and the initiation of fish movements corresponds to increased temperatures in the pond outlet creek. All coho smolts also leave the ponds during this May and June high-flow period. Just under 50% of the smolts are age 1+, with the remainder predominantly 2+ and a few age 3+ fish. Mark-recapture estimates conducted just prior to smolt outmigration indicate populations in the ponds ranged from just over 200 coho prior to the main pond construction to between 900 and 2800 coho after construction. The estimates of coho production range from 11 to 32 pre-smolts/100 m^2. The measured smolt movements from the pond complex averaged 12 smolts/100 m^2, suggesting the productivity in this interior pond complex is lower than the range of estimates from coastal ponds reported in the literature. Despite poor fry recruitment into the ponds during some years, the recruitment of yearlings into the ponds has helped smooth fluctuations

Bustard, D. 2004. Juvenile coho off-channel pond habitat development adjacent to an interior British Columbia glacier-fed river. Pages 173-174 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

in smolt outmigration from the ponds. A series of tests and observations indicated that coho smolts had difficulty locating the pond outlet when the culvert and beaver control screen and box were inappropriately installed. Special fencing and debris rafts to reduce beaver problems at the inter-connecting stream channel, while initially effective, have had problems since 2000. The development of the ponds, in addition to increasing coho smolt production has added habitat biodiversity along this section of the Telkwa River, particularly for amphibian and bird populations. Funding for the monitoring of this project has been largely derived from Forest Renewal B.C. through West Fraser Mills Ltd. and Fisheries and Oceans Canada.

Stream Crossing Inventories in the Smoky and Simonette River Watersheds of Northwestern Alberta

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Extended Abstract

Detrimental effects to aquatic environments, resulting from stream crossings, are well documented. When properly installed, monitored and maintained, stream crossings such as culverts and bridges allow for the connectivity of waterways and minimized aquatic impacts. Improperly installed culverts have been shown to cause encroachment of stream channels. This leads to increased water velocities, scouring, sedimentation and hanging culverts, all of which can impact habitat connectivity and result in reduced viability of fish populations. Prioritizing culvert crossings for remediation requires information on each structure's status and effectiveness at providing fish passage, while maintaining road integrity. From this information, road managers can make informed decisions that will provide the greatest benefit to fish populations.

Canfor Grande Prairie identified 79 crossing sites, within their Forest Management Area (FMA), to be assessed based on their associated fish and habitat qualities. Using a Geographical Information System and spatial data acquired in 1997, we estimated stream-crossing density in the Simonette watershed to be approximately 0.11 crossings/kilometer of stream. This value is likely an underestimate, as recent road construction information was not included on our data layers. Of the original 79 identified crossings, 78 were assessed for fish passage potential and 45 were deemed suitable for fish and habitat assessments. Data were collected in adjacent stream reaches, upstream and downstream of the crossing structure. We defined fish passage

Doran, M.A., Johns, T.W.P., Tchir, J.P. and Hvenegaard, P.J. 2004. Stream Crossing Inventories in the Smoky and Simonette River Watersheds of Northwestern Alberta. Pages 175-176 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

barriers as either: "outfall" (i.e., culvert outlet not flush with water level), "potential velocity" (i.e., water velocity greater than fish passage ability), "debris" (i.e., logs, beaver dams, etc), or "damaged pipe" (i.e., physical distortion to culvert shape making the structure impassable to fish).

Potential barriers were identified at 43 of the 78 assessed sites (i.e., 55%). "Outfalls" represented an overwhelming majority of these (i.e., 79%, n = 34), while "potential velocity" and "debris" were each observed at four sites (i.e., 9%) and "damaged pipe" was recorded at one crossing location (i.e., 2%). Assessed sites were prioritized for remediation based on: Strahler Stream Order; fish presence at crossing; proximity to fish bearing water; usable upstream habitat; and type of potential barrier. Although a large proportion of the assessed culverts did not accommodate potential fish passage, the majority of crossings (i.e., 86%) were located on first-order and second-order reaches. Due to limited habitat suitability and water availability in these orders, it was assumed that there was a relatively low probability of fish occurrence and stream crossing structures were ranked accordingly.

Given the high likelihood of culvert crossings resulting in potential fish passage barriers, it would appear necessary for responsible watershed stakeholders to monitor, maintain and/or replace culverts more frequently than structures that provide consistent fish passage over longer durations (i.e., open bottom arches or bridges). The data collected will assist forest planners in developing site-specific crossings plans, and will help to minimize the risk of negative impacts on aquatic environments. Canfor's proactive approach, and collaborative effort with the Alberta Conservation Association towards managing culvert crossings, is a positive step towards sustainable development and reducing their ecological footprint.

The Kakwa River Bull Trout Project: Establishing Ecological Baselines to Evaluate Environmental Impacts

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Extended Abstract

The Kakwa River Bull Trout Program is a multi-year research initiative to establish ecological baselines and to quantify the overall status of bull trout (*Salvelinus confluentus*) in the Kakwa River watershed in northwest Alberta. Bull trout within the Kakwa River, a tributary to the Smoky River (54°N 118°W), represent a northern population that has, until recently remained relatively unexploited largely due to its remote location and limited road access. Both fluvial and resident life history forms of bull trout are present.

However, recent and continued expansions in both the oil and gas and forestry sectors has increased access to the watershed and resulted in overall changes in forest cover attributes in the watershed. Since its inception in 1995 the program has developed baseline GIS digital layers describing linear (e.g., roads, seismic lines) and patch disturbances (e.g., harvest blocks, well sites) and related GIS layers to quantify levels of industrial activity which have been combined with focal studies that have quantified and assessed: i) critical winter habitats in the main stem, ii) locations of spawning habitats and their use by adults, iii) spawning frequency and spawning site fidelity in one of the two predominant spawning tributaries, iv) density and size composition of adults in the main stem and density and size composition of juveniles in one benchmark sub-basin.

Results from fish inventories throughout the basin have shown that while bull trout are a predominant species in the river basin and occur at about 40% of study reaches, they occur at relatively low densities of about 0.3 fish per 100 m^2 of river. Data from radio telemetry suggests that adult bull trout occupy relatively few over wintering areas in the main stem, rather than in the smaller tributaries. A predominant over wintering reach of about 15 km in length was identified in the upper reaches of the main stem. Radio telemetry has also shown that bull trout reproduce predominantly in three stream tributaries and that adults may not reproduce on an

Hvenegaard, P. and Tchir, J. 2004. The Kakwa River bull trout project: establishing ecological baselines to evaluate environmental impacts. Pages 177-178 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

annual basis. Annual monitoring of juveniles in one of the dominant spawning tributaries over a six year period has shown that the density of juveniles is moderately variable ranging between 0.6 to 2.1 individuals per 100 m^2.

These and recently developed studies provide the opportunity to evaluate the effects of industrial activities on bull trout and are the basis of a scientific rigorous database that can assist with the continued develop and adoption of best management practices. Our poster presentation provides an overview of activities and results to date and their importance to sustainable forest management.

Headwater Stream Temperature Responses to Clearcut Logging in North Central British Columbia

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Extended Abstract

Most of our knowledge about the effects of streamside timber harvesting comes from studies conducted in coastal regions, but there remains some uncertainty as to whether results from these studies are applicable to regions that differ with respect to climate and topography. In 2000 a project was initiated to determine if riparian harvesting along headwater streams in north-central British Columbia (a region characterized by temperate, interior geo-climatic conditions) maintained the necessary ecological attributes for healthy fish populations. One project objective concerned the quantification of the temporal, spatial and among stream variations in summertime (June to August) temperatures and any change resulting from riparian harvesting. Pre-harvest data were collected in 2001 and 2002 from five small, headwater streams. Logging operations were conducted around three of the streams in 2003. Streamside clear-cut logging consisted of the removal of only mature commercial timber, with the majority of non-commercial and deciduous timber continuing to provide shade and future recruitment of large organic debris. This harvesting treatment resulted in the loss of ~50% of the streamside canopy cover. By comparison, the remaining cutblock areas were completely clear-cut.

Pre-harvest temperature data indicated all streams were relatively cold during the summer months. Daily temperatures ranged from 5.8° to 11.5°C with relatively small diurnal fluctuations (daily maximum minus daily minimum) averaging 1.1°C. Post logging changes in daily mean, maximum, and minimum temperatures, as well as in daily fluctuations, were <1.5°C. The application of a model to predict the average summertime daily warming or cooling (defined as the difference between upstream and downstream mean daily temperatures) revealed an average absolute deviation from the line of correspondence of < 0.75°C for both

Maloney, D., Mellina, E. and Chamberlist, L. 2004. Headwater stream temperature responses to clearcut logging in north central British Columbia. Pages 179-180 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

pre- and post-harvesting seasons. The model may be useful to managers wishing to forecast potential changes in stream temperatures following the removal of riparian timber because it requires average maximum temperatures recorded at an upstream boundary and canopy cover as predictors.

Our data indicate that these small, headwater streams located in a temperate, interior region are generally cold, and that the riparian harvesting treatments we applied resulted in relatively small increases in temperature (compared to increases of ~5–7 °C that have previously been reported in the predominantly coastal literature). These differences may be a result of differences in climate, topography, forest cover and logging methods that exist between the two regions. Lastly, the relatively small temperature changes observed in our streams are not likely to be detrimental to resident salmonids.

Historical Changes in Rocky Mountain Foothills Stream Fish Communities: Evaluating the Use of Fish Abundance and Size as Ecological Indicators

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Abstract

We evaluated the use of two aquatic indicators to detect changes in fish abundance and size composition as part of an adaptive management system to support sustainable forest management. We conducted the evaluation by comparing attributes of fish abundance and size composition at a median 20 year interval using data from within a number of watersheds representing a broad range of biophysical and land-use conditions in west-central Alberta. Our first indicator, a measure of fish abundance, was the average catch rate using backpack electrofishing from multiple sites within a given watershed. Our second indicator was the proportion of fish of catchable size by angling (> 149 mm), at an individual site. Our data suggest that the use of the fish abundance indicator is contingent on obtaining sufficient replication and minimizing or accounting for between-site variation in natural factors. These problems were however not associated with the use of proportion of catchable size fish. Our findings suggest that in certain foothills streams, angling history has influenced both fish abundance and size. Connections between these indicators and land-use require further exploration including discerning the effects of angling from land-use and identifying of specific habitat impacts. Development of habitat indicators will likely become an increasingly important part of an adaptive process to support sustainable forest management.

McCleary, R., and Bambrick, C. 2004. Historical changes in Rocky Mountain Foothills stream fish communities: evaluating the use of fish abundance and size as ecological indicators. Pages 181-187 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

Sustainable forest management initiatives advocate the establishment of an adaptive management system that comprises a set of indicators representing major values (CCFM 2003). Our development of such a program with aquatic indicators through the Foothills Model Forest (FMF) presented two challenges. First, in order to evaluate the status of fish populations and their habitats, we needed to establish the natural range of spatial and temporal variation of each potential indicator. Second, detection and identification of plausible causes of an observed change needs occur in a timely manner so that management activities can be adjusted prior to the onset of long-lasting impacts. Fortunately, within our 10 000 km^2 study area within the Foothills ecoregion, fish inventories were initiated in the early 1960's near the onset of wide-spread resource development. Therefore, as one of the first steps towards development of a set of aquatic resource indicators, we undertook a comparison of the findings from historic and recently replicated fish inventories from adjacent streams. Within our study area, a number of factors may influence the distribution and abundance of stream fish; therefore we compared any changes from our historic-replicate comparison with changes in land-use and angling regulations from the corresponding time period.

Historically, stream assessments were completed as a result of a variety of initiatives including road and pipeline development, forestry-related research and government studies. However, findings from these studies had never been compiled. Preliminary analyses of these compiled studies suggested that the number and extent of historic samples sites might represent a wide range of human-use activities and ecological conditions. For this study, we postulated that if sufficient inventories had been completed within the study area, valuable information could be gained by replicating the historic surveys to quantify spatial variation in attributes of stream fish communities and to gain insight on whether changes in fish community structure are related to changes in land-use.

Study area

Our study area lies within the Forest Management Agreement Area held by Weldwood of Canada Ltd., Hinton Division. The study area (10,000 km^2) includes watersheds that originate in the Subalpine, Upper Foothills and Lower Foothills natural subregions in west-central Alberta. Forest harvest, initiated in 1956, focused on specific areas dominated by over-mature timber (Bott et al. 2003) and harvest was not evenly dispersed across the landscape. As a result, we were able to locate watersheds within all natural subregions with varying levels of harvest and road development. We selected a total of twelve watersheds (3rd to 6th order, 49 to 337 km^2) representing the range of biophysical characteristics (McCleary et al. 2003), land-use conditions (Sherburne and McCleary 2003) and angling efforts (Bambrick and McCleary 2003). Of these watersheds, we expected to identify detectable increases in the density and size of individual sport fish species in Lambert Creek, which until recently (pre 1995) experienced high angling effort as a result of its close proximity to the towns of Edson and Robb, coincident with consistent low levels of land-use within the watershed. Similar results were expected in Solomon Creek as a result of its close proximity to the town of Brule and corresponding low levels of land-use. Given the various combinations of land-use and angling effort within the remaining 10 catchments, no other trends in density and size of sport fish were forecast.

Methods

As a first step, we undertook a literature search for all stream fisheries projects and stream fish databases completed within the study watersheds. We defined historic information as data collected in 1992 or earlier and current information as any study completed after 1992. We considered a number of data sources for historic and current information including those held by the Government of Alberta, and those completed by the Foothills Model Forest.

Information on the abundance and structure of stream fish communities and fish habitat from historic and current reports was entered into a single database. The majority of information was produced by stream fish surveys between 50 and 350 m lengths of stream (i.e., the sampling site). The locations of historic surveys were identified in a variety of ways including UTM coordinates, legal land descriptions, schematic maps (average scale 1:50,000), or written descriptions. Using GIS, we converted locations of this historical information into a standard UTM format, which were imported into the common database.

Once all sites were mapped, comparisons were made between historic sites and current sites to identify sites that may have been sampled on multiple occasions. Historic sites and current sites were considered to be the same location if they were within close spatial proximity (i.e., 300 m), and (1) were located in a stream reach with similar slope and drainage area size, (2) had the same stream order, and (3) were not bisected by a potential barrier such as a road or waterfall. Where a historic site and an FMF site shared the same location and sampling season, no further sampling was necessary to establish a historic-current pair. Historic sampling sites lacking a current pair were selected for inventory in 2001. Inventories of stream fish communities and related habitat data were collected using standard inventory methods (McCleary et al. 2001).

We selected fish catch rate as an indicator of relative abundance, which was calculated by dividing the number of fish captured greater than 50 mm in fork length by the sample area (number of 100 m^2 units) and then by the sample effort (number of minutes electro-fished).

For 10 of the watersheds, we tested the null hypothesis that the mean replicate catch rate equals the mean historic catch rate, using a two-tailed paired t-test ($\alpha = 0.10$). For Lambert and Solomon Creeks, we tested the null hypothesis that the mean replicate catch rate is not greater than the mean historic catch rate, using a one-tailed paired t-test ($\alpha = 0.05$). The sample size for this test was equal to the number of paired sites within a watershed.

Providing an opportunity for angling is an important objective of both fishery and fish habitat managers. Previous studies have identified a minimum length of 150 mm to be sufficient for sport fishing (Koning and Keeley 1997). Given that previous studies within the region have found fish with a fork length as small as 120 mm in spawning condition, the number of fish > 149 mm may also provide information on the spawning or reproductive capacity of a population.

For individual sites within 10 of the watersheds, the null hypothesis that the proportion of catchable size sport fish (> 149 mm) on the historic date and current date do not differ was tested using a two-tailed Z-test ($\alpha = 0.10$) for 10 of the watersheds. For individual sites within Lambert and Solomon Creeks, we tested the null hypothesis that the proportion of catchable size sport fish (> 149 mm) on the replicate date is not greater than the historic date using a one-tailed Z-test ($\alpha = 0.05$). The sample size for this test was the total number of fish greater than 50 mm that were captured at an individual site. Note that pooling of the data from a number of sites was not required. The Z-test was selected over a t-test because the former has been adapted to test for differences in proportions from two populations. For example, wildlife researchers use proportional analyses to assess changes in dichotomous (0,1) variables and have developed a power analysis as a function of sample size (Krebs 1989). We tested for differences if at least ten fish greater than 50 mm fork length were captured in both the historic survey and the current survey.

An increase in the proportion of fish of catchable size would be considered a positive indicator only if this occurred with the maintenance of juvenile size classes. The case where an increase in the proportion of catchable size occurs while juvenile fish are poorly represented would be indicative of recruitment failure. Therefore, to confirm the presence of juveniles, we generated fork length frequency distributions for all cases that met the criteria for the proportional analysis. Based on findings from previous studies in the region, fish with a fork length of a minimum of 120 mm were considered adults and smaller fish were considered juveniles.

For all watersheds where historic-current comparisons were completed, the findings were compared to changes in angling regulations (Bambrick and McCleary 2003) and changes in land-use (Sherburne and McCleary 2002) over the corresponding time period. Changes in angling regulations were based on a review of all published regulations since 1952. The changes in land-use included percent harvested (derived from a GIS analysis of harvest inventory information) and road density (km/km^2 derived from a GIS analysis of road inventory information verified with a review of digital orthophotos). We rated the level of commercial forest harvesting within individual watersheds based on the percent of harvest as: i) low (< 10% of the entire watershed harvested), ii) moderate (10-30 %) or iii) high (> 30 %). Similarly, we rated road density as: i) low (< 0.3 km/km^2), ii) moderate (0.3-0.4 km/km^2) or iii) high (> 0.4 km/km^2).

Results

Fish surveys and comparison of fish size and abundance

Our search of historic databases revealed fish inventory data for a total of 59 stream reaches. Using our stream reach criteria, we subsequently identified 33 suitable historic sites, of which 21 were paired with inventories since 1992 (i.e., replicate sites). As a result, our comparisons of fish abundance (measured as catch per unit effort [number of fish/100 m/minute electrofishing]) and size (proportion of total fish caught

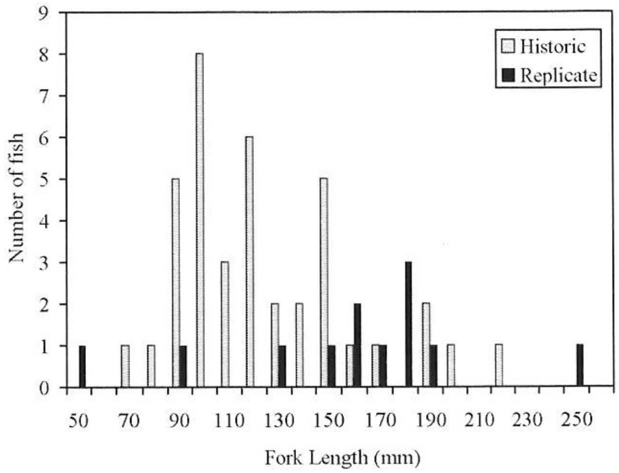

Figure 1. Length-frequency distributions of rainbow trout captured at one location within the Mackenzie Creek watershed during historic (1983) and replicate (2001) surveys. Juvenile age classes = fork length < 120 mm.

> 149 mm) were based on a maximum of 21 paired observations with sample sizes varying with fish species and indicator (i.e. abundance, size). Within the 21 paired observations, the median year of the historic and replicate surveys was 1982 and 2001 respectively. At the watershed scale, analyses of catch rates (Lambert, MacKenzie, Moon, and Pinto Creek) and catchable size fish (Lambert, MacKenzie, Solomon, and the Upper Erith River) were completed for only a single species in each of four watersheds. Given the differences in dominant natural sub region among these four watersheds (Lambert: Lower Foothills; MacKenzie: Subalpine; Solomon: Subalpine/Upper Foothills; Upper Erith: Upper Foothills), we expected differences in fish abundance by species and therefore grouped sites for catch rate analysis by watershed.

A total of 14 fish species were captured at the historic and replicate sites with results presented for sport fish species including brook trout (*Salvelinus fontinalis*, BRTR), bull trout (*Salvelinus confluentus*, BLTR), and rainbow trout (*Oncorhynchus mykiss*, RNTR).

Our analysis identified a detectable difference in catch rate ($P < 0.10$) of rainbow trout only in Lambert and MacKenzie Creeks (Table 1). These findings were derived from a sample size of three sites. Differences in the abundance of other species between the historic and replicate surveys were not statistically significant for any of the other species in any of any of the four watersheds.

We found statistically significant differences ($P < 0.10$) in the proportion of catchable size fish between the historic and current surveys at one location (Table 2). The increase in proportion of catchable size fish at the location in MacKenzie Creek was accompanied by poor representation of juvenile age classes (Figure 1).

Relationships between changes in catch rates, angling regulations and land-use

Significant changes in catch rates between historic and current surveys were detected in two of four watersheds (Table 3). In the Lambert Creek watershed, an increase in catch rate coincided with the implementation of catch and release angling regulations. In the Mackenzie Creek watershed, a decrease in catch rate of rainbow trout corresponded to the implementation of zero catch limit of bull trout in 1995 and full angling closure in this watershed in 2000. Harvest and road development levels were low throughout the study in both the Lambert and Mackenzie Creek watersheds. In the Moon Creek watershed, changes in catch rate were not detected, despite implementation of restrictive angling regulations. There was little change in harvest levels and there was a decrease in road density from high to medium. In the Pinto Creek watershed, changes in catch rate were not detected, despite an increase in angling restrictions, harvest and road development.

Relationships between changes in proportion of catchable size fish and land-use

Changes in the proportion of catchable size fish were detected in one of four watersheds. In the Mackenzie Creek watershed, an increase in the proportion of catchable rainbow trout (> 149 mm) coincided with very low juvenile recruitment and therefore should be considered an indicator of concern for the health of that population. This observation corresponded to the implementation of more restrictive angling regulations including the zero catch limit on bull trout in 1995 and full angling closure of this watershed in 2000. In the Lambert Creek watershed, statistically significant changes in the proportion of catchable rainbow trout were not detected, despite the implementation of catch and release angling regulations during a period where harvest and road development levels also remained low. In the Solomon Creek watershed, no statistically significant changes in the proportion of catchable size brook trout were detected, despite the more restrictive angling regulations and lack of increase in land-use. In the Upper Erith River watershed, no

Table 1. Summary of mean (coefficient of variation) catch rates between historic (before 1992) and replicate (after 1992) surveys in four watersheds of Alberta. BLTR = bull trout; RNTR = rainbow trout. Lambert Creek P < 0.05 (one-tailed), all other watersheds P < 0.10 (two-tailed). * = significant difference.

Watershed	Sites	Species	Historic catch rate (fish/100m²/min)	Replicate catch rate (fish/100m²/min)	Catch rate change (fish/100m²/min)	P-Value
Lambert	3	RNTR	0.003 (131%)	0.008 (45%)	0.005 (35%)	0.02*
Mackenzie	3	RNTR	0.071 (18%)	0.022 (21%)	-0.049 (17%)	0.08*
Moon	2	BLTR	0.056 (111%)	0.023 (111%)	-0.033 (111%)	0.42
Pinto	2	RNTR	0.025 (43%)	0.065 (119%)	0.040 (220%)	0.63

change in proportion of catchable size rainbow trout was detected, despite the moderate increase in angling restrictions, low forest harvest and a high increase in road development.

Discussion

Sustainable forest management and assessment of aquatic indicators

The practice of adaptive forest management requires the creation of linkages between valued ecosystem components (e.g. aquatic resources) and particular forest management activities. For aquatic resources, these linkages could be established in two ways: differentiating angler harvest from land-use effects; and determining effects of land-use on instream fish habitats. Within this study, we detected both positive and negative changes in fish abundance and size that coincided with changes in angling regulations. We failed to detect changes in fish abundance and size that coincided with increases in land-use. Therefore, future studies on aquatic ecosystems changes within this study area should include a methodology to differentiate the effects of angler harvest from those associated with land-use. Habitat evaluations offer a second potential linkage between aquatic resources and land-use. Unfortunately within this study, most of the historic surveys did not contain habitat data that could have been replicated. For future studies, specific hypotheses related to habitat features and land-use should be formulated prior during study design. Road crossings, channel widths, bank characteristics, residual pool volumes and beaver activity could be considered.

The use of catch rates as an indicator of fish population status presented several limitations including the very low sample size (n = 2 or 3) and high variability between sites (coefficient of variation in catch rates in Moon and Pinto Creek watersheds = 111% and 220%), even within a watershed. As a result, the possibility of both Type 1 and Type 2 errors likely remained high. Overcoming these challenges was a key component of other successful fish population studies (Cao et al. 2001). In contrast, these problems were not as strongly associated with the use of proportion of catchable size fish and its utility as an indicator needs to be more fully evaluated. While our evaluation were based on data collected at two time intervals, future studies should also consider the natural temporal variation associated with fish abundance and size.

Considerations from observed relationships

Via a major highway, Lambert Creek is in close proximity to the towns of Edson and Robb and as a

Table 2. Summary of size distributions of rainbow trout and brook trout at individual sites within four watersheds of Alberta. RNTR = rainbow trout; BKTR = brook trout. Large fish = > 149 mm fork length. Lambert Creek and Solomon Creek P < 0.05 (one-tailed), all other watersheds P < 0.10 (two-tailed). * = significant difference.

Site	Species	Sample size		Proportion of large fish		P-value
		Historic	Replicate	Historic	Replicate	
Lambert	RNTR	17	83	0.00	0.13	0.06
Mackenzie	RNTR	39	12	0.31	0.75	0.01*
Solomon	BKTR	75	67	0.20	0.28	0.12
Upper Erith	RNTR	27	19	0.22	0.32	0.48

Table 3. Summary of changes in catch rates, fish size, angling regulations, harvest and road development in six study area watersheds. Large fish = fork length > 149 mm. * = low juvenile recruitment.

Watershed	Significant change in catch rate (+/- to indicate increase or decrease)	Significant change in proportion of large fish	Related angling regulation changes	Percent harvest			Index of road density		
				Historic	Current	Change	Historic	Current	Change
Lambert	Yes (+) RNTR	No RNTR	Catch and release	low	low	low	low	low	low
Mackenzie	Yes (-) RNTR	Yes (+)* RNTR	• Zero BLTR limit 1995 • Full closure 2000	low	low	low	low	low	low
Moon	No BLTR	NA	Catch and release	med	med	low	high	med	low
Pinto	No RNTR	NA	Catch and release	low	med	med	low	med	med
Solomon	NA	No BKTR	Catch and release	low	low	low	low	low	low
Upper Erith	NA	No RNTR	Catch and release	low	low	low	low	high	high

result may have historically had the highest angling pressure of all the study watersheds. Therefore, it seems plausible that rainbow trout in this watershed may have responded most strongly to reductions in ongoing angling effort mediated by increased restrictions in angling regulations.

In Mackenzie Creek, the decrease in rainbow trout catch rate is more difficult to explain. One possible explanation is that the recent province-wide ban on bull trout harvest may have resulted in an increase in bull trout use of Mackenzie Creek (an important bull trout spawning stream), which accompanied increased predation by bull trout on rainbow trout in this watershed. Additional efforts to gain insights on the potential causal mechanisms producing these observations are warranted.

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Long-Term Effects of Riparian Harvest on Fish Habitat in Three Rocky Mountain Foothills Watersheds

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Abstract

In 2001, we completed a detailed fish habitat assessment in the Tri-Creeks experimental basin to quantify long-term changes in habitat structure as a result of clear-cut harvesting that occurred between 1979 and 1984. We compared channel morphology surveys from 1985 and 2001. The surveys were completed at six sites representing one of three different forest harvest treatments of i) minimal disturbance; ii) forty percent harvest with retention of riparian buffer strips; and iii) forty percent harvest without retention of riparian buffers. In 1985, no short-term changes to fish habitat elements were detected in response to the various treatments. Some researchers expected that establishment of trees and shrubs would replace the bank protection afforded by the roots of the harvested streamside-trees and that major changes in stream bank stability would not occur following harvest. We found that long-term effects of clear-cut harvest in riparian areas included channel widening and expansion of channel area. Stream channel width increased by 60 and 16 percent at the two riparian harvest sites. While channel widening was a local effect, the large volumes of sediment generated through expansion of channel cross-section area have the potential to affect the quality of fish habitat in downstream areas. These habitat impacts may have taken almost two decades to become detectable.

McCleary, R., Sherburne, C. and Bambrick, C. 2004. Long-term effects of riparian harvest on fish habitat in three Rocky Mountain Foothills watersheds. Pages 189-198 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

Introduction

Riparian harvest can result in the loss of structural elements, including streambanks and large woody debris (LWD), which can translate into a decrease in productivity of salmonid-bearing streams (Harmon et al. 1986, Keeley and Walters 1994). In the short-term, felling, skidding and removal of LWD from riparian areas can damage these elements (MacDonald et al. 1991). In the long-term, riparian harvest diminishes the future reservoir of LWD (McClure et al. 2004) and can also de-stabilize dynamic landforms including alluvial valley bottoms and alluvial fans (Nistor et al. 2001). While these views are widely held, there is surprisingly little empirical data quantifying long-term effects of riparian harvest on fish habitat, particularly from harvested sites where mechanical streambank damage and LWD removal were avoided during harvest. Such information can be difficult to obtain because the specific cause of long-term impacts resulting from riparian harvest may be difficult to discern from other forestry related impacts that may include: i) hydrologic changes associated with extensive clearing and ii) increases in sediment inputs conveyed by networks of roads (MacDonald et al. 1991).

In 1965, a major study was initiated within the Tri-Creeks experimental basin in west-central Alberta. The objective of the study was to determine if increased erosion as a result of extensive timber harvest, riparian logging, and scarification ultimately lead to the degradation of water quality and fish habitat (Bodnaruk 1987). The study area comprised a portion of the land base of Alberta's first forest management agreement, established in 1956. From the inception of this agreement, operating rules required retention of buffer strips along permanent streams during harvest (Bott et al. 2003). Despite this requirement, the effectiveness of this strategy for conserving the productivity of these streams remained a topic for debate. Foresters argued that given the inherent cold temperatures of streams within the region, productivity of the systems would increase with greater solar energy inputs (Bott et al. 2003). This question was addressed as part of the Tri-Creeks experimental study where treatments included clear-cut harvesting with and without buffer strips and minimally impacted control sites. Direct damage to the banks, channel and LWD was avoided during the riparian harvest treatment. While data collection was initiated a decade before treatment (i.e. 1966-1976), post-treatment data collection ended within two years after completion of the harvest treatments (i.e. 1984-1985) and findings were limited to short-term effects.

The Tri-Creeks researchers found that buffers provided a number of benefits including reduced changes to water temperature and possibly reduction of erosion and suspended sediment concentrations (Bodnaruk 1987). As a result of these findings, researchers recommended additional protective measures including: i) making buffers wider to prevent blow-down; ii) extending buffers beyond the permanent streams to include tributaries and gullies; iii) prohibiting scarification of these smaller watercourses and iv) emphasizing erosion control in roadway cuts (Bodnaruk 1987). After the Tri-Creeks study, the provincial operating ground rules continued to require buffer strips along permanent watercourses. In 2002, researchers within the Rocky Mountain foothills found that riparian areas are not protected from fire and that fire serves as an important disturbance event within these ecosystems (Andison and McCleary 2002). Given these recent developments, the effectiveness of buffer strips along streams was once again called into question by the industry and government agencies that negotiate the operating ground rules for commercial forest harvesting.

The habitat data from the Tri-Creeks study provided an opportunity to document longer-term effects of clear-cut harvesting within riparian areas on fish habitat. Our results were intended to support sustainable management of forest and aquatic resources by providing pertinent information to evaluate and if required, revise the current operational ground rules.

The objective of this study was to quantify long-term changes in fish habitat attributes, including channel cross-section characteristics, undercut banks and the preponderance of pools by comparing habitat conditions in 2001 with those documented in 1985 (1-6 years after riparian harvest) (Andres et al. 1987). Previous studies on short-term changes to stream bank and other habitat attributes were completed using data collected in 1985 (i.e. 1-6 years after harvest) and failed to detect changes in stream habitat (Andres et al. 1987). While some researchers expected that establishment of trees and shrubs would replace the bank protection afforded by the roots of the harvested streamside-trees and that major changes in stream bank stability would not occur following harvest (Andres et al. 1987), others noted that changes were likely to develop over a longer time frame than afforded by the initial study (Bodnaruk 1987).

Study Area

Tri-Creeks refers to Eunice, Deerlick and Wampus Creeks located in the Rocky Mountain Foothills approximately 40 km southeast of Hinton, Alberta. The three adjacent watersheds have similar topography, elevation ranges, geology, soils and vegetation. The topography is gently rolling with slopes ranging between 10 and 30 percent with all basins draining to the north and northeast. Elevations range 425 m, with a minimum elevation of 1,260 m and a maximum of 1,685 m. Shale and sandstone bedrock is covered by glacial till comprised of sandy loam, cobble and gravel. Below an elevation of 1,370 m, the till is overlain by an erodible lacustrine silt and clay deposit up to 8 m thick. Vegetation in Tri-Creeks generally changes with elevation zone. Black spruce occurs on the low wet lacustrine soils, lodgepole pine in the mid-elevations and white spruce in the higher elevations. Shrub meadows occur in higher elevation valley bottoms.

Wampus Creek is the largest watershed, followed by Eunice and Deerlick basins (Table 1). In 1985, the three watersheds included an non-harvested control (Eunice Creek), a clear-cut treatment with buffer strip retention (Wampus Creek) and a clear-cut treatment with no buffer strip retention (Deerlick Creek). Wampus and Deerlick basins were logged with an alternating clear-cut pattern until approximately 40 percent of the basins were harvested. In Wampus Creek, harvest was initiated in 1977 and completed in 1983. In Deerlick Creek, harvest was initiated in 1981 and completed in the spring of 1984. Since completion of the experiment, limited harvest has occurred in Eunice Creek watershed (Table 1).

Although several stream channel assessments were completed before and after the experimental treatment, only one survey could be duplicated with sufficient detail to allow accurate paired habitat comparisons. This survey, completed during 1984 and 1985, included channel cross-sections and channel profiles collected at two locations within each of the three watersheds including: i) a lower site located within 0.5 km of the stream mouth and ii) an upper site located approximately halfway between the stream mouth and source (Table 1). The channel profiles at the lower sites of Wampus, Deerlick and Eunice Creeks were 425, 375 and 190 m respectively, while the profiles at the middle sites were 180, 300 and 125 m respectively. There was no explanation provided for the variation in survey length among the sites, however the length was sufficient to include between three and seven pools. At each of the six locations, between 23 and 28 cross-sections were surveyed. The cross-sections were collected using standard rod and level surveying methods, and the surveyors also measured the undercut on both banks at each cross-section. In most cases, benchmarks (iron pins or iron spikes pounded into the base of a tree) were established at both cross-section end points, however a number of cross-sections were referenced only by a single benchmark. While the original survey data was never located, we found a report that included detailed graphical representations. For each site, three types of graphical information were available: i) a plan view including all cross-sections, benchmarks, the channel thalweg and segments of the left and right banks, ii) the longitudinal channel profile including the minimum bed level and water surface slope and iii) channel cross-section profiles including undercuts, benchmarks and water surface.

Methods

Channel classification

The sensitivity of a stream channel to change is

Table 1. Summary of basin and stream reach characteristics of the six survey sites from the Rocky Mountain Foothills, Alberta.

Site	Area (km²)	Bankfull Discharge (m³/s)	Summer Baseflow (m³/s)	Riparian Vegetation	Slope (%)	Harvest (%)	Riparian Harvest (km)
Upper Wampus	12.3	2.1	0.1	Shrub meadow	0.8	33	0
Lower Wampus	28.3	4.8	0.3	Spruce	1.6	40	0
Upper Deerlick	11.3	1.8	0.1	Spruce	1.5	38	1.7
Lower Deerlick	15.0	2.4	0.2	Spruce	1.3	42	4.5
Upper Eunice	6.9	0.8	0.1	Pine	1.0	17	0
Lower Eunice	16.3	1.9	0.2	Spruce	1.3	18	0

influenced by a complex set of site factors including valley confinement, gradient, bedrock control, riparian vegetation type and proximity to disturbance. Therefore, we classified the channels based on these attributes and where necessary, we divided individual study sites to meet uniform channel characteristics.

Channel cross-section geometry

From each paired cross-section (1984 and 2001), we calculated four variables to provide information on channel stability and fish habitat quality including: channel width, streambed elevation, cross-section area and streambank undercuts. Channel width was measured horizontally from the break in slope corresponding to the top of the vegetated bank. Increasing channel width is indicative of bank instability and decreasing amount and quality of fish cover (MacDonald et al. 1991). Change in streambed elevation was defined as the difference in elevation of the deepest point in the channel between the two dates. Increasing bed elevation, or in-filling of the channel, is indicative of an excess of sediment, while decreasing elevation can indicate bed scouring due to higher peak flows (MacDonald et al. 1991). For each cross-section, the difference in area between the two dates was calculated using WinXSPRO software (WEST Consultants, Inc. 1998). An increase in channel area occurs with scour of the channel, while a decrease in channel area occurs with channel in-filling. Both changes may be associated with a simplification or decrease in fish habitat quality (MacDonald et al. 1991). Streambank undercuts provide cover for fish and was measured as the distance (m) from the vertical line extending down from the furthest point of the bank protrusion to the furthest undercut of the bank (Platts et al. 1983). The undercuts from the left and right banks were summed to provide a measure of total undercut at each date for each cross-section.

From these cross-section parameters, we also calculated three erosion descriptors of: annual bank erosion rate, annual bed erosion rate and estimated net erosion. Annual erosion rates were calculated for each site by dividing the average changes in width and elevation by the number of years between the surveys. We estimated net erosion for each site for a uniform length of channel (100 m), by multiplying the average change in channel area by 100.

In 2001, using the plan view from each of the six 1984-1985 surveys, we searched for benchmarks and reconstructed each of the 23-28 cross-sections using a string tied between cross-section end points. Our approach to re-surveying individual cross-sections initially completed in 1984-1985 was likely confounded to a certain degree by a number of factors including: i) benchmarks were not labeled with a number corresponding to the cross-section number in the report, ii) due to frost-heaving, a number of bench marks appearing taller than in original surveys, iii) at Upper and Lower Deerlick extensive bank erosion resulting in the loss a number of the benchmarks and iv) at Upper Wampus site, beaver dam and lodge construction obscuring a number of benchmarks. Once all cross-sections were reconstructed to the best of our ability at each site, we used a total station and prism pole to re-survey all benchmarks, cross-sections and channel profiles. For our analyses, we confirmed cross-section pairs using two cross-referencing procedures. First, for each cross-section, we calculated the horizontal and vertical distances separating the two benchmarks and then determined the differences between the two survey dates. Second, we linked the coordinate files from each date through a common stable point and simultaneously viewed the graphical representations. Only those cross-sections that were confirmed to share the same location were used in the detailed analyses (91 out of 150 candidate pairs).

To obtain the 1985 data, we extracted distance-elevation information from the initial graphical descriptions of approximately 150 cross-sections among the six locations. For each graphical analysis, we determined the scale for both elevation and distance, and labeled each point used to create the cross-sections. For each point, we measured the elevation and distance to the nearest millimeter on the figure, then converted chart distances to real distances based on scale conversions. We expect that our estimations are associated with a measurement variance of 0.05 m for elevation and 0.1 m for horizontal distance.

Widths of undercut banks (cross-section view) were measured in 2001 to the nearest 1 cm with a metre stick. The historic channel cross-sections included undercut banks and we extracted undercut measures from the diagrams while we collected the other distance-elevation measures from each cross-section. We expect that our undercut estimations are associated with a potential measurement error of 0.1 m.

Pool depth and spacing

We identified residual pool depth and pool spacing as indicators of fish habitat quality that could be readily obtained from data collected in both the 1984-1985 and

Figure 1. Upstream view of an eroding bank on the outside meander bend at Cross-section 12, in Upper Deerlick Creek, October 2001. Note the bank erosion in the vicinity of the stumps and lateral deformation of roots as potential indicators of the 1985 bank location.

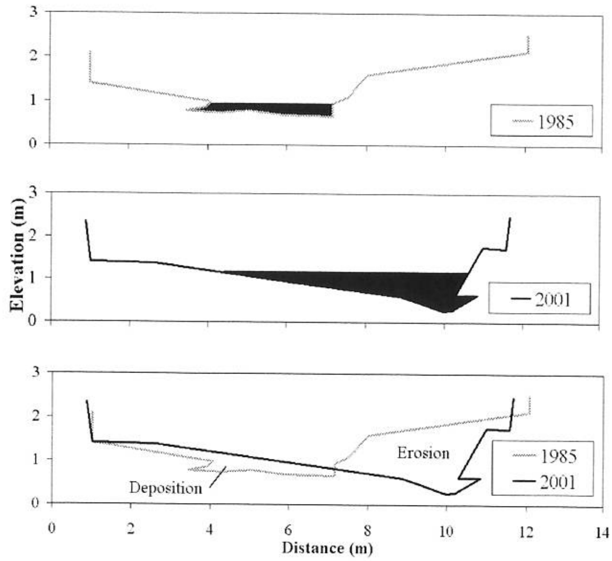

Figure 2. Upstream view of Cross-section 12 Creek, Upper Deerlick in 1985, 2001 and 1985-2001 pair. Shaded areas indicate the non-vegetated channel. Our analysis revealed a 2.6 m and 2.0 m² increase in channel width and cross-sectional area from 1985 to 2001.

2001 surveys. Residual pool depth (cm) was defined as the difference between the maximum pool depth and the riffle crest depth, or pool outlet depth. For analyses, we only included pools with a residual depth of 30 cm or greater. Pool spacing was measured as the distance separating like-features, including riffle crest or pool bottom, between successive pools (Rosgen 1996).

We extracted historic information on pool depth and pool spacing from the drafted profiles for each of the six locations. For each drafted diagram, we determined the scale for both elevation and distance, and labeled channel profiles with riffle crests, pool bottoms, and potential pool surfaces. We then measured residual pool depth and the distance between pools to the nearest millimeter on the figure and calculated real distances using the same methods as described for undercuts. We expect that our estimations are associated with a measurement variance of 0.05 m for elevation and 1.0 m for horizontal distance.

In 2001, we determined residual pool depth and distance between pools from the survey data. We imported digital easting, northing, and elevation data into a database and calculated distances between individual points along the longitudinal profile from the easting and northing data using the Pythagorean theorem. We calculated cumulative distances along the survey to produce a longitudinal profile for 2001. The profiles were labeled with riffle crests, pool bottoms, and potential pool surfaces. Residual pool depths and pool spacing were calculated from differences in appropriate distance and elevation data and recorded in a spreadsheet.

Data analysis

Candidate paired cross-sections included those with horizontal closure less than 0.5 m between benchmarks. Two-sample paired t-tests or one-sample t-tests ($\mu = 0$) were used to test for differences in channel width, undercuts, changes in bed elevation and changes in cross-section area between 1985 and 2001 (SPSS 1999). To test for differences in pool depth and pool spacing, we used a two sample t-test with individual measures pooled within each site. The detailed habitat survey results obtained in 2001 were compared with changes in land-use (Sherburne and McCleary 2003). Where data was available, results were also compared with trends in abundance of rainbow trout (*Oncorhynchus mykiss*) over the corresponding time period (McCleary et al. 2003).

Results and Discussion

Channel classification

Due to changes in site attributes, both the Upper and Lower Wampus sits were subsequently divided into two separate reaches. At the Upper Wampus site, a beaver dam with a 2 m crest elevation above the streambed was constructed and subsequently failed between the

Table 2. Mean (± 1 SD) change in bankfull channel width, streambed elevation, cross-section area and undercuts between 0-7 years after harvest (1984) and 17-24 years after harvest (2001) from paired cross-sections in eight locations in the Tri-Creeks experimental area, Alberta. P = Probability based on paired tests. Symbol abbreviations (streambed elevation and cross-section area): - denotes channel scour and a loss of channel material; + denotes infilling and the addition of channel material.

		Cross-Section Variable										
		Bankfull Channel Width (m)			Streambed Elevation (cm)		Cross-Section Area (m²)		Undercut (cm)			
Site	n	Mean 1984 Width	Change in 2001	P	Change in 2001	P	Change in 2001	P	Mean 1984 Undercut	Change in 2001	P	
Upper Wampus 1 (below beaver dam)	6	4.5	1.2 ± 2	0.19	+0.1 ± 0.2	0.27	-0.5 ± 0.7	0.18	27	-27 ± 28	0.06	
Upper Wampus 2 (previously flooded)	12	4.5	1.7 ± 1.2	0.00	+0.2 ± 0.2	0.01	-0.2 ± 1.4	0.58	20	-13 ± 29	0.16	
Lower Wampus 1 (alluvial fan)	6	8.4	0.1 ± 1.6	0.90	+0.2 ± 0.1	0.01	+2.3 ± 1.0	0.00	2	7 ± 8	0.09	
Lower Wampus 2 (confined channel)	9	7.1	-0.1 ± 0.9	0.67	-0.1 ± 0.1	0.04	-0.1 ± 0.4	0.48	4	13 ± 16	0.04	
Upper Deerlick[1]	13	3.5	2.1 ± 0.8	0.00	0.0 ± 0.3	0.75	-1.2 ± 2.1	0.05	45	-13 ± 38	0.24	
Lower Deerlick	15	4.4	0.6 ± 0.6	0.00	0.0 ± 0.1	0.58	-0.1 ± 0.7	0.49	7	18 ± 29	0.03	
Upper Eunice	12	3.1	0.4 ± 1.6	0.37	0.0 ± 0.2	0.73	-0.6 ± 0.8	0.03	23	5 ± 25	0.53	
Lower Eunice	18	4.1	0.2 ± 0.7	0.32	0.0 ± 0.1	0.26	-0.2 ± 0.4	0.08	9	23 ± 33	0.01	

[1] Original data collected in 1985

1984 and 2001 surveys. The dam was located near the 30 m mark and would have back-watered all upstream cross-sections. As a result, we divided the stream site into two reaches at the intersection of the relic beaver dam. Similarly, we divided the Lower Wampus site into two reaches at the 75 m mark where a confined canyon opened into an unconfined alluvial fan.

Channel cross-section geometry

Our analyses revealed significant changes in channel width ($P \leq 0.10$) at three of the locations of Upper Wampus 2, Upper Deerlick and Lower Deerlick (Table 2). Among these sites, annual bank erosion rate was greatest at Upper Deerlick, followed by Upper Wampus 2 and Lower Deerlick (Table 3, Figures 1 and 2). Responses at the riparian harvest sites in Deerlick Creek were consistent with other published findings where changes in channel geometry occurred as a result of loss of rooting strength from direct modification to riparian vegetation (e.g., Montgomery and Buffington 1993). Widening of the channel at Upper Wampus 2 may have occurred after dam failure from the loss of channel capacity due to sediment deposition and death of riparian vegetation, both which can occur with inundation. Channel widening has been shown to occur from sediment deposition and channel infilling (e.g., Montgomery and Buffington 1993). These findings illustrate that within the study area, riparian vegetation plays an important role in the maintenance of channel geometry.

We detected significant changes in bed elevation ($P \leq 0.10$) at three of the eight sites (Table 2). While channel infilling occurred at Upper Wampus 2 and Lower Wampus 1, channel scour occurred at Lower Wampus 2 and the greatest rate of annual change was observed at Lower Wampus 1 (Table 3). Alluvial fans are inherent zones of deposition, therefore additional analyses of bedload transport and deposition rates would be required to determine if upstream forest harvest has

Table 3. Annual stream bank and stream bed erosion rates and estimated net erosion 0-7 years after harvest (1984) and 17-24 years after harvest (2001) in six locations in the Tri-Creeks experimental area, Alberta. Symbol abbreviations (bed erosion rate and net erosion): - denotes channel scour and a loss of channel material; + denotes infilling and the addition of channel material.

	Erosion Variable		
Site	Annual Stream Bank Erosion Rate (cm/yr)	Annual Stream Bed Erosion Rate (cm/yr)	Estimated Net Erosion (m^3 / 100 m channel)
Upper Wampus 1 (below beaver dam)	7.3	+0.7	-46.7
Upper Wampus 2 (previously flooded)	10.1*	+0.9*	-22.5
Lower Wampus 1 (alluvial fan)	0.5	+1.2*	+233.3*
Lower Wampus 2 (confined channel)	-0.8	-0.4*	-8.9
Upper Deerlick[1]	13.4*	-0.2	-123.1*
Lower Deerlick	3.5*	-0.1	-13.3
Upper Eunice	2.5	+0.1	-56.7*
Lower Eunice	1.0	-0.2	-17.8*

[1] Original data collected in 1985
* Significant change in original cross-section variable at P < 0.10

affected Lower Wampus 1. The scour documented at Lower Wampus 2 may be similar to that described by other studies where channel incision occurred from increased run-off associated with clear-cut harvest (e.g., Montgomery and Buffington 1993). Our findings indicate that changes in bed elevation were relatively minor compared to changes in channel width.

Significant changes in cross-section area ($P \leq 0.10$) were detected at four locations (Table 2). Channel in-filling occurred at Lower Wampus 1, whereas channel area increased at the Upper Deerlick, Upper Eunice and Lower Eunice sites. While the estimated net erosion was highest at Lower Wampus 1 (Table 3), this change could not be attributed to harvest because deposition is an inherent natural process on alluvial fans, such as the one at this site. However, at Upper Deerlick, net erosion of 123 m^3 /100 m of stream occurred since 1985 (Table 3). Given that riparian harvest occurred adjacent to 4.5 km of stream channel within the basin (Table 1), erosion related to riparian harvest may be the dominant sediment source within the Deerlick Creek watershed. The lack of vegetation within the recently over-widened channel is likely to result in ongoing channel widening. In over-widened streams, the establishment of vegetation can initiate channel stabilization over a period of several decades (Friedman et al. 1996).

Changes in the width of undercut streambank (cross-section view) were detected at five of the eight sites (Table 2). At Upper Wampus 1, the average change was equivalent to a 100% decrease in undercut banks. At Lower Wampus 1, an increase in the width of undercut coincided with a statistically significant increase in bed elevation. At Lower Wampus 2, an increase in undercut width occurred along with a decrease in bed elevation. At Lower Deerlick, both undercut width and channel width increased. At this site, the root mat associated with the shrub and herbaceous riparian vegetation may have helped to retain the physical integrity of the upper bank to provide some cover during the channel widening process. At Lower Eunice, undercuts continued to develop without a corresponding increase in channel width. The increase in width of undercut banks at the Lower Eunice Creek site suggests that these habitat features likely evolve over a long period (i.e. multiple decades) that corresponds to the development of the forest stands. While undercuts typically provide a good indicator of the degree of bank protection afforded during land use (Platts et al. 1983), we observed undercut banks in both stable and actively eroding stream channels. These findings suggest that within the

Table 4. Comparison of mean (± 1 SD) residual pool depths and pool spacing 0-7 years after harvest (1984) and 17-24 years after harvest (2001) in six locations in the Tri-Creeks experimental area, Alberta.

	Habitat Variable					
	Residual pool depth (cm)			Pool Spacing (m)		
Site	1984	2001	P	1984	2001	P
Upper Wampus	65 ± 21	59 ± 13	0.62	49 ± 12	47 ± 20	0.90
Lower Wampus	30 ± 0	36 ± 3	0.01	109 ± 117	99 ± 120	0.92
Upper Deerlick[1]	45 ± 16	42 ± 8	0.72	46 ± 16	25 ± 8	0.22
Lower Deerlick	56 ± 16	56 ± 16	1.00	48 ± 16	55 ± 16	0.79
Upper Eunice	37 ± 7	38 ± 8	0.84	38 ± 7	36 ± 8	0.88
Lower Eunice	36 ± 9	35 ± 3	0.89	36 ± 3	23 ± 3	0.26

[1] Original data collected in 1985.

study area, the development of undercut banks is likely a complex process influenced by a number of factors. Therefore, evaluations of the effects of land use on fish habitat should include measures of channel width and bank stability, in addition to streambank undercuts.

Pool attributes

Due to the low number of pools, Upper and Lower Wampus were not divided into separate reaches for analyses. We detected a significant change in pool attributes at only one location – Lower Wampus Creek and this change comprised a significant increase in mean residual pool depth (Table 4). Changes in pool spacing were not detected; these results are not consistent with other studies that report the decrease in frequency of pools and pool size after riparian harvest (e.g., Sedell et al. 1988). Previous studies have found that LWD generated during riparian harvest may persist two decades (McClure 2004). This quantity of LWD eventually declines and lack of recruitment will result in a decrease in the functioning LWD pool (McClure 2004). The lack of a response in Deerlick Creek is difficult to explain, especially given the sediment release from the channel widening and potential for pool infilling. The increase in residual pool depth in Lower Wampus was also unexpected and difficult to explain, but is unlikely a measurement error since the magnitude of the detected change was similar to our measurement accuracy. Further studies, perhaps based on a more spatially and temporally intensive sampling design, are required to better understand dynamics of pool structure. Within the study area, ongoing fluvial processes may be able to maintain pool depth and spacing, even with changes in sediment load and channel cross-section geometry. In addition to other efforts, a follow-up study is needed to evaluate the sensitivity of pool volume as an indicator of land use impacts on fish habitat.

Relationships between changes in habitat parameters, changes in land-use and trends in rainbow trout abundance

While land use change in the study area was limited to a moderate increase in harvest within Eunice Creek (Table 5), changes in fish habitat attributes were detected at all study sites in the Tri-Creeks region. Habitat changes in Wampus Creek were associated with beaver activity and ongoing land forming processes, and additional investigations are required to establish land use relationships. At Lower Wampus, the decrease in abundance of rainbow trout (Table 5) warrants such studies. Habitat changes in Deerlick Creek coincided with riparian harvesting in 1981-1984. At Lower Deerlick, changes in habitat did not coincide with a decline in rainbow trout abundance. The magnitude of changes was greatest in Upper Deerlick, unfortunately recent estimates of fish abundance are not available at this site. Such information could improve our understanding of the relative importance of fish habitat, angling and climatic variables in explaining the dynamics of rainbow trout populations. With the initiation of harvest within Eunice Creek, Tri-Creeks no longer has a true control. However, given the varying harvest magnitudes among Eunice, Wampus and Deerlick Creeks, the study area offers opportunities for additional follow-up studies.

Table 5. Summary of changes in harvest extent, road density and rainbow trout abundance at six locations within the Tri-Creeks study area. Harvest Information: < 10% = low, 10-30% = medium, > 30% = high (Sherburne and McCleary 2003). Index of Road Density: ≤ 0.2 km/km^2 = low, 0.3-0.4 km/km^2 = medium, ≥ 0.5 km/km^2 = high (Sherburne and McCleary 2003). Detectable change in rainbow trout abundance at $P < 0.10$ (McCleary et al. 2003).

	Harvest information			Index of road density			
Site	**Historic % harvested**	**Current % harvested**	**Change**	**Historic**	**Current**	**Change**	**Detectable change in rainbow trout abundance**
Upper Wampus	high	high	low	high	low	med	not detected
Lower Wampus							decrease
Upper Deerlick	high	high	low	med	low	low	no data
Lower Deerlick							not detected
Upper Eunice	low	med	med	low	med	low	no data
Lower Eunice							no data

Conclusions

While previous studies focused on changes in water quality following harvest (Andres et al. 1987), our findings indicate that actual long-term effects can differ from forecasted long-term effects. Given that managing for unanticipated long-term effects presents a major challenge for current adaptive forest management frameworks, there may be important benefits to following up historical studies to provide insights on long-term impacts. Researchers currently engaged in short-term studies should consider that the quality of follow-up studies would depend upon their initial study design and data archiving.

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Effects of Streamside Forest Harvesting on Stream Temperatures in the Central Interior of British Columbia: The Moderating Influence of Groundwater and Lakes

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Extended Abstract

Interactions between forestry practices and the physical and biological processes occurring in small streams are numerous and potentially complex. Water temperature is one of the most important factors regulating biological processes in small streams, and it is often a key consideration when planning timber harvesting activities around streams. Evidence from the literature is overwhelming in supporting the conclusion that streamside timber harvesting results in increased summer stream temperatures, whether the magnitude of the increase is large or small. As a result, there is often a conflict between fisheries managers wishing to protect stream fish populations on the one hand, and foresters wishing to harvest the most valuable timber that is typically found within stream riparian zones. Although the future timber supply in the northern hemisphere is expected to come from boreal and sub-boreal forests, little research has been conducted in these regions that examines the temperature responses of small streams to streamside clear-cut logging. Furthermore, studies investigating the impacts of forestry activities on water temperatures tend to focus on headwater streams, with little attention being paid to streams headed by lakes. We present data from a field study conducted in the central interior of British Columbia, Canada, a region characterized by a temperate sub-boreal climate,

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that examined the temperature patterns of two small lake-headed streams and their responses to clear-cut logging. In the first three years following timber harvesting, we found only modest changes (averaging 0.05 - 1.1 °C) with respect to summer daily maximum and minimum temperatures, diurnal fluctuations, and stream cooling, compared to increases of ~5 - 7 °C that have been reported in the literature for headwater streams. A comparative survey also revealed that downstream cooling was most commonly observed in streams headed by lentic waterbodies (lakes and swamps), regardless of whether or not logging had taken place within the stream riparian zones, and that this cooling was attributed largely to the presence of the lakes and swamps, as well as to groundwater inputs. By comparison, headwater streams generally exhibited downstream warming (again regardless of whether their riparian zones were unharvested or logged). Our data lead us to conclude that the presence of lentic waterbodies may therefore moderate stream temperature responses to streamside clear-cut logging. A multiple regression model was developed to predict summertime downstream cooling or warming in headwater and lake-outlet streams using easily measured predictor variables (maximum upstream temperatures and canopy cover). Lastly, we examined the fisheries implications of increased post-logging stream temperatures on juvenile stream-dwelling rainbow trout (*Oncorhynchus mykiss*), and found that their growth was enhanced (and the emergence of trout fry accelerated) in the two logged streams when compared to a cold, unharvested control stream.

Quantifying the Distribution and Relative Abundance of Stream Fish Communities in Alberta: The Cooperative Fisheries Inventory Program

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Extended Abstract

Established in 1994, the Cooperative Fisheries Inventory Program (CFIP) is a comprehensive, cooperative venture to describe the distribution and relative abundance of stream fish communities throughout much of Alberta's forested regions. The program has been, and continues to be, funded by divisions of Local, Provincial and Federal Governments, and companies within the Energy and Forestry sectors. Based on this multi-stakeholder approach, the objectives of the program are to contribute to the stewardship and management of stream fish resources by quantifying: i) the distribution and relative abundance of stream fish and ii) characterizing select stream reach habitats. These data provide resource managers information on which to evaluate potential effects of industrial activity on stream fish communities, and are used to assess species status according to Species At Risk guidelines.

The program is applied using the four guiding principles of: i) the application of standardized and cost effective fish and fish habitat sampling methods, ii) integration of data into leading-edge geographic information systems platforms, iii) continued use of the resulting information by resource managers within government, industry, research and conservation organizations, iv) the philosophy that the responsibility to collect baseline information on stream fish

Osokin, L., Fitzsimmons, K.M., Gardiner, K.G., Johnson, C. and Hvenegaard, P.J. 2004. Quantifying the distribution and relative abundance of stream fish communities in Alberta: the cooperative fisheries inventory program. Pages 201-202 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

communities is shared by many stakeholder groups including multiple levels of government, industry and research and conservation organizations.

After the successful completion of nine years of programming, information gained through the CFIP continues to be a critical source of information used by fisheries managers, land-use planners, non-government organizations, and industry partners for landbase planning and completion of environmental scoping exercises. The resulting information has also heightened the public's awareness of watershed management issues across the landscape.

This program augments the provincial Fisheries Management Information System (FMIS) database pertaining to fish and fish habitat from priority areas within the Northwest, East Slopes, and Southern regions of Alberta. To date, the CFIP has completed stream fish inventories within select sub-basins of the major watersheds located within Alberta's Aspen Parkland, Montane, Subalpine, Alpine, Boreal Cordilleran and Boreal Mixed wood ecoregions. Analyses of some of the data collected through the CFIP suggests that stream fish communities within Alberta forested ecosystems differ among ecoregions and can be predicted using a suite of stream habitat, watershed and landscape-level characteristics.

To date investments of $2,800,000 contributed by 19 stakeholder organizations have provided baseline information of stream fish communities at > 4,000 stream sites. We expect that information gathered through the CFIP will continue to provide resource managers with important baseline data that can be used to manage Alberta's natural resources.

The Environmental Effects Monitoring Program: An Overview of the Adult Fish Survey

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Extended Abstract

The objective of the Environmental Effects Monitoring (EEM) program is to evaluate the effects of human activity on fish, fish habitat and the use of fisheries resources. This is done through periodic monitoring of biota (fish and benthic invertebrate surveys), supporting environmental variables, and sublethal toxicity tests. Currently, the metal mining and pulp and paper industries conduct EEM studies.

The EEM provision of the Metal Mining Effluent Regulations (MMER) came into effect December 2002. The 23 metal mines in our administrative region are required to complete effluent characterization and water quality monitoring (four times annually), sublethal toxicity testing (twice annually) and biological monitoring. The frequency of the biological monitoring studies varies from 2 to 6 years depending on the results of the previous phase. EEM has also been required under the Pulp and Paper Effluent Regulations (PPER) since 1992. Ten mills in our administrative region perform sublethal toxicity tests on their effluent twice yearly and conduct biological monitoring once every cycle (4 years).

The endpoints measured in the adult fish survey component of the EEM program reflect energy use, energy storage and age, and include age, condition, liver weight and gonad weight. An effect in this program is defined as a statistically significant difference between reference and exposure area endpoints. Facilities must also include a statistical power analysis in order to verify that sample size was sufficient for the detection of an effect.

Nationally, species used as sentinel organisms have shifted over time from a reliance on large bodied species to smaller bodied monitoring organisms. This was primarily a response to concerns about fish mobility, exposure to effluent and harvest pressure on populations of larger bodied fish. Pulp and Paper facilities in the Prairie and Northern Region have included small bodied fish in their monitoring programs with increasing frequency, and success. For example, spoonhead sculpin (*Cottus ricei*), trout-perch (*Percopsis omiscomaycus*), longnose

Siwik, P. 2004. The Environmental Effects Monitoring Program: an overview of the adult fish survey. Pages 203-204 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land - Fish II Conference – Ecosystem Stewardship through Collaboration. Proc. Forest Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

dace (*Rhinichthys cataractae*) and lake chub (*Couesius plumbeus*) were some of the species used in Cycle 1 and 2. The Pulp and Paper industry submitted their Cycle 3 Interpretative Data Reports on April 1, 2004, so that data will not be included on this poster.

The paucity of basic biological information for some small bodied species has resulted in some debate over their suitability. Lack of information on spawning, diet and mobility are often cited as a concern by facilities and consultants. This poster will summarize the program requirements for the EEM adult fish survey, present some of the Cycle 1 and 2 results, and outline some of the data gaps and research questions related to small bodied fish that are pertinent to our region.

Winter Dissolved Oxygen Monitoring in a Small Aerated Lake Stocked With Rainbow Trout

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Extended Abstract

Lakes in the Edmonton area receive excessive fishing pressure due to the large number of anglers targeting a restricted number of recreational fisheries. Local non-profit fishing organizations are working in co-operation with regional and provincial government agencies to develop additional recreational fisheries in waterbodies which currently do not support viable sport fish populations.

The consortium investigated several small "pothole lakes" west of Edmonton, in order to locate a suitable candidate for development of a quality rainbow trout fishery (i.e. application of a special regulation to allow catch of large trout). Muir Lake was selected, although previous attempts at creating a viable trout fishery in the lake (beginning in 1956) have failed due to frequent, and occasionally severe, winterkills. Lake aeration has proven to be effective in reducing the frequency and severity of winterkill in several lakes in the Peace Region and eastern slopes of Alberta. It was hoped that installation of mechanical aeration units in Muir Lake would provide similar benefits.

Muir Lake is a small (0.29 km^2), relatively shallow (76.4 % of total lake is < 3 m deep) waterbody, with a drainage area of 2.36 km^2, and currently no active inlet or outlet.

Approximately 14000 rainbow trout fingerlings were stocked in May 2003; two mechanical aeration units were installed in the lake in November 2003. Golder Associates Ltd. volunteered the services of fisheries staff to monitor the effectiveness of the aeration units in improving dissolved oxygen conditions in the lake. One of the objectives during the under ice period is to determine the vertical and horizontal dispersal of dissolved oxygen throughout the lake, particularly into bays potentially isolated by either distance from the aeration units or basin configuration. A secondary objective is to track the survival and growth of the 2003 and subsequent stocked cohorts.

Monthly sampling of dissolved oxygen and water temperature has been conducted since the

Stefura, C. 2004. Winter dissolved oxygen monitoring in a small aerated lake stocked with rainbow trout. Pages 205-206 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest-Land Fish II Conference – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf., April 26-28, 2004, Edmonton, Alberta.

beginning of May 2003. Preliminary analysis of the dissolved oxygen levels measured in December 2003 indicated a wide dispersal of dissolved oxygen radiating from the aerators. Field measurements in early March 2004 indicated oxygen levels exceeding 5.0 mg/L in the upper 2.0 m of the water column (excluding ice cover). Dissolved oxygen levels recorded during the same time period in Chickakoo Lake, a non-aerated lake approximately 8 km west of Muir Lake, were less than 0.5 mg/L in each of the basins measured.

The preliminary findings suggest that the aeration efforts have been successful in providing dissolved oxygen conditions in Muir Lake favorable to the survival of the stocked rainbow trout cohort. Additional field studies will be required to determine the overwinter survival and growth of the 2003 stocked cohort.

Managing Fish and Aquatics Data Using the ArcHydro Data Model

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Extended Abstract

The Foothills Model Forest (FMF) is a non-profit partnership dedicated to providing practical solutions for stewardship and sustainability on Alberta forest lands. As part of its focus, the Fish and Watershed Program and its partner agencies have collected a large amount of data that was historically stored in spreadsheets and database systems. The Fish and Watershed program and its partner agencies have traditionally stored field-collected data in ad-hoc, non-spatial spreadsheet and database systems which has resulted in problems with data correctness, data duplication and inefficiencies in summarizing, reporting and mapping inventory data. To better consolidate all field-collected data and to enforce correct spatial placement of field sampling locations, a database was designed using traditional entity-relationship techniques and the spatial data model template called ArcHydro (developed in the United States) to tie together spatial and non-spatial features in a single database management system. The resulting database functionality includes data entry in Microsoft Access, fine-scale mapping, analysis and querying capabilities in Environmental Systems Research Institutes (ESRI's) ArcMap desktop GIS system.

The FMF has been completing fish inventory studies in the west central foothills regions of Alberta for the past seven years and currently manages data from more than 1300 surveys on the FMF landbase. These data include the locations of survey sites, attributes of riparian and instream habitats, and fish community structure including fish presence and abundance and individual fish lengths and weights. These data were stored in a Microsoft Access database and in accompanying Microsoft Excel files. The Microsoft Access data entry and reporting components of this system performed well initially, but the database design did not enable expanding collection needs, did not constrain field values to ensure data were logical, and did not effectively capture the spatial dimension of the data. Over time this tool resulted in several chronic and typical data management problems including: i) data errors due to the

Weik, C.R. 2004. Managing fish and aquatics data using the ArcHydro data model. Pages 207-208 in G.J. Scrimgeour, G. Eisler, B. McCulloch, U. Silins and M. Monita. Editors. Forest Land–Fish Conference II – Ecosystem Stewardship through Collaboration. Proc. Forest-Land-Fish Conf. II, April 26-28, 2004, Edmonton, Alberta.

lack of logistical constraints; ii) survey locations not falling on hydrographic features due to differing accuracies in capture methods and the inability to place points using GPS; iii) significant time demands to enter spatial reference information (e.g., management units, watersheds, stream name and stream tributary); iv) inability of end users to build queries due to poor database design; v) inability or difficulty to effectively map in-stream single point or linear sections representing surveyed areas; and vi) an inability or difficulty to create maps describing fish species presence and absence. The problems related to spatial reference were particularly apparent given the complex nature of stream network systems. My poster presentation will describe the main objectives of developing a new database management system, its ability to solve the historical data management challenges and its overall applicability of the ArcHydro spatial data model template for the FMF landbase.

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