Monitoring salmon habitat in small streams using streambed profiling and the importance of large woody debris for juvenile chinook salmon habitat in small Yukon streams

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Abstract

In the Yukon Territory, monitoring, protecting, and restoring salmon habitat in streams affected by placer gold mining are critical activities because salmon fisheries and placer gold mining have important economic and social values. Placer mining refers to the recovery of fine gold from gravel deposits associated with streams. To provide information to help develop new placer mining guidelines, I conducted a field study to 1) evaluate a new tool to monitor fish habitat in small streams, including streams affected by placer mining, and 2) determine the importance of large woody debris (LWD) in providing fish habitat in small, boreal forest streams.

Thalweg profiles are longitudinal profiles of streambed elevation measured along the deepest portion of a stream. This technique has recently been suggested as a monitoring tool for fish habitat. For sections of 14 small, undisturbed Yukon streams, I examined whether thalweg profiles provided useful information on fish habitat and whether a corresponding relation between fish abundance and metrics calculated from thalweg profiles was present. Thalweg metrics, which reflect features such as pools, provided useful information on juvenile chinook salmon (*Oncorhynchus tshawytscha*) habitat and chinook salmon density was correlated with thalweg metrics. Thalweg metrics were higher in these undisturbed streams compared to metrics from two streams affected by placer gold mining. These results suggest that thalweg profiling provides a useful tool to monitor fish habitat and can further be used to monitor the recovery of fish habitat in placer-mined streams.

LWD was abundant and formed 28% of the pools in these streams. Pools provide important habitats for juvenile chinook salmon. Chinook salmon density was also correlated with LWD abundance. My findings suggest that LWD performs a similar function in creating fish habitat as shown in more southerly latitudes. To better protect fish habitat, new placer mining guidelines should aim to 1) protect riparian forests in order to maintain sources of LWD to the stream, and 2) restore LWD in streams affected by placer mining using the abundance and characteristics of LWD in these undisturbed streams as a guide.

Dedication

To my colleagues, family and friends who have helped me throughout my academic career.

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Introduction

A better understanding of fish habitat in streams is important to help protect fisheries resources in the face of increasing land-use pressures. In the Yukon Territory, monitoring, protecting, and restoring salmon habitat in streams affected by placer gold mining are critical activities. Placer gold mining, which refers to the recovery of fine gold from gravel deposits associated with streams, occurs on many Yukon streams, and its large effects on streams can potentially have detrimental effects on salmon fisheries. Both mining and fishing generate important economic and social values in the area. An unavoidable consequence of placer mining, as currently practiced, is the disruption of nearby terrestrial and aquatic habitats. Diverting streams to access placer deposits, discharging sediment-laden effluent to streams, and disrupting the stream channel during mining are examples of activities that might affect fish habitat. These impacts can negatively affect streams that provide important rearing habitat for juvenile chinook salmon (*Oncorhynchus tshawytscha*) and other fishes (LaPerriere and Reynolds 1997).

Placer mining is currently regulated under the Yukon Placer Authorization (YPA 1993). To better protect fish and fish habitat, the Department of Fisheries and Oceans (DFO) plans to replace the YPA and phase in new mining regulations. This new policy represents a shift from regulating the Yukon placer mining industry as a whole to site-specific appraisals. As part of this shift, DFO will develop new guidelines and standards to help placer miners comply with the federal Fisheries Act of 1977.

This field study of small streams in the upper Yukon River basin has two components, each included as an individual, stand-alone chapter. The first evaluates the use of thalweg profiling as a tool for monitoring fish habitat in small streams. My objective was to determine whether thalweg profiles provide useful information on fish habitat and whether there is a corresponding relation between juvenile chinook salmon abundance and metrics calculated from the thalweg profiles. The second component of this research investigates the role of large woody debris (LWD) in providing important fish habitat in small boreal forest streams. My objective was to describe the abundance, characteristics, and function of LWD in these Yukon streams. The concluding section links both chapters together and provides specific management recommendations for the development of new placer mining guidelines.

DFO's Policy for the Management of Fish Habitat (DFO 1986) has a guiding principle of 'no net loss of the productive capacity of fish habitat', and strives to balance unavoidable habitat losses with equivalent habitat replacement or compensation. Given the habitat losses associated with placer mining, stream restoration channels (i.e., channels built after mining) have been and will likely remain an important management tool for placer mining. To date approximately 500 to 1000 restoration channels have been built at Yukon placer mines under existing mining regulations (A. von Finster, DFO Whitehorse, personal communication). However, there has been little follow-up monitoring as to the success of these channels. What little data there are suggest that monitored channels are ineffective as they were often unstable, built too small to handle high flows, and lacked the required fish habitat structures (Chilibeck 1993). These results indicate that current restoration practices are inadequate. Fisheries managers need a better understanding of chinook salmon habitat to help guide channel construction, as well as improved methods to monitor the effectiveness of such channels. Both the thalweg profiling and LWD components of this research project address these two information gaps.

Results of this research will also help to manage streams outside of the Yukon. In the Pacific Northwest, for example, large investments are made in stream habitat restoration and better methods are needed to monitor changes in fish habitat and evaluate the benefits of this work (Roni et al. 2002). Thalweg profiling has recently been advocated as a new tool to monitor habitat in small streams and this research addresses some critical questions related to its development. Similarly, relatively little is known about the role of LWD in boreal forest streams (Clarke et al. 1998) despite increasing land-use pressures in boreal forests (Schindler 1998). Information on LWD in Yukon streams will increase our understanding of LWD in boreal forest streams in general and assist habitat managers to protect and restore fish habitat.

While this research focuses on managing fish habitat in placer-mined streams, the management implications are also relevant to other land uses in Northern Canada such as forestry or oil and gas exploration. For example, forest harvest in the southern portion of the Yukon is increasing (Anonymous 2000) and can potentially affect fish habitat. Similarly, a proposed gas pipeline along the Alaska Highway would involve numerous stream-crossings, which could also adversely affect fish habitat.

Chapter 1
Thalweg profiling

Abstract

Thalweg profiles are longitudinal profiles of the streambed elevation measured along the deepest portion of the stream. This technique has recently been advocated as a tool to monitor fish habitat in streams because profiles can provide useful information on such habitat, and measurements are both repeatable and independent of stream flow. However, a relation between fish abundance and metrics calculated from thalweg profiles has not been established. To test this relation, I surveyed thalweg profiles and sampled juvenile chinook salmon (*Oncorhynchus tshawytscha*) density in 14 reaches of small, undisturbed streams in the Yukon Territory. Chinook salmon density was correlated with three thalweg metrics. Two of these metrics—length in residual pools and mean residual pool depth—provided useful information on chinook salmon habitat. Thalweg metrics differed between these undisturbed streams and reaches in streams affected by placer gold mining. These results suggest that thalweg profiling provides a useful tool to monitor fish habitat in small streams.

Introduction

Methods to assess and monitor fish habitat in streams are important for managing fish populations because it is generally easier to measure habitat variables than it is to estimate fish abundance. Furthermore, abundance can vary in response to factors unrelated to habitat. Interest in predicting fish yield from streams is evident by the number and diversity of existing habitat models (see reviews in Fausch et al. 1988; Shirvell 1989) and corresponding habitat assessment techniques (reviewed in Bain and Stevenson 1999). Such models are used to measure habitat change, assess environmental impacts, predict the benefits of enhancement projects, and determine optimal flows in regulated rivers.

A fundamental problem in monitoring fish habitat is the lack of standardized, systematic approaches to collect data (Bauer and Ralph 2001). For example, habitat models that predict standing crop in streams (e.g., number or biomass of fish per unit of stream area) using physical habitat variables such as wetted area, substrate, or cover can be useful for predicting changes to fish habitat and fish populations. However, these models often use physical habitat variables that are dependent on stream flow (i.e., pool size and depth) or variables that are difficult to define objectively (i.e., cover, boundaries between pools, glides and riffles). Subsequently, these variables are difficult to measure consistently, thereby limiting the utility of such models for management (Poole et al. 1997), especially for long-term monitoring.

Thalweg profiling has recently been advocated as a useful tool to assess and monitor fish habitat in small streams (e.g., Bauer and Ralph 2001). Thalweg profiling involves surveying the streambed elevation along the deepest portion of the stream (the thalweg) to produce a two-dimensional, longitudinal profile of streambed elevations. Depressions in the profile represent pools or deeper habitats with low current velocity during low flow periods, while crests in the profile represent riffles. Thalweg profiling uses standard survey techniques and thus the measurements should be repeatable with high precision (i.e., within and among survey crews). Furthermore, the profile is independent of stream flow because only the streambed is measured.

Thalweg profiling has long been used to study stream channel morphology (e.g., Richards 1976). More recent studies on fish habitat and channel morphology have suggested calculating metrics from the profile that would provide useful indices of fish habitat (Lisle 1995; Madej 1999a). For example, Madej (1999b) suggested using an index of morphological diversity to monitor channel conditions based on the assumption that a more variable morphology reflects more complex aquatic habitat and, therefore, better fish habitat. However, the relation between fish populations and such thalweg metrics has not been examined and, therefore, the utility of the thalweg profiling for assessing and monitoring fish habitat has yet to be tested.

This study evaluates thalweg profiling as a tool that could be used to monitor fish habitat over time in small streams, by first determining whether there is a relation between the abundance of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and thalweg metrics in small, undisturbed streams in the Yukon. I then compare thalweg metrics from these undisturbed streams with streams that have previously been placer mined for gold to determine whether thalweg metrics reflect the impacts of land use on channel morphology. These Yukon streams are well suited to test for a relation between juvenile chinook salmon abundance and thalweg metrics because, as explained in the following section, juvenile chinook salmon abundance seems limited by physical habitat, a fundamental assumption of habitat models (Fausch et al. 1988).

Materials and Methods

Study area and habitat conditions

I examined 12 relatively undisturbed and two previously placer-mined small streams in the Yukon. Placer mining refers to the recovery of fine gold from alluvial deposits and involves major disturbances to the valley bottom and stream channel (LaPerriere and Reynolds 1997). Streams were grouped into three sub-regions: Whitehorse, Minto and Dawson City (Fig. 1.1). The topography of the stream valleys differs among sub-regions. The Dawson sub-region was not glaciated during the last glacial maxima of 40 000 to 14 000 years BP (Oswald and Senyk 1977); consequently, the stream valleys are more steeply incised. In contrast, the stream valleys near Minto and Whitehorse were previously glaciated and, as such, exhibit a more gentle topography. Croucher Creek near Whitehorse has incised itself into sediments deposited by ancient glacial lakes and rivers.

All streams are located in the boreal cordillera eco-zone (Wiken 1986) and boreal forest dominated by white spruce (*Picea glauca* (Moench) Voss) covers the valley bottoms. While climate varies among sub-regions, all three have long cold winters and brief summers. Whitehorse has a continental climate and receives approximately 270 mm of precipitation annually while air temperature ranges from summer highs of >20°C to winter lows less than -40°C. Minto and Dawson receive approximately 247 and 325 mm of annual precipitation, respectively (Oswald and Senyk 1977). Compared to Whitehorse, Minto and Dawson have shorter summers and colder winter temperatures. Discontinuous permafrost is found throughout the region (Oswald and Senyk 1977). Smaller streams may freeze solid and can be covered with 1 to 3 m of ice during the winter. Approximately 40% of the precipitation falls from June to August in all sub-regions. Therefore, discharge in small streams fluctuates following summer rainstorms.

Streams that I surveyed are direct tributaries to the Yukon River mainstem with the exception of the two placer-mined streams that flow into the Klondike River, a large tributary of the Yukon River. Reaches had pool-riffle or plane-bed morphology, as described by the system of Montgomery and Buffington (1998) (Table 1.1). All valleys were alluvial and some of the plane-bed streams had areas with exposed bedrock.

Such small streams provide non-natal (i.e., non-spawning) rearing habitat for juvenile chinook salmon prior to their downstream migration to the Bering Sea. Juvenile chinook salmon colonize small streams up to several km upstream of the stream's confluence with the Yukon River (Bradford et al. 2001), provided that there are no barriers to fish passage. Some surveyed streams contain juvenile chinook salmon exclusively, although others contain arctic grayling (*Thymallus arcticus*), slimy sculpin (*Cottus cognatus*), longnose sucker (*Catostomus catostomus*), lake chub (*Couesius plumbeus*), and juvenile northern pike (*Esox lucius*) (Bradford et al. 2001). Non-native rainbow trout (*O. mykiss*) are present in Croucher Creek (Moodie et al. 2000). In August, most juvenile chinook salmon in these streams are age 0 (mean fork length = 65 mm), although a few age 1 chinook salmon may also be present (Bradford et al. 2001).

Juvenile chinook salmon in these streams have specific habitat preferences. Chinook salmon densities (fish/m^2) are highest in pools and deeper habitats, yet are generally less abundant or absent in habitat units less than 25 cm deep (i.e., individual pools, glides, and riffles; Bradford et al. 2001). Therefore, I assumed that, for a given length of stream, more juvenile chinook salmon colonized streams with better quality habitat.

Although the undisturbed watersheds appear pristine, with the exception of sparse road developments or small homesteads, many watersheds were in fact logged for cordwood to fuel steamboats that navigated the Yukon River during the 1890s to 1950s. Steamboats switched from burning wood to coal in the latter part of this period. No detailed records are available for the exact timing and extent of such logging or the fire history for individual watersheds. Valley bottoms of the two mined streams (Hunker and Bonanza Creeks) have been placer-mined repeatedly throughout much of their length since the Klondike gold rush of the 1890s. Streams were diverted while the underlying gold-bearing gravels were mined and mining techniques ranged from mining by hand to the use of huge dredges to extract and process these deposits. Placer mines were active in both watersheds when I surveyed them.

Field data

I surveyed thalweg profiles, measured fish habitat and sampled juvenile chinook salmon densities from one reach[1] on 10 of the 12 recently undisturbed (i.e., not previously placer-mined) streams, and two separate reaches on each of the remaining two undisturbed streams, for a total of 14 surveyed reaches. All reaches were at least 19 bankfull channel widths long (84 to 190 m; Table 1.1). Reaches were surveyed near the stream's confluence with the Yukon River, except for the upper reach on McCabe Creek (~3 km upstream of the Yukon River), and both reaches on Croucher Creek (~1.0 and 1.5 km upstream of the Yukon River). Fish habitat and thalweg surveys took place from late June to early August 2001. Chinook salmon densities were sampled in August over several years as described below.

Data collected from two previously placer-mined streams were used to compare thalweg metrics in mined and undisturbed streams. I surveyed Bonanza and Hunker Creeks roughly 9 and 14 km, respectively, upstream from their confluence with the Klondike River. I did not measure habitat or sample fish in mined reaches because a direct comparison of chinook salmon densities between mined and undisturbed reaches was not feasible. The abundance of juvenile chinook salmon in the Klondike River is lower than in the Yukon River mainstem (M. Bradford, DFO, unpublished data). Consequently, there were fewer chinook salmon to colonize these mined streams compared to the undisturbed streams. Differences in chinook salmon density between mined and undisturbed streams may, therefore, be attributed to factors unrelated to habitat. In addition, high sediment loads discharged from active placer mines likely impact chinook salmon density in mined reaches.

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[1] I use the simplest definition of the term 'stream reach' to refer to a specified length of stream (Armantrout 1998). The two surveyed reaches on Croucher Creek could also be described as two separate sections of the same 'reach' or homogeneous stretch of stream.

Chinook salmon densities

I sampled juvenile chinook salmon densities in each reach during August 2001 and 2002. All but two reaches were also sampled in August between 1998 and 2000 as part of a previous study (M. Bradford, DFO, unpublished data). Ten of these reaches were sampled twice between 1998 and 2000 while two reaches were sampled once.

Reaches were sampled using either multiple-pass depletion electrofishing or mark-recapture methods; the latter were similar to those described in Bradford et al. (2001). Chinook salmon were first captured using minnow traps baited with salmon roe held in perforated plastic bags. Minnow traps were placed every 10 to 20 m along the reach, then wetted for roughly 24 hours. Captured chinook salmon were marked on the caudal fin with a Pan-Jet inoculator and Alcian Blue dye (Thedinga and Johnson 1995), allowed to recover in buckets, and then released back into the reach. Chinook salmon were re-captured by electrofishing roughly 24 hours later. In reaches lacking suitable locations for minnow trapping, I employed multiple-pass depletion electrofishing (Zippin 1956; Zippin 1958) instead of mark-recapture. Crews made two or three passes, each consisting of one upstream pass, with at least 30 minutes between passes. For both methods, nets to prevent the movement of fish into or out of the reach were not employed because previous tagging studies found that chinook salmon movement in such streams was limited (Bradford et al. 2001).

I calculated reach-scale population estimates using standard Petersen formulas (Ricker 1975) for mark-recapture sampling and the ‘maximum weighted likelihood estimation’ technique from Carle and Strub (1978) for multiple-pass sampling. Densities were calculated by dividing the population estimate by the wetted area in the reach, which was measured during habitat surveys (see Ch. 2). For each reach, I calculated average chinook salmon densities across years. I excluded the 2001 data from this calculation because chinook salmon densities in 2001 were, on average, approximately 80% lower than in other years. These low densities may have been due to low recruitment of juveniles to the streams. Parent spawners in 2000 were low in abundance and were infected with a parasite (*Ichthyophonus hoferi*), which may have affected their spawning success (M. Bradford, DFO, personal communication).

Thalweg profiling

I employed standard surveying techniques (Harrelson et al. 1994) and generally followed the thalweg survey methods described in Madej (1999a) to survey thalweg profiles. I used a three-person crew consisting of a surveyor, rod-person and data recorder. Surveys began and ended at riffle crests (the location in a riffle with the highest elevation) for streams with a pool-riffle structure or ended mid-riffle for streams lacking pool-riffle morphology. Distance along the channel was measured from a center tape that ran as straight as possible along the channel. Streambed elevation and water depth were measured using a survey level and stadia rod at a systematic interval every 0.25 to 1.0 bankfull widths (1 to 5 m) as well as at every break in streambed slope. Reaches with more variable bed morphology were sampled using a shorter interval whereas reaches with less variable beds were sampled at a longer interval.

Numerous metrics can be calculated from a thalweg profile. I examined two groups of metrics; those that directly measure pool habitat and others that measure variability of the profile. Pools that would be present during low flow periods are represented as depressions in the thalweg profile. Such depressions can be measured by the residual pool depth (Fig. 1.2a), which is equal to the elevation of the deepest point in a pool minus the elevation of the riffle crest immediately downstream (Lisle 1987). Residual pool depth represents the depth of a pool that would theoretically occur if there was no flow in the stream. Since more than one streambed elevation can be measured along a residual pool (e.g., the 4 streambed elevations that form the bottom of Pool A in Fig. 1.2a), I refer to 'residual depth' as any residual depth measured along a residual pool, and 'residual pool depth' as the maximum residual depth in a pool.

I expected chinook salmon densities to be higher in streams with abundant, deep pools because a positive relation between chinook salmon density in a pool and pool depth was found in a previous study (Bradford et al. 2001). Therefore, I examined one metric of pool extent and another of residual pool depth. 'Length in residual pool' (Lisle 1986) measures pool abundance or extent and is expressed as the proportion of the stream length in residual pools. It is calculated by first calculating the length of all stream sections in residual pools (e.g., the length of Pools A, B and C in Fig. 1.2a) and then dividing by the total surveyed stream length. 'Mean residual pool depth' reflects pool 'quality' and is calculated as the average of the residual pool depths in a reach (e.g., the average of the residual pool depths for Pools A and B in Fig. 1.2a). A similar metric, mean residual depth, is noted in Madej (1999a). I applied a minimum residual depth criterion of 0.1 m to this metric to ensure that residual depths reflected major morphological features and not simply small irregular features in the streambed (e.g., 'Pool' C in Fig. 1.2a). Chinook salmon in these streams were most abundant in habitat units greater than 0.25 m deep (Bradford et al. 2001), which was roughly equivalent to pools and glides with residual depths greater than 0.1 m.

I examined two metrics of profile variability. The ‘variation index’ (Madej 1999a) is an index of profile variability calculated as the standard deviation of the population of residual depths. Higher variation index values indicate a more variable profile. To account for differences among sizes of streams, Madej (1999a) recommended standardizing the metric by dividing by the bankfull depth (stream depth at bankfull flow). For example, a larger stream having the same relative profile ‘shape’ as a smaller stream would have deeper residual depths and a larger standard deviation of residual depths. I did not employ this transformation because my purpose was to examine metrics to monitor a given stream over time; consequently, I did not need to standardize for differences in stream size. As well, stream size varied little among surveyed streams (Table 1.1). ‘Mean square error’ (MSE) is another index of profile variability, which is calculated as the MSE from a linear regression line fit to the thalweg data (Fig. 1.2b). Higher MSE values should represent more variable profiles. Stream gradient generally decreases as one moves downstream (Leopold et al. 1964) and a second order polynomial fit to the thalweg data would account for such curvature in the profile. However, a linear regression provided an appropriate fit to these profiles because gradient was relatively constant over these relatively short reaches.

While I hypothesized that streams with deeper pools would have larger MSE and variation index values, a potential mechanism for a correlation between chinook salmon density and these metrics of variation was less apparent than for the two metrics of residual pools. All metrics were calculated in meters.

I correlated chinook salmon density with each thalweg metric. However, the interpretation of these relations is potentially confounded by correlations between chinook salmon density and other habitat characteristics. Therefore, I also assessed the correlation between chinook salmon density and three habitat variables that have been shown to influence salmonid abundance in previous studies (e.g., Rosenfeld et al. 2000; Roni and Quinn 2001): 1) channel gradient, 2) abundance (pieces/100 m) of large woody debris (LWD; dead wood greater than 10 cm in diameter and 1 m in length), and 3) the percent of the wetted channel area occupied by pools ('% wetted area in pools'; data from Ch. 2). I also assessed the collinearity among thalweg metrics and these three habitat variables. I then performed stepwise multiple regression to estimate the separate effects of thalweg metrics and other habitat variables on chinook salmon density.

I employed principal components analysis (PCA) to reduce the number of variables used in subsequent regression analyses with chinook salmon density by combining highly correlated variables with little loss of information. The PCA also provided an alternative method to assess the correlation among thalweg metrics and habitat variables. In the PCA, I included variables from Table 1.2 that had a significant ($P < 0.05$) or nearly significant correlation with chinook salmon density: gradient, $\log_{10}$-LWD abundance, $\log_{10}$-% wetted area in pools, length in residual pool, mean residual pool depth and the variation index. I then used the scores for principal components (PCs) with eigenvalues greater than 1 (Kaiser 1960) in subsequent regression analyses with chinook salmon density.

Where appropriate, variables were $\log_{10}$-transformed to standardize variances and improve normality. Analyses were performed using JMPIN (SAS Institute Inc. 2001).

Results

Thalweg profiles and metrics in undisturbed reaches

Differences among thalweg profiles are qualitatively apparent in the example profiles in Figure 1.3. Figure 1.3a (Quebec Creek) is typical of steeper gradient reaches with straight, uniform profiles. Figure 1.3c (Hoocheekoo Creek) is typical of lower gradient reaches with more variable or ‘bumpier’ profiles and Figure 1.3b (Croucher Creek Upper) is intermediate between the two. Thalweg metrics provide a quantitative representation of these qualitative differences among these three profiles. All metrics except MSE were highest in Hoocheekoo Creek, intermediate in Croucher Creek Upper and lowest in Quebec Creek (Table 1.1). The MSE values were less consistent with qualitative differences seen in the profiles; values were higher in Croucher Upper Creek than in Hoocheekoo Creek and lowest in Quebec Creek.

Values for all thalweg metrics varied at least three-fold among the undisturbed reaches (Table 1.1). The coefficient of variation (across all reaches) was similar among metrics and ranged from 34 to 49% (Table 1.1).

Thalweg metrics were positively correlated with each other and they all decreased with increasing gradient in the undisturbed streams (Table 1.2). The latter trend is apparent in Figures 1.2a-c as steeper streams had less variable profiles with fewer residual pools.

Chinook salmon density and thalweg metrics

Chinook salmon densities varied greatly among reaches (Table 1.1) and thalweg metrics explained up to 64% of the variation in the $\log_{10}$-transformed data (Fig. 1.4,

Table 1.2). All statistics in this section are calculated based on a sample size of 14 stream reaches. Log_{10}-chinook salmon density was strongly correlated with three of the four thalweg metrics, length in residual pool, mean residual pool depth, and the variation index, but was not correlated with the MSE.

Log_{10}-chinook salmon density was positively correlated with log_{10}-LWD abundance and log_{10}-% wetted area in pools, and negatively correlated with gradient (Fig. 1.5, Table 1.2). Thalweg metrics, log_{10}-LWD abundance and log_{10}-% wetted area in pools were negatively correlated with gradient (Table 1.2).

To separate the effects of gradient and thalweg metrics on chinook salmon density, I used a stepwise multiple regression model with gradient included as the first predictor variable and a thalweg metric as the second predictor variable. My purpose here was to better understand the collinearity among thalweg metrics and gradient, not to develop models to predict fish density in streams. Gradient was included in the model first because gradient is a function of topography and probably plays a major role in stream habitat. Gradient explained 51% of the variation in log_{10}-chinook salmon density and the addition of a thalweg metric (length in residual pool) to the model explained at most 13% more of the variation (Table 1.3). Length in residual pool, mean residual pool depth, the variation index and MSE explained 13%, 10%, 4% and 1% more of the variation, respectively. In the two-variable models, length in residual pool was the only thalweg metric that was significant ($P < 0.1$; Table 1.3). The general lack of significance for individual variables in the two-variable models is partially a result of the relatively small sample size, and the correlation between thalweg metrics and gradient (Table 1.2). Based on the Akaike Information Criterion (corrected for small sample sizes), models that included a thalweg metric performed the same or slightly worse than the model that included only gradient (Table 1.3).

The PCA showed that all six analyzed variables were highly correlated. PC1 explained 76% of the variation among variables and was the only PC with an eigenvalue greater than 1. Principal component 1 was composed of a roughly equal combination of all six variables, with a slightly lower loading for $\log_{10}$-LWD abundance (component loadings: gradient = -0.43, length in residual pool = 0.45, mean residual pool depth = 0.42, variation index = 0.44, $\log_{10}$-% wetted area in pools = 0.38 and $\log_{10}$-LWD abundance = 0.30). Log_{10}-chinook salmon density was strongly correlated with the PC1 scores ($r = 0.79$, $P = 0.0007$). This result suggests that all six variables contained essentially the same information. Furthermore, there is no second component that can explain some of the residual variation in $\log_{10}$-chinook salmon density not explained by PC1.

Thalweg profiles and metrics in mined streams

Thalweg profiles for the two mined reaches were less variable than for undisturbed streams of similar gradient. Bonanza Creek (Fig. 1.3d) had a uniform profile despite having a low gradient, similar to Croucher Creek Upper (Fig. 1.3b), which had a more variable profile. Thalweg metrics for the two mined reaches were 8 to 71% the size of metrics for undisturbed reaches of similar gradient (Table 1.1). For example, mean residual pool depth in Hunker Creek (0.17 m) was roughly one third that of the three undisturbed reaches with gradients less than 1% (Table 1.1). This difference was more pronounced in Bonanza Creek where no residual pools deeper than 0.1 m were present. Such residual pools were common in undisturbed streams with similar gradients.

Discussion

The main result of this study is that thalweg profiles can provide useful information on fish habitat. Unlike other thalweg profiling studies, I further related fish density to thalweg metrics. Juvenile chinook salmon density was correlated with thalweg metrics. Lower thalweg metrics in mined streams indicate that thalweg profiles can measure the effects of land use on channel morphology. These two results suggest that thalweg profiling is a useful tool to monitor fish habitat in Yukon streams, and can also likely be adapted to monitor fish habitat in other regions.

Geomorphology and thalweg metrics

Steeper gradient streams had less variable profiles and fewer residual pools than lower gradient streams. This result is consistent with geomorphic theory. Pool-riffle features develop as secondary currents (i.e., currents at an angle to the direction of the primary downstream current) cause the thalweg to move both laterally and vertically, and travel in a three-dimensional wave along the channel (Newbury and Gaboury 1993). This three dimensional flow creates channel meanders and a sequence of deeper pools and shallow riffles, with pools scoured along alternate banks of the channel. In steeper, more turbulent channels, pool-riffle morphology does not develop because this three-dimensional flow is decomposed due to the lower width-to-depth ratios and greater relative roughness in such channels (Montgomery and Buffington 1998).

A continuum of morphologies was present in these undisturbed study streams ranging from the most defined pool-riffle morphology in the lowest gradient stream to more featureless morphologies composed of one long continuous riffle in the steepest stream. Based on the classification system of Montgomery and Buffington (1998), these morphologies can be grouped into lower gradient (less than roughly 2%) pool-riffle morphologies and steeper gradient (greater than roughly 2%) plane-bed morphologies. A gradient threshold near 2% separated morphologies in these reaches. Thus, the strong negative correlation between thalweg metrics and gradient can be explained by the above geomorphic trends associated with gradient. The variation index and length in residual pools also decreased with increasing gradient in other studies (Robison 1997; Madej 1999a).

Channel obstructions such as LWD and boulders also influence channel morphology and form pools (Lisle and Kelsey 1982), thereby influencing thalweg profiles. Obstructions form pools by impounding water upstream, creating an eddy or re-directing flow to scour the streambed (Zimmerman et al. 1967; Keller and Swanson 1979; Montgomery et al. 1995; Beechie and Sibley 1997; Martin 2001). These obstructions form pools in addition to those formed by secondary currents. Obstructions can increase the pool frequency in pool-riffle reaches or form pools in plane-bed reaches that might otherwise lack pools (Montgomery and Buffington 1998). The influence of obstructions on thalweg profiles was more pronounced in pool-riffle channels where obstructions formed relatively large, channel-spanning pools (data presented in Ch. 2). In both plane-bed and pool-riffle reaches, obstructions influenced the channel without forming pools (e.g., scoured a portion of a glide) and formed smaller pools that did not span the channel or contain the thalweg (and thus were not measured in the thalweg profiles).

Chinook salmon density and thalweg metrics

Chinook salmon density was correlated with three thalweg metrics: length in residual pool, mean residual pool depth, and the variation index. Higher chinook salmon density in pools provides a potential mechanism for such correlations. Length in residual pool and mean residual pool depth provide indices of available pool habitat and the higher chinook salmon densities in pools can explain the correlation between chinook salmon density and these two metrics.

While the variation index was designed as an index of profile variability (Madej 1999a), it also indirectly provides an index of pool habitat. Reaches with higher variation index values had deeper and more abundant pools (Table 2.1). The variation index is strongly correlated with mean residual pool depth. These two metrics are similar because the standard deviation of residual depths is sensitive to the deepest residual depth in a pool. Thus, the correlation between chinook salmon density and the variation index may be attributable to this metric's indirect measurement of available pool habitat rather than the variation in residual depths. Given this alternative explanation, the correlation between chinook salmon density and the variation index does not necessarily support the assumption of Madej (1999b) that a more variable morphology reflects better habitat for fish.

Chinook salmon density was not correlated with MSE, possibly because this metric does not measure available pool habitat. Unlike the other metrics, MSE is not calculated based on residual pools. Mean square error is the mean of the squared residuals above or below a linear regression line. Therefore, a profile can have large deviations from the regression line and still have few residual pools. Deadwood Creek, for example, had a low variation index, length in residual pool, and mean residual pool depth yet had a high MSE. Deadwood Creek was a steep plane-bed stream with no residual pools, and had low chinook salmon density.

These streams appear to meet an important biological assumption of habitat models, that habitat, rather than biological factors such as recruitment, predation, interspecific competition or food availability, limits abundance (Fausch et al. 1988). Recruitment of juveniles to the stream does not appear to limit abundance in individual streams because there is no spawning in these streams and streams are colonized by a pool of juvenile chinook salmon moving in from the Yukon River mainstem. I interpret my results to suggest that, for a given length of stream, more juveniles colonize streams with better quality habitat. Interspecific competition and predation are limited or absent because few fish other than juvenile chinook salmon are present and few avian or terrestrial predators are present. I do not, however, have data to determine whether food availability limits abundance (Chapman 1966).

I had to correlate stream habitat data (thalweg metrics and other habitat variables) collected during 2001 with chinook salmon densities sampled during other years and averaged over one to four years of sampling. Therefore, I had to assume that habitat conditions during 2001 were similar to other years. This assumption is reasonable because, while thalweg profiles will change and LWD will move over time, these natural changes should average out over the entire stream reach and yield similar reach-averaged values across years. Therefore, correlating chinook salmon density averaged over one to four years of sampling with habitat data collected during one year is also reasonable. In addition, fish density estimates are inherently variable, due to both natural variability and sampling error, and the benefits to the analysis of having better density estimates from a longer time-series will likely outweigh the effects of relatively small changes in habitat. For a northwestern California watershed, Madej (1999a) showed that changes across years in the thalweg metrics of mean residual pool depth, percent of channel in riffles (opposite to % of channel in residual pools) and the variation index were consistent among streams (i.e., values increased from one year to the next in all streams and streams kept their same relative ranking). Furthermore, there were no major land uses in these watersheds that may have affected stream habitat.

Chinook salmon density and gradient

This study employed a design that 'substituted space-for-time'; I surveyed streams across space to evaluate thalweg profiling as a tool that could be used to monitor habitat over time and to compare the effects of human activities on fish habitat. A potential limitation of this approach was that collinearity among thalweg metrics and other habitat variables could confound the interpretation of the correlation between chinook salmon density and thalweg metrics. Collinearity among habitat variables is a common problem in habitat models and direct cause-and-effect relations are also difficult to determine because measured variables may be proxies for other habitat features (Shirvell 1989). The correlation between thalweg metrics and gradient is of particular interest because it is unclear whether chinook salmon are responding to characteristics measured by the profile, other habitat characteristics related to gradient, or a combination of the two.

The correlation analysis, multiple regression analyses, and PCA all showed that the thalweg metrics and habitat variables were strongly correlated (Tables 1.2, 1.3) and that chinook salmon density, thalweg metrics and other habitat variables were negatively correlated with gradient (Table 1.2). Based on the geomorphic theory described above, I interpret these correlations as gradient having a large overall physical control on fish habitat (including thalweg profiles and LWD), which, subsequently, affects chinook salmon density. Chinook salmon densities are responding to changes in available habitat that are largely controlled by gradient; chinook salmon are not responding directly to differences in gradient. Therefore, I infer that the correlations between chinook salmon density and thalweg metrics or habitat variables represent biologically relevant relations.

The correlation between chinook density and channel gradient suggests that measuring stream gradient can provide a rapid assessment (i.e., less labour intensive than surveying the thalweg profile) of fish habitat and chinook density in undisturbed Yukon streams. However, gradient cannot be used to monitor habitat over time because large changes in channel morphology and fish habitat due to land use can occur with little corresponding change in channel gradient. Thalweg metrics can be used to monitor such changes in fish habitat.

Comparison with mined streams

In a given reach, human activities on land can potentially have large impacts on channel morphology while having little or no effect on channel gradient. Therefore, when evaluating thalweg profiling as a tool to monitor channels over time, the relation between thalweg metrics and gradient is less important than the relation between thalweg metrics and land use. Given the correlations among chinook salmon density, thalweg metrics and gradient, comparing thalweg metrics in mined and undisturbed streams of similar gradient is important in determining the validity of using thalweg profiling to monitor fish habitat. Mined streams had lower thalweg metrics than undisturbed streams of similar gradient. Since chinook salmon density is correlated with thalweg metrics, I infer that chinook salmon habitat in these mined streams was inferior to habitat in undisturbed streams of similar gradient.

Placer-mined streams had plane-bed morphologies despite having gradients where we would expect pool-riffle morphologies to occur naturally. While this comparison is provisional because I only surveyed two placer-mined streams, this result is consistent with other studies on the effects of placer mining on small streams. Gilvear et al. (1995) analyzed aerial photographs from a small placer-mined stream in central Alaska and noted a change from abundant pools and riffles prior to mining to extensive shallow habitat during mining. Their results further suggested that geomorphological recovery after mining stops is slow; four years after mining, the wetted area of pools and deep-water habitats (> 50 cm) was only 6% of pre-mining values. Given the results of Gilvear et al. (1995), the morphology of undisturbed streams of similar gradient, and the geomorphic theory described above, we can expect that these mined streams in the Yukon should eventually develop pool-riffle morphology after mining stops. However, recovery will be affected by other changes to the channel from placer mining such as changes in sediment supply and substrate size or channelling of the stream. For example, the reach on Hunker Creek was constrained by piles of mining tailings on one bank. These tailings will prevent the lateral movement of the channel and the formation of pools.

Given our understanding of channel morphology, I conclude that thalweg profiling would be an effective tool to monitor geomorphological recovery and the subsequent recovery of fish habitat in mined streams. While thalweg profiling has been used to monitor geomorphological recovery in other studies, my results support the suggestion that the recovery of fish habitat can also be inferred.

Advantages of thalweg profiling

The major advantage of using thalweg profiling to monitor fish habitat is that measurements are repeatable and independent of stream flow. A visual survey of wetted area in pools, riffles and glides, referred to as ‘habitat-typing’, is an alternative approach that has been used to monitor habitat. Poole et al. (1997) discuss the limitations and potential consequences for management of monitoring fish habitat using habitat-typing. These limitations include subjective definitions of habitat units and dependence on flow (Poole et al. 1997). Thalweg profiling is more objective; arbitrary boundaries between pools and glides, for example, need not be determined visually. Residual pool boundaries are determined from the profile, which can be measured relatively precisely (discussed in the following section). Thalweg profiling is also not dependent on flow. I measured thalweg profiles in some Yukon streams during high flows when the standard habitat-typing approach, which must be preformed during low flow, was not feasible. Given these characteristics, surveys by different survey crews or under different flow conditions can yield the same profile (provided that the profile has not changed between surveys). In addition to these technical advantages of thalweg profiling, thalweg metrics provide a better measure of chinook salmon habitat compared to habitat-typing. Chinook salmon density was more strongly correlated with thalweg metrics than habitat-typing metrics (% wetted area in pools; Table 1.2). Thalweg profiling is also a relatively rapid survey technique. On average, our three-person crew surveyed a 150 m reach in three hours.

Thalweg profiling may be particularly useful in situations where fish abundance is difficult to estimate reliably and where chemical and biological factors do not limit fish abundance. Fish populations in streams are affected by physical, chemical and biological factors. Thalweg profiles provide a good measure of physical habitat and likely represent an upper limit to fish abundance that can be expected in a stream if other limiting factors, such as water temperatures or food availability, do not affect fish abundance. Thus, thalweg profiling can be incorporated as one component of a more comprehensive monitoring program that includes measurements of chemical and biological habitat features.

Challenges of thalweg profiling

Thalweg profiles provide only a two-dimensional view of fish habitat and, as such, will provide only a partial description. Profiling does not measure important morphological features or habitats that do not include the thalweg (i.e., the deepest part of a stream), such as backwater pools near the channel margin behind LWD or boulders. Despite this limitation, chinook salmon density was strongly correlated with thalweg metrics. However, this limitation may be more important in situations where pools at the channel margin comprise a large portion of the available pool habitat.

While profiling is repeatable and uses standard techniques, surveys will be sensitive to measurement error. Minimizing measurement error is important in order to maintain high statistical power to detect changes over time (Peterman 1990). Keim and Skaugset (2002) examined how errors in survey rod placement affected calculations of residual pool volume. Calculations were particularly sensitive to errors measuring riffle crest elevations. Errors in riffle crest measurements translated into large (13 to 54%)

errors in the volume of individual pools. However, the errors averaged out across several pools along a reach, resulting in reach-scale errors of less than 5%. To help ensure that the riffle crest is accurately measured, I recommend increasing the sampling frequency at riffle crests. My results should exhibit similar errors to those described by Keim and Skaugset (2002) because my survey techniques were comparable (though I employed different survey equipment) and the metrics I calculated (with the exception of the MSE metric) are equally sensitive to riffle crest elevations.

General recommendations

Based on my results, I recommend calculating both length in residual pool and mean residual pool depth to monitor chinook salmon habitat in Yukon streams. I suggest using metrics that measure pool habitat rather than profile variability such as the variation index or MSE. Length in residual pool measures the extent of pools while mean residual pool depth measures pool depth or ‘quality’. While these metrics could be combined into one metric—residual pool area—this metric would not be able to distinguish between changes in pool depth and pool frequency or extent. For example, increased sediment supply in pool-riffle channels from land use upstream can fill in pool bottoms. Mean residual pool depth will decrease with possibly little or no change to length in residual pools. Mean residual pool depth in undisturbed streams also had the greatest contrast with mined streams. Conversely, the addition of LWD or boulders to form pools in a pool-riffle channel would increase the length in residual pools, while the change in mean residual pool depth would be uncertain because newly formed pools could be deeper, shallower or the same depth as existing pools.

Thalweg profiling is better suited to monitor habitat in lower gradient pool-riffle streams for the following reasons. First, thalweg profiling can monitor the larger, channel-spanning habitat features that were common in pool-riffle reaches, such as free-formed or LWD-formed pools. In steeper plane-bed streams, these larger habitat features were less frequent, whereas smaller backwater pools near the channel margin (which are not measured in the thalweg profile), for example, were more common. Second, pool-riffle reaches are more sediment transport-limited (Montgomery and Buffington 1998) and thus would be more sensitive to changes in land use such as an increase in sediment supply.

Future research

While fish density was correlated with these thalweg metrics in small Yukon streams, we cannot assume that such a relation will exist in new geographic locations. Shirvell (1989) showed that habitat models were 'location-specific' rather than 'species-specific'. While habitat models explained much of the variation in fish abundance in the geographic location from which they were developed, models explained much less variation when applied to a new location. Therefore, the relation between fish abundance and thalweg metrics must be recalibrated for new locations. Developing a relation between fish abundance and thalweg metrics may be more challenging in systems where predation, recruitment, and interspecific competition can have a stronger influence on fish abundance than in Yukon streams. More controlled environments such as experimental channels or restored side-channel rearing habitats may be beneficial for such research.

Chapter 2
Large woody debris

Abstract

The importance of large woody debris (LWD) in forested stream ecosystems is well documented. However, little is known about LWD in northern boreal forest streams. I investigated the abundance, characteristics and function of LWD in small tributary streams of the upper Yukon River. Such streams provide non-natal rearing habitat for juvenile chinook salmon (*Oncorhynchus tshawytscha*) prior to their downstream migration to the Bering Sea. I measured LWD, surveyed fish habitat and sampled chinook salmon densities in 15 reaches 100 to 200 m in length among 13 study streams. Median LWD abundance was 22 pieces per 100 m, which is within the range reported for other studies. Median LWD diameter was 14 cm and LWD loading was 0.27 m^3 per 100 m^2. These values are well below values reported from Pacific coastal studies but are more similar to values from studies outside the Pacific coastal region. LWD formed 28% of the pools, which was slightly lower than in other studies. I focused on the function of LWD in forming pools because pools are the preferred habitat for these chinook salmon. Individual pool-forming pieces were generally 11 to 26.5 cm in diameter and ring counts on fallen riparian trees indicated that these pieces were at least 62 to 126 years old. Compared to Pacific coastal studies, pool-forming pieces were slightly smaller yet much older. Although pool-forming LWD in the Yukon was smaller relative to other studies, 86% of the pool-forming pieces were stabilized in the stream either in jams or by being rooted to the stream bank. Reach-scale chinook salmon density was correlated with LWD

abundance. I conclude that despite the unique climate and forest type of these northern boreal forest streams, LWD appears to perform a similar function in creating fish habitat as shown in more southerly latitudes.

Introduction

In-stream large woody debris (LWD) is recognized as an important component of stream ecosystems in forested watersheds. LWD can affect channel morphology and form pools (Keller and Swanson 1979; Lisle and Kelsey 1982), retain organic matter, gravel and sediment (Bilby and Likens 1980; Bilby 1981; Cederholm et al. 1989), influence invertebrate abundance (Hilderbrand et al. 1997) and provide cover and a velocity refuge for fish (McMahon and Hartman 1989; Shirvell 1990). LWD affects fish populations via the above mechanisms and a positive association between LWD and the abundance of juvenile salmonids has been documented (Murphy et al. 1986; Fausch and Northcote 1992; Cederholm et al. 1997; Roni and Quinn 2001). Most of the above research has been conducted in the Pacific Northwest (see summaries in Harmon et al. 1986; Bilby and Bisson 1998) where new forestry practices and stream restoration techniques have been implemented to protect and restore LWD in streams.

Similar research has rarely been done in boreal forests, even though they cover vast areas of North America, Europe and Asia between the tundra to the north and temperate forest or grassland to the south. These sparsely populated sub-arctic forests are also subject to increasing land-use pressures. Boreal forest streams in Canada, for example, are affected by forestry, mining, hydroelectric developments, and oil and gas exploration (Schindler 1998). Given these pressures, understanding the role of LWD in these streams will assist habitat managers to protect and restore fish habitat.

Only Clarke et al. (1998) have examined LWD in boreal forest streams. They investigated LWD dynamics and its relation to brook trout (*Salvelinus fontinalis*) densities in four small Newfoundland streams. Clarke et al. (1998) suggested that the function of LWD in boreal forest streams might differ from southern regions due to differences in climate, forest composition, fish fauna, and biophysical conditions. Relative to streams of the Pacific Northwest, for example, trees in boreal forests are small and slow-growing, ice conditions that are capable of moving LWD are more severe, and fish may have different habitat preferences.

LWD forms pools in streams by impounding water upstream, creating an eddy or re-directing flow to scour the streambed (Zimmerman et al. 1967; Keller and Swanson 1979; Montgomery et al. 1995; Beechie and Sibley 1997; Martin 2001). Such pools are the preferred rearing habitat for juveniles of several salmonid species (Bisson et al. 1982). Studies from the Pacific coastal region, for example, have outlined the importance of LWD in forming pools with cover that provide over-wintering habitat and refuge from high stream flows for juvenile coho salmon (*Oncorhynchus kisutch*) (Bustard and Narver 1975; Heifetz et al. 1986; Murphy et al. 1986; McMahon and Hartman 1989; Cederholm et al. 1997). The ability of LWD to form pools depends on its size and stability in the stream relative to the stream's size and hydraulic energy and, on average, pool-forming LWD are larger than non pool-forming pieces (Montgomery et al. 1995; Richmond and Fausch 1995; Beechie and Sibley 1997).

The characteristics of LWD and its relative importance to fish and fish habitat are not known for the Yukon River, the major river of central Alaska and the Yukon Territory. The Yukon River is an important producer of chinook salmon (*O. tshawytscha*) and juvenile chinook salmon make extensive use of small streams in the Yukon River basin. Chinook salmon densities in such streams are often highest in pools, especially those associated with LWD (Bradford et al. 2001).

This study describes the abundance, characteristics and function of LWD in northern boreal forest streams of the Yukon. A major objective was to compare metrics of LWD abundance (pieces/100 m), loading (volume/100 m^2 of channel area) and function in Yukon streams with data from other studies in more southerly locations. I also relate fish abundance to LWD abundance and evaluate the relative importance of LWD in forming pool habitat in these small streams.

Materials and Methods

Study area and growing conditions

I examined 13 small Yukon streams grouped into three sub-regions: Whitehorse, Minto and Dawson City (Fig. 1.1). All streams are located in the boreal cordillera eco-zone (Wiken 1986). While climate varies among sub-regions, all three have long cold winters and brief summers resulting in a relatively short growing season for these forests. Whitehorse has a continental climate and receives approximately 270 mm of precipitation annually while air temperature ranges from summer highs of >20°C to winter lows less than -40°C. Minto and Dawson receive approximately 247 and 325 mm of annual precipitation, respectively (Oswald and Senyk 1977). Compared to Whitehorse, Minto and Dawson have shorter summers and colder winter temperatures. Discontinuous permafrost is found throughout the region (Oswald and Senyk 1977). Smaller streams may freeze solid and can be covered with 1 to 3 m of ice during the winter. Approximately 40% of the precipitation falls from June to August in all sub-regions. Therefore, discharge in small streams fluctuates following summer rainstorms.

Boreal forest dominated by white spruce (*Picea glauca* (Moench) Voss) covers the valley bottoms of these streams. Grey alder (*Alnus incana* (L.) Moench ssp. *tenuifolia* (Nutt.) Breitung) and willow (*Salix* spp.) generally occupy the riparian areas within the first few meters from the stream bank. Balsam poplar (*Populus balsamifera* L. ssp. *Balsamifera*), the occasional paper birch (*Betula papyrifera* Marsh.) and trembling aspen (*Populus tremuloides* Michx.) often mix in with the white spruce in riparian areas further from the stream. White spruce, the only conifer species found near the study streams, would be the climax species in the absence of stand-replacing fires, which allow several early-successional deciduous species to re-establish (Cody 2000).

The topography of the stream valleys differs among sub-regions. The Dawson sub-region was not glaciated during the last glacial maxima of 40 000 to 14 000 years BP (Oswald and Senyk 1977); consequently, the stream valleys are more steeply incised. In contrast, the stream valleys near Minto and Whitehorse were previously glaciated and have a more gentle topography. Croucher Creek near Whitehorse has incised itself into sediments deposited by ancient glacial lakes and rivers.

Streams that I surveyed are directly tributary to the Yukon River mainstem, with the exception of Stony Creek, which flows into the Takhini River, a large tributary of the Yukon River. Average channel widths in surveyed streams were 3.4 to 7.7 m and gradients were 0.7 to 4.2% (Table 2.1). Streams had pool-riffle or plane-bed morphologies, according to the system of Montgomery and Buffington (1998) (Table 2.1). Streams of this size enter every 5-10 km along both banks of the Yukon River and along larger tributaries in the basin. Such small streams provide non-natal (i.e., non-spawning) rearing habitat for juvenile chinook salmon prior to their downstream migration to the Bering Sea. Many streams contain juvenile chinook salmon exclusively, although some also contain a few native arctic grayling (*Thymallus arcticus*), slimy sculpin (*Cottus cognatus*), longnose sucker (*Catostomus catostomus*), lake chub (*Couesius plumbeus*), and juvenile northern pike (*Esox lucius*) (Bradford et al. 2001). Non-native rainbow trout (*O. mykiss*) are present in Croucher Creek (Moodie et al. 2000).

Although these watersheds appear pristine, with the exception of sparse road developments or small homesteads, many watersheds were in fact logged for cordwood to fuel steamboats that navigated the Yukon River during the 1890s to 1950s. Steamboats switched from burning wood to coal in the latter part of this period. No detailed records are available for the exact timing and extent of such logging or the fire history for individual watersheds.

Field data

I surveyed one reach[2] on 11 of the 13 study streams, and surveyed two separate reaches on each of the remaining two streams, for a total of 15 reaches. All reaches were $\geq$20 bankfull channel widths in length (80 to 200 m). Reaches were surveyed near the stream's confluence with the Yukon River with the exception of Stony Creek (~3 km upstream of the Takhini River), the upper reach on McCabe Creek (~3 km upstream of the Yukon River), and both reaches on Croucher Creek (~1.0 and 1.5 km upstream of the Yukon River).

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[2] I use the simplest definition of the term 'stream reach' to refer to a specified length of stream (Armantrout 1998). The two surveyed reaches on Croucher Creek could also be described as two separate sections of the same 'reach' or homogeneous stretch of stream.

I defined LWD as dead wood (logs, limbs and rootwads) $\geq$10 cm diameter and $\geq$1 m long, at least partially within or directly above the bankfull stream channel. Dead standing pieces, rootwads (roots without a stem attached) and pieces still rooted in the ground were also included. For each item, I measured the diameter at each end using calipers, total piece length and length within the channel. For pieces with a root attached, the dimensions of the root and diameter at the junction of the bole and rootwad were measured. I recorded whether the piece was part of a jam (two or more pieces in contact with one another), whether the piece held the structure of the jam, and, where possible, identified the tree species. In conjunction with the habitat unit classification (described below), I recorded whether the piece influenced the stream channel or formed a pool.

Stream habitat units were visually classified according to Bisson et al. (1982) with two modifications; I added a category for both free-formed pools and under scour pools (Appendix 2.1). Pools are classified based on the pool-forming mechanism and comprise six pool types plus the two additional pool types mentioned above (definitions for pool types provided in Appendix 2.1). I defined pools as deeper areas relative to the rest of the channel with little flow. Pools were either free-formed or had an obstruction that dammed the pool, created an eddy or forced the flow to scour the pool. Obstructions included LWD, boulders, bank projections and bedrock outcrops, as in other studies (Montgomery et al. 1995). Small woody debris (pieces <10 cm in diameter or 1 m in length) and live trees were also considered pool-forming obstructions. Pools were classified as 'obstruction-formed' when the obstruction dammed the pool, created an eddy or forced flow in a direction consistent with the scour in the pool. Wetted areas of all habitat units were measured using a tape measure. The number of pools and riffles in a reach increases as stream flow decreases because pools break-up long riffles that are present at high-flow (Hilderbrand et al. 1999). Many Yukon streams were higher than base flow discharge (i.e., flows were influenced by direct surface runoff from rainfall or snowmelt) when they were surveyed and, therefore, I likely underestimated the number of habitat units present relative to base flow.

To estimate the age of woody debris, I sampled fallen riparian trees on the stream banks. One 5-cm-thick disk was cut from 87 pieces near stump height (at the junction of the bole and rootwad, approximately 30 cm above the ground) for subsequent aging. I assumed that in-stream LWD had experienced similar growing conditions and growth rates as aged pieces. I sampled a range of piece sizes for all species present and in all three sub-regions.

Data analyses

I calculated the volume of individual pieces from piece length and average diameter using the equation for the volume of a cylinder (Grette 1985). For pieces with an attached root, I added the root volume, calculated as the volume of a frustum with one diameter at the junction of the bole and rootwad and the other diameter at the base of the root. For rootwads, I used the equation for the volume of a cone. For each stream, I calculated LWD abundance as the number of pieces per 100 m and LWD loading as the LWD volume per 100 m^2 of bankfull channel area. For the loading calculations, I included only the portion of the piece that was within or above the bankfull channel.

I performed simple exploratory linear regression analyses to evaluate relations between selected LWD metrics and stream characteristics. To reduce the chance of type I errors, I limited the regression analyses to those commonly reported in the literature. For each stream, I used geometric means as the measure of central tendency for piece diameter and length because the distribution of these data was positively skewed (Bilby and Ward 1989). I also correlated reach-scale densities of juvenile chinook salmon (data from Ch. 1) with LWD abundance. Where appropriate, variables were $\log_{10}$-transformed to standardize variances and improve normality. Analyses were performed using JMPIN (SAS Institute Inc. 2001).

LWD and pools

I calculated the total pool area, number of pools and the percent pools formed by LWD ($\%P_{LWD}$) or by other obstructions. I also summarized the characteristics of pool-forming LWD and categorized them as either single pieces (not part of a jam) or pieces that held the structure of a jam (one or two structural pieces per jam). Median values are reported for the size (diameter, length and volume) of pool-forming and non pool-forming LWD because these data were positively skewed.

Aging LWD

I prepared and aged wood samples following the methods of Smiley and Stokes (1968). I counted rings under a binocular microscope and used up to 35x magnification in order to distinguish between closely spaced rings on samples that exhibited slow growth.

Age estimates from ring counts near stump height (or any distance above the root collar) will underestimate tree age because growth rings laid down between the root collar and stump height are not counted (see Wong and Lertzman 2001; Gutsell and Johnson 2002). Gutsell and Johnson (2002) examined errors from aging trees at various heights above the root collar in boreal stands in central Saskatchewan. Using their data, I estimated approximate height corrections of 15 years for white spruce and 5 years for aspen and paper birch. These estimates are supported by seedling height-age data from studies monitoring post-fire regeneration in forests near the Yukon-British Columbia border (Jill Johnstone, University of Alaska Fairbanks, unpublished data; Oswald and Brown 1990). A height correction of 5 years was also used for the other deciduous species based on the above height-age data.

To obtain a broad understanding of the age-diameter relation for woody debris, I calculated an age-diameter relation across all species. Analyses by species were not feasible because of low sample sizes.

Results

Characteristics of LWD

LWD abundance, loading and piece size varied greatly among stream reaches (Table 2.2). LWD abundance ranged from 4.9 to 78.0 pieces/100 m (16-fold variation) with a median abundance of 22.3 pieces/100 m. LWD loading ranged from 0.05 to 1.27 $m^3/100\ m^2$ (25-fold variation; median loading 0.27 $m^3/100\ m^2$). Abundance and loading were highest at the two Croucher Creek sites—approximately twice that of any other creek. Geometric mean piece diameter (measured at the larger end of a piece) and length (both within and outside of the stream channel) in individual streams ranged from 12.3 to 19.5 cm and 2.62 to 7.09 m, respectively. Median diameter and median total length of all pieces measured was 14.0 cm and 4.0 m, respectively. Many pieces were stabilized in the stream in one or both of the following ways; 42% of the pieces were part of jams and

38% were still rooted in the ground. The largest measured piece was a white spruce of 50 cm in diameter and 24.5 m in length. Small woody debris was present in all streams.

Six species of LWD were present (Fig. 2.1), of which white spruce was the most abundant (40%). White spruce and poplar pieces were larger than the other species and together comprised 74% of the total LWD volume.

LWD and pools

Pools were either free-formed or an obstruction such as LWD, boulders, small woody debris, bedrock, live trees or bank projections formed the pool (Table 2.3). Pools did not necessarily span the channel and many were found along the channel margin. My definition of pools included many small pools and calm backwater areas including pools behind large boulders in mid-channel. LWD formed 28% of all pools and between 3 to 73% of the pools in individual reaches (Table 2.1). Most LWD-formed pools were backwater pools (58%); the remaining LWD-formed pools were either dammed (17%), under scour (14%), plunge (7%), lateral scour (3%) or secondary channel (2%) (Appendix 2.2). LWD formed 49% and 10% of the pools in pool-riffle and plane-bed streams, respectively. Some of the bank projections that formed pools were armoured by a tree root or LWD on the bank but these pools were still classified as bank-formed. LWD also influenced the morphology of pools formed primarily by other mechanisms. For example, LWD increased the depth of many free-formed pools by partially damming the pool or deflecting flow to scour a portion of the pool.

Pool-forming LWD was larger than non pool-forming pieces (Table 2.4). The median diameter of pool-forming LWD was 26% larger than non pool-forming pieces. A two-way ANOVA for $\log_{10}$-LWD diameter with stream reach as one factor and the ‘pool forming or non pool-forming' characteristic of each piece as the second factor showed that this difference was significant (pool-forming: $F_{1,584} = 18.9$, $P < 0.0001$; stream reach: $F_{14,584} = 3.4$, $P < 0.0001$; pool-forming x stream reach: $F_{14,584} = 2.3$, $P = 0.005$). A similar pattern was observed for $\log_{10}$-LWD length (pool-forming: $F_{1,584} = 6.3$, $P = 0.01$; stream reach: $F_{14,584} = 3.2$, $P < 0.0001$; pool-forming x stream reach: $F_{14,584} = 1.6$, $P = 0.07$) and $\log_{10}$-LWD volume (pool-forming: $F_{1,584} = 17.6$, $P < 0.0001$; stream reach: $F_{14,584} = 3.7$, $P < 0.0001$; pool-forming x stream reach: $F_{14,584} = 1.9$, $P = 0.03$). Log_{10}-transformed LWD size data were not normally distributed because the minimum criterion for LWD diameter and length removes the lower tail of each distribution. However, ANOVA is robust to departures from normality when sample sizes are large (Zar 1996).

Although pool-forming LWD was larger than non-pool-forming LWD, small pieces (i.e., pieces 10 to 12 cm in diameter) also formed an important number of pools (29% of the LWD-formed pools; Appendix 2.2). However, 86% of all pool-forming pieces were stabilized in the stream either in LWD jams or by being rooted in the ground. LWD jams formed 61% of the LWD-formed pools while single pieces formed the remainder (Table 2.4). Of the pool-forming pieces, 55% of the single pieces and 45% of the pieces primarily responsible for holding the structure of the jam were rooted in the ground, either inside or outside of the stream channel. Only 17% of the pools were formed by pieces that were not stabilized in the stream (single, non-rooted pieces). All species formed pools and the proportion of pools formed by individual species was generally proportional to species abundance (Appendix 2.2).

LWD and stream characteristics

LWD was larger, longer and LWD loadings were higher in larger streams. Both geometric mean piece diameter and total length were positively correlated with stream drainage area (Fig. 2.2a,b; diameter: $r = 0.80$, $P = 0.0003$, $n = 15$; total length: $r = 0.54$, $P = 0.04$, $n = 15$). However, neither was correlated with channel width (diameter: $r = 0.06$, $P = 0.83$, $n = 15$; total length: $r = -0.20$, $P = 0.48$, $n = 15$). Log_{10}-LWD loading was correlated with drainage area (Fig. 2.2c; log_{10}-Volume/100 m: $r = 0.51$, $P = 0.05$, $n = 15$), whereas log_{10}-LWD abundance was not (log_{10}-LWD abundance: $r = 0.28$, $P = 0.32$, $n = 15$). The relation between channel width and either log_{10}-LWD abundance or loading (m^3/100 m) was not significant (log_{10}-LWD abundance: $r = 0.18$, $P = 0.53$, $n = 15$; log_{10}-LWD loading: $r = 0.12$, $P = 0.67$, $n = 15$). Correlations with gradient were opposite in slope to those reported above for drainage area because gradient was negatively correlated with drainage area ($r = -0.75$, $P = 0.001$, $n = 15$).

The age of LWD

Aged pieces were 25 to 224 (median 70) years old and were 7 to 38 (median 12) cm in diameter (Fig. 2.3). White spruce were generally older and larger than the deciduous species. A linear regression on the log_{10}-transformed data was used to summarize the age-diameter relation (Fig. 2.3; $r = 0.59$, $P < 0.0001$), although some non-linearity was present in the data. This non-linearity is attributable to the lack of correlation between age and diameter among the white spruce samples. Using this relation I estimated that the median age of LWD was 76 years, the median age of pool-forming LWD was 89 years and the interquartile range for pool-forming LWD was 62 to 126 years.

Chinook salmon density and LWD

Reach-scale $\log_{10}$-juvenile chinook salmon density increased with $\log_{10}$-LWD abundance (Fig. 2.2 d; $r = 0.61$, $P = 0.02$, $n = 14$).

Discussion

The main result of this study is that LWD in Yukon streams appears to perform a similar function in creating fish habitat as shown in more southerly latitudes. The relation between chinook salmon density and LWD abundance suggests that LWD provides important fish habitat in these northern boreal streams.

Characteristics of LWD

LWD abundance in Yukon streams was similar to values reported from studies in southern regions (Fig. 2.4). This comparison is approximate because I compare the range of LWD abundances observed in individual streams (Fig. 2.4). Nevertheless, the range of LWD abundances in Yukon streams was similar to values reported from studies in southern regions. Yukon streams with high LWD abundance had up to four times the abundance reported for small boreal forest streams in Newfoundland (Fig. 2.4).

LWD diameter was much smaller than values for Pacific coastal studies and slightly smaller than values for studies outside the Pacific coastal region. Geometric mean diameter of LWD in small western Washington streams ranged from 22 to 38 cm (Beechie and Sibley 1997), approximately twice the diameter of LWD in Yukon streams

(Table 2.2). Median piece diameter in subalpine Colorado streams ranged from 16 to 22 cm (Richmond and Fausch 1995), which was 13% larger than in Yukon streams. LWD loading (m^3 volume/100 m^2 channel area) in Pacific coastal streams was 5 to 16 times higher than in Yukon streams while loadings in studies outside the Pacific coastal region were up to twice as high (Fig. 2.5). Lower LWD loading in the Yukon can be attributed primarily to smaller LWD diameter because LWD abundance was similar to other studies.

No standard criterion exists for minimum LWD diameter and length; criteria vary according to study objectives, stream size, and tree size in the riparian forest. I selected the smallest commonly used criterion of 10 cm in diameter and 1 m in length because trees in the Yukon were relatively small. Nevertheless, based on cursory observations, I still suspected that small pieces would likely influence channel morphology. This criterion was appropriate; 89% of the pools formed by woody debris were pieces that met the LWD criterion.

LWD characteristics were highly variable among Yukon streams (e.g., the 25-fold variation in LWD loading; Table 2.2), as in other studies. LWD loading is dynamic and is affected by numerous input and output processes operating at different spatial and temporal scales (Harmon et al. 1986). My streams also spanned a large geographic area and differences in climate, topography, and possibly the timing and extent of cordwood collection. These factors may account for some variation among streams. Very high loadings in Croucher Creek, for example, may be partially a result of higher LWD input rates due to past flooding from beaver ponds and extensive bank undercutting as the channel migrates through the soft glacial sediments of the valley floor.

LWD and pools

LWD formed an important number of pools (28%), which provide important chinook salmon habitat in these streams (Bradford et al. 2001). The combined influence of vegetation—LWD, live trees and small woody debris—on pool habitat was greater than the percentage of pools formed by LWD ($\%P_{LWD}$) alone because live trees and small woody debris also formed pools (5% of the pools) and LWD frequently increased the depth of pools formed by other mechanisms.

The $\%P_{LWD}$ was, on average, roughly half the reported values for studies of streams in the Pacific coastal region: 70% in Andrus et al. (1988), 73% (range 11 to 95%) in Montgomery et al. (1995), 48% (range 8 to 84%) in Beechie and Sibley (1997), and 54% in Martin (2001). My results were also lower than the 76% (range 30 to 90%) reported for subalpine Rocky Mountain streams (Richmond and Fausch 1995). However, the range of $\%P_{LWD}$ among Yukon streams (3 to 73%) was similar to the values reported in these studies. In Yukon streams, $\%P_{LWD}$ was higher in pool-riffle streams than in steeper plane-bed streams, similar to small Washington streams (Rot et al. 2000). The $\%P_{LWD}$ in Yukon pool-riffle streams was lower than the 67% reported for small Washington streams (Rot et al. 2000), although this result was more similar than the comparisons across all channel types presented above. The $\%P_{LWD}$ in Yukon plane-bed streams, however, was still much lower than the 40% reported in Rot et al. (2000).

Pool-forming LWD in Yukon streams were slightly larger and approximately double the volume of non pool-forming pieces, similar to results for other studies (Richmond and Fausch 1995; Beechie and Sibley 1997). Pool-forming LWD in the Yukon was, on average, smaller than in other regions. In small Pacific coastal streams the ‘diameter of the smallest piece that formed a pool’ increased with stream size and measured 12 cm in the smallest stream (Beechie and Sibley 1997). The smallest pool-forming LWD in the Yukon was 10 cm in diameter. It is difficult to compare this metric among regions because it is only a minimum and does not include a range or central tendency. Interestingly, however, it indicates that even relatively small pieces can form pools in small Pacific coastal streams, as I have shown for the Yukon. Pool-forming LWD in subalpine Colorado streams (Richmond and Fausch 1995) had similar median length (4.4 m) yet had larger median diameter (22 cm) and volume (0.186 m^3) than pool-forming LWD in Yukon streams. However, the median size of ‘single, non-rooted’ (see below) pool-forming LWD in Yukon streams was similar to the size of pool-forming LWD in Colorado streams (most of which were single pieces).

LWD must be stable in the stream in order to form pools. While pool-forming LWD in the Yukon was relatively small, most pieces (86%) were stabilized in the stream either by their roots or in jams. Conversely, most pieces not stabilized by their roots or in jams were too small to be stable in the stream and subsequently did not form pools. I cannot directly compare this result with other regions because jam and rooting information for pool-forming LWD is not commonly reported.

On Merrice Creek, the stream channel had aggraded with a large volume of gravel accumulated upstream of a large, channel-spanning jam. This jam also formed a large plunge pool downstream. Though not a focus in this study, these observations suggest that jams can exert a major control on channel morphology as shown for Pacific Coastal streams (Keller and Swanson 1979; Hogan et al. 1998a; Luzi 2000).

Chinook salmon density and LWD

Studies from southern regions have documented the importance of LWD for fish and fish habitat (reviewed in Crook and Robertson 1999). My correlation between reach-scale chinook salmon density and LWD abundance suggests that LWD is also important in Yukon streams. A potential mechanism for this correlation is the role of LWD in forming deeper pool habitats. Chinook salmon rearing in small Yukon streams have similar habitat preferences to coastal coho salmon; both species are present in higher densities in pools, especially those associated with LWD (e.g., Bradford et al. 2001; Bisson et al. 1982). Correlations between reach-scale LWD abundance and coho salmon density are commonly reported (Murphy et al. 1986; Fausch and Northcote 1992; Cederholm et al. 1997; Roni and Quinn 2001) and this correlation is attributable to increased pool habitat. Conversely, densities of young-of-the-year (YOY) brook trout (*Salvelinus fontinalis*) were negatively correlated with LWD abundance and loading in four small boreal forest streams in Newfoundland (Clarke et al. 1998). Those authors indicate, however, that YOY brook trout reared in shallow habitats lacking LWD. Chapter 1 provides a more extensive discussion on chinook salmon habitat associations in Yukon streams.

A few large LWD jams formed barriers to the upstream migration of juvenile chinook salmon and, thus, LWD can limit the available rearing habitat in these streams. We observed some of these jams to break-up over a few years (M. Bradford, DFO, unpublished data), suggesting that they may be only transient features of the stream.

LWD characteristics and stream size

A positive relation between channel width and both LWD diameter and length, and a negative relation between channel width and LWD abundance is commonly reported in studies from the Pacific coastal region (Bilby and Bisson 1998). The proposed mechanism for this relation is that, in larger streams, only larger pieces remain stable in the stream while smaller pieces are transported downstream, causing an increase in average size and lower abundance (Bilby and Ward 1989).

I did not find a relation between channel width and geometric mean LWD diameter or length in Yukon streams; however, both were positively correlated with drainage area. Similarly, there was no significant correlation between channel width or drainage area and LWD abundance, whereas, there was a positive correlation between LWD loading and drainage area. Drainage area provides another measure of stream power because bankfull discharge, the discharge that moves LWD, is proportional to drainage area (Leopold et al. 1964). Consequently, drainage area may be a more appropriate measure of stream size for such analyses in Yukon streams. I examined only a small range of channel widths relative to other studies but examined a large range of drainage areas (Table 2.1) and, therefore, may have been better able to detect trends with drainage area. Alternatively, channel width may be a poor indicator of stream size in these boreal streams, possibly due to the influence of ice conditions on the shape of the stream cross-section; unlike other studies, channel width was not correlated with drainage area (e.g., Bilby and Ward 1989; Richmond and Fausch 1995).

The positive correlation between drainage area (as it affects stream power) and LWD diameter or length is consistent with previous studies and the proposed mechanism for this relation. This result also suggests that despite differences in the timing and magnitude of peak, channel-forming flows (e.g., peak flows from heavy rains in Pacific coastal streams vs. the break-up of ice during the spring in the Yukon), similar physical processes affect average LWD size in these Yukon streams as seen in other regions. The lack of correlation between LWD abundance and stream size is not consistent with the proposed mechanism. However, this relation is not consistent across all studies (e.g., Ralph et al. 1994; Richmond and Fausch 1995).

Beechie and Sibley (1997) found a negative correlation between both LWD loading and abundance, and channel width in small Washington streams. They suggested that the decrease in LWD abundance with channel width offset the associated increase in LWD diameter and length. Conversely, the increase in LWD loading with drainage area in the Yukon can be attributed to an increase in LWD diameter and length, not an increase in LWD abundance, because LWD abundance was not correlated with drainage area.

The age of LWD

My results indicate that trees in the riparian zones of Yukon streams grow more slowly than those in Pacific coastal forests. I am not aware of other LWD studies that report LWD age; therefore, I compared my results with those from a model for Pacific coastal streams (Beechie et al. 2000). For a coastal watershed in Washington State, the model estimated that it takes red alder (*Alnus rubra* Bong.) 7 years and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) 15 years to grow to a breast-height diameter of 13 cm (the smallest diameter of pool-forming LWD in 5 m wide streams) and be recruited to a stream as debris if some force causes the tree to fall. My data indicate that, in the Yukon, it takes at least 25 years and generally 40-70 years for grey alder and other deciduous species to reach this size and at least 70 years for white spruce (Fig. 2.3). The implication of this slower tree growth is that it will take longer for the natural supply of functional (i.e., pool-forming) LWD to become available to the stream following a disturbance in the riparian area of Yukon streams. For example, if I assume that tree growth rates and species composition in disturbed sites are similar to those for the pieces that I aged, then it may take at least 62 to 126 years, on average, for trees to grow to a size where they would form pools once recruited into the stream.

Estimated LWD ages are minima for the following two reasons. First, LWD diameter was not necessarily measured at the base (e.g., for broken LWD pieces) and the bark had been eroded from many LWD pieces, whereas all aged pieces were measured at the base and had intact bark. Therefore, the measured diameter of LWD would likely underestimate the piece’s true diameter at the base and, therefore, would be older than estimated using the age diameter relation. Second, I am unable to rule out the occurrence of locally absent[3] or double rings (Stokes and Smiley 1968) without employing techniques from dendrochronology. Ages would be underestimated if locally absent rings were present and overestimated if double rings were present. Locally absent rings were present in approximately 25% of the trees sampled (up to 30 absent rings per tree) in central Saskatchewan stands (Gutsell and Johnson 2002).

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[3] Locally absent rings are rings that are not present around the entire circumference of the tree.

Management recommendations

My results have several implications for the management of small streams similar to those examined in this study. Although LWD in Yukon streams is generally small, slow-growing, and present at lower loading relative to other regions, LWD is as abundant as in other regions and also forms important fish habitat. Therefore, sources of LWD recruitment to the stream, such as riparian forests, should be maintained. For disturbed streams, management plans should aim to restore and maintain LWD abundance, as well as the size and stability characteristics of LWD, within the range reported for the relatively undisturbed streams examined in this study. Adding LWD is especially important in streams where riparian vegetation has been removed because it will take at least 62 to 126 years for the natural supply of functional (i.e., pool-forming) LWD to become available. Adding LWD to disturbed Yukon streams, including pieces with the same size and stability characteristics of pool-forming LWD, will create pools and provide improved habitat for juvenile chinook salmon.

Conclusions

The major result of the thalweg profiling study is that thalweg profiles provide useful information on fish habitat. Unlike other thalweg profiling studies, I further related fish density to thalweg metrics and found relations between juvenile chinook salmon density and certain thalweg metrics. Therefore, thalweg profiling may be an effective tool to estimate and monitor fish habitat in streams. The major advantages of using thalweg profiling compared to existing methods to monitor fish habitat are that measurements should be repeatable with high precision, and are independent of stream flow. The major result of the LWD study is that LWD is abundant and provides important habitat for juvenile chinook salmon. Together, this information can be used to help better manage fish habitat in small Yukon streams, as well as outside of the Yukon.

Linking LWD and thalweg profiling

The influence of LWD on channel morphology is reflected in the thalweg profiles. Large woody debris formed pools that were measured on the thalweg profile, particularly in lower gradient streams with pool-riffle morphology. When restoring streams with LWD, thalweg profiling can be used to measure the influence of LWD on channel morphology over time.

Broader applicability of these results

Thalweg profiling is potentially a widely applicable tool that could be used to monitor fish habitat over time in small streams. Thalweg profiling can be used to monitor the effects of land use or the benefits of habitat restoration. While I have shown a relation between chinook salmon density and thalweg metrics in Yukon streams, such a relation may also be present in other systems. Coho salmon in Pacific Northwest streams, for example, have similar habitat preferences to Yukon chinook salmon and, therefore, the density of juvenile coho salmon in small streams may be correlated with the thalweg metrics examined here. Such studies have not been done for coho salmon or other fish species. Again, however, the relation between fish abundance and these thalweg metrics will need to be recalibrated for new geographic locations.

Results of the LWD study provide important information for boreal forest streams in general. My results show that despite differences in climate and forest type between boreal and temperate forests, LWD is abundant and forms important habitat in Yukon streams. Given this result, LWD may be equally important in boreal forest streams outside of the Yukon.

Specific management recommendations related to placer mining

In addition to the general management recommendations provided in each chapter, I suggest the following management recommendations to protect and restore chinook salmon habitat for placer-mined streams in the Yukon:

1) Use thalweg profiling to monitor fish habitat and geomorphological recovery in small placer-mined streams with gradients less than 2%. Thalweg profiling is not as well suited to monitor channel morphology in steeper (>2 % gradient) plane-bed streams because channel-spanning habitat features are less common in such channels. I recommend calculating length in residual pools and mean residual pool depth to monitor the recovery of chinook salmon habitat. Ideally, profiles should be surveyed

prior to mining to provide reference conditions to guide restoration. In the absence of such pre-mining information, metrics from the recently undisturbed streams studied here with similar gradient can be used as a guide.

2) Protect riparian vegetation, where possible, because it provides an important source of LWD to the stream channel. Given the slow tree growth in the Yukon, planting trees in disturbed riparian zones will not provide a source of LWD to the streams for at least 62 to 126 years. These estimates assume that tree growth rates and species composition in disturbed sites are similar to those for the pieces of LWD that I aged.

3) Add LWD to restored stream channels to create important chinook salmon habitat. Such LWD is needed until the natural supply of LWD to the stream from riparian forests can be restored. The abundance and characteristics of LWD in the undisturbed streams studied here provide reference conditions that can be used as a guide. The current white book guideline (Hardy BBT Ltd. 1991), which equates to roughly 1 piece/100 m for streams of this size, is well below what is found naturally in these streams. I suggest that an appropriate new guideline for LWD abundance should be within the middle 50% of abundance values observed naturally in these streams, from 13 to 38 pieces/100 m. My data indicate that LWD abundance varies naturally among streams, and thus there is no single or optimal LWD loading for these streams. While LWD abundance fluctuates over time, restoration plans should aim to maintain LWD abundance within the range shown for these undisturbed streams. Although the use of whole LWD pieces is preferred, bundled alder or willow cuttings can be used if LWD of adequate size is not available on-site (Karle and Densmore 1994).

4) Anchor a sufficient number of LWD pieces in restored channels in order to form pools. While the current white book recommends anchoring LWD, the stated purpose there is to provide cover and not to form pools. The potential benefits to chinook salmon habitat of anchoring LWD are high and the cost of doing so is low. During the restoration of 'type IV fish' channels (non-salmonid rearing streams), the cost of installing each anchored LWD piece was only $80 (1991 dollars) with anchored LWD comprising 0.1% of the total restoration cost (Jacobsen and Korn 1991).
5) Consider the natural geomorphology of stream channels when designing restoration plans. This study showed that chinook salmon density, thalweg metrics, and LWD abundance decreased with increasing channel gradient. Channel features also differed markedly between pool-riffle channels (gradients less than 2%) and plane-bed channels (gradients greater than 2%) because the processes that create pools varied by channel type. Thus, the pattern of channel recovery will in part depend on channel type and gradient. Pool-riffle channels should be engineered to allow pool-riffle characteristics to develop naturally (e.g., maintain the gradient, sinuosity, and substrate size of natural channels). Since such pool-riffle characteristics may take a long time to develop (Montgomery et al. 1995), obstructions (e.g., LWD and boulders) should also be included in pool-riffle channels to form pools and increase pool-frequency. Obstructions formed all of the pools in plane-bed streams and pool-forming obstructions should be included in the restoration of these channels. The type of obstruction employed will also depend on channel type. Boulders were the dominant obstruction in plane-bed channels while LWD dominated pool-riffle channels.

Table 1.1. Thalweg metrics and physical and biological data for 14 undisturbed and two previously placer-mined stream reaches. Thalweg metrics are defined in the text and are based on measurements made in meters. Channel type is based on the system of Montgomery and Buffington (1998): pr = pool-riffle and pb = plane-bed. Wetted area in pools is the percent of the wetted channel area occupied by pools. Observed densities of age 0 chinook salmon are averaged over 1 to 4 years of sampling (not all streams sampled for fish in all years). Bankfull width, bankfull depth and channel type are not ‘natural’ for placer-mined streams and thus are not directly comparable to undisturbed streams.

		Thalweg metrics				Stream physical and biological characteristics								
Stream	Location	Variation index	Mean square error	Length in residual pool (proportion)	Mean residual pool depth (m)	Gradient (%)	Reach length (m)	Bankfull width (m)	Bankfull depth (m)	Drainage area [a] (km^2)	Channel type	Large woody debris abundance (no. / 100m)	Wetted area in pools (%)	Chinook salmon density (no. / m^2)
Undisturbed streams														
Croucher Upper	Whitehorse	0.14	0.028	0.83	0.30	1.1	147	4.6	0.98	122	pr	78.0	17	0.49
Croucher Lower	Whitehorse	0.16	0.031	0.80	0.30	1.3	126	5.7	1.12	123	pr	70.6	14	0.41
Hoocheekoo	Minto	0.15	0.027	0.91	0.46	0.7	84	4.5	0.31	85	pr	39.8	59	2.26
McCabe Upper	Minto	0.17	0.039	0.79	0.46	0.7	168	5.6	1.00	193	pr	24.5	8	2.42
McCabe Lower	Minto	0.19	0.041	0.79	0.41	0.9	190	5.0	0.93	197	pr	12.6	21	0.41
Merrice	Minto	0.07	0.030	0.28	0.27	1.9	164	5.5	0.74	136	pb	10.3	8	0.13
Williams	Minto	0.17	0.039	0.78	0.35	1.2	148	5.3	0.73	91	pr	22.3	30	1.14
Baker	Dawson	0.05	0.016	0.17	0.16	3.6	169	6.1	0.82	26	pb	21.3	4	0.26
Caribou	Dawson	0.03	0.011	0.25	0.14	4.2	100	3.4	0.77	18	pb	4.9	4	0.05
Deadwood	Dawson	0.07	0.030	0.37	0.19	3.1	178	4.5	1.01	30	pb	13.5	5	0.11
Ensley	Dawson	0.10	0.020	0.68	0.26	1.3	129	5.4	1.01	74	pr	25.2	21	0.84
Garner	Dawson	0.10	0.018	0.67	0.24	1.3	100	4.7	0.79	73	pr	35.2	3	1.01
Mechem	Dawson	0.08	0.024	0.55	0.23	2.0	137	4.7	0.88	59	pr	37.7	6	0.98
Quebec	Dawson	0.04	0.021	0.27	0.13	2.1	126	5.3	1.05	37	pb	13.0	5	0.03
Mean		0.11	0.027	0.58	0.28	-	-	-	-	-	-	-	-	-
cv		49%	34%	45%	40%	-	-	-	-	-	-	-	-	-
Placer-mined streams														
Bonanza	Dawson	0.03	0.003	0.32	0[b]	1.0	201	11.1	0.7	206	pb	-	-	-
Hunker	Dawson	0.06	0.017	0.56	0.17	0.8	200	6.2	0.8	93	pb	-	-	-
Mean		0.04	0.010	0.44	0.17	-	-	-	-	-	-	-	-	-
cv		55%	98%	39%	N/A	-	-	-	-	-	-	-	-	-

[a] Area upstream of sampled reach

[b] No pools with a residual depth greater than 0.1m deep were present in Bonanza Creek

Table 1.2. Pairwise correlation coefficients among thalweg metrics, habitat variables and chinook salmon density in 14 stream reaches. Light shading represents r > 0.55 and P < 0.05, dark shading represents r < -0.55 and P < 0.05.

	Gradient	Log_{10}-Chinook salmon density	Log_{10}-Large woody debris abundance	Log_{10}-% Wetted area in pools	Mean square error	Mean residual pool depth	Length in residual pool	Variation index
Gradient	-							
Log_{10}-Chinook salmon density	-0.71	-						
Log_{10}-LWD abundance	-0.60	0.64	-					
Log_{10}-% Wetted area in pools	-0.66	0.53[a]	0.37	-				
Mean square error	-0.66	0.42	0.19	0.53[a]	-			
Mean residual pool depth	-0.82	0.77	0.37	0.74	0.76	-		
Length in residual pool	-0.87	0.80	0.67	0.74	0.59	0.83	-	
Variation index	-0.83	0.70	0.50[a]	0.73	0.81	0.89	0.90	-

[a] 0.05 < P < 0.10

Table 1.3. Statistics for multiple linear regression models describing the relations between log_{10}-chinook salmon density and channel gradient, length of channel in residual pool (L resid), mean residual pool depth (M resid), variation index (Var), and mean square error profile (MSE) in 14 stream reaches. Shown are parameter estimates (and SE) and asterisks denote parameters where P < 0.1. AIC_c = Akaike's Information Criterion corrected for small sample sizes.

	Independent variables						
	Variable 1		Variable 2				
Intercept	Variable	Estimate	Variable	Estimate	P	R^2	AIC_c
0.28 (0.22)	Gradient	-0.37 (0.11)*			0.004	0.51	-19.8
-1.29 (0.81)	Gradient	-0.03 (0.19)	L resid	1.65 (0.82)*	0.004	0.64	-20.1
-0.98 (0.77)	Gradient	-0.13 (0.17)	M resid	2.95 (1.73)	0.006	0.61	-19.0
-0.42 (0.74)	Gradient	-0.22 (0.19)	Var	3.82 (3.89)	0.01	0.55	-17.0
0.50 (0.70)	Gradient	-0.41 (0.15)*	MSE	-6.00 (17.92)	0.02	0.51	-15.9

Table 2.1. Characteristics of surveyed stream reaches, pool habitat and juvenile chinook salmon density in 15 Yukon stream reaches. '% Wetted area in pools' is the percent of the wetted channel area occupied by pools.

Stream	Location	Reach length (m)	Channel gradient (%)	Drainage area (km^2)	Bankfull width (m)	Channel type[a]	% Wetted area in pools	% Pools formed by LWD	Chinook density[c] (no./m^2)
Croucher U.	Whitehorse	110	1.1	122	4.7	pr	17	47	0.49
Croucher L.	Whitehorse	126	1.3	123	5.7	pr	14	67	0.41
Stony	Whitehorse	201	2.0	97	7.7	pr[b]	20	17	N/A[d]
Hoocheekoo	Minto	83	0.7	85	4.5	pr	59	60	2.26
McCabe U.	Minto	168	0.7	193	5.6	pr	8	17	2.42
McCabe L.	Minto	190	0.9	197	5.0	pr	21	3	0.41
Merrice	Minto	172	1.9	136	5.5	pb	11	20	0.25
Williams	Minto	162	1.2	91	5.2	pr	30	24	1.14
Baker	Dawson	169	3.6	26	6.1	pb	4	6	0.26
Caribou	Dawson	102	4.2	18	3.4	pb	4	10	0.05
Deadwood	Dawson	178	3.1	30	4.5	pb	5	30	0.11
Ensley	Dawson	131	1.3	74	5.4	pr	21	70	0.84
Garner	Dawson	111	1.3	73	4.7	pr	3	73	1.01
Mechem	Dawson	146	2.0	59	4.7	pr	6	17	0.98
Quebec	Dawson	131	2.1	37	5.3	pb[b]	5	35	0.03

[a] Channel type from Montgomery and Buffington (1998), pr = pool-riffle, pb = plane-bed

[b] Channel type intermediate between pool-riffle and plane-bed

[c] Reach-scale chinook salmon densities in August, averaged over 1 to 3 years of data from 1998, 1999, 2000, 2002 (not all streams sampled each year)

[d] Chinook salmon not present in Stony Creek during summer 2001 and 2002 sampling

Table 2.2. Characteristics of LWD in and along 15 Yukon stream reaches. Mean diameter and length are geometric means. Diameter is measured at the larger end of a piece and length includes piece length within and outside of the stream channel. 'No./100 m' is the number of LWD pieces per 100 m of stream. 'Vol. /100 m^2' is the volume of LWD per 100 m^2 of bankfull channel area (note: the volume calculation includes only the portion of pieces that are within and above the bankfull channel).

	LWD Characteristic			
Stream	Mean diameter (cm)	Mean length (m)	No./100m	Vol./100m^2 (m^3)
Croucher U.	14.8	6.16	78.0	1.27
Croucher L.	14.6	3.75	70.6	0.78
Stony	15.0	2.64	21.9	0.13
Hoocheekoo	15.2	3.32	39.8	0.51
McCabe U.	17.2	3.35	24.5	0.29
McCabe L.	19.5	7.09	12.6	0.28
Merrice	17.1	4.02	16.3	0.20
Williams	14.3	3.72	22.3	0.18
Baker	14.0	3.05	21.3	0.11
Caribou	14.2	2.62	4.9	0.05
Deadwood	14.2	3.58	13.5	0.13
Ensley	13.2	5.07	25.2	0.14
Garner	16.4	3.66	35.2	0.45
Mechem	13.5	3.95	37.7	0.35
Quebec	12.3	3.55	13.0	0.27

Table 2.3. Pool-forming mechanisms in 15 Yukon stream reaches. 'N' is the number of pools formed by a given pool-forming mechanism across all 15 stream reaches. '% Total N' and '% Total pool area' is the percentage of all pools and the percentage of the total pool area formed by given pool-forming mechanism, respectively. 'Average pool area' is the average wetted area of individual pools.

Pool-forming mechanism	N	% Total N	Area (m^2)	% Total pool area	Average pool area (m^2)
Free-formed	21	9.9	617.7	59.2	29.4
LWD	59	27.8	239.0	22.9	4.1
Boulder	78	36.8	90.6	8.7	1.2
Bank [a]	34	16.0	63.2	6.1	1.9
Bedrock	10	4.7	17.0	1.6	1.7
SWD [b]	7	3.3	6.5	0.6	0.9
Live trees	3	1.4	8.8	0.8	2.9

[a] The banks for many of the bank formed pools were armoured with LWD

[b] SWD is small woody debris, or wood <10 cm in diameter or <1 m long. Most were ~8 cm in diameter or slightly <1m long.

Table 2.4. Characteristics of LWD that formed pools and non pool-forming LWD in 15 Yukon stream reaches. Length, diameter and volume values are medians with the interquartile range in parentheses. 'Pool-forming single' are pool-forming pieces that were not part of a LWD jam. 'Pool-forming jam' are the key structural pieces in a LWD jam that formed a pool. '% Rooted' refers to pieces that were rooted either on the streambank or in the channel.

LWD Type	N	Length (m)	Diameter (cm)	Volume (m^3)	% Rooted	# Pools formed
Non pool-forming	518	4.0 (2.1-6.6)	13.5 (11.0-18.0)	0.032 (0.018-0.075)	36	n/a
All pool-forming pieces	66	5.4 (3.3-8.7)	17.0 (11.0-26.5)	0.067 (0.024-0.208)	48	59
Pool-forming single	20	5.5 (3.6-8.9)	18.0 (11.0-35.0)	0.094 (0.023-0.378)	55	23
Pool-forming jam	46	6.0 (3.2-8.7)	16.0 (11.0-24.0)	0.064 (0.024-0.207)	45	36

Fig. 1.1. Map of the Yukon River and study sites, which are grouped into three sub-regions: Whitehorse, Minto, and Dawson City.

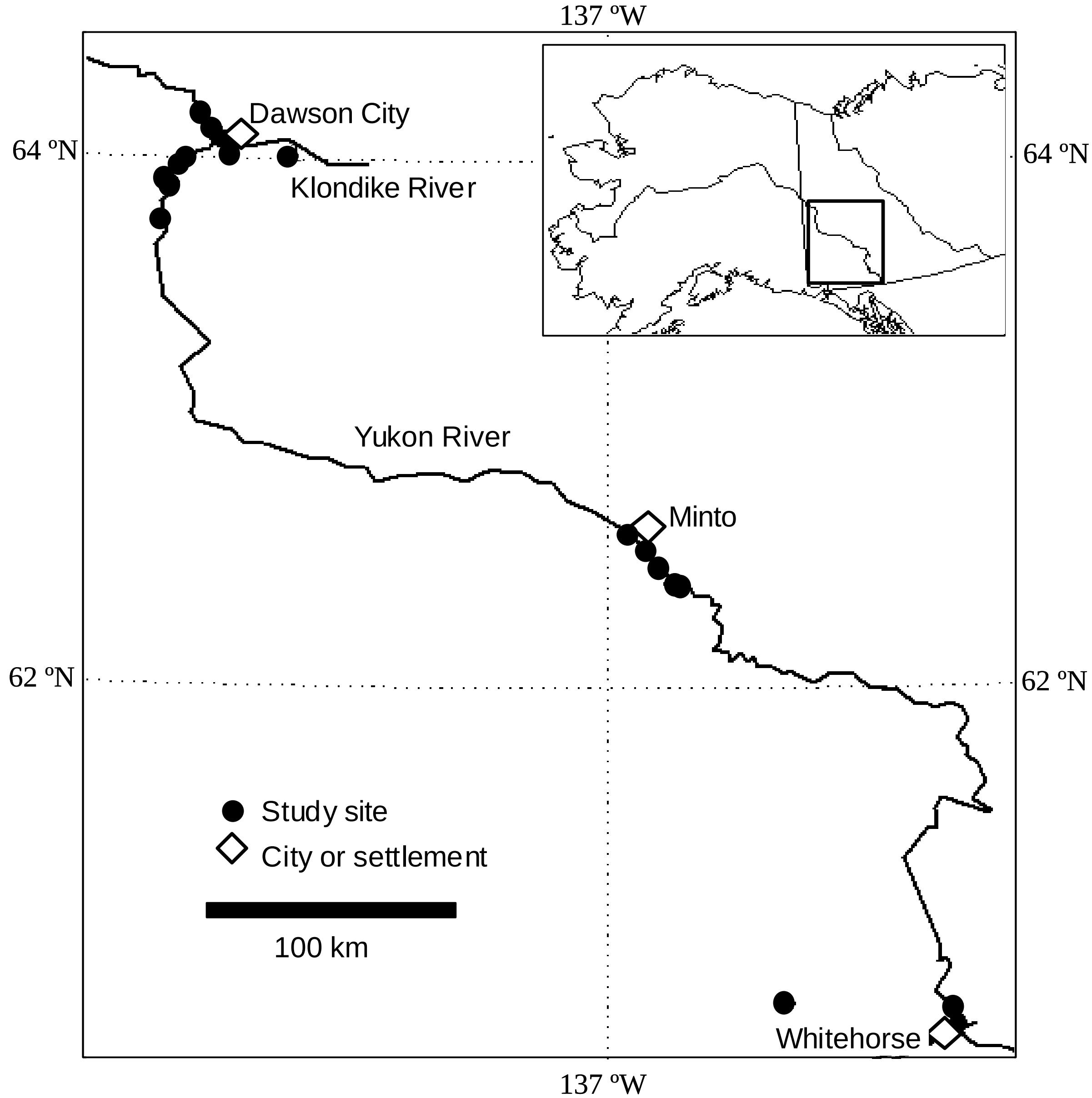


Fig. 1.2. Hypothetical thalweg profiles to illustrate calculations for the thalweg metrics, which are defined in the text. Profile (a) highlights residual pools. Pools A and B meet the 0.1 m minimum depth criterion. ‘Pool’ C illustrates a small morphological feature that is shallower than the minimum depth criterion and, therefore, is not included in the calculation for the thalweg metric ‘mean residual pool depth’. Profile (b) illustrates a linear regression line fit to the thalweg data.

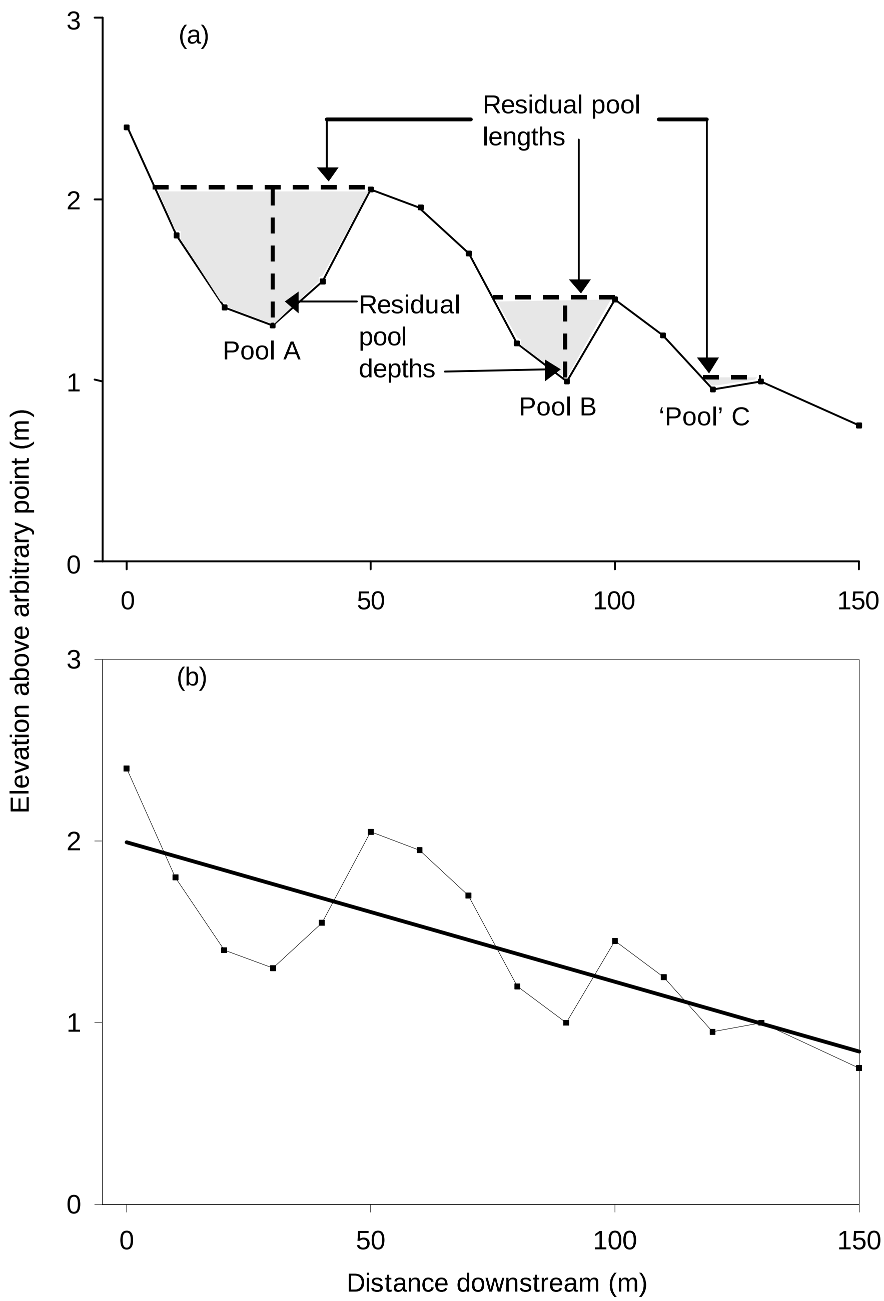
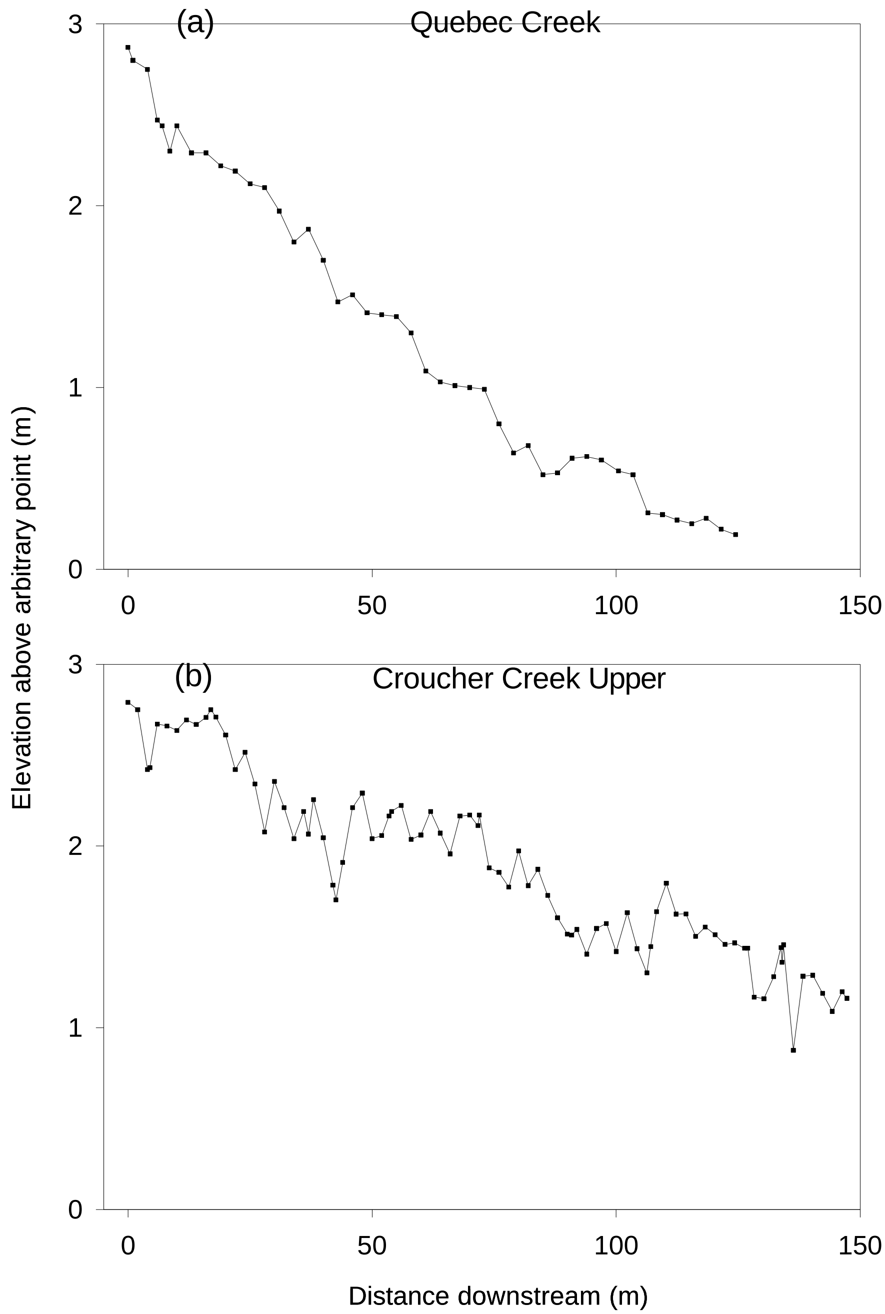
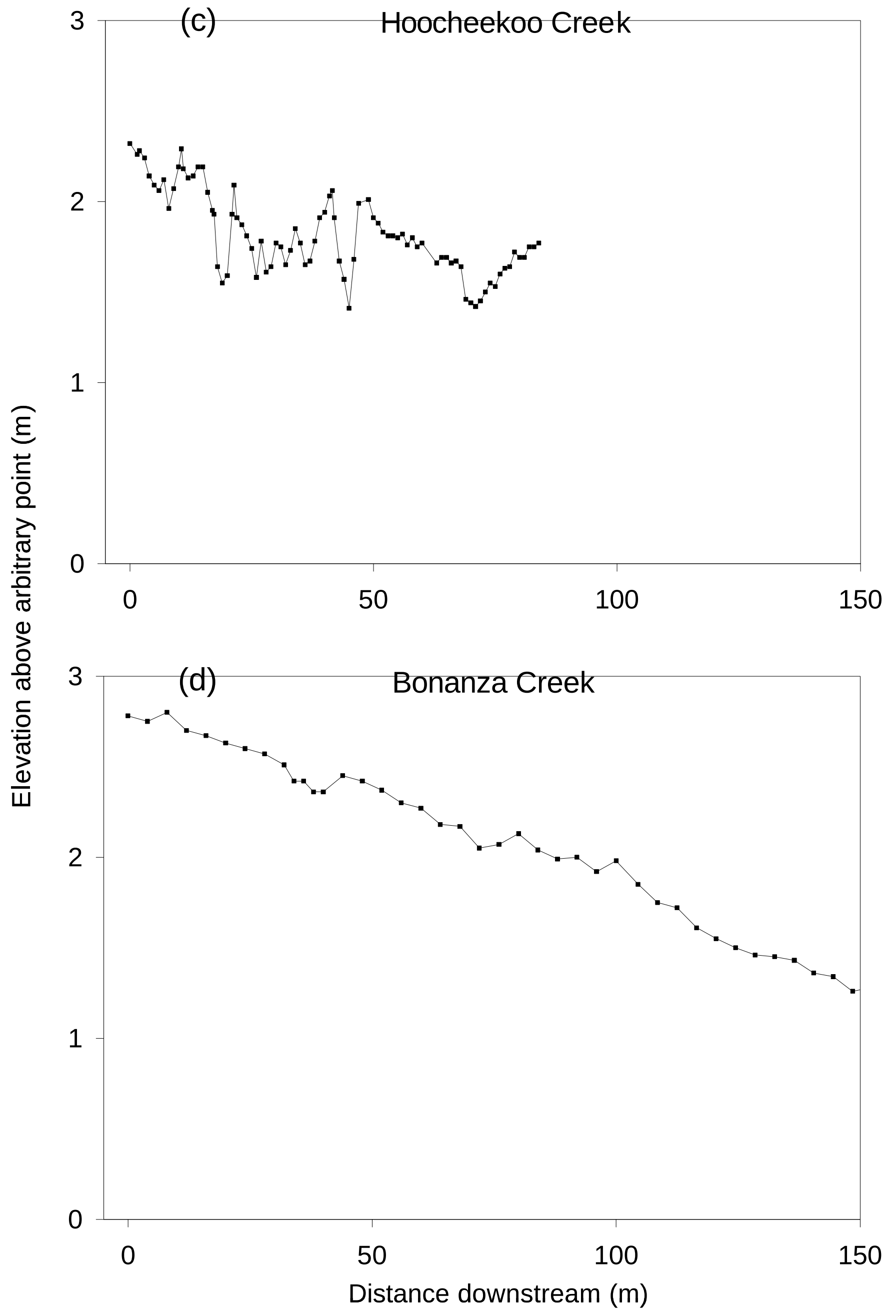


Fig. 1.3. Thalweg profiles for three undisturbed stream reaches and one previously placer-mined reach. (a) Quebec Creek is high gradient with a uniform profile and low chinook salmon density. (b) Croucher Creek Upper is medium gradient with a moderately variable profile and medium chinook salmon density. (c) Hoocheekoo Creek is low gradient with a highly variable profile and high chinook salmon density. (d) Bonanza Creek was previously placer mined, has medium gradient but a uniform profile similar to the steeper gradient streams.


(a)
Quebec Creek
(b)
Croucher Creek Upper
Elevation above arbitrary point (m)
Distance downstream (m)
0
1
2
3
0
50
100
150



(c)
Hoocheekoo Creek
(d)
Bonanza Creek
Elevation above arbitrary point (m)
Distance downstream (m)
0
1
2
3
0
50
100
150

Fig. 1.4. Correlation between juvenile chinook salmon density and selected thalweg metrics from 14 stream reaches. All correlations are significant (P < 0.05); correlation coefficients are provided in Table 1.2.

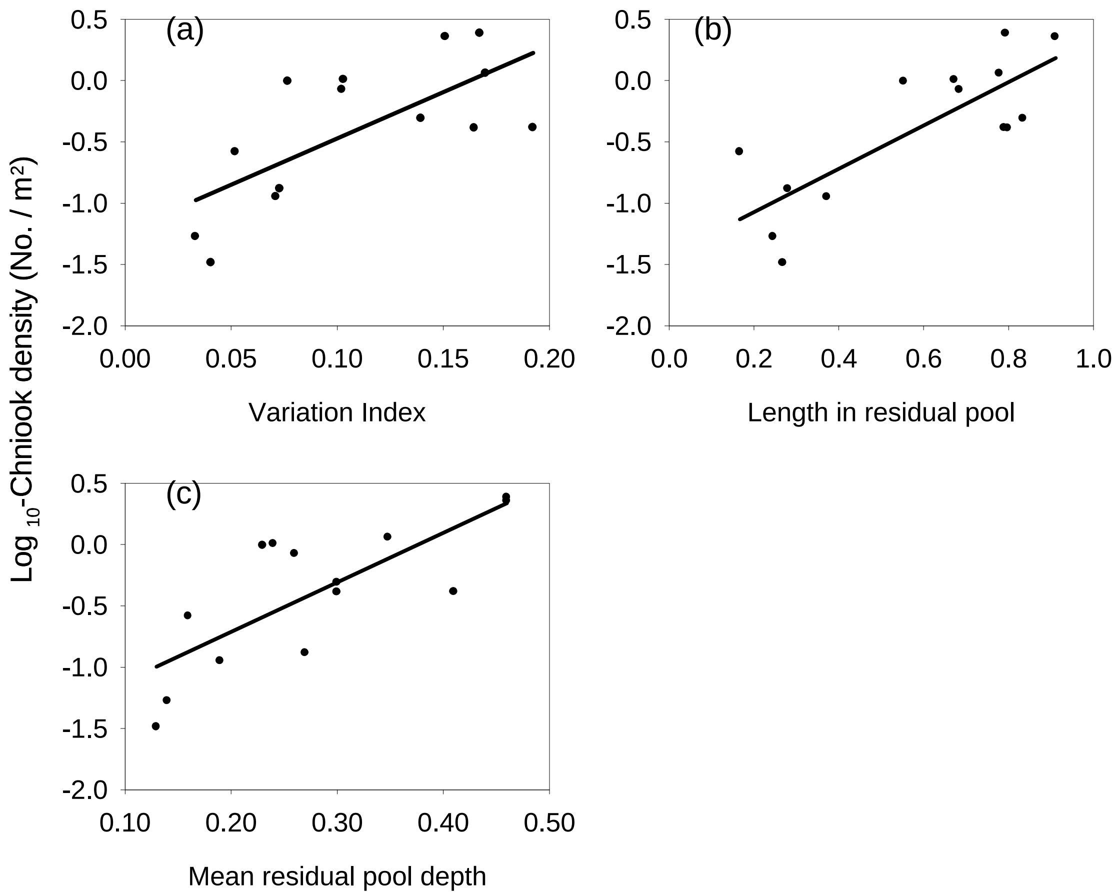


Fig. 1.5. Correlation between chinook salmon density and selected habitat variables in 14 stream reaches. All correlations are significant ($P < 0.05$) with the exception of plot (b) ($P = 0.051$). Correlation coefficients are provided in Table 1.2.

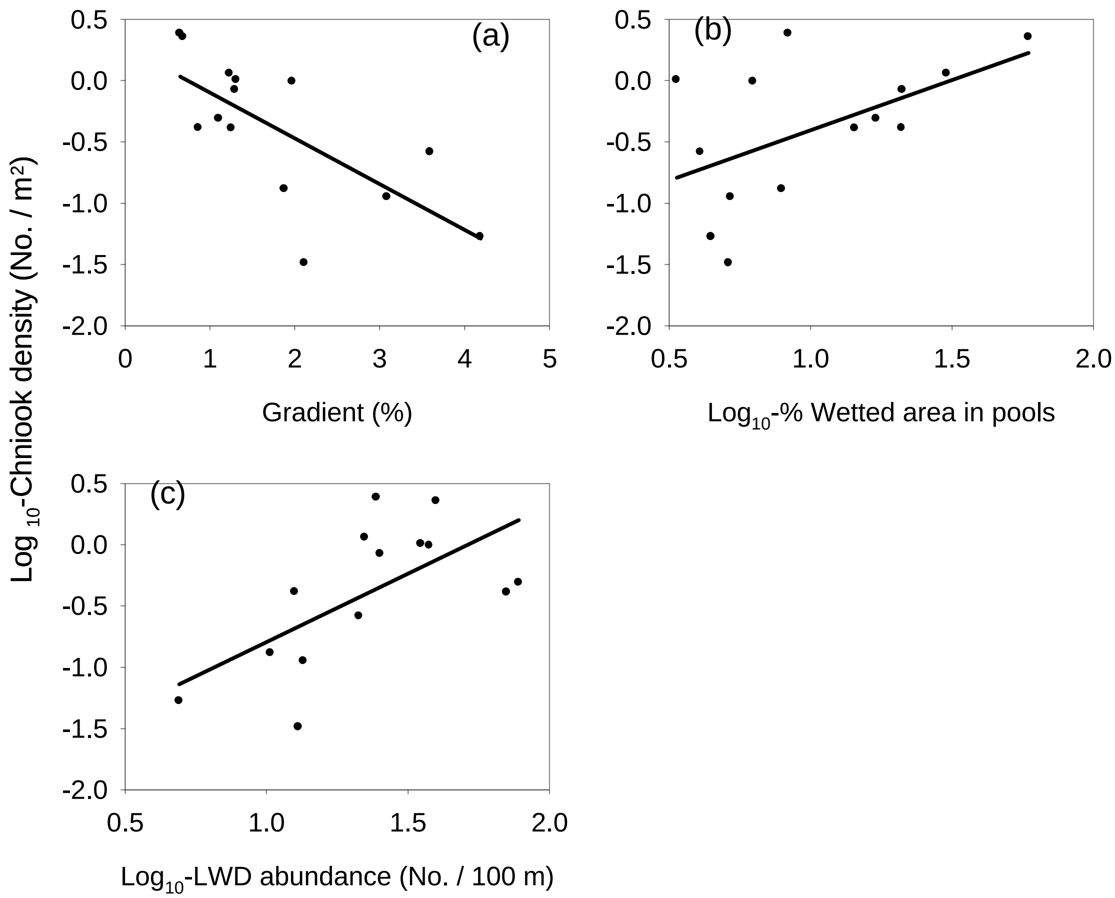


Fig. 2.1. Number of LWD pieces by species for 15 Yukon stream reaches. Pieces labelled 'Unknown' or 'Unknown Deciduous' could not be identified to species.

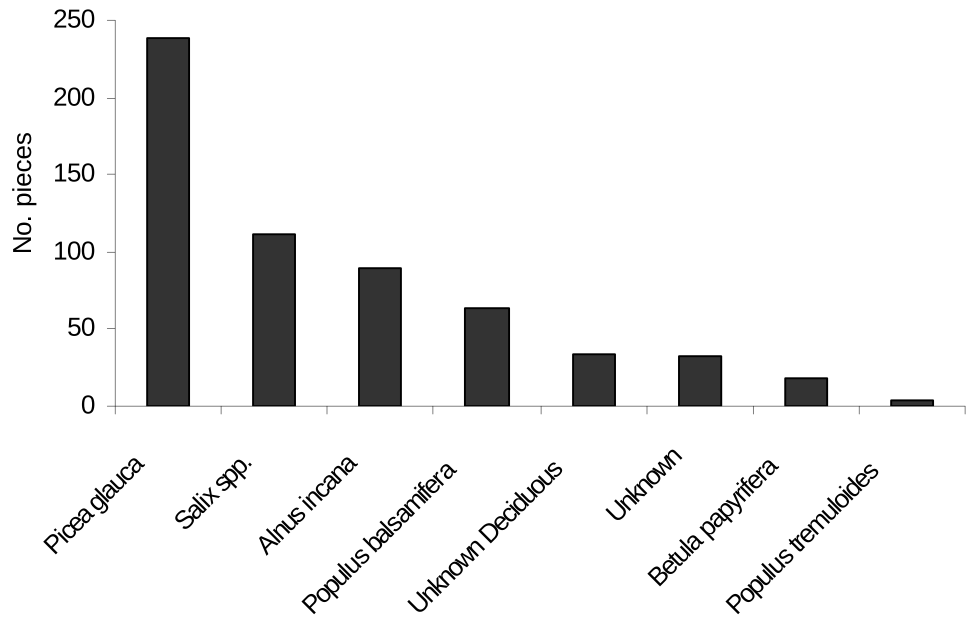


Fig. 2.2. Plots for selected regression analyses. Data are for 15 Yukon stream reaches except for plot (d), which is for 14 reaches. All correlations are significant (P < 0.05); statistics are provided in the text. Mean diameter and length are geometric means.

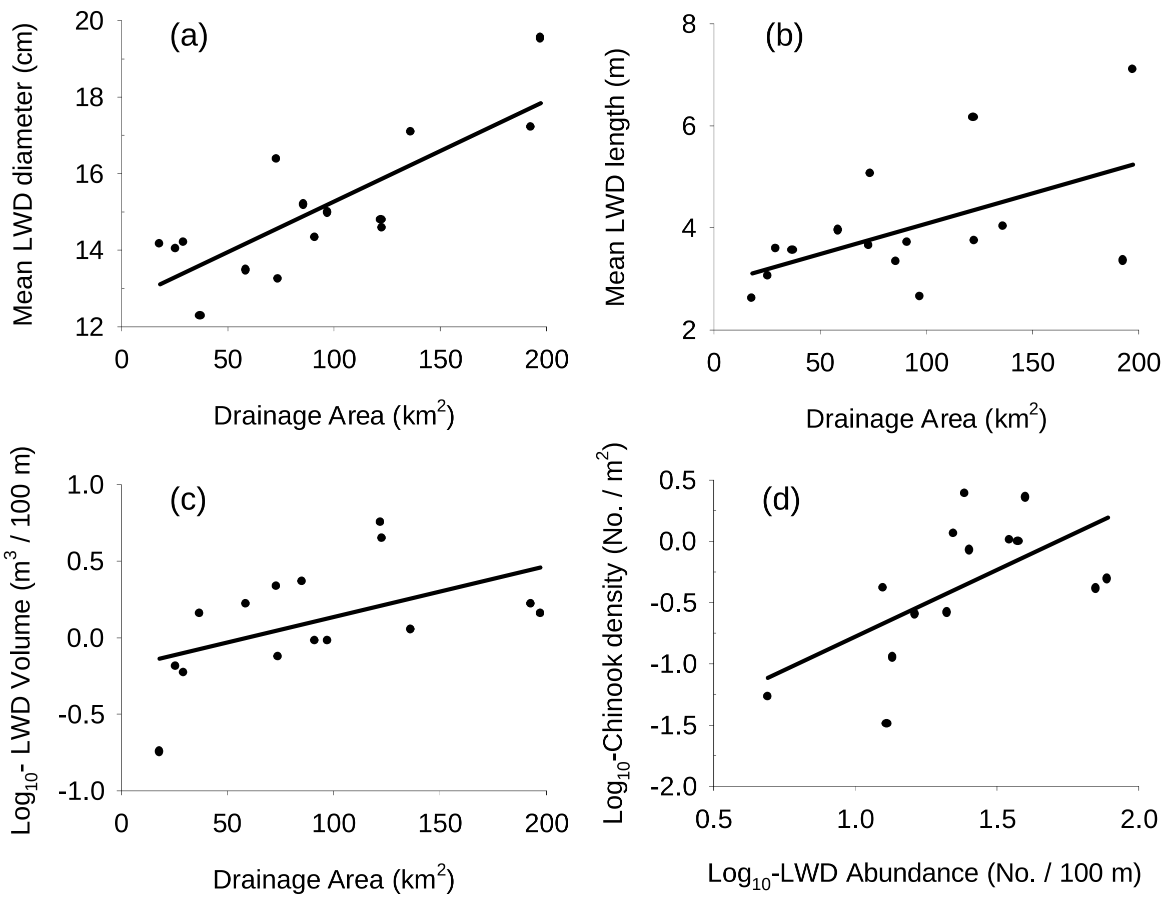


Fig. 2.3. Relation between diameter at stump height and tree age estimated from ring counts on 87 fallen riparian trees on the banks of Yukon streams. Regression equation: Log_{10}-age = 0.96 + 0.80 x log_{10}-diameter.

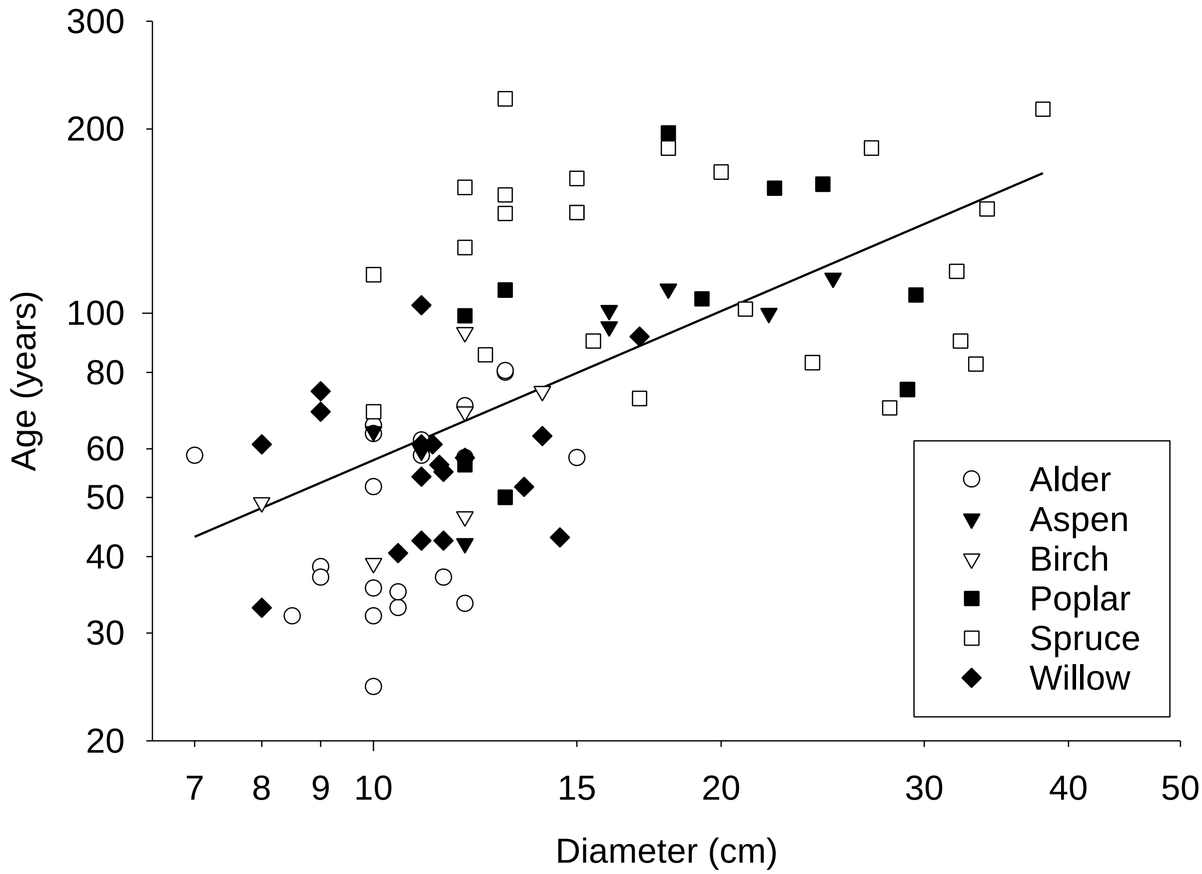
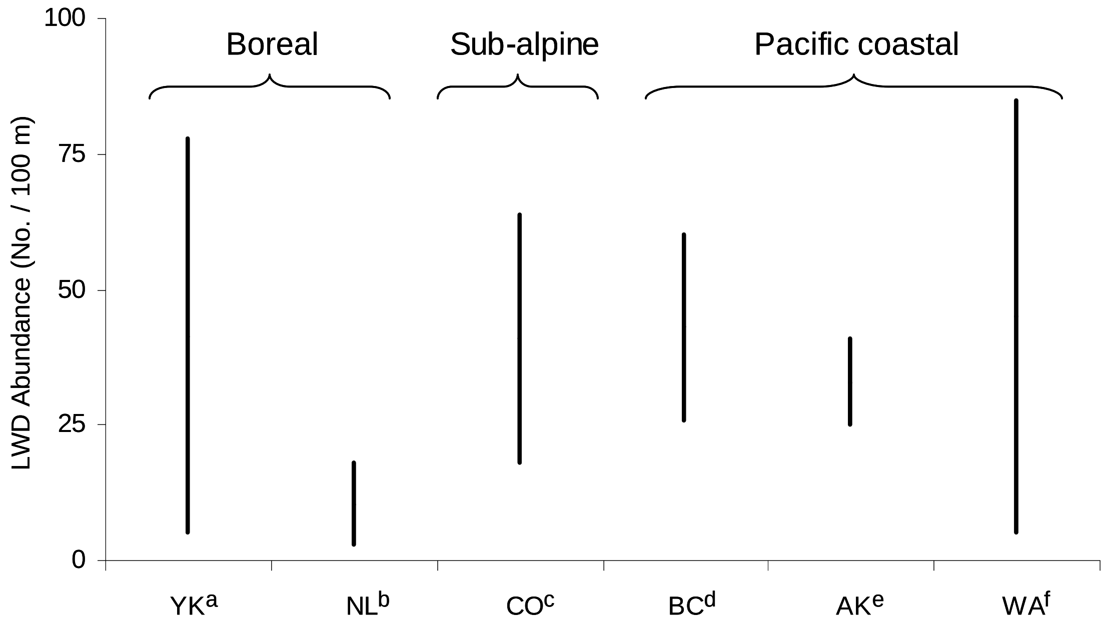
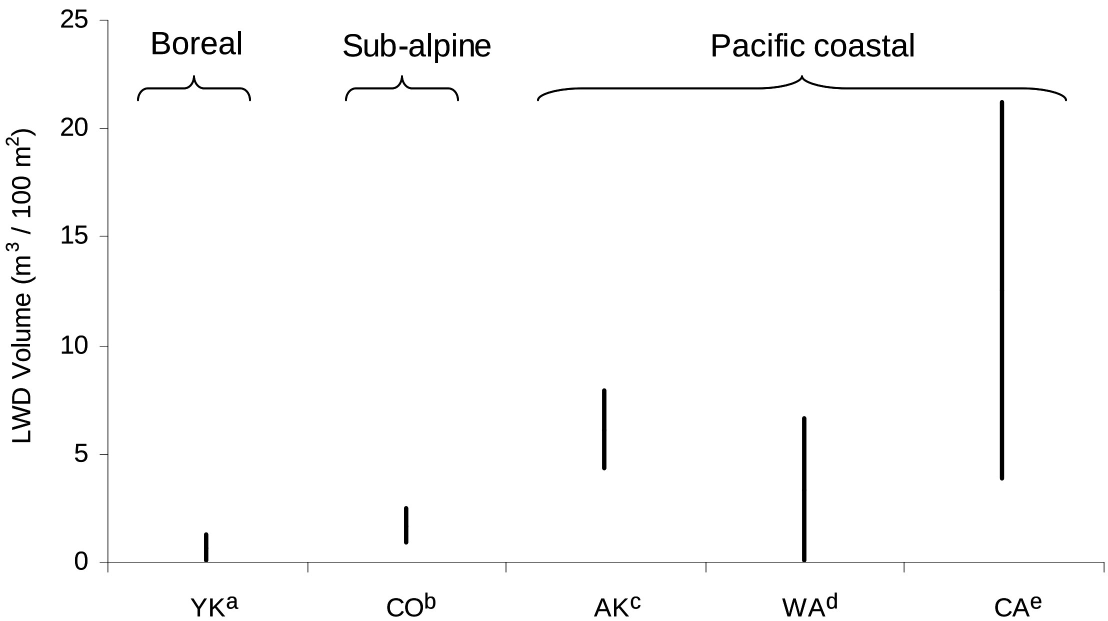


Fig. 2.4. LWD abundance for Yukon (YK) streams in the present study (range of values for individual streams) and those reported for undisturbed streams from the following regions: Newfoundland (NL); Rocky Mountains, Colorado (CO); Vancouver, BC (BC); coastal southeast Alaska (AK); and northwest Washington (WA). These data include only those streams that are of similar size and gradient to those in the present study (i.e., within the range of gradients and drainage areas in Table 2.1). Comparisons are approximate because studies used different size criteria for LWD, as indicated in the notes below.



[a] Present study — LWD $\geq$10 cm /1 m
[b] Clarke et al. (1998) — LWD $\geq$8 cm /any length
[c] Richmond and Fausch (1995) — LWD $\geq$10 cm /1 m
[d] Fausch and Northcote (1992) — LWD $\geq$10 cm /1 m
[e] Robison and Beschta (1990) — LWD $\geq$20 cm /1.5 m
[f] Beechie and Sibley (1997) — LWD $\geq$10 cm /2 m

Fig. 2.5. LWD loading volume (m^3/100 m^2) for Yukon (YK) streams in the present study (range of values for individual streams) and those reported for undisturbed streams from the following regions: Rocky Mountains, Colorado (CO); coastal southeast Alaska (AK); northwest Washington (WA); and northwest California (CA). These data include only those streams that are of similar size and gradient to those in the present study (i.e., within the range of gradients and drainage areas in Table 2.1). Comparisons are approximate because studies used different size criteria for LWD, as indicated in the notes below. However, size criteria will have a small effect on LWD loading because larger pieces make up most of the total volume.



[a] Present study	LWD ≥10 cm /1 m
[b] Richmond and Fausch (1995)	LWD ≥10 cm /1 m
[c] Robison and Beschta (1990)	LWD ≥20 cm /1.5 m
[d] Beechie and Sibley (1997)	LWD ≥10 cm /2 m
[e] Keller et al. (1995)	LWD ≥10 cm /unspecified length

Appendices

Appendix 2.1. Pool types in 13 Yukon streams. All terms except for 'under scour pool' and 'free-formed pool' are from Bisson et al. (1982). No trench pools were present because there were no bedrock walls. 'N' is the number of pools for a given pool type across all 13 streams, 'Area' is the total wetted area for all pools of a given pool type,'% Total pool area' is the percentage of the total pool area made up by a given pool type. 'Average pool area' is the average wetted area of individual pools.

Pool type	Formation and characteristics	N	Area (m^2)	% Total pool area	Average pool area (m^2)
Backwater	Eddies behind obstructions	162	274	26	1.7
Free-formed	Not formed by an obstruction	21	618	59	29.4
Under scour	Scoured by flow forced under an obstruction	8	57	5	7.2
Lateral scour	Scoured from flow deflected towards one bank by an obstruction	2	29	3	14.6
Dammed	Impounded upstream of an obstruction	14	27	3	2.0
Trench	Long with bedrock walls	0	0	0	-
Secondary channel	Pools in a secondary channel	1	6	1	5.9
Plunge	Scoured by flow falling vertically over an obstruction	4	32	3	7.9

Appendix 2.2. Characteristics of pool-forming LWD in 13 Yukon streams. Abbreviations for 'Pool type': BP = backwater pool, DP = dammed pool, LSP = lateral scour pool, SP = under scour pool, PP = plunge pool and SCP = secondary channel pool. 'Piece type' includes 'single' pieces which are not part of a jam, and '1° jam' and '2° jam' pieces, which are primarily and secondarily responsible for the structure of a LWD jam. Species abbreviations: Al = grey alder, Sp = white spruce, Wil = willow, Pop = poplar, Asp = aspen, Bir = paper birch and 'Dec?' = deciduous, unknown species. Pieces without a diameter indicated are rootwads. Shaded cells indicate that two pools were formed by the same piece or jam (e.g., a dammed pool upstream of the jam and a backwater pool downstream).

Stream	Pool type	Pool #	Pool area (m^2)	Piece type	Species	Diameter (cm)	Total length (m)	Total Volume (m^3)	Rooted
Caribou	BP	P14	1.0	Single	Wil	13	1.58	0.019	Yes
Croucher Lower	BP	P3	1.8	Single	Sp	40	23.50	1.031	No
Croucher Lower	BP	P5	3.0	Single	Sp	18	6.30	0.084	No
Croucher Lower	DP	P6	2.6	Single	Sp	36	8.70	0.517	Yes
Croucher Lower	LSP	P1	9.8	Single	Sp	40	23.50	1.031	No
Croucher Lower	SP	P11	17.8	Single	Sp	15	5.00	0.057	No
Croucher Upper	BP	P3	4.5	Single	Al	10	5.63	0.023	Yes
Croucher Upper	BP	P14	4.8	Single	Al	11	3.49	0.030	Yes
Croucher Upper	BP	P7	0.2	Single	Sp	17	9.50	0.112	Yes
Deadwood	BP	P2	1.6	Single	Pop	10.5	4.80	0.020	Yes
Deadwood	DP	P13	0.5	Single	Wil	10	5.29	0.020	No
Ensley	BP	P9	1.1	Single	Wil	19	5.46	0.069	Yes
Ensley	DP	P8	2.3	Single	Wil	19	5.46	0.069	Yes
Garner	BP	P3	1.4	Single	Al	10	4.04	0.020	Yes
Garner	DP	P8	0.8	Single	Pop	35	8.92	0.511	Yes
Garner	PP	P9	1.0	Single	Wil	23	6.45	0.118	Yes
McCabe Lower	BP	P2	1.8	Single	Asp	29	12.40	0.429	No
McCabe Upper	BP	P3	4.2	Single	Sp	39	5.60	0.450	No
McCabe Upper	BP	P4	5.1	Single	Pop	22.5	5.40	0.112	No
Mechem	SP	P9	5.6	Single	Sp	17	12.10	0.105	Yes
Stony	BP	P3	4.8	Single	Sp	11	2.00	0.017	No
Williams	BP	P5	2.5	Single	Dec?	19	1.00	0.024	Yes
Baker	BP	P15	4.2	1^0 jam	Pop	48	3.16	0.340	Yes
Baker	BP	P15	4.2	2^0 jam	Wil	15	2.46	0.021	No
Baker	DP	P14	2.0	1^0 jam	Pop	48	3.16	0.340	Yes
Baker	DP	P14	2.0	2^0 jam	Wil	15	2.46	0.021	No
Baker	BP	P7	1.5	1^0 jam	Wil	18	8.48	0.060	Yes
Baker	BP	P7	1.5	2^0 jam	Wil	15	1.48	0.024	No
Baker	DP	P6	2.5	1^0 jam	Wil	18	8.48	0.060	Yes
Baker	DP	P6	2.5	2^0 jam	Wil	15	1.48	0.024	No

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Appendix 2.2 continued

Stream	Pool type	Pool #	Pool area (m^2)	Piece type	Species	Diameter (cm)	Total length (m)	Total Volume (m^3)	Rooted
Croucher Lower	BP	P14	2.1	1^0 jam	Sp	41	18.25	0.792	No
Croucher Lower	SP	P15	1.0	1^0 jam	Sp	31	15.40	0.370	No
Croucher Upper	BP	P1	1.1	1^0 jam	Al	11.5	7.43	0.023	Yes
Croucher Upper	BP	P2	1.0	1^0 jam	Sp	11	10.10	0.029	Yes
Croucher Upper	BP	P9	0.6	1^0 jam	Sp	21.5	13.20	0.643	No
Croucher Upper	LSP	P5	19.5	1^0 jam	Sp	38	21.60	0.645	Yes
Croucher Upper	LSP	P5	19.5	2^0 jam	Sp	18	4.30	0.076	No
Croucher Upper	PP	P8	0.4	1^0 jam	Sp	20	4.00	0.080	No
Croucher Upper	PP	P8	0.4	2^0 jam	Sp	16	12.27	0.101	Yes
Croucher Upper	BP	P10	0.5	1^0 jam	Wil	10	3.88	0.023	Yes
Deadwood	DP	P29	0.6	1^0 jam	Sp	50	1.04	0.115	No
Deadwood	DP	P29	0.6	2^0 jam	Al		1.32	0.226	No
Ensley	BP	P7	6.7	1^0 jam	Wil	10	2.72	0.012	Yes
Ensley	BP	P7	6.7	2^0 jam	Bir	20	9.00	0.086	Yes
Garner	BP	P4a	1.3	1^0 jam	Al	11	8.74	0.029	Yes
Garner	BP	P1	3.2	1^0 jam	Sp	14	2.00	0.025	Yes
Garner	BP	P7	1.5	1^0 jam	Sp	36	17.93	0.535	No
Garner	DP	P6	1.3	1^0 jam	Sp	36	17.93	0.535	No
Hoocheekoo	SP	P4	11.31	1^0 jam	Pop	26	4.80	0.208	No
McCabe Upper	SP	P1	8.6	1^0 jam	Pop	31	12.57	0.338	No
McCabe Upper	SP	P1	8.6	2^0 jam	Sp	35	2.93	0.214	No
Mechem	DP	P2	0.5	1^0 jam	Pop	17	3.61	0.207	No
Mechem	BP	P8	0.5	1^0 jam	Sp	21	8.24	0.136	Yes
Mechem	BP	P8	0.5	2^0 jam	Sp	16	8.17	0.085	Yes
Mechem	SP	P7	0.5	1^0 jam	Sp	21	8.24	0.136	Yes
Mechem	SP	P7	0.5	2^0 jam	Sp	16	8.17	0.085	Yes
Mechem	BP	P3	1.9	1^0 jam	Wil	13	6.44	0.051	Yes
Mechem	BP	P3	1.9	2^0 jam	Wil	11	5.04	0.022	Yes
Mechem	BP	P4	1.1	1^0 jam	Wil	10	5.23	0.012	Yes
Mechem	BP	P4	1.1	2^0 jam	Al	12	2.00	0.021	No
Mechem	BP	P6	1.6	1^0 jam	Wil	13	6.33	0.064	No
Mechem	BP	P6	1.6	2^0 jam	Wil	11	7.65	0.064	No
Mechem	SCP	P10	5.9	1^0 jam	Wil	22	5.36	0.077	Yes
Mechem	SCP	P10	5.9	2^0 jam	Wil	11	6.84	0.023	Yes
Merrice	PP	P1	27.36	1^0 jam	Asp	33	11.40	0.605	No
Merrice	PP	P1	27.36	2^0 jam	Asp	21	13.50	0.193	No

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Appendix 2.2 continued

Stream	Pool type	Pool #	Pool area (m^2)	Piece type	Species	Diameter (cm)	Total length (m)	Total Volume (m^3)	Rooted
Quebec	BP	P10	0.2	1^0 jam	Al	16	5.86	0.056	No
Quebec	PP	P1	3.0	1^0 jam	Sp		1.00	0.087	No
Quebec	PP	P1	3.0	2^0 jam	Sp		1.00	0.084	No
Quebec	BP	P19	0.5	1^0 jam	Wil	10	6.25	0.015	Yes
Quebec	BP	P19	0.5	2^0 jam	Wil	10	6.05	0.014	Yes
Stony	BP	P2	6.0	1^0 jam	Sp	10	3.70	0.024	No
Stony	BP	P2	6.0	2^0 jam	Sp	13	2.20	0.025	No
Stony	BP	P4.5	6.8	1^0 jam	Sp	37	3.45	0.277	No
Stony	BP	P5	11.0	1^0 jam	Sp	14	2.50	0.033	No
Stony	DP	P1.5	11.0	1^0 jam	Sp	10	3.70	0.024	No
Stony	DP	P1.5	11.0	2^0 jam	Sp	13	2.20	0.025	No
Stony	SP	P1	8.5	1^0 jam	Sp	28	4.70	0.195	No
Williams	SP	P7	4.1	1^0 jam	Wil	18	6.00	0.057	Yes
Williams	SP	P7	4.1	1^0 jam	Sp	16	2.30	0.038	No

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