EXAMINING THE IMPACTS OF STREAM CHANNELIZATION ON SALMONID AND AQUATIC INVERTEBRATE COMMUNITIES OF A FIFTH-ORDER MONTANE RIVER

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Abstract

Stream channelization is a common form of anthropogenic disturbance, whose impacts on cold water salmonid communities have received little attention in comparison to the body of work demonstrating its negative effects on low land, warm water systems. Here, I compared the effects of stream channelization on fish and invertebrate communities and their habitats in disturbed and undisturbed cold-water mountain streams in southern Alberta. I demonstrate that stream channelization has imposed significant alterations to stream habitat, most notably a loss of deep habitat, and that these alterations have led to a statistically significant decline in abundance and biomass of Rainbow Trout and Mountain Whitefish, as well as a significant decline aquatic invertebrate biomass. Because of the importance of monitoring fish abundance in deep pools, I used snorkel surveys instead of electrofishing. In shallow, disturbed streams, however, snorkel surveys may be less effective and should be used with caution.

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Chapter 1

The Issue of Stream Channelization, with a Focus on the Crowsnest River

1.0 Abstract

Mountain streams evolve over time alongside a suite of geomorphological processes, which govern the creation of habitat and provide stream residents with the conditions and resources necessary to persist and thrive. However, where anthropogenic demand comes into conflict with such streams, their natural properties are often compromised, imposing significant effects on aquatic communities. An example of this conflict can be found on the Crowsnest River, where stream channelization resulting from European settlement has altered a significant portion of the river's main stem. In order to properly manage such systems and ensure the long term persistence of aquatic biota, it is crucial that we have a complete understanding of the relationship between aquatic organisms and their habitat, the natural processes which give rise to such habitat, and how these have two previous components may be altered by the disturbance in question. Here, I provide a detailed review of past works describing the processes that create a variety of stream habitat in natural streams, and why these habitats are crucial to the various salmonid and invertebrate species which inhabit the Crowsnest River. I also provide a review of the research which has been done to demonstrate the effects of channelization on aquatic habitat and communities, as well as past attempts to mitigate such deleterious impacts.

1.1 Site Description

The Crowsnest River is a fifth order stream (Strahler Method - 1:50 000 resolution), which drains eastward along the foothills of the Rocky Mountains, in Southern Alberta. It is located within the Oldman River Watershed, where it begins at the mouth of Crowsnest Lake and flows to the Oldman Reservoir. The study area includes the section of the river from Crowsnest Lake (elev. 1355 m) to Lundbreck Falls (elev. 1172 m), which is an impassable barrier for fish located 9 km upstream from the Oldman Reservoir. This 45 km section passes through Coleman, Blairmore, Frank, Bellevue and Hillcrest. At baseflow, the wetted width of the river is 11.9 m, with a thalweg depth of 0.86 m at its head, and widens and deepens downstream to 21.7 m and 1.19 m (Blackburn 2010). The thermal regime varies little along the study area (MacDonald 2011).

The Crowsnest River, above Lundbreck Falls, once supported a salmonid community consisting of Westslope Cutthroat Trout *Oncorhynchus clarki lewisii*, Mountain Whitefish *Prosopium williamsoni*, and Bull Trout *Salvelinus confluentus*, which is the typical native community for Southern Alberta headwater streams. In the 1930-1940's, however, Rainbow Trout *Oncorhynchus mykiss*, were stocked intensively and the native cutthroat trout became restricted to the upper tributaries; bull trout are now found only below Lundbreck Falls, leaving mainly rainbow trout and mountain whitefish in the study area. Other species such as Lake Trout *S. namaycush*, Brook Trout *S. fontinalis* and Brown Trout *Salmo trutta* have also been introduced and, along with the few remaining Westslope Cutthroat Trout, make up a small fraction of the current salmonid community (Blackburn 2010).
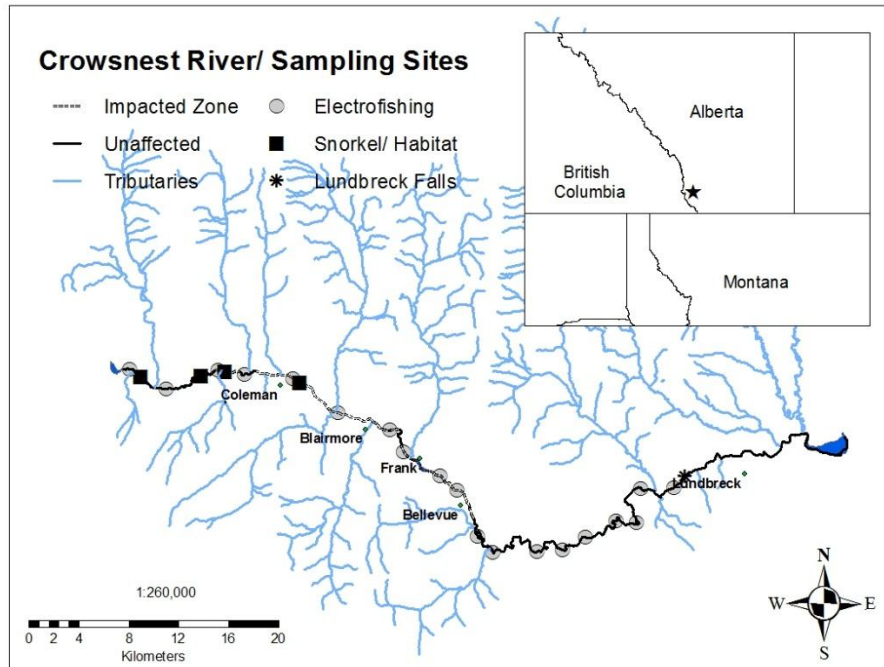
*Electrofishing conducted by Alberta Conservation Association

Figure 1-1. Map of Crowsnest River, Study Reaches and Impacted Zone.

Since the beginning of European settlement in the late 1800's, the Crowsnest River underwent a series of changes, associated with the building of the Canadian Pacific Railroad, and the development of coal mining communities in the river valley. As a result, large segments of the river have been diverted into artificial channels, greatly altering the aquatic habitat. Impacted reaches are found throughout the study area, from Coleman to Hillcrest (Fig 1-0).

1.2 Introduction to Crowsnest River Taxa and Community Structure

1.2.1 Invertebrate Community

The Crowsnest River, in its unaltered reaches, presents a heterogeneous compliment of habitat featuring a variety of substrates, depths and flow velocities, providing a suitable environment for a diversity of aquatic invertebrates. The aquatic invertebrate community

of the Crowsnest River is dominated by members of the orders Ephemeroptera,

Plecoptera, Trichoptera, Diptera and Oligochaeta, and also feature in less abundance,

various members of Amphipoda, Coleoptera, Hirudinea, and rarely, Megaloptera

(personal observation). Aquatic invertebrates are classified based on characteristics

describing the adult life stages of invertebrates, and as such, during the larval stage of the

life cycle, great variance may exist in the habitat or food requirements of members of the

same taxonomic family (Resh and Rosenberg 1984). The aquatic invertebrates of the

Crowsnest River will be treated in terms of four functional feeding groups (FFG):

scrapers, shredders, collectors and predators (Cummins 1973, Cummins and Klug 1979).

 Aquatic invertebrates belonging to the scraper FFG feed by scraping algae and detritus

from the surfaces of large substrates, facilitated by specialized mouthparts. An example

of such specialization is the lining of stiff hairs along the labial or maxillary palps in

some Ephemeropterans and Plecopterans (Resh and Rosenberg 1984). In addition to their

role as primary consumers in the river food web, scrapers dislodge particulate organic

matter making it available for other organisms, such as collectors (Clifford 1991). The

most abundant scrapers in the Crowsnest River are members of the families

Oligoneuridae and Limnephilidae. Although not much is known about the specific habitat

preferences of these families, scrapers are generally found in greater abundance in

shallow areas of streams, in both fast and slow moving conditions (Clifford 1991).

The shredder FFG feed on coarse plant material matter, mostly leaves and other detritus

from terrestrial sources, which at certain times of the year can make up a significant

fraction of the organic matter in the river (Graffius and Anderson 1980). Like scrapers,

the shredders can generate a resource subsidy for other feeding groups by breaking down

coarse particulate matter to a size suitable for smaller organisms (Cummins and Klug 1979). Shredders in the Crowsnest River include members of the families of Perlidae, Chloroperlidae, Pteronarcidae and Tipulidae. The most abundant shredders in this system are those members belonging to the order Plecoptera, however, Cummins and Klug (1979) noted that as they increase in size, Plecopterans such as Perlidae and Chloroperlidae will become increasingly more predacious, shifting them into a separate FFG. The habitat preferences of shredders are quite variable. Due to high oxygen requirements, members of Plecoptera are likely to be most abundant in swift moving oxygen rich flows and larger, cleaner substrate (Cummins 1973, Brusven and Prather 1974), while members of Diptera within this FFG are more tolerant of low oxygen levels, and may be found in greater abundance in slower moving, depositional habitats (Clifford 1991).

The collector FFG feed on the fine particulate matter found throughout most habitat types. Fine particles of algae, detritus, and bacteria can be collected from free flowing sources in the water column, or from deposited sources, either in the interstitial areas of larger substrates or on the bottom of pools and backwaters (Resh and Rosenberg 1984). Collectors have developed a variety of strategies to exploit this resource, and can be further subdivided into two groups based on the collection methods they utilize (Cummins 1973, Cummins and Klug 1979). The first group are the deposit-collectors, which are commonly associated with slower water velocities and finer substrates (Hynes 1970), typical of most depositional habitats. The deposit-collectors are well represented in the Crowsnest River by the families Brachycentridae, Baetidea, Ephemerellidae, Oligoneuridae, Siphloneuridae, Naididae, Lumbriculicidae and Chironomidae. The filter-

collectors are the other group, and as implied by their name feed on the particulate matter

which is free flowing in the water column. This sub-group is represented by the Dipteran

Simuliidae and the caddisfly Hydropsychidae in the Crowsnest River. Hydropsychids

construct capture nets from spun silk, which they place perpendicular to the flow of the

stream to capture particulate matter suspended in the stream flow, and as such generally

prefer faster moving flows which increase the delivery of food resources (Wallace and

Merritt 1980, Clifford 1991, Jowett et al. 1991). They have also been shown to select for

larger substrate sizes (12-25 mm) rather than pebbles or sand (Brusven and Prather 1974),

possibly due to greater suitability for anchoring nets. Simuliidae also use spun silk,

however rather than constructing nets, they use their silk to anchor themselves to

substrate while positioning their posterior ends in the flowing water where they collect

free flowing particulate matter with specialized filtering fans on their mouthparts

(Clifford 1991).

The fourth FFG are the predators, which feed primarily on primarily on other aquatic

invertebrates, and are thus secondary, and not primary consumers, like most other aquatic

invertebrates (Resh and Rosenberg 1984). Predators are often large, and rapidly growing

and active making them important prey for salmonids (Cummins and Klug 1979, Clifford

1991). In general, members of this FFG are intolerant of low oxygen concentrations in

the water, and are therefore most commonly found in swift flowing oxygen rich habitats

featuring clean substrates (Jenkins et al. 84), such as riffles or runs. One exception to this

would be the largely predacious family of leeches, Glossiphonidae, commonly found in

the oxygen poor debris accumulations in rivers (Clifford 1991). Large predatory

invertebrates may select for clean substrate with large interstitial spaces which serve as

refugia from larger predators such as salmonids. Brusven and Rose (1981) observed an increase in invertebrate predation by salmonids in substrates with decreasing pore size, supporting this (Brusven and Rose 1981). The predator FFG in the Crowsnest River is comprised mostly of the stoneflies Perlidae, Chloroperlidae, Perlodidae and the mayfly Heptageniidae, with other families such as Rhyacophilidae and Glossiphonidae occurring in less abundance (personal observation).

1.2.2 Salmonid Community

The Crowsnest River supports a highly productive salmonid community, dominated by native mountain whitefish and introduced rainbow trout, which account for approximately 95% of the total fish community (Blackburn 2010). *O. mykiss* occur naturally along the Pacific Coast, as well as inland through the Peace and Liard River drainages in British Columbia and the Athabasca River drainage in the Northern Alberta (Nelson and Paetz 1992). The widespread stocking of *O. mykiss* over the past century, however, has led to introduced communities occurring in most streams and rivers across Southern Alberta, including the Crowsnest River. The native home range of *P. williamsoni* encompasses most of Alberta, and extends west throughout British Columbia and south as far as Nevada. As such, *P. williamsoni* occur naturally in the Crowsnest River (Nelson and Paetz 1992).

At fall low flow, *O. mykiss* are primarily foraging on invertebrate drift, and will occupy habitats which present optimal foraging opportunities. Juvenile *O. mykiss* are commonly found in habitats such as shallow runs or riffles, and as their size increases so too will the size of the habitat they select for (Grant and Kramer 1990). Larger, mature individuals

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belonging to the invertebrate orders Ephemeroptera, Plecoptera and Trichoptera (EPT) are thought to be an important component of the diet of *O. mykiss* (Tippets and Moyle 1978). It follows that habitat types which optimize EPT abundance likely provide the greatest foraging opportunities for *O. mykiss*.

During fall low flow *P. williamsoni* tend to feed more directly on benthic, rather than drifting, invertebrates (Pontius and Parker 1973, DosSantos 1985). This was observed in *P. williamsoni* in the Sheep River, Alberta, which fed mostly within 10 cm of the stream bed and never from the surface (Davies and Thompson 1976). Juvenile *P. williamsoni* have been found to be most abundant in shallower habitats of streams and in areas of lower velocity (Pettit and Wallace 1975), and similar to *O. mykiss*, individuals will seek out larger and deeper habitats as they increase in size (McPhail and Troffe 1988). Several studies have reported strong schooling behaviour in *P. williamsoni* communities, observing clusters of fish representing multiple age/ size groups congregating in deep pools (Davies and Thompson 1976, McPhail and Troffe 1988), making these important habitats for *P. williamsoni* communities. This preference for slower habitats coincides with a preference for these habitat conditions exhibited by the invertebrate family chironomidae, a major food source of *P. williamsoni* (Pontius and Parker 1973, Overton et al. 1978, McPhail and Troffe 1988).

Where two species which occupy similar niches are brought to co-exist and niche overlap occurs, competition for optimal habitat or food resources is likely to arise, and such is the case in many mountain streams where *O. mykiss* have been introduced (Fausch 1988). *O. mykiss* are infamously known for their ability to outcompete and displace native coldwater salmonids, attributed to physiological and morphological differences such as

higher growth rates and the ability to tolerate relatively warm temperatures (Bear et al. 2007, Seiler and Keeley 2009). Such competitive advantages have led to the well documented, widespread displacement of many native coldwater salmonid populations along the Eastern slopes of the Canadian Rocky Mountains (Fausch 1988, Gresswell 1988, Donald and Alger 1993). In the Crowsnest River, the introduction of *O. mykiss* paired with over the fishing of native species led to the ultimate displacement of native Westslope Cutthroat and Bull Trout (Blackburn 2010). However, despite the decline of these two native species, *P. williamsoni*, which are also native to the Crowsnest River, continue to thrive with *O. mykiss*. The persistence of *P. williamsoni* may be explained by niche segregation. For example, in a coldwater Newfoundland stream, *O. mykiss* occupied faster velocities in the open channel, whereas Brook Char preferred slower water velocities and greater cover opportunities, such as pools, enabling the sympatric trout to co-exist without competing (Cunjak and Green 1983). Like the Brook Char, *P. williamsoni* prefer pools to faster open channel habitats, potentially enabling them to co-exist with introduced *O. mykiss*. In three Montana streams *P. williamsoni* and *O. mykiss* were able to co-exist largely due to differences in the diets of adults despite major overlap in juvenile diets (DosSantos 1985). This study may provide further insight to the mechanisms which allow sympatric *O. mykiss* and *P. williamsoni* to co-exist in the Crowsnest River.

1.3 Stream Channelization

1.3.1 History of the Crowsnest

The Crowsnest River runs through the municipality of the Crowsnest Pass (Coleman, Blairmore, Frank, Bellevue and Hillcrest) whose history reflects the development of the

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railroad, and the exploitation of lumber, coal and other resources in this watershed. Initially, the railway resulted in major alterations to the meandering river channel, creating many channelized sections (C.P.H.S. 1979), and secondary alterations then came from flood control measures associated with urban development. In 1909, a retaining wall along the east bank of Lyon Creek (C.P.H.S. 1979) near the growing municipality of Blairmore, and in the following 30 years further flood control measures involving further channelization. The straightened channels were built adjacent to the flood prone meandering reaches of the river, and were used to divert water out of these natural flood prone reaches. As a result, the Crowsnest River channel has lost a considerable portion of its natural meander.

1.3.2 Ecological Consequences of Channelization and the Importance of Habitat

Anthropogenic disturbances and their residual effects, while altering the natural state of our streams and rivers, are a driving force in aquatic research and fisheries management. Such disturbances, constitute experiments that can yield valuable insight into ecosystem function, revealing important mechanisms of river function. For the manager, concerned for the productive capacity of the system, anthropogenic disturbances, though representing major ecosystem stresses, have led to innovative measures of mitigation and river management. Indeed, much attention has been focussed on stream channelization by researchers and managers alike (Nunnally and Keller 1979, Brookes et al. 1983, Brookes 1985, Brookes 1987a, Moerke et al. 2004).

Stream channelization, the artificial straightening of an existing stream, or the diversion of flow from a naturally existing stream into a straight, man-made channel, is used to

divert water from the river flood plain (Emerson 1971). Besides the most obvious effect shortening reach length and reducing sinuosity (Hansen 1971, Brookes 1987a), other impacts such as removal of bankside vegetation, widening and deepening of the channel profile, and homogenization of substrate have also been associated with channelization either as a direct result of the stream manipulation itself, or as a secondary result of the altered channel processes, many of which only appear decades later (Chapman and Knudsen 1980, Reily and Johnson 1982, Brooker 1985, Williamson et al. 1992, Landwehr and Rhoads 2003, Moerke et al. 2004, Lau et al. 2006, Pedersen 2009, Duncan et al. 2011). Direct reductions in available fish and invertebrate habitat resulting from reduced stream length are quite easy to quantify, as the amount of stream length lost translates to a direct unit of habitat area lost for lotic organisms (Brooker 1985, Cramer and Ackerman 2009). Straightening can also produce many indirect effects on productive capacity (Montgomery and Buffington 1997), for example, those associated increasing the elevational gradient (slope) (Brooker 1985, Brookes 1987a). A stream's natural gradient is an evolved outcome of the stream's geomorphological history, and tends to a stable state (Williamson et al. 1992, Rabeni and Jacobson 1993, Hooke 2008). Increasing the gradient (slope) of a stream generally increases flow velocity, triggering channel widening or lowering (incision or entrenchment), which tend to restore equilibrium (Rhoads 1990, Williamson et al. 1992, Landwehr and Rhoads 2003). Channel incision or entrenchment lowers the stream bed, and occurs when the ability of a stream to transport sediment increases, without a corresponding increase in sediment load. Although channel incision itself is not known to directly affect stream biota (Duncan et al. 2011), lowering the water table has been shown to reduce growth of bankside vegetation (Reily and

Johnson 1982), which is important for both fish and invertebrates by providing cover from sunlight and overhead predation, providing a nutrient supply in the form of allochthonous input, and by playing a role in thermal regulation (Beschta 1997, Wallace et al. 1997, Johnsson et al. 2004, Laeser et al. 2005, Roth et al. 2010). In a study examining the value of overhead cover with respect to predation risk in four salmonids, Johnsson et al. (2004) found that territory owners showed a significant preference for cover habitat, and also that owners of cover territories were more aggressive in defending those territories than those with non-cover territories (Johnsson et al. 2004). Channel incision also reduces connectivity to side channels, oxbows and other floodplain habitats, inherently limiting the available habitat for fish and invertebrates as well as disconnecting these organisms from potential resources.

The importance of allochthonous inputs as an energy subsidy for both aquatic invertebrate and fish communities has also been well documented (Wallace et al. 1997, Kawaguchi et al. 2003, Rasmussen 2010), as has the significance of bankside vegetation as a supplier of that subsidy (Laeser et al. 2005). Quinn et al. (1992) found a significant reduction in terrestrial litter input in streams which had been cleared of bankside vegetation due to channelization (Quinn et al. 1992), and others have found similar declines in instream large woody debris and coarse organic matter in correspondence to a loss of adjacent riparian vegetation (Gregory et al. 1991). Wallace et al.(1997) performed an experiment whereby they excluded terrestrial litter from a stream for a period of three years and noted a major drop in both abundance and biomass across a variety of invertebrate taxa (Wallace et al. 1997).

Overhanging vegetation also provides shade, lowering stream temperatures during daylight hours by reducing the effect of solar radiation (Beschta 1997, Roth et al. 2010). Modelling studies have shown that the removal of bankside vegetation can result in a 0.7°C increase in stream temperature, while the establishment of a dense riparian community can lower stream temperature by as much as 1.2°C (Roth et al. 2010). Furthermore, the effect solar radiation has on raising stream temperatures is magnified in streams susceptible to widening, due to an increase in surface area to volume ratio (Beschta 1997). With respect to biota, it has been shown that increasing stream temperatures can influence community structure by providing a competitive advantage for those species which are more tolerant of warmer waters, such as common invasive species like *O. mykiss* (Paul and Post 2001, Bear et al. 2007). In fact, due to higher thermal tolerances, *O. mykiss* gained a significant survival advantage over a native coldwater trout at warmer temperatures (Bear et al. 2007), illustrating the potential for shifts in community structure in correspondence to increased water temperature.

Streams also tend to re-establish equilibrium by widening, which occurs through bank erosion, which dissipates energy resulting from increased velocity horizontally through its banks (Nunnally 1978). Channelized stream banks are generally less stable than those of natural streams due to a lack of a riparian root matrix (Reily and Johnson 1982), and as such are very susceptible to stream bank erosion.

Bank erosion may also increase the sediment load of the stream, impacting the physical components of the lotic environment and incurring deleterious effects on the biological community (Kroes and Hupp 2010). Undercutting of banks, resulting from stream straightening and channel incision, can be a major l contributor to erosion and increasing

sediment load (Williamson et al. 1992). Increased sediment load in the channel flow also increases the amount of sediment available to deposit during low flow seasons (Landwehr and Rhoads 2003), altering the state of the stream subclass by either changing the particle size distribution of the stream bed, or by increasing the embededness of the substrate (Culp et al. 1983, Erman and Erman 1984). Stream bed alterations can greatly influence invertebrate community structure (Lenat et al. 1981, Minshall 1984, Zweig and Rabeni 2001). Low levels of deposition reduced invertebrate density, but had little effect on community structure, however, high levels of deposition led to an increase in invertebrate density, and a shift in community structure from an Ephemeroptera, Trichoptera, Plecoptera (EPT) dominated community to an Oligochaeta dominated community (Lenat et al. 1981). Similarly, EPT density and richness were negatively correlated to sediment deposition across study streams (Zweig and Rabeni 2001).

Erosion and deposition are also greatly influenced by flow regime as it relates to peak and low flows, which differ substantially among channelized and unaffected meandering reaches. Channelized stream reaches are engineered with high, often reinforced banks to reduce the access of flow to the flood plain (Emerson 1971, Brookes et al. 1983, Brookes 1987a). Thus, during peak spring runoff events, larger volumes of discharge remain within the main channel rather than being allowed to dissipate its energy over its banks, creating a system which is much more powerful (Rhoads 1990, Kroes and Hupp 2010). In contrast, during late summer or fall this effect of increased power is actually reversed due to the already low amount of flow being evenly spread over the monotonous channel profile (Nunnally 1978, Brookes et al. 1983). This is an important consideration with respect to seasonal fluctuations of stream flow and channelization, as channelized reaches

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feature a very homogeneous channel profile with little difference in depth or velocity across the stream (Emerson 1971, Keller 1976). Meandering reaches, however, are much more heterogeneous in profile, featuring some shallow depositional zones on the insides of bends, and fast flowing deeper sections to the outside margin of bends (Rabeni and Jacobson 1993, Fukushima 2001, Hooke 2008). During the low flow season, physical heterogeneity is crucial as these deeper sections retain sufficient power to transport the sediment within the water column, and in their absence, channelized reaches suffer a great reduction in sediment transport capability during low flow conditions, and can at these times be net depositional zones (Landwehr and Rhoads 2003).

Flow patterns play a key role in sediment transportation, but they are also crucial to the formation of habitat (Rabeni and Jacobson 1993). The force and patterns of flow in an unaffected meandering stream differ greatly from those in a channelized stream due to the presence of bends, and the interaction between channel flow and these bends is what gives rise to a variety of micro-habitat types (Rabeni and Jacobson 1993). For example, when the flow of a stream encounters a bend the greatest portion of its flow is concentrated to the outside, exposing the outer bank to stronger water velocities which induce erosion or scour effects. In opposition to this, the inside of the bend experiences a much more gentle flow, and is therefore characterized as a more depositional environment (Rabeni and Jacobson 1993).

Micro-habitat types associated with the outside of bends are generally deeper due to the scouring processes which give rise to them, and include such habitat as pools, deep runs and deep undercuts (Rosgen 1994). Deep pools are characterized by slower moving waters (despite the fact they are created by fast moving water), and offer salmonids good

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resting positions from higher velocities as well as cover from overhead predation (Pettit and Wallace 1975, Keller 1976, Smith and Brannon 2007). Pools tend also to have finer sediment resulting from net deposition during low flow seasons (Landwehr and Rhoads 2003), and as a result will host a different invertebrate community then what exists in adjacent habitats (Duan et al. 2009). It follows that deep pools offer not only an alternate food source for salmonids, but due to lower velocities may also favour benthic feeding strategies rather than drift feeding (Davies and Thompson 1976, DosSantos 1985).

Deep runs are created through similar processes as deep pools, however, runs are positioned in areas where flow is much greater, either on the downstream edge of lateral scour pools or mid channel where the thalweg creates a deeper profile (Rabeni and Jacobson 1993). These habitats feature greater velocities than pools, and are generally characterized by coarser substrates (Montgomery and Buffington 1997). Thus, deep runs support a different invertebrate community than exists in the finer substrate composition of pools (Jowett 2003, Jowett et al. 2005), supplying a different food source for salmonids. Furthermore, it has been shown that drift densities increase with increasing flow velocities (Waters 1965), and thus, deep runs may present an advantage for species which favour higher swimming velocities as well as drift feeding strategies.

Both deep pools and runs which form on the lateral edges of streams are also often associated with deep undercuts (Rabeni and Jacobson 1993). Undercut habitat is created when the bank of a stream is scoured away, but root masses of established bankside vegetation keep part of the bank intact, creating and overhanging ledge which offers increased overhead cover. Although little is known about how undercut habitat may affect invertebrate communities, it is well understood that many salmonid species show a

preference for the cover value provided by deep undercut banks (Magoulick and Wilzbach 1997, Myers and Resh 2000). Deep water habitats are very important to larger individuals of a community (Grant and Kramer 1990), and often permit fish to partition the water column in the vertical dimension (Cramer and Ackerman 2009), making deep-water habitats important for their overall contribution to stream biomass as well.

Along the inner banks of channel bends the depositional environment present, gives rise to habitats such as riffles, shallow runs and slackwater habitat (Rabeni and Jacobson 1993). Riffles generally feature clean gravels with little to no fine sediment (Montgomery and Buffington 1997), and favour invertebrates that prefer well aerated, moderately-sized substrates (cobble and pebble) such as Ephemeroptera, Plecoptera and Trichoptera (EPT) (Resh and Rosenberg 1984, Jowett 2003, Jowett et al. 2005). For salmonids, riffles represent quality foraging positions for a variety of species and life stages and may provide some cover value for smaller individuals (Smith and Brannon 2007).

Shallow runs, also referred to as glides occur at intermediate depths and typically feature intermediate velocities (Rosgen 1994), and variable substrates, and during low seasonal flows can be quite susceptible to deposition and sedimentation (Landwehr and Rhoads 2003). Thus, invertebrate communities of glides are also quite variable (Resh and Rosenberg 1984). Shallow runs offer little to no salmonid cover, and likely do not represent the optimal feeding positions in a stream (Fausch 1984, Smith and Brannon 2007).

Slackwater habitats generally occur directly leeward of point bars formed along the inner banks of stream meanders, where eddies branching off from the main current create zones

of little or no flow in shallow, marginal habitat (Rabeni and Jacobson 1993). Although generally shallower, these habitats are similar to pools in that they feature low flow velocities and finer substrate composition than adjacent faster water habitats (Montgomery and Buffington 1997), and favour invertebrates that are well adapted to fine sediments and low water velocities, such as Chironomidae and Oligochaeta. Slackwater habitats are also important refugia from spates or flood events (Lancaster and Hildrew 1993, Negishi et al. 2002). Although slackwater areas are generally not of major importance for salmonids, it has been suggested that these areas may play a significant retention role for organic matter such as woody debris and nutrients (Webster et al. 1994, Brookshire and Dwire 2003, Daniels 2006), which may then have indirect, though delayed, benefits for all aquatic biota (Bilby and Likens 1980, Bilby 1981, Lamberti et al. 1989). Where slackwater habitats accumulate large amounts of woody debris, it provides a unique substrate type available for colonization by filter feeding and detritivorous invertebrates (Reice 1980), adequate cover from predation, for many different invertebrates (Wallace et al. 1997).

Fluvial processes resulting from stream meander thus provide the diversity of habitat characteristic of unaltered stream reaches. Streams impacted by channelization, devoid of bends and stripped of their habitat forming processes, feature a contrastingly monotonous habitat compliment which would be expected to impact aquatic communities as well as stream carrying capacity (Nunnally and Keller 1979, Brookes et al. 1983, Brookes 1985, Cramer and Ackerman 2009, Pedersen 2009). Impacts such as increased sedimentation, erosion, or the many others which have been summarized above, can have severe deleterious effects on the salmonid communities which are valued for recreation or

aesthetics, as well as their invertebrate food resources (Emerson 1971, Duvel et al. 1976, Quinn et al. 1992, Negishi et al. 2002).

1.3.3 Mitigation

While channelization may negatively affect one taxa, it may simultaneously provide benefits for another (Beechie and Bolton 1999), making it difficult to design optimal management strategies to mitigate effects on diverse aquatic communities. As such, when developing effective mitigation techniques it is crucial that managers consider not only the compliment or severity of impacts, but also the fact that micro-habitats are not used equally across all taxa and life stages present within a stream, therefore certain impacts may weigh more heavily than others depending on the stream's community structure. Such is the case with many physically based (rather than biologically based) habitat restoration techniques, which generally benefit one specific taxonomic group while potentially harming many others (Reeves et al. 1991).

When considering the restoration of habitat, it is believed that a focus on restoring natural processes to a stream is far more beneficial than in-stream habitat manipulation (Beechie et al. 1996, Kauffman et al. 1997). Roni et al. provide a review of restoration techniques on streams impacted by a variety of land uses in the Pacific Northwest and outline the importance of process restoration as well as instream habitat manipulation, but note that in order for specific instream manipulation treatments to be optimally effective, natural processes should first be restored (Roni et al. 2002). With respect to fish and invertebrates, it is well understood that the stream processes most affected by channelization are those which are responsible for micro-habitat creation (Nunnally 1985,

Brookes 1987b) and those associated with erosion and deposition of sediment (Rhoads 1990, Landwehr and Rhoads 2003).

Impacts associated with erosion and deposition can be addressed by many bank stabilization methods, including re-enforcing unstable banks with live vegetation, or re-establishment of a riparian zone, which help to increase channel stability, decrease erosion, and reduce sediment load in a stream. Restoring micro-habitat forming processes to a channelized reach, however, is a much more difficult task due to the fact that restoration of stream sinuosity requires large scale channel modifications and is often not economically feasible. Nunnally is among the few to address this issue, but did so from a preventative stand-point, providing a set of provisions for future channelization projects which would allow for micro habitat forming processes to be maintained within the reach (Nunnally 1978). These guidelines included minimal straightening, limited removal of bankside vegetation and the integration of bank stabilization techniques, and would ideally create a hybrid channelized stream reach which satisfied anthropogenic needs while maintaining the equilibrium and flow processes of a natural stream (Keller 1976, Nunnally 1978, Nunnally and Keller 1979, Nunnally 1985, Brookes 1987b). In 1978, Brookes was the first to attempt to restore sinuosity to a channelized stream section on a large scale (Brookes 1987b). The experiment involved an 800 m reach of a low energy stream in Denmark which had been channelized. The original meandering channel which existed in the 19th century was plotted using historical maps, and then excavated. To restore the reach's historical stream bed, substrate composition was determined from historical geological information as well as upstream sources, and then placed in pre-determined intervals along the stream to be distributed naturally by high flow events.

This would ensure that the distribution of the substrate emulated what would exist as a result of natural processes. To ensure bank stability and reduce erosion, natural vegetation was planted along the stream banks, and rip rap was used where necessary (Brookes 1987b). Although Brookes did not comment on the cost effectiveness of the project, or estimates of feasibility for future projects, it is important to note that this work was done on a small stream (width < 2 m), and it could be assumed that the cost of such projects would certainly increase as stream sizes become larger. Initial monitoring of the stream two years post-manipulation suggested that the desired ecological results had been achieved (Brookes 1987b). Several other projects have since attempted to restore sinuosity to channelized streams, although at smaller scales, and have similarly reported benefits such as increased fish and invertebrate density and diversity upon initial monitoring (Iversen et al. 1993), but to my knowledge no long term monitoring of these systems (> 3 yrs post-meander restoring) has been undertaken. Although short term monitoring programs are not without benefit (Bayley 2002), they may also be misleading if insufficient time has been given for stream biota to fully adjust to a new system (Bisson et al. 1992). This point is illustrated by Moerke at al. (2004), who performed both short and long term monitoring of a sinuosity restoration project in Indiana. Similar to Brookes, stream substrate was added to reflect what existed pre-channelization and stream banks were re-enforced with natural vegetation. After a full year post-meander construction, invertebrate and fish density and diversity within the restored reaches were either equal or greater than reference channelized reaches. Monitoring was then conducted for a second time five years post-meander construction to determine long term success of the restoration, and revealed that although invertebrate density remained

higher, invertebrate diversity as well as fish abundance and diversity were all equal to or lower in the restored reaches compared to the channelized reaches. Following their observations, Moerke et al.(2004) determined restoration masked the effects of sediment input from upstream channelized reaches in the short term which required several years to once again impose deleterious effects on stream biota (Moerke et al. 2004).

As dictated by the inherent difficulty in restoring micro habitat forming processes to channelized reaches, most micro-habitat restoration efforts in channelized reaches most typically resort to in-stream manipulation methods such as the addition of boulders, large woody materials, or artificial structures to the reach. The objective of these structures is to create obstructions in the stream flow which produce diverse flow conditions (Thompson 2006) and heterogeneous substrates (Laasonen et al. 1998), improve retention of organic material (Muotka and Laasonen 2002) and may also provide cover (Brittain et al. 1993). Gowen and Fausch (1996) determined that the installation of large woody debris in mountain streams effectively increased pool depth, pool volume and proportion of fine sediment in treated areas (Gowan and Fausch 1996). Following the placement of instream habitat features, Rosenberg and Huato (2003) reported an increase in riffles and pools, as well as habitats which offered cover and foraging opportunities for salmonids (Rosenfeld and Huato 2003).

While it is widely accepted that the addition of instream features is an effective method of restoring a variety of micro-habitats within a stream, it has yet to be clearly shown that these methods provide any measurable benefits for fish or invertebrate communities. Recent research assessing the response of fish and invertebrate communities following the addition of instream features found no statistical differences in fish abundance, fish

biomass, invertebrate abundance or invertebrate taxa richness across impacted and restored reaches (Muotka et al. 2002, Lepori et al. 2005). Broad literature reviews undertaken in the past decade have suggested that due to widely variable results (Stewart et al. 2009), lack of evidence (Thompson 2006) or inadequate monitoring (Bayley 2002), previous studies have failed to provide unambiguous support for these restorative techniques. Further to the uncertainty surrounding the effectiveness of instream features, criticism has also been afforded to the durability, or long term persistence of these structures in larger streams and rivers. Roni et al.(2002) noted that due to the magnitude of peak flow events in streams wider than 12 m, artificial instream features would likely not persist beyond 20 years, creating even greater scepticism when considering the long term effectiveness of these methods (Roni et al. 2002).

The most recent habitat rehabilitation effort to occur on the Crowsnest River was part of the fish mitigation program following the completion of the Oldman River Dam in 1991. To help meet the objective of "no net loss of fishing opportunity", boulder clusters were placed in a reach of the Crowsnest River, downstream of Lundbreck falls, with the intention of creating pool habitat suitable for game size fish to rest or overwinter in. In 1995, the structures were damaged and required repair following a flood event. In 2001, the Oldman River Dam Environmental Advisory Committee evaluated the mitigation efforts on the Crowsnest River. They determined that although the structures themselves had been repaired, the pre-designed distribution of the structures had been significantly altered. As a result of this altered distribution, the committee further determined that the target habitat had therefore also not been sufficiently maintained. The final recommendation of the committee was that, due to a lack of supporting evidence, it could

not be concluded that the objective of "no net less of fishing opportunity" had been achieved (O.R.D.E.A.C. 2001).

The need for effective measures of stream rehabilitation remains a pressing issue in lotic ecology, especially those which apply specifically to the mitigation of stream channelization, one of the most common anthropogenic disturbances affecting streams. There is currently a debate among the scientific community as to the effectiveness of many stream rehabilitation techniques (Bayley 2002, Thompson 2006, Stewart et al. 2009), therefore it is crucial that we continue to generate new and innovative techniques in order to ensure the long term persistence of our river ecosystems.

In the chapter to follow, I will discuss the impacts of stream channelization on a variety of channel processes, stream habitat, and aquatic communities, as well as introduce an innovative restoration project which will attempt to mitigate these negative impacts, with a strong focus on process restoration.

Chapter 2

Impacts of Stream Channelization on Salmonid and Invertebrate Communities and their Habitat, in the Crowsnest River

2.0 Abstract

Stream channelization is a form of anthropogenic disturbance, common to warm water low land streams where agriculture is the dominant land use. While much work has been done to assess the negative effects of channelization on stream habitat and fish communities, a large percentage of studies focus on warm water systems, and as a result, much less attention has been afforded to impacts on cold water salmonid streams. Here I assess the impacts of channelization on stream habitat, and determine how these impacts affect the salmonid and aquatic invertebrate communities of a 5th order cold water river in Southern Alberta. I demonstrate that stream channelization has imposed significant alterations to stream habitat, most notably a loss of deep habitat, and that these alterations have led to a statistically significant decline in abundance and biomass of Rainbow Trout and Mountain Whitefish, as well as a significant decline in their food source (aquatic invertebrates). While these findings should be applicable for management across a wide range of cold water salmonid species, I suggest that they may be of special significance to the management of Mountain Whitefish populations, which may be more sensitive to channelization than other coldwater species.

2.1 Introduction

Where streams and rivers come into conflict with urban land use they are often subject to alterations which can have profound effects on the physical structure and biotic communities of the ecosystem. One form of anthropogenic disturbance associated with urban land use and common across a wide range of landscapes, is stream channelization. Channelization can be most simply defined as the artificial straightening of streams (Emerson 1971), and is used as an effective method to impede water from accessing the flood plains adjacent to rivers, thereby reducing the risk of flooding in areas where other anthropogenic land uses would require such action.

Physical stream alterations which result from channelization include elevated velocities and discharge, increased levels of erosion and sedimentation, less instream cover and decreased allochthonous input (Lau et al. 2006), all of which can significantly impact invertebrate and fish assemblages (Townsend 1989, Englund 1991, Smock et al. 1992). Channelized streams have also been described as having homogenous habitat, having lost the mosaic of different habitat types characteristic of a pristine meandering channel (Keller 1976, Rambaud et al. 2009).

Among the most notable channelization-induced impacts that affect aquatic invertebrates are those that result in habitat loss, through either loss of useable area due to straightening (Hansen 1971, Brookes 1987a) or changes in water velocities, depths and/or substrate composition which may limit the carrying capacity of instream habitats (Minshall 1984, Jowett et al. 2005, Kroes and Hupp 2010). Flow refugium is best described as those habitats which offer protection from high flow events and minimize the risk of stream inhabitants being flushed away throughout high flow events, but this crucial habitat is

typically lost where channelization occurs. These habitats are known to be of great importance for the purpose re-colonization of adjacent habitat following high flow events (Lancaster and Hildrew 1993), and research has shown that loss of refugia habitat following channelization can impose significant effects on the aquatic invertebrate community (Negishi et al. 2002).

Another important component of invertebrate habitat, typically lost through channelization but often ignored, is the deposition of woody debris and other organic matter, which provide aquatic invertebrates with an alternate substrate for colonization as well as a source of nutrients (Bilby and Likens 1980, Reice 1980, Bilby 1981, Quinn et al. 1992, Wallace et al. 1997). Any loss of such habitat resulting from channelization will likely have negative effects on the invertebrate community.

Channelization has been shown to reduce the carrying capacity of streams for fish by homogenizing habitat, transforming riffle-pool sequences characteristic of natural meandering reaches into a riverscape dominated by run or glide-type habitat (Keller 1976, Rambaud et al. 2009) with few deep pools (Duvel et al. 1976, Nunnally and Keller 1979, Cramer and Ackerman 2009). Although bank undercutting is accelerated by channelization, and stable undercut banks provide key fish habitat, riparian vegetation in straightened reaches is usually not sufficient to keep banks stable (Rhoads 1990, Rabeni and Jacobson 1993, Magoulick and Wilzbach 1997, Myers and Resh 2000).

Efforts to mitigate habitat loss often focus on restoring individual micro-habitats via the placement of artificial instream features, although there is no general agreement on the effectiveness of such techniques (Muotka and Laasonen 2002, Roni et al. 2002, Moerke

et al. 2004). These methods focus on small-scale physical structure rather than large scale processes (Oscoz et al. 2005), which would of course be much more difficult to manage and likely require more resources. Examples of process restoration, although limited, are nevertheless available (Brookes 1987b, Iversen et al. 1993, Moerke et al. 2004), and such restoration projects should be considered whenever they are a feasible option.

The Crowsnest River supports a very productive recreational salmonid fishery dominated by Mountain Whitefish (*Prosopium williamsoni)* and Rainbow Trout (*Oncorhynchus mykiss)*. Although approximately 40% of the main channel of the Crowsnest River above Lundbreck Falls is affected by channelization, the potential fishery effects are unquantified.

2.2 Objectives and Hypothesis

This chapter will describe how salmonid and invertebrate communities use habitat in Crowsnest River, and assess how channelization on the Crowsnest River has altered the availability of that habitat and impacted these biota. Furthermore, the potential opportunity to mitigate the effects of channelization, through the application of a unique process- based restoration project, will be evaluated. Specifically, the chapter tests the hypothesis that channelization has reduced habitat heterogeneity, and has resulted in reduced abundance of both aquatic invertebrates and salmonids.

2.3 Materials and Methods

2.3.1 Assessment of Impacts on Habitat

The Crowsnest River is a fifth order river (Strahler) which supports well established

communities of aquatic invertebrates as well as native and introduced salmonid species, despite being heavily impacted by channelization. I selected four study reaches (two non-channelized, two channelized) to assess the impacts of channelization on stream habitat (Table 2-1).

Table 2-1. Summary of Location and Elevation of Study Reaches (Head) Used to Assess Habitat, Invertebrate Biomass and Salmonid Habitat Use and Abundance.

Site Name	Reach Type	Zone	Easting	Northing	Elevation
CR1	Non-channelized	11U	673142	5500432	1352 m
CR2	Non-channelized	11U	676361	5500413	1337 m
CR3	Channelized	11U	677409	5500781	1332 m
CR4	Channelized	11U	681264	5500483	1312 m

Because impacts resulting from channelization, such as increased sediment load and deposition, are likely to affect not only channelized reaches but also downstream reaches, the two non-channelized reaches were selected in areas upstream of the "impacted zone", and the two channelized reaches were selected in areas in the up-stream most sections of the "impacted zone" (Figure 2-1). To ensure that study reaches were representative, all reach lengths were determined by a 750 m valley length, which corresponded to approximately 40x average wetted width (Lyons 1992). Valley length was used rather than stream length to account for a reduction in stream length associated with stream channelization. Once I selected my study reaches, photographs were taken to characterize bank and riparian differences among channelized and non-channelized stream reaches. (Figure 2-2A, 2-2B).
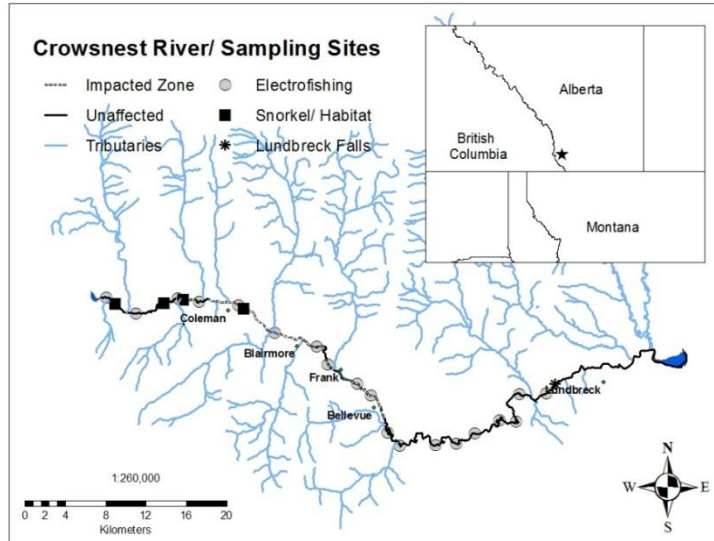
Figure 2-1. Map of Crowsnest River, Study Reaches and Impacted Zone.

Prior to data collection, a suite of habitat metrics was selected based on literature, which I

deemed to be ecologically significant for invertebrate and salmonid communities. For

each given metric, the entire in-stream area of all four reaches was recorded to determine

the distribution of habitat, as well as the range of values for each metric across all sites.

Depth measurements were taken with a 1.5 m measuring stick. Velocity measurements

were made using a surface float method (Bain and Stevenson 1999). Measurements of

substrate were made using a Vernier caliper, and reflect the dominant substrate size of the

patch. Embededness measurements were made from visual observation and reflect the

average of the patch. Cover was measured from visual observation. Once the range of

values was determined, each metric was then subdivided into ecologically significant,

ordinal categories (Table 2-2). Using these criteria, distinct units of habitat (patches) were

then delineated. This was done by first delineating patches, by depth, and then further

sub-dividing those patches based on all other metrics until a mosaic of distinct patches

had been outlined. Multiple methods were used to describe differences in habitat between

channelized and non-channelized reaches, facilitated by GIS. As a measure of habitat heterogeneity, number of total patches for non-channelized and channelized reaches were expressed as patches per 100 m^2 and compared among reach types.

A- Channelized reach CR4
B- Channelized reach EF6

Figure 2-2. Photos of Channelized Reaches and Riparian Vegetation.

To determine how the availability and distribution of each metric within a reach was affected by channelization, the average proportion of patches that each individual class accounted for within each metric for was calculated for both channelized and non-channelized reaches. These proportions were then compared among channelized and non-channelized reaches using chi-square contingency table analysis to determine if the proportions of class within metrics were significantly different among non-channelized and channelized reaches. Total area available for each individual class within each habitat metric, per reach (750 m valley length), was calculated and compared between channelized and non-channelized reaches. This value was expressed as area per unit valley length, rather than river length, to account for the loss of habitat area as a result of decreased sinuosity from channel straightening.

Table 2-2. Habitat Metrics and Sub-Class.

Depth	Velocity	Cover	Embededness	Substrate [a]
0 cm	0-0.5 m/s	Absent	< 5 %	Gravel
1-20 cm	0.51-1.0 m/s	Bank Undercut	5-25 %	Pebble
21-60 cm	1.01-1.5 m/s	Other [b]	25-75 %	Cobble
61-100 cm	>1.5 m/s	Undercut and	> 75 %	Boulder
>100 cm		Other [c]		

[a] Values derived from (Cummins 1962)
[b] Over head vegetation, boulder or large downed wood
[c] Combination of bank undercut and other cover

2.3.2 Assessment of Impacts on the Invertebrate Community

Biomass

To assess the impacts of channelization on the aquatic invertebrate community, the same

four reaches described above (Table 2-1) were sampled with a variety of quantitative

benthic techniques, using a random stratified sample design (Murphy and Willis 1996),

with depth as the initial metric for stratification, sampling at random points generated by

GIS. A Surber sampler was used for patches less than 60 cm in depth, any patches deeper

than 60 cm required sampling with a D-handle kick net, and a Hess sampler was also

used to sample patches consisting of substrate with high woody debris accumulation and

in patches featuring little or no flow. Individual patches were considered as a sampling

unit. To ensure adequate representation of each patch, and due to the variation in size and

shape of patches, patch samples were composite, consisting of three sampling points

randomly distributed within the patch. Each separate sampling point consisted of a

30cmx30cmx10cm plot. Any large rocks (diameter > 10 cm) within the sample area were

scraped clean (scrapings left to drift into the sampler) and removed prior to disturbing the plot for a 60 second period. Samples were transferred to plastic Ziploc bags, labeled, and stored in a freezer until analyzed. Reaches were sampled bi-weekly during the months of July, August and September during fall low flows in 2010 and 2011.

In the lab, invertebrates were identified to family, dried for 48 hours in a 60°C oven and weighed to the nearest 0. 1 mg to determine biomass for each patch type (habitat type) and reach type, which were then compared within and among channelized and non-channelized reaches. As well as total invertebrate biomass, I included two additional response variables, Ephemeroptera-Plecoptera-Trichoptera (EPT) biomass and chironomidae biomass, in light of their importance as a food source for the salmonids in the Crowsnest River. Biomass was expressed as dry weight per unit area. Biomass estimates were Log_{10} transformed to improve homoscedascity of the data set, and nested ANOVAs were performed to determine variation in biomass among reach type and within sites among reach type for comparative analysis.

Density

To determine the relative importance of individual habitat types, I examined the density of habitat in the same four reaches that were examined in the habitat component of this study (Table 2-1, Figure 2-1). Habitat metrics hypothesized to be of greatest importance to the invertebrate communities were determined prior to analysis. Initially, the metrics of substrate and bed velocity were chosen on the basis of previous literature, but due to sampling logistics and a strong observed relationship between velocity and depth, depth was used rather than bed velocity. Since all habitats considered in these analyses were

available within all four study reaches, data were pooled from all reaches and total

invertebrate, EPT, and Chironomidae biomass estimates were compared among class

within habitat metrics with two-way ANOVAs to determine the density of various

habitats. I included reach type (channelized/non-channelized) as a categorical variable in

my analysis to determine if reach type had a significant effect on the invertebrate-habitat

relationship. To compare the density of woody debris/ organic matter patches to other

habitats, a third ANOVA was carried out to compare total biomass of woody debris/

organic matter patches to other classes within the metric of substrate. The data from

channelized reaches was excluded from this analysis because this form of habitat was not

available in channelized reaches. Prior to all analyses, biomass estimates were Log_{10}

transformed to improve homoscedascity of the data set.

2.3.3 Assessment of Impacts on the Salmonid Community

Abundance

Snorkel Surveys were conducted on three study reaches (CR1, CR2 and CR4, see Table

2-1, Figure 2-1) to determine salmonid abundance in channelized and non-channelized

reaches on the Crowsnest River. Access to resources limited snorkel surveys to only one

snorkeler and one on shore data recorder, which on fifth order river such as the

Crowsnest required special design considerations in order to ensure quality data. It was

not possible for one observer to observe the entire stream width, nor was it possible to

develop a lane by lane system due to a lack of multiple observers, therefore an alternative

method which reduced the probability of double counts yet retained a definitive sample

area had to be developed. After completing several trial runs, a repeatable transect line in

downstream progression was developed and mapped in a GIS. This transect was then

applied with the depth patch data acquired from the habitat component of the study to

determine which patches could be adequately observed from the transect line. Those

patches which met the criteria were included in the sample area, those that did not were

excluded. In late July, August, and early September a total of three underwater

observations were made for each of the three reaches, between the hours of 10:00 and

16:00 always under good visibility conditions (>5 m). The observer worked in a

downstream progression, recording species, size class and location on a large slate

attached to their arm, stopping at pre-determined points spaced approximately 50 m apart

to relay information to the on shore data recorder. Any fish whose behaviour appeared to

be affected by the presence of the observer was recorded for abundance estimates, but

was not included in the analysis of habitat use. Abundance was calculated as the mean

value of fish/100 m^2 from the three observational passes for each reach, and was reported

for both juvenile and adult salmonids (>300 mm). T-tests were used to determine

significant differences in total and adult abundance among non-channelized and

channelized reaches.

Habitat Use vs. Availability

To determine habitat use by salmonids, fish locations on two non-channelized reaches

 (CR1 and CR2, Table 2-1, Figure 2-1), were compared to randomly distributed locations

generated by a GIS. Observations made in the channelized section (CR4) were omitted

from these analyses due to a lack of habitat variability. Juvenile salmonids and adult

salmonids (>300 mm), were compared to determine if these two groups used habitat

differently. Individual observations were considered sampling units. The number of

random locations to be generated for comparison was determined by the average number

of actual observations for that reach. Mean values of the three passes for each reach were

then compared among actual and randomly generated data using t-tests to determine

significant differences in use of individual classes within the habitat metrics of depth,

velocity and cover. If the number of observations for each habitat type from underwater

surveys were significantly different from those randomly generated by the GIS, I

determined that salmonids were actively selecting for those habitats.

Salmonid Density

To determine the density of individual habitats with respect to the salmonid community

of the Crowsnest River, I used data from observations performed on the two non-

channelized reaches (as described above) to determine abundance estimates (fish/100 m^2)

of individual classes within the habitat metrics depth, velocity and cover. As with the

analysis of habitat use vs. availability, I divided the data into two response variables,

juvenile salmonids and adult salmonids (>300 mm), to determine if density in various

habitats were different among the two groups. Mean values were derived from the three

observational passes of each reach, and compared among classes within each habitat

metric to determine which habitats supported the greatest abundance of both total and

adult salmonids.

Biomass

 I used abundance counts obtained from underwater observations to calculate salmonid

biomass estimates for channelized and non-channelized reaches, using methods as

outlined by Garcia and Associates (GANDA 2008). Because lengths were not recorded for each individual fish, I treated all fish within a distinct size class as measuring the length of the mean size for that category (ie for the size class 150-300mm, all fish were treated as 225mm). I then applied those lengths to *O. mykiss* and *P. williamsoni* weight to length relationship curves which were derived from electrofishing data of the Crowsnest River during the summer of 2010 (Blackburn 2010) (Figure 2-3A,2-3B). Once weights were derived, I multiplied those estimates by the number of fish observed within each size class to obtain biomass estimates of salmonids for each size class.

Figure 2-3A. *O. mykiss* Weight to Length

$$\log weight = -5.036607 + 3.0229763 \text{ x } \log length$$

Figure 2-3B. *P. williamsoni* Weight to Length

$$\log weight = -5.665674 + 3.2639904 \text{ x } \log length$$

Biomass (g/m^2) was estimated for all size classes for each study reach, and ANOVAs were performed to test for significant variation among treatments (channelized and non-channelized).

2.4 Results

2.4.1 Impacts of Channelization on Stream Habitat

In total, 557 habitat patches were delineated throughout the 4 study reaches (Table 2-1). Mean number of patches/$100m^2$ were higher in non-channelized reaches CR1 and CR2 reaches (17.40 and 17.87, respectively) when compared to channelized reaches CR3 and CR4 (9.10 and 10.35, respectively). Channelized reaches featured elevated, reinforced

banks, and had been cleared of most vegetation along the side of the river closest to urban development (Figure 2-2A). Riparian vegetation remained on the opposite banks, although many other reaches within the impacted zone which were surrounded on both sides by urban development, had been cleared of vegetation along both banks (Figure 2-2B).

Chi-square contingency analysis revealed a significant difference between channelized and non-channelized reaches in the proportions of individual depth classes (p=0.02, d.f.=4). The largest difference was the proportion of >100 cm patches, which were almost twice as common in non-channelized reaches compared to channelized reaches (Figure 2-4A). In terms of average total area per reach, non-channelized reaches featured, on average, 3187 m^2 more area >100 cm depth per km of valley length than channelized reaches (Figure 2-4B). For the metric of substrate, only 524 of the total 557 patches were considered, as patches which featured a depth of 0 cm (instream islands) were not considered for this analysis. Chi-square contingency analysis revealed a significant difference in the proportions of individual classes per reach across channelized and non-channelized reaches (p=2.39E-07, d.f.=3). Channelized reaches featured a higher proportion of patches of gravel and pebble compared to that in non-channelized reaches, while non-channelized reaches featured a higher proportion of cobble in comparison to channelized reaches (Figure 2-5A). The most abundant patches for all reaches were those represented by cobble substrate size, with the exception of CR4, which was most abundantly represented by patches of pebble sized substrate (Figure 2-5B).
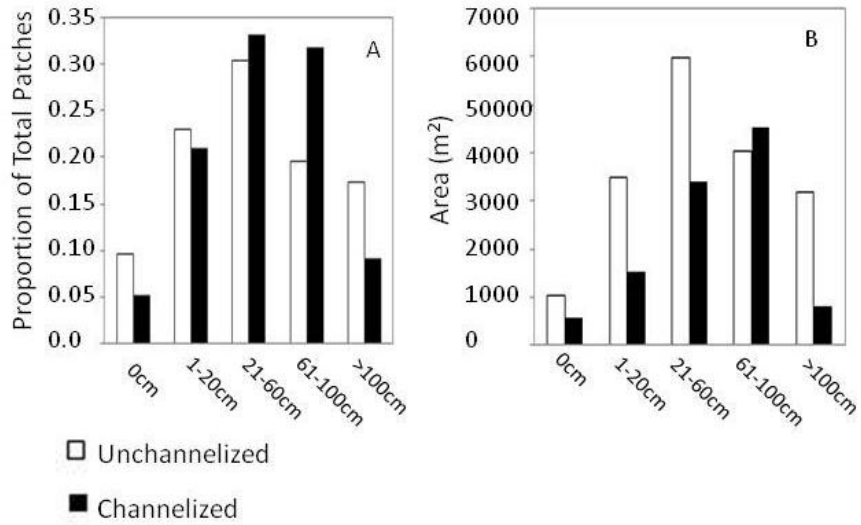
A- Average proportion of total patches per each for all depth classes
B-Average area of patches per reach for all depth classes

Figure 2-4. Proportion of Patches and Total Area per Reach of Depth Class across Reach Type.
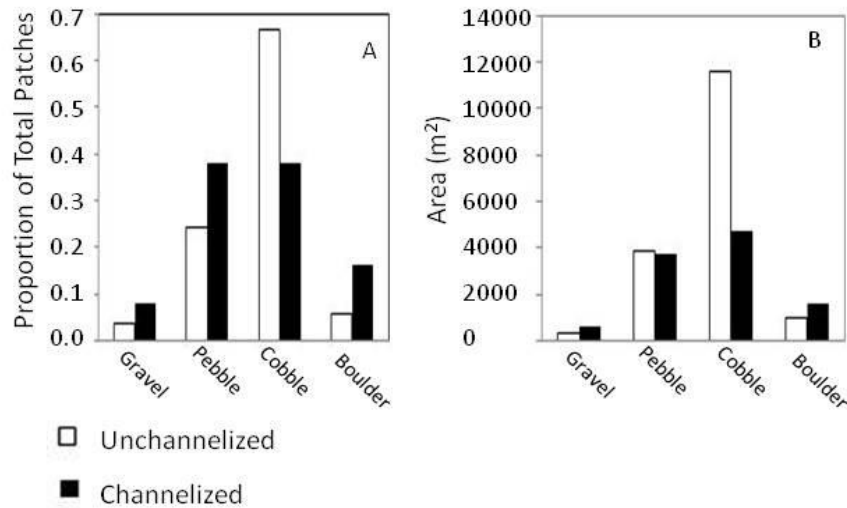
A- Average proportion of total patches per each for all substrate classes
B-Average area of patches per reach for all substrate classes

Figure 2-5. Proportion of Patches and Total Area per Reach of Substrate Class across Reach Type.

A total of 520 patches were considered for the analysis of the embededness across

channelized and non-channelized reaches. Chi-square contingency analysis revealed a

39

significant difference in the proportions of individual classes per reach across

channelized and non-channelized reaches (p=3.68E-06, d.f.=3). The largest difference

was observed in the proportion of patches with negligible (<5 %) embededness, with a

higher proportion present in non-channelized reaches than in channelized reaches (Figure

2-6A). At the opposing end of the embededness spectrum were those patches which were

heavily (>75 %) embedded, which were more common in channelized reaches than in

non-channelized reaches. Moderately (5-25 %) embedded substrate was the most

common type within channelized reaches, whereas negligible embededness were the

most common in non-channelized reaches (Figure 2-6B).
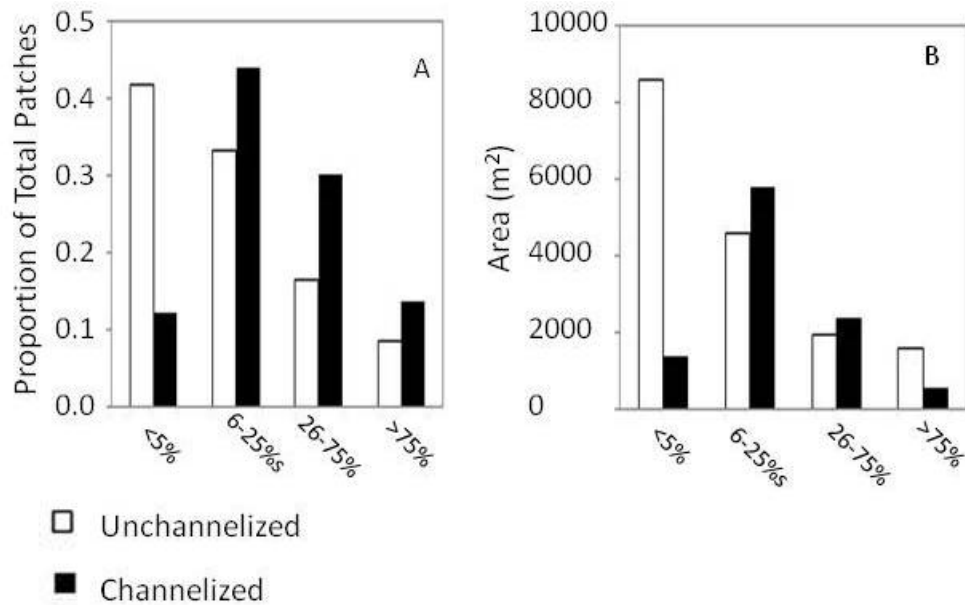
A- Average proportion of total patches per each for all embededness classes
B-Average area of patches per reach for all embededness classes

**Figure 2-6. Proportion of Patches and Total Area per Reach of Embededness Class
across Reach Type.**

A total of 520 patches were considered for the analysis of velocity across channelized and

non-channelized reaches. Chi-square contingency analysis revealed a significant

difference in the proportions of individual classes per reach across channelized and non-

channelized reaches (p=6.09E-10, d.f.=3). Non-channelized reaches contained

approximately triple the proportion of low velocity(<0.5 m/s) habitat found in

channelized reaches, whereas, moderate (0.51-1.0 m/s) velocity patches were the most

common type within channelized reaches(Figure 2-7A & 2-7B).
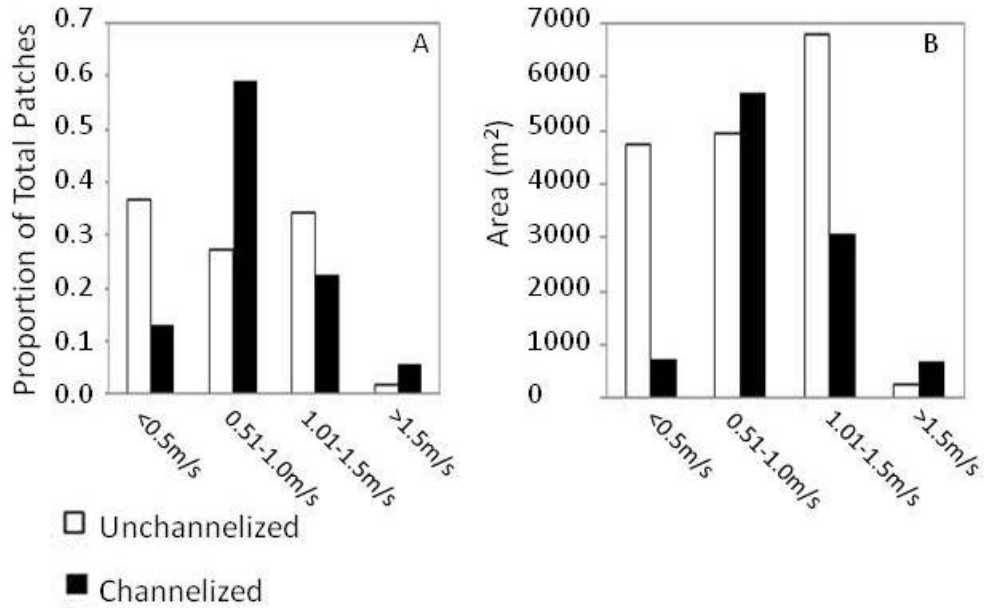
A- Average proportion of total patches per each for all velocity classes
B- Average area of patches per reach for all velocity classes

Figure 2-7. Proportion of Patches and Total Area per Reach of Velocity Class across Reach Type.

For the final habitat metric investigated in this study, cover value, 524 patches were

considered for the analysis. Chi-square contingency analysis revealed a significant

difference in the proportions of individual classes per reach across channelized and non-

channelized reaches (p=3.36E-06, d.f.=3). Non-channelized reaches featured

proportionately four times greater the number of patches which had both forms of cover

and two times the number of patches which featured only bank undercut cover, compared

to channelized reaches (Figure 2-8A).
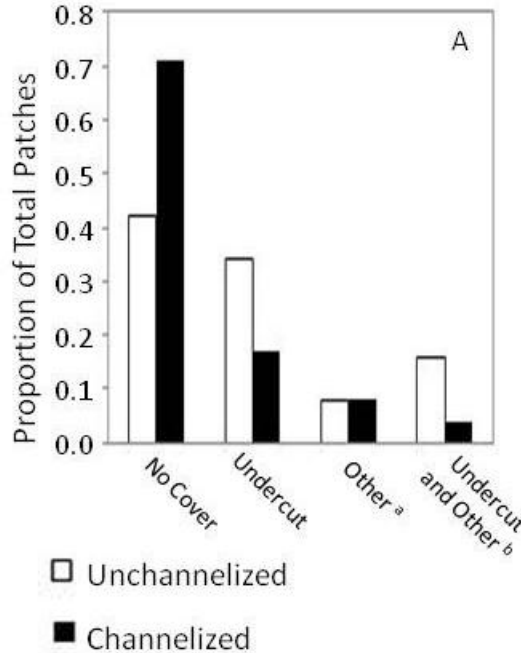
A- Average proportion of total patches per each for all depth classes
[a] Over head vegetation, boulder or large downed wood
[b] Combination of bank undercut and other cover

Figure 2-8. Proportion of Patches of Cover Class across Reach Type.

2.4.2 Impacts of Channelization on the Invertebrate Community

Density

Two-way ANOVA used to determine the effect of depth on the three response variables

revealed significant variation in total invertebrate biomass ($F_{5,110}$=9.44, p<0.0001), EPT

biomass ($F_{2,110}$=7.73,p<0.0001) and chironomidae biomass ($F_{5,110}$=3.21,p=0.0096) among

classes. An interaction term was included in each analysis to determine if reach type had

an effect on the relationship among depth and biomass, and was in each circumstance not

significant. Total invertebrate biomass was significantly higher in patches of 1-20cm

depth compared to depths of 20-60 cm (p<0.0001) and depths of 60-100 cm (p=0.0189),

as determined from post hoc Tukey analysis following the initial ANOVA (Figure 2-9A).

EPT biomass exhibited a similar trend, with biomass at depths of 1-20cm significantly

greater than both 20-60cm (p<0.0001) and 60-100cm (p=.0022) depths, determined from

post hoc Tukey analysis following the initial ANOVA (Figure 2-9B). Chironomidae

biomass was also highest at the shallowest depths of 1-20cm, however only significantly

so when compared to depths of 60-100 cm (p=0.0407), as determined from post hoc

Tukey analysis following the initial ANOVA (Figure 2-9C).
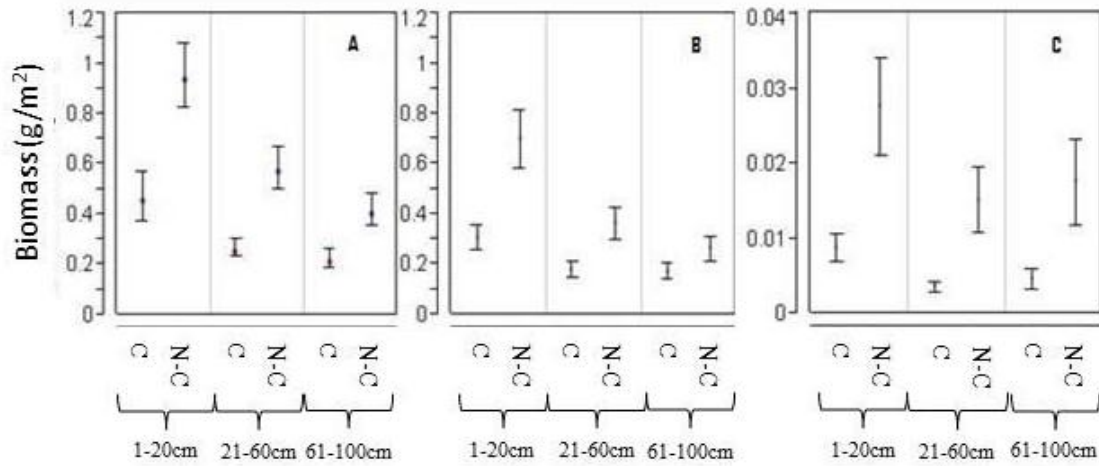
A-Total Invertebrate Biomass
B-EPT Biomass
C-Chironomidae Biomass
C=Channelized, N-C=Non-Channelized

Figure 2-9. Invertebrate Biomass, by Depth and Reach Type.

Two-way ANOVA used to determine the effect of substrate on the three response

variables revealed significant variation in total invertebrate biomass ($F_{7,108}=6.01$,

p<0.0001), EPT biomass ($F_{7,108}=4.28$,p=0.0003) and chironomidae biomass

($F_{7,108}=8.73$,p<0.0001) among classes. An interaction term was included in each analysis

to determine if reach type had an effect on the relationship among substrate and biomass,

and was in each circumstance not significant. Total invertebrate biomass was highest in

pebble sized substrate, and was significantly lower in boulder sized substrate when

compared to pebble (p=0.0019), gravel (p=0.0231) and cobble (p=0.0225) sized substrate

determined from post hoc Tukey analysis following the initial ANOVA (Figure 2-10A).

EPT biomass was also highest in pebble sized substrate, and significantly lower in

boulder sized substrate compared to pebble (p=0.0010), cobble (p=0.0091) and gravel

(p=0.0486) sized substrates determined from post hoc Tukey analysis following the

initial ANOVA (Figure 2-10B). Chironomidae biomass was highest in gravel sized

substrate, and was significantly higher in gravel sized substrate compared to boulder

(p<0.0001), cobble (p<0.0001) and pebble (p=0.0001) sized substrates determined by

post hoc Tukey analysis following the initial ANOVA (Figure 2-10C).
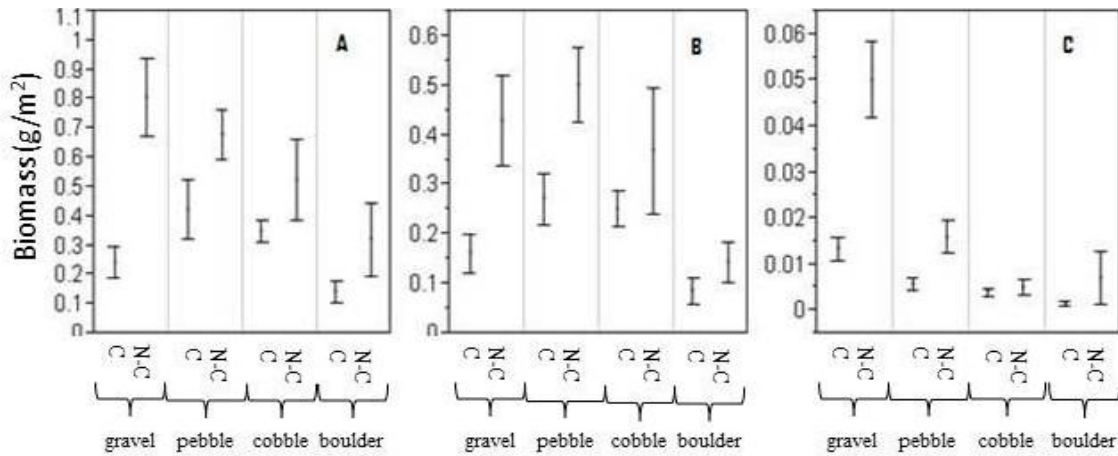
A-Total Invertebrate Biomass
B-EPT Biomass
C-Chironomidae Biomass
C=Channelized, N-C=Non-Channelized

Figure 2-10. Invertebrate Biomass, by Substrate and Reach Type.

Total invertebrate biomass in patches with woody debris/ organic matter depositions were

higher than in patches lacking such deposition, post-hoc Tukey analysis revealed

estimates of these patches to be significantly higher than patches with gravel (p=0.0047),

pebble (p=0.0022), cobble (p=0.0002) and boulder (p<0.0001) sized substrates (ANOVA,

$F_{4,131}$=10.59, p<0.0001).

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Impacts of Channelization on Biomass

In total, 18721 aquatic invertebrates were collected, representing 24 distinct families, from a sample size of 148 habitat patches. Biomass estimates from pooled invertebrate data for each of the four study reaches are available in Table 2-3.

Table 2-3. Total Invertebrate, EPT and Chironomidae Biomass by Reach.

Site Name	Reach Type	Total Invertebrate Biomass (g/m^2)	EPT Biomass (g/m^2)	Chironomidae Biomass (g/m^2)
CR1	Non-channelized	0.60 ± 0.09	0.44 ± 0.08	0.019 ± 0.001
CR2	Non-channelized	0.70 ± 0.08	0.44 ± 0.06	0.021 ± 0.001
CR3	Channelized	0.26 ± 0.03	0.17 ± 0.02	0.005 ± 0.001
CR4	Channelized	0.40 ± 0.08	0.28 ± 0.05	0.006 ± 0.002

*Average biomass (g/m^2)/reach $\pm$ 1 s.e.

Nested ANOVA (CR1, CR2 nested within non-channelized, CR3, CR4 nested within channelized), revealed a significant variation in the three response variables, total invertebrate, EPT and chironomidae biomass, among reach types ($F_{3,128}=12.73$, p<0.0001, $F_{3,112}=6.99$, p=0.0002, $F_{3,112}=2.94$, p=0.0361, respectively). When testing for variation in biomass among sites within reach types, no significant differences were found for any of the response variables, total invertebrates, EPT or chironomidae (p=0.0791, p=0.1020, p=0.9781, respectively).

2.4.3 Impacts of Channelization on the Salmonid Community

Habitat Use vs. Availability

A total of 436 observations of salmonid habitat use were made throughout six

observational surveys. At site CR1, juvenile salmonids were observed using the deepest

depths (>100 cm) more than what would be expected from the given availability of

habitat (t-test, p=0.0447, df=3). Similarly, adult salmonids were also observed using the

deepest depths more than what would be expected from the given availability of habitat

(t-test, p=0.0600, df=3), however significant only with 90% confidence. For all other

depths, both total juvenile and adult salmonids were observed using habitat less often or

equal to what would be expected from the given availability of habitat (Table 2-4). For

site CR2 I found similar results, with both total salmonids (t-test, p=0.0072, df=3) and

adult salmonids (t-test, p=0.0124, df=3) being observed using the deepest depths more

than what would be expected from the given availability of habitat. For all other depths,

both total salmonids and adult salmonids were observed using habitat less often or equal

to what would be expected from the given availability of habitat, which was also

consistent with observations from site CR1 (Table 2-4).

Table 2-4. Underwater Observations of Depth Habitat Use per Reach and Randomly Generated Locations of Juvenile and Adult Salmonids.

Site Name	Depth	Juvenile Salmonid Use		Adult Salmonid Use	
		Snorkel Observation	Random Generated*	Snorkel Observation	Random Generated*
CR1	20-60 cm	1.67 ± 0.33	13.00± 1.52	0.00 ± 0.00	11.33 ± 0.88
	60-100 cm	4.00 ± 1.50	7.00 ± 1.73	4.00 ± 0.58	5.00 ± 1.53
	>100 cm	25.00 ± 3.40	11.00 ± 1.15	27.33 ± 3.53	15.67 ± 2.40
CR2	20-60cm	6.67 ± 0.33	15.67 ± 2.40	0.00 ± 0.00	7.67 ± 1.76
	60-100cm	10.33 ± 0.88	27.33 ± 3.33	4.33 ± 0.67	9.33 ± 1.86
	>100cm	43.00 ± 3.05	17.00 ± 1.15	19.00 ± 2.08	6.00 ± 1.00

*fish locations randomly distributed among patches which were included in underwater snorkel observations

Use of velocity habitat at site CR1 revealed a preference among adult salmonids for low (<0.5 m/s) velocities, with fish being observed using these velocities more than what would be expected from the given availability of habitat (t-test, p=0.0039, df=3). Similarly, juvenile salmonids were also observed using low velocities more than what would be expected from the given availability of habitat, however this difference was significant only with 90% confidence (t-test, p=0.0669, df=3). For all other velocities, both juvenile and adult salmonids were observed using habitat less often or equal to what would be expected from the given availability of habitat (Table 2-5).

Table 2-5. Underwater Observations of Velocity Habitat Use per Reach and Randomly Generated Locations of Juvenile and Adult Salmonids.

Site Name	Velocity	Juvenile Salmonid Use		Adult Salmonid Use	
		Snorkel Observation	Random Generated*	Snorkel Observation	Random Generated*
CR1	<0.5 m/s	17.67 ± 3.18	6.00 ± 0.00	20.30 ± 1.20	8.67 ± 1.45
	0.5-1.0 m/s	1.33 ± 0.33	2.67 ± 0.88	0.67 ± 0.67	2.67 ± 0.33
	1.0-1.5 m/s	10.33 ± 2.33	20.00 ± 1.53	9.67 ± 2.67	18.33 ± 1.20
	>1.5 m/s	2.33 ± 2.33	2.33 ± 1.20	1.00 ± 0.00	2.33 ± 0.67
CR2	<0.5 m/s	39.00 ± 3.51	13.33 ± 1.86	15.67 ± 3.28	4.33 ± 0.67
	0.5-1.0 m/s	5.33 ± 2.02	12.00 ± 2.52	0.33 ± 0.33	8.00 ± 1.73
	1.0-1.5 m/s	15.33 ± 2.03	35.33 ± 2.84	7.00 ± 2.08	10.00 ± 1.53
	>1.5 m/s	0.33 ± 0.33	0.67 ± 0.67	0.33 ± 0.33	0.00 ± 0.00

*fish locations randomly distributed among patches which were included in underwater snorkel observations

For site CR2, juvenile salmonids were observed using pool velocities more than what would be expected from the given availability of habitat (t-test, p=0.0072, df=3). Adult salmonids were also observed using low velocities more than what would be expected

from the given availability of habitat (t-test, p=0.0668, df=3), however the difference being significant only with 90% confidence. For all other velocities, both juvenile and adult salmonids were observed using habitat less often or equal to what would be expected from the given availability of habitat (Table 2-5).

Use of cover habitat at site CR1 revealed no significant preference among juvenile or adult salmonids for any cover habitat above what would be expected from habitat availability.

Table 2-6. Underwater Observations of Cover Habitat Use per Reach and Randomly Generated Locations of Juvenile and Adult Salmonids.

Site	Cover	Juvenile Salmonid Use		Adult Salmonid Use	
		Snorkel Observation	Random Generated*	Snorkel Observation	Random Generated*
	no cover	0.00 ± 0.00	1.33 ± 1.45	0.00 ± 0.00	3.00 ± 0.58
	undercut	24.00 ± 6.93	22.67 ± 1.86	28.33 ± 2.85	20.67 ± 1.20
CR1	other [a]	2.33 ± 1.86	3.00 ± 1.53	0.67 ± 0.33	2.00 ± 1.52
	undercut and other [b]	5.67 ± 1.20	4.00 ± 0.58	2.33 ± 1.45	6.33 ± 1.45

*fish locations randomly distributed among patches which were included in underwater snorkel observations
[a] over head vegetation, boulder or large downed wood

For all other cover habitat, both juvenile and adult salmonids were observed using habitat less often or approximately equal to what would be expected from the given availability of habitat (Table 2-6).

Salmonid Density and Biomass

In total, 273 habitat patches spanning 30630 m^2 were observed and included in the

analysis of productive capacity of depth and velocity habitats. At both sites CR1 and CR2, depths of >100 cm produced the highest amount of total fish/100m^2, with the deepest depths at CR1 producing an average 2.74 ($\pm$ 0.15) fish/100 m^2 and at CR2 producing an average 3.36 ($\pm$ 0.16) fish/100 m^2. Depths greater than 100 cm also produced the highest amount of adult salmonids at both sites CR1 and CR2, producing an average of 1.43 ($\pm$ 0.18) and 1.03 ($\pm$ 0.11) fish/100 m^2, respectively. The shallowest of depths (20-60 cm) in both CR1 and CR2 produced the lowest numbers of juveniles (0.09 $\pm$ 0.01 and 0.45 $\pm$ 0.02 fish/100 m^2, respectively), and contained no adults in any of the underwater observations at either sites (Figure 2-11).
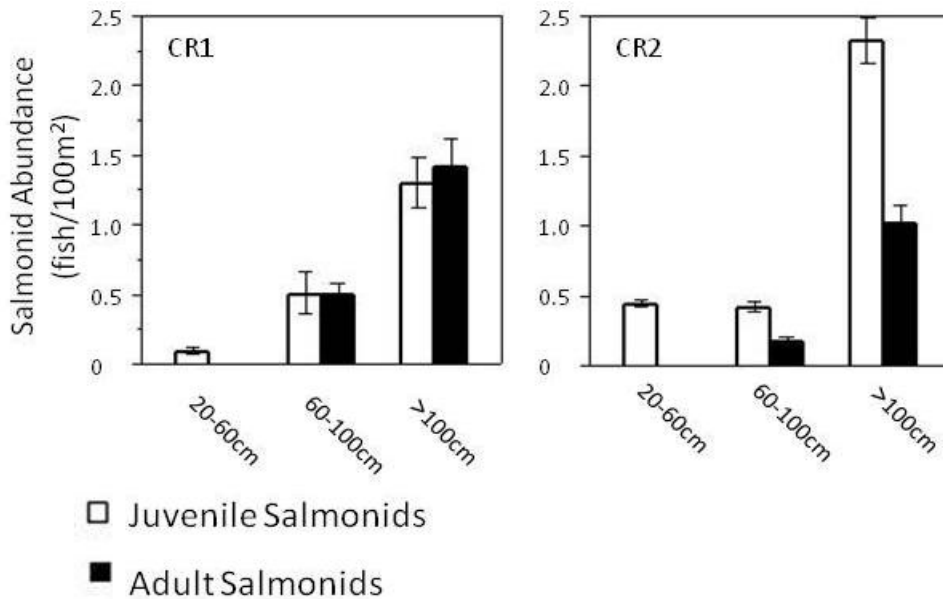
*Average abundance (fish/100m^2) $\pm$ 1 s.e.

Figure 2-11. Juvenile and Adult Salmonid Density of Depth Class.

Within the various velocity classes, low velocities (<0.5 m/s) produced the highest number of juvenile salmonids at site CR1 (1.40 $\pm$ 0.25 fish/100 m^2) and CR2 (3.31 $\pm$ 0.30 fish/100 m^2). Similarly, these same low velocities also produced the highest numbers of

adult salmonids at CR1 (1.61 ± 0.10 fish/100 m^2) as well as CR2 (1.32 ± 0.28 fish/100 m^2) (Figure 2-12).
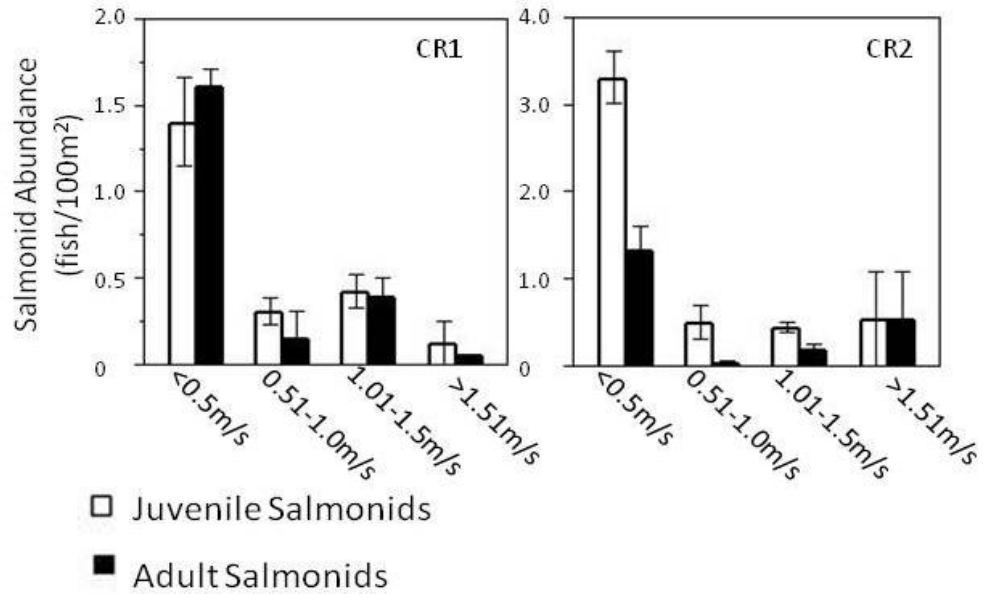
*Average abundance (fish/100m^2) ± 1 s.e.

Figure 2-12. Juvenile and Adult Salmonid Density of Velocity Class.
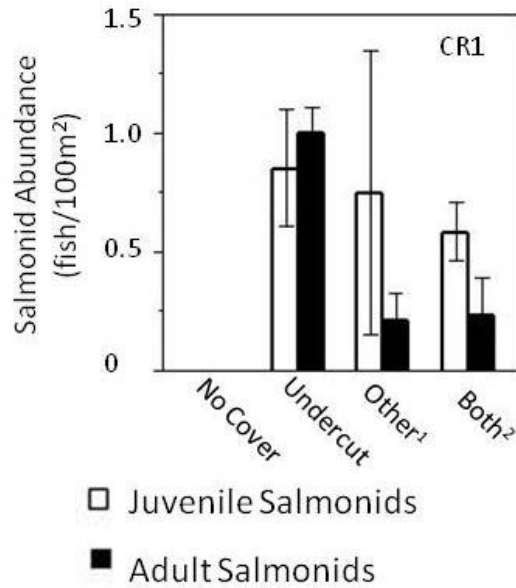
*Average abundance (fish/100m^2) ± 1 s.e.

Figure 2-13. Juvenile and Adult Salmonid Density of Cover Class.

A total of 132 patches covering 13290 m^2 were observed and included in the analysis of salmonid density of cover habitats. At site CR1, habitat patches with access to only bank undercut habitat produced the greatest number of both juvenile (0.85 ± 0.25 fish/100 m^2) and adult (1.01 ± 0.10 fish/100 m^2) salmonids (Figure 2-13).

Impacts of Channelization on Abundance

A total of 537 fish were observed throughout the study period. Total fish abundance was significantly higher in both CR1 (t-test, p=0.0006) and CR2 (t-test, p<0.0001) (non-channelized reaches) when compared to that of CR4 (channelized reach). Similarly, adult abundance was significantly higher in both CR1 and CR2 in comparison to that of CR4 (t-test, p<0.0001, p=0.0006, respectively) (Figure 2-14).
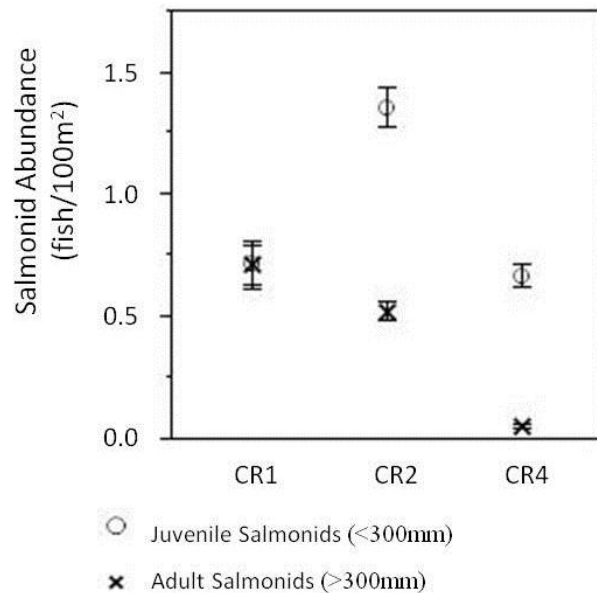
*Average abundance (fish/100m^2)/reach $\pm$ 1 s.e.
** CR1 and CR2 non-channelized, CR4 channelized

Figure 2-14. Juvenile and Adult Salmonid Abundance by Reach.

For CR1 (non-channelized), an average of 1.43 ($\pm$ 0.10) total fish/100 m^2 was estimated after three passes, with the adults of the population accounting for 0.71 ($\pm$ 0.07) fish/100

m^2. Total abundance for CR2 (non-channelized) was higher than in CR1, supporting an

average 1.88 ($\pm$ 0.07) fish/100 m^2, however featured fewer adults (0.53 $\pm$ 0.07

fish/100m^2) compared to its non-channelized counterpart.CR4 (channelized) featured the

lowest total abundance (0.73 $\pm$ 0.05 fish/100 m^2) and adult abundance (0.06 $\pm$ 0.01

fish/100 m^2), among the three sites.

ANOVA revealed significant variation in total fish biomass across the three study sites

($F_{2,8}$=52.58,p=0.0002) (Figure 2-15). Post-Hoc Tukey analysis was used to compare all

pairs, and revealed site CR4 (channelized reach) featured significantly lower biomass

than either CR1 (p=0.0002) or CR2 (p=0.0005) (non-channelized reaches).
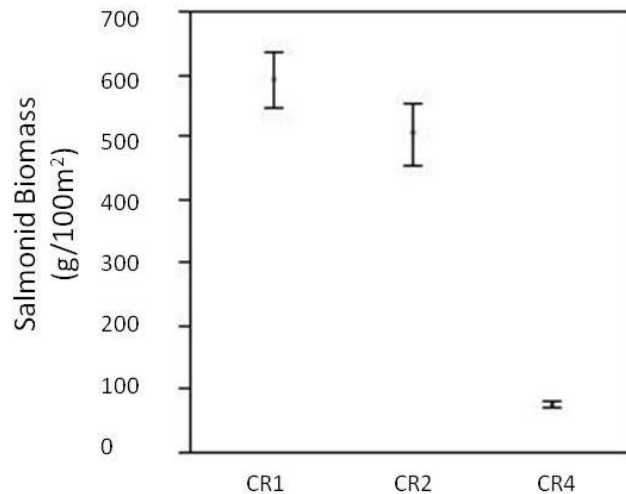
*Average biomass (g/100m^2)/reach $\pm$ 1 s.e.
** CR1 and CR2 non-channelized, CR4 channelized

Figure 2-15. Average Salmonid Biomass by Reach.

2.5 Discussion

2.5.1 Impacts of Channelization on the Stream Habitat

Channelized reaches featured decreased total habitat patch diversity and patch

abundance. Compared to the non-channelized meandering reaches, channelized reaches

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were monotonous in composition, and where any habitat variation did occur seemed to be due to processes caused by instream islands resulting from increased deposition rather than from a meandering flow. Riparian vegetation had been removed from most banks of channelized reaches. The banks in channelized reaches were also elevated and reinforced with rip-rap in certain areas where erosion appeared to be at its worst. The effects of stream channelization were apparent across each habitat metric I investigated.

Depth seemed to demonstrate the effects of channelization on stream habitat better than any other metric. This was due to not only a lack of deep patches (>100cm), which is commonly reported in the literature (Emerson 1971, Duvel et al. 1976, Keller 1976, Chapman and Knudsen 1980, Brooker 1985, Brookes 1985, Lau et al. 2006), but also a lack of the shallowest patches (<20 cm) in channelized reaches. From personal observations, it was apparent that the lack of pool habitat in these sections was due to a lack of stream processes which normally result from meandering morphology (Nunnally and Keller 1979, Nunnally 1985). Several mid-channel islands had formed in multiple locations along the channelized reaches as a result of prolonged exposure to increased sediment deposition, which altered the course of the flow and created some deep patches via scouring, as well as some shallower patches which had formed along the perimeter of the islands. Apart from these localized areas, channelized reaches featured mostly mid-range, glide-like depths, which has also been reported quite extensively in the literature (Emerson 1971, Duvel et al. 1976, Keller 1976, Chapman and Knudsen 1980, Brooker 1985, Brookes 1985, Lau et al. 2006).

With respect to the impacts of stream channelization on substrate revealed in this study, similar findings are offered from Smiley and Dibble (2006), who reported an increase in

percentage of gravel and smaller sized substrates, and a decrease in woody debris and leaf litter in channelized streams compared to non-channelized streams (Smiley and Dibble 2008). The absence of woody debris depositions was likely in part due to a reduction of adjacent bankside vegetation. Finally, channelized reaches featured a higher percentage of boulder sized substrates compared to non-channelized reaches, however, rip-rap used to fortify the banks of channelized reaches accounted for most of the boulder sized substrates in these reaches.

The degree of embededness among patches was also significantly different among channelized and non-channelized reaches.. While many studies have reported similar results to mine, in relation to embededness following recent channelization (1-20 years), my findings would suggest that these effects may be persistent over much longer periods of time (80-100 years). Patches which were severely embedded in the non-channelized reaches were mostly pools and backwaters, as one would expect as these habitats are generally separated from the main stream flow to some degree (Nunnally 1985, Rabeni and Jacobson 1993). However, in channelized reaches it was the glide-like habitats, exposed to similar flow as every other patch present in the stream, which accounted for those severely embedded patches, demonstrating the reduction of stream power in channelized reaches and the inherent net deposition of sediment during fall low flows, as previously found by Landwehr and Rhoads (Landwehr and Rhoads 2003).

The proportion of different habitat patch velocities was significantly different among channelized and non-channelized stream reaches. In non-channelized reaches, the spectra of various velocities considered for this study was relatively evenly distributed among low (<0.5 m/s), moderate (0.5-1.0 m/s), and high (1.0-1.5 m/s) velocities, with only few

patches characterized by very high (>1.5 m/s) velocities. Channelized sections were not characterized by such an even distribution, but rather featured a very high percentage of patches with moderate velocities, accounting for more than half of the proportion of all patches in the channelized reaches (approximately double the percentage of non-channelized reaches). These results support findings by Nunnally (1979) and Brookes et al. (1983) who reported uniform and weakened fall flows in streams post-channelization (Nunnally and Keller 1979, Brookes et al. 1983). Channelized reaches also featured a very low proportion of depositional areas, which likely inhibits any opportunity for upstream sources of woody debris and organic matter to accumulate in these reaches (Webster et al. 1994, Daniels 2006). Along with a lack of adjacent riparian vegetation, this could certainly help explain the total absence of woody debris deposits from channelized sections.

Lastly of the habitat metrics considered in this study, the ratio of the various forms of cover differed significantly among channelized and non-channelized stream reaches. Habitat patches which featured cover in the form of bank undercut were much fewer, proportionately, in channelized reaches, which was to be expected due to the unstable nature of channelized stream banks (Rhoads 1990). Surprisingly, channelized reaches featured a higher percentage of habitats with other forms of cover, which I characterize here as overhanging vegetation, boulders or other large artificial features on the stream bed which broke the surface of the water. I attribute this to the fact that although non-channelized reaches featured more adjacent riparian vegetation, this vegetation rarely provided cover directly above the stream (perpendicular to the water surface), and as such was not considered as overhead cover. Furthermore, artificial features such as concrete

rip-rap provided adequate instream cover and as such were treated within the designation of other cover features, but were not present in non-channelized reaches.

2.5.2 Impacts of Channelization on the Invertebrate Community

Density

We compared the three response variables (total invertebrate biomass, EPT biomass, chironomidae biomass) individually among the various classes of two predetermined habitat metrics (depth and substrate), and found significant variation among each response variable for both metrics. My analysis of the biological density of patches of various depths revealed the highest biomass estimates for each of the three response groups in the shallowest depths (1-20 cm), which was consistent across both channelized and non-channelized reaches. This preference for shallow habitat was expected among the chironomids, who as deposit collectors are known to prefer shallow, slow moving, depositional habitat (Hynes 1970, Clifford 1991), however, was somewhat surprising for the EPT group. Members of this group, such as Perlidae, Chloroperlidae and Hydropsychidae, which were all abundant in my samples, have all been known to prefer swifter flows (Brusven and Prather 1974, Wallace and Merritt 1980, Jenkins et al. 1984), and as such I anticipated higher biomass estimates for this group in deeper habitats. One possible explanation for this could be that while I note that depth and velocity were correlated in this study, these velocity measurements were taken from the surface of the stream and not the surface of the substrate, therefore the differences I observed in surface velocity among shallow (1-20 cm) and moderate (21-60 cm) depths may not have been

present along the stream bed, where they likely have a greater effect on benthic invertebrates.

Also consistent among channelized and non-channelized reaches, and also among total, EPT, and Chironomidae biomass, was that the lowest biomass estimates were observed in the deep patches. One explanation for this difference is the increased susceptibility of invertebrates to being carried away from the benthos and becoming drift, in deeper, swifter habitat (Waters 1965). Another possible explanation for higher invertebrate biomass in the shallow patches could be due to an increase in primary productivity. Although no direct measurements of periphyton growth along the substrate were made (due to large fluctuations throughout the sampling season), from personal observations it was apparent that the presence of such accumulations were more common in shallow patches than in deep patches. Invertebrate productivity has been shown to be positively correlated with periphyton growth (Feminella and Hawkins 1995), and as such may explain the low biomass estimates in habitats lacking this resource.

As I previously stated, my analysis of the density of patches of various substrate types revealed significant variation among classes for each of the three response variables. For total invertebrates and EPT, mean biomass was highest in pebble substrate, and for Chironomidae, mean biomass was highest in gravel or finer substrate. These findings are consistent with previous research, which have noted high abundances of Chironomidae in fine or even embedded substrates, while larger members of Ephemeroptera, Plecoptera and Trichoptera prefer larger and cleaner substrates (Brusven and Prather 1974, Jenkins et al. 1984). Although it was expected that EPT biomass would be highest in substrate sizes larger than what produced the highest biomass estimates for Chironomidae, it was

nevertheless surprising that boulder substrates produced significantly lower biomass estimates then all other substrate types. This would seem to contradict previous literature which suggests that larger invertebrates such as Plecoptera and Trichoptera prefer larger substrates which offer greater interstitial spaces and well oxygenated flow (Wallace and Merritt 1980, Brusven and Rose 1981). Perhaps the association of boulder substrates with deep water in the Crowsnest favours salmonids, resulting in predation and reduced EPT biomass.

The overall effect of substrate types on biomass estimates was not significantly different across channelized and non-channelized, however, there was a significant difference in the biomass of total invertebrates in gravel sized substrate. This difference is quite interesting, as it is the result of gravel or finer habitats in channelized reaches featuring the second lowest of biomass estimates of all the substrate types, but featuring the highest biomass estimates of all the substrate types in non-channelized reaches. These types of fine substrate habitat patches are most commonly found in depositional areas with low velocities (personal observation), and I theorized that they may be playing a key role as flow refugia for aquatic invertebrates. If this theory is true, then it may help explain why these patches feature higher biomass of total invertebrates than any other substrate type in non-channelized reaches (excluding woody debris/ organic matter patches), as it has been reported that refugia habitat exhibit an increase in aquatic invertebrate abundance following the occurrence of high flow events (Lancaster and Hildrew 1993). Although I did not only sample these patches following the occurrence of freshets, the frequent occurrence of such events throughout the season may have been a factor in the high biomass estimates of these patches. However, this would not explain why gravel or finer

substrate patches do not feature equally high biomass estimates in channelized reaches, where they in fact produce the second lowest biomass among substrate types. For this I offer an explanation which involves the degree of embededness of patches in channelized reaches. We've previously shown that channelized reaches featured a greater proportion of slightly, moderately and severely embedded patches than non-channelized reaches, demonstrating the effects of increased sedimentation in channelized reaches. Furthermore, as I have implied that these gravel and finer substrate patches are mostly associated with depositional environments, it follows that these patches are also likely to experience high degrees of embededness, relative to others in the reach. Invertebrates are not likely able to burrow into highly embedded gravel and thereby escape the high flows in the channelized reaches. Consequently, their biomass is reduced.

The habitats which featured the highest total invertebrate and EPT biomass estimates were the woody debris/organic matter deposition patches. These patches were comprised of all types of woody debris, sediment, dislodged macrophytes and many other forms of allochthonous input which create a unique substrate for colonization as well as a nutrient source for a wide variety of invertebrates (Reice 1980, Wallace and Merritt 1980). These patches were also limited to depositional environments, likely making them important sources of flow refugia.

Impacts of Channelization on Invertebrate Biomass

As I hypothesized, mean biomass (g/m^2) of total Invertebrates, EPT, and Chironomidae pooled across all habitat types, were each significantly higher in non-channelized reaches of the Crowsnest River when compared to channelized reaches. From my results

characterizing the effects of channelization on habitat, coupled with data describing aquatic invertebrate habitat density which was just summarized, I offer a number of possible explanations for the reduction in biomass found in channelized reaches;

The total area of patches of 1-20 cm depths per 750 m of valley length account for 3505 m^2 in non-channelized reaches, and only 1520 m^2 in channelized reaches (Figure 2-4B). Patches of 1-20 cm depths support significantly higher total invertebrate biomass than any other depth in the Crowsnest River (Figure 2-9), and as such, this drastic loss of habitat (1985 m^2) is potentially detrimental to the invertebrate community.

The habitat patches which held the highest density, in terms of total invertebrate biomass, among all patches considered in this study were the woody debris/ organic matter patches. A combination of diverse habitat for colonization, high nutrient value, and being located in areas of the stream which offer refuge from high flow events make these unique habitats a crucial component of habitat for invertebrate communities (Bilby and Likens 1980, Reice 1980, Wallace and Merritt 1980, Negishi et al. 2002). These patches account for an average 1088 m^2 of highly productive habitat in non-channelized reaches but are completely absent from channelized reaches. This absence from channelized reaches is likely the result of a reduction of inputs from lateral riparian sources as well as a lack of habitat which is depositional in nature throughout the seasons, and could certainly be an important factor in explaining why non-channelized reaches feature higher biomass than channelized reaches.

Another possible explanation for the reduced invertebrate biomass estimates observed in channelized stream reaches becomes apparent when we consider the role of flow refugia

as it applies to aquatic invertebrate communities. Flow refugia offer protection from high flows, during either spring runoff or individual high flow events, where most invertebrates would otherwise become quite susceptible to being swept away downstream by the increased flow (Waters 1965). It then follows, that the ability of an invertebrate community to re-colonize a stream reach could be partially reliant on the occurrence of flow refugia within that reach. This was found to be true by Negishi et al. (2002), who reported that aquatic invertebrates were able to re-colonize riffle and run habitats more effectively if refugia habitat was present in the reach than if refugia were absent from the reach (Negishi et al. 2002). In the preceding paragraphs I have demonstrated that habitat which could qualify as invertebrate flow refugia, such as woody debris/ organic matter patches or patches with gravel or finer substrate, are either completely absent or have possibly been rendered ineffective due to sedimentation and embededness in channelized reaches. As such, it is possible that invertebrate communities in channelized reaches are simply not able to recover from spring runoff or other high flow events, or at least not to the degree to which they can in non-channelized reaches, which may then explain why non-channelized stream reaches feature higher invertebrate biomass than do channelized reaches.

2.5.3 Impacts of Channelization on the Salmonid Community

Use vs Availability and Salmonid Density in Various Habitats

To determine if salmonids were using habitat unequivocally to what would be expected from habitat availability, I compared my underwater observations of habitat use by salmonids to randomly generated fish positions. I first looked at how the total salmonid

population utilized habitat patches of various depths, and found that salmonids were observed using the deepest depths (>100 cm) almost twice as often as was expected from habitat availability at site CR1, and nearly three times as often at site CR2. For all other depths, and across both study reaches, I found total salmonids used habitat less often than what would have been expected from availability. I then broke down the data to examine only adult (>300 mm) habitat use and found the same trends, with adults using the greatest depths more often and using all other depths less often than what was expected from the availability of habitat in both reaches. To strengthen my analysis, I also calculated the density of each depth across both study reaches, and found that the greatest depths also contained the highest numbers of fish/100m^2 for both juvenile salmonids as well as adult salmonids across both CR1 and CR2 reaches.

These results were not surprising to find with respect to the adult response group, as many others have revealed greater usage and noted higher carrying capacity of deeper habitats among both *O. mykiss* and *P.* williamsoni (DosSantos 1985, McPhail and Troffe 1988, Cramer and Ackerman 2009). However, with respect to the juvenile population (<300 mm), I expected to find a much more even distribution among various depths as both juvenile *O. mykiss* and *P. williamsoni* generally prefer shallower habitats which present less competition from adults (Pettit and Wallace 1975, Grant and Kramer 1990). These somewhat surprising findings, with respect to juveniles, were likely the result of a strong schooling behavior which was observed among *P. williamsoni* in the Crowsnest River. Schooling behavior is not common among salmonid species, and as such, optimal habitat patches are often found to be occupied by one larger individual who will defend that stream position from other individuals (Johnsson et al. 2004). However, *P.*

williamsoni are one such exception to this phenomenon, and adults and juveniles can often be found congregating in deep pools (Davies and Thompson 1976, McPhail and Troffe 1988), making these deeper habitats of equal importance to *P. williamsoni* of all size classes. Furthermore, although much less common, these congregations on occasion contained a number of juvenile *O. mykiss*, indicating that juveniles of both dominant species in the Crowsnest River were utilizing these deeper habitats.

When I analyzed habitat use vs. availability among the various classes of velocity habitat, I found that all salmonids seemed to actively select for the slowest velocities (<0.5 m/s), and that these velocities were also associated with higher salmonid density than any other. Results such as these are characteristic of many *P. williamsoni* populations who seem to prefer slower moving habitats (DosSantos 1985, McPhail and Troffe 1988). With respect to *O. mykiss* however, these results are somewhat surprising, as I anticipated a greater preference for faster moving habitats among these individuals, which should have produced a more even distribution of habitat use observed among the total population. Similar to my depth metric results, I attribute this to the schooling behaviour of *P. williamsoni* in pools, and the fact that this behaviour seems to be shared, at least to some degree, with the symbiotic juvenile *O. mykiss*.

The final habitat metric I analyzed with respect to the salmonid community was cover value of habitat. Undercut habitat contained the highest salmonid density among all cover classes considered, demonstrating the importance of such cover features. Another notable observation, with respect to the importance of cover, was that not a single fish was ever observed using a habitat patch which featured no cover. These results support the work of many others who have demonstrated a preference among various species of salmonids for

cover features, and specifically a strong preference for bank undercut cover (Magoulick and Wilzbach 1997, Myers and Resh 2000).

Impacts of Channelization on Salmonid Abundance and Biomass

Non-channelized reaches CR1 and CR2 both featured significantly higher total and adult salmonid abundance (fish/100m^2) than the channelized CR4, as I hypothesized. Furthermore, when I compared biomass estimates from non-channelized reaches vs channelized reaches, I found that both unaffected reaches featured significantly higher salmonid biomass estimates than the channelized reach. These findings support much of the previous work which has attempted to link the effects of channelization induced degradation of habitat to impacts on fish communities. However, much of the existing body of work has focused on describing only the impacts to habitat and then making inference to how this may be linked to changes in the fish community based on known habitat preferences from the literature or using existing habitat indices. While I included some analysis similar to these (impacts on salmonid biomass), here I have also used my "in situ" observations of salmonid habitat use to make more direct linkages as to how this habitat loss associated with channelization is impacting the salmonid community of the Crowsnest River (impacts on salmonid abundance).

The greatest disparity in habitat among channelized and non-channelized stream reaches, with respect to the salmonid community, was the presence of deep (>100 cm) and slow velocity (<0.5 m/s) habitat patches (pools), as outlined in my results of the impacts of channelization on instream habitat. I have determined that not only were these habitats used more frequently than would be expected from their availability (inferring selection

of these habitats), but they also presented the greatest density among all types of patches considered. It follows, that this lack of high capacity and actively selected-for habitat in channelized streams is likely a large contributing factor to the reduced abundance of salmonids observed in these reaches. These results seem to support much of the past research undertaken in describing the ecological effects of channelization on fish communities, which often attribute the greatest losses to a reduction of pool habitat in channelized reaches (Emerson 1971, Keller 1976, Nunnally and Keller 1979, Brookes et al. 1983, Lau et al. 2006). However, research which has specifically focused on the impacts of channelization on salmonid communities have commonly reported that this loss of pool habitat affects only the adults of the population, leading to only insignificant effects on juveniles which are presumably unable to compete for such optimal habitat as pools (Duvel et al. 1976, Chapman and Knudsen 1980). In the current study, I have determined that these affects are not limited only to the adults of the population, but rather affect the population as a whole (juveniles included), which would appear to be somewhat contradictory to past studies. I attribute this contradiction to the fact that these optimal habitats are not limited only to the adult salmonid population of the Crowsnest River, but are rather used by all members of the population, and I conclude that these results may be unique to populations which exhibit schooling behavior. *P. williamsoni* are among the most commonly occurring fish along the eastern slopes of the Rocky Mountains, but unlike most other species of salmonids common to this region, exhibit strong schooling behavior (Davies and Thompson 1976, DosSantos 1985, McPhail and Troffe 1988). I therefore propose that the effects of channelization are likely of greater consequence to *P. williamsoni* populations, as they are likely to affect not only adults, but

also younger age classes, which in turn may affect recruitment and lead to profound effects on the persistence of *P. williamsoni* populations.

Although not likely to have such pronounced effects on the salmonid population as the loss of deep and slow moving habitat, the reduction of bank undercut cover may also be a noteworthy contributor to the diminished abundance of salmonids in channelized reaches. As I reported previously, due to a lack of riparian vegetation and its associated root matrices in channelized reaches, stream banks in such impacted areas are unstable, which inhibits the creation of bank undercuts. Habitat such as this plays an important role as refuge from overhead predation, and is therefore thought to be a preferred form of habitat among many salmonid species (Myers and Resh 2000, Johnsson et al. 2004). In the absence of cover, the risk of predation may outweigh the potential benefits of even optimal habitat, and as such, it is possible that salmonids are avoiding even the most optimal of habitat available in channelized reaches, simply due to the overlying risk of predation.

During late summer and early fall (our sampling period), both *O. mykiss* and *P. williamsoni* are primarily foraging and building up reserves for spawning and overwintering, and as such are likely actively seeking out habitat which optimizes foraging opportunities. With respect to these species of salmonids, previous research has shown that aquatic invertebrates belonging to the families Ephemeroptera, Plecoptera and Trichoptera (EPT), as well as Chironomidae, make up a significant portion of their diets (Pontius and Parker 1973, Overton et al. 1978). As I have shown previously with the results describing the impacts of channelization on the aquatic invertebrate community, channelized reaches support significantly lower biomass of both EPT and Chironomidae

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than do non-channelized reaches, and as such present limited foraging opportunities. It follows, that much like I discussed above in relation to a lack of cover opportunities, that salmonids may be actively avoiding even the most optimal of habitats available in channelized reaches simply due to a lack of quality foraging opportunity, contributing to an overall reduction in salmonid abundance in channelized reaches.

Stream Channelization as a Potential Barrier for Salmonids

To further investigate the impacts of channelization on the salmonid community of the Crowsnest River, I analyzed data from 19 electrofished reaches disbursed along the river's main stem above Lundbreck Falls (Figure 2-1) (Blackburn 2010). I compared biomass estimates from stream reaches within the impacted zone, to those both upstream and downstream of the impacted zone. When I analyzed these results, I observed a significant difference among the channelized section and the downstream non-channelized section, which I expected, but surprisingly found no significant difference among the channelized section and the non-channelized reaches upstream of the channelized section (Figure 2-16). Furthermore, I also found salmonid biomass in reaches in the upstream section to be significantly lower than reaches downstream of the impacted zone, despite having no discernible difference in stream habitat (Blackburn 2010).

When attempting to understand why biomass estimates were so high in non-channelized reaches downstream of the impacted zone compared to channelized reaches, but why non-channelized reaches upstream of the impacted zone were not significantly higher than those same channelized reaches, I came to the conclusion that it could be possible

that an impassible barrier may be restricting fish movement to upstream reaches. Upon

further investigation I determined that no such barrier existed throughout these reaches,

however, I entertained the thought that perhaps it was the limited habitat availability

throughout this large section that was in fact acting as a form of barrier.
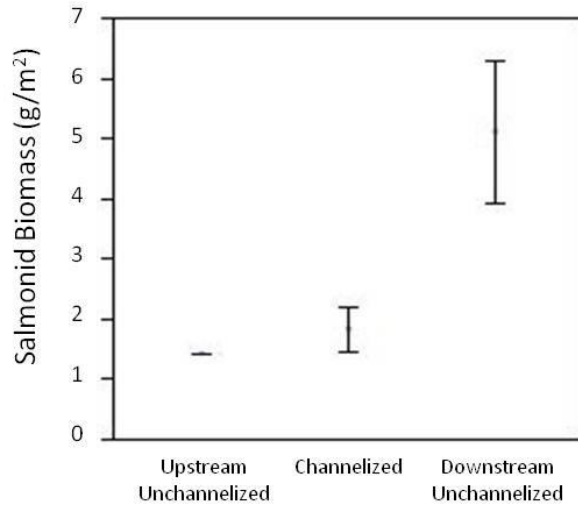
*Average biomass (g/m^2) ± 1 s.e.

Figure 2-16. Average Salmonid Biomass of Upstream Non-channelized, Channelized and Downstream Non-channelized Reaches.

The length of the Crowsnest River which is contained within the impacted zone is

approximately 18 km. It is therefore conceivable that this is the minimum distance a

salmonid inhabiting an non-channelized reach downstream of the impacted zone would

need to move in search of further optimal habitat. While this distance falls well within the

movement range for many adult salmonids during the spring-summer months, it may be

out of the range for adult *O. mykiss* and *P. williamsoni* during the summer-fall months,

and is quite likely well out of the range of movements for juveniles of the same species

(Pettit and Wallace 1975, Gowan et al. 1994, Young 2011). As a result of these

limitations to salmonid movements in rivers, it is possible that rather than continue

searching for habitat in the channelized sections of the river, they are simply returning to

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the optimal habitat available in the downstream non-channelized reaches from which they came, as they are unable to find the optimal habitat which exists in the non-channelized reaches upstream of the impacted zone.

Stream Reactivation: An Alternative Method to Mitigate the Effects of Channelization

As I stated in the introduction, there appears to be a need for more effective methods of mitigation of deleterious effects on salmonid populations, such as those brought forth by stream channelization. The current standard seems to be the addition of instream features to re-create crucial micro-habitat types such as scour pools, an example being the addition of boulders to lower sections of the Crowsnest River. However, as we have seen on the Crowsnest River, these techniques are often ineffective in creating any measurable benefits with specific respect to either fish or invertebrate populations and also have limited longevity. As such, it would seem only logical that further mitigation efforts assume a new direction.

An interesting aspect of the Crowsnest River is that, in multiple areas where channelization has occurred, artificial channels were created which then received the diverted water which once flowed through the pre-existing channel, but the pre-existing channels were never filled in and to this day remain quite apparent in the adjacent flood plains of the channelized reaches. Areas such as these account for approximately 4.5 km of "de-activated" meandering stream channel, and may hold the key to a new direction of mitigation techniques unique to this system. In 2009, Golder Associates Ltd., in collaboration with Trout Unlimited Canada, proposed to re-activate these channels, instead of adding instream features to the channelized reaches. While the project has

stalled due to a lack of information concerning risks pertaining to hydrological aspects of the re-activation, such as flooding potential and initial sediment pulse following the re-activation, the potential benefits to the biotic communities of the Crowsnest River have provided aquatic biologists, such as ourselves, with much optimism.

The first and most simple aspect to describe in terms of benefit to the biotic community is the potential increase of useable area. While the meandering, inactive channels would account for approximately 4.5 km of stream habitat, the current channelized reaches through which the river now flows, which spans the same length of valley as the de-active channels, provide only 2.3 km of habitat. Thus, should the water be diverted back into the historic channels it would nearly double the amount of instream habitat currently available over the same valley length distance. Furthermore, not only would re-activation increase the total amount of area, but as I have shown throughout this paper, the new habitat created would also support a much higher biological density for both invertebrates and salmonids than is currently available in the presently active channelized reaches.

Another anticipated benefit for the salmonid community of the Crowsnest River, which at this point is only theoretical, addresses the issue that these channelized sections may be effectively acting as a barrier which is limiting the potential biological density of the non-channelized reaches upstream of the impacted zone. As I have demonstrated, these reaches feature excellent habitat for both *O. mykiss* and *P. williamsoni*, however may be unable to realize their potential density due to the inability of the abundant downstream salmonid inhabitants to find these reaches as a result of the length of impacted river. If I am correct in hypothesizing this hindrance to upstream movement, then by shortening the length of this impacted section and at the same time implanting optimal habitat within the

impacted zone, we could potentially improve the connectivity between the reaches

upstream and downstream of the channelized reaches. This improved connectivity would

then allow large numbers of fish to once more colonize these upstream non-channelized

reaches, improving the overall density of the already productive and renowned fishery

which is the Crowsnest River.

With this research, I have demonstrated that stream channelization has imposed

significant alterations to stream habitat of reaches within the impacted zone, and that

these alterations have led to a statistically significant decline in abundance and biomass

of *O. mykiss* and *P. williamsoni*, as well as a significant decline in their food source.

Chapter 3

Comparing Snorkel Surveys and Single Pass Electrofishing in their Estimation of Salmonid Abundance and Biomass across Simple and Complex Habitat Reaches of a Fifth Order Stream

3.0 Abstract

Using snorkel surveys, rather than electrofishing, to estimate fish abundance is becoming more popular among fisheries ecologists Consequently, it is important to determine the environmental conditions that optimize the accuracy of each method's estimates. Here I demonstrate that snorkel surveys offer a distinct advantage over electrofishing surveys in the assessment of salmonid abundance in deep water habitats which frequently occur in large, unaffected streams, however are perhaps disadvantageous in the assessment of shallow, marginal habitat, and therefore should be applied with caution. In resource limited studies, snorkel surveys can and should be used in combination with electrofishing to provide a more accurate assessment of salmonid community abundance than would be available from electrofishing alone.

3.1 Introduction

Electrofishing is a commonly used tool in the assessment of salmonid populations in lotic environments, and can provide managers and researchers with confident estimations of population parameters from only several passes of a stream reach (Nordwall 1999). This, in combination with new innovations in technology which make electrofishing gear more efficient and user friendly (Nordwall 1999), has led to an increase in the popularity and use of this technique despite the fact that this method also has several flaws and associated biases (Gardiner 1984, Cunjak et al. 1988, Rodgers et al. 1992, Thompson et al. 1997a, Thompson et al. 1997b, Reynolds et al. 2003). Electrofishing requires a crew of personnel, costly equipment and sufficient time to both plan and execute the sample design, therefore resource limitations are often a concern for any electrofishing project (Gardiner 1984, Mullner et al. 1998). Also of concern, is crew safety, as electrofishing is considered hazardous work and can result in significant injury to crew members, should the proper precautions not be undertaken (Reynolds et al. 2003). Electrofishing can also cause significant harm to fish, both in the short term through injury (Thompson et al. 1997a, Nielsen 1998) and in the long term by reducing growth rates (Thompson et al. 1997b).

The utility of electrofishing can also be compromised by the physical structure and water quality of stream reaches. A factor that can significantly impact electrofishing effectiveness is water conductivity (Cunjak et al. 1988, Reynolds et al. 2003). The physical complexity of stream reaches can also affect the efficiency of electrofishing, by making certain areas difficult to access (e.g. large woody debris), or by increasing escape opportunities for fish (e.g. boulders, bank undercut) (Kruse et al. 1998). Deep, wide pools

are also very difficult to electrofish (Gardiner 1984, Rodgers et al. 1992), and in addition, are a crucial habitat for large salmonids (Cramer and Ackerman 2009). Alternative methods, that can more efficiently sample deep pools and other complex habitats can therefore be of considerable importance, and as such, the inability to adequately assess these crucial habitats may be the greatest advocate for an alternative method to examine fish populations in streams with complex habitat and frequently occurring large, deep pools.

One alternative to depletion electrofishing is single pass electrofishing. The advantage of this method is that it is less demanding of resources, in that it requires less time and effort to plan and execute only a single pass rather than multiple passes which is required when using the depletion method (Kruse et al. 1998). Also, because fish are subject to only one pass of the electrofisher, the possibility of fish being harmed by the equipment is reduced (Kruse et al. 1998). However, similar to depletion methods, single pass electrofishing is subject to reduced efficiency as a result of variable water quality, and with consideration to the physical properties of streams, specifically stream width and cover, Kruse suggests that single pass electrofishing may not be suitable for complex streams with widths greater than 8 m (Kruse et al. 1998). Furthermore, single pass electrofishing is not nearly as efficient as multiple pass depletion electrofishing, and can provide only an index of what would be estimated from multiple pass efforts (Meador et al. 2003).

Another alternative to depletion electrofishing is underwater observation with snorkel surveys. Snorkel surveys can be an effective method to assess salmonid abundance and population size structure (Zubik and Fraley 1988, Thurow and Schill 1996, Mullner et al. 1998), and presents a variety of solutions to some of the shortcomings of electrofishing.

Snorkel surveys require less time, less effort and fewer crew members than electrofishing surveys, and therefore can be viable option for researchers with limited resources (Gardiner 1984, Cunjak et al. 1988, Mullner et al. 1998). Another advantage of snorkel surveys is that observations and assessments can be made with less effect on fish or their behavior, allowing for more accurate measurements of habitat (Heggenes et al. 1990, Thurow 1994), without risk of harming the fish (Kruse et al. 1998). Snorkel surveys also make it possible to make accurate observations of fish in deep water pools, such as those which are often not accessible to electrofishing crews.

Like electrofishing, however, snorkel surveys can be compromised in certain types of habitats, and when water is turbid (Schill and Griffith 1984). In shallow water with rough substrate, fish may be hard to see, since cobble and boulders can provide a great deal of cover (Thurow 1994). Perhaps the most notable criticism of snorkel surveys is that while researchers have demonstrated significant correlations among snorkel survey and electrofishing population estimates underwater observations generally produce lower abundance estimates (Mullner et al. 1998, Wildman and Neumann 2003). Much like single pass electrofishing, snorkel surveys can only provide an index of what would be expected from multiple pass depletion electrofishing, and this is likely the reason for their limited use in research (Cunjak et al. 1988, Rodgers et al. 1992).

The Crowsnest River is a fifth order river, with an average width greater than 15 m, and in its naturally meandering reaches, features very complex and diverse habitat with many deep pools. As such, it poses a significant challenge for managers which require accurate estimates of population parameters such as salmonid abundance and size-class structure, since electrofishers are unable to effectively sample deep pools and other habitat

complexities such as bank undercuts. Snorkel surveys may present an alternative method to assess the salmonid population in the relatively clear waters of the Crowsnest River, and may provide a more accurate estimation of fish abundance in stream reaches which feature an abundance of large, deep pools.

This chapter compares the efficacy of two relatively low cost, low resource-requiring methods (snorkel surveys and single pass electrofishing) to estimate fish abundance and community size structure in a fifth order stream. The comparison spans reaches with complex habitat and many deep pools, as well as reaches with monotonous habitat and no deep pools. I hypothesize, that in reaches with simple habitat, single pass electrofishing and snorkel surveys will produce similar abundance estimates and size class frequency distributions, since there is little habitat to hinder either technique. In complex reaches, however, snorkel surveys should be expected to yield higher abundance estimates than single pass electrofishing due mainly to more efficient sampling in deep pools. These differences in estimates should be more pronounced for large, adult fish which inhabit the deep pools, and should be most significant for biomass estimates.

3.2 Materials and Methods

3.2.1 Site Description

The Crowsnest River is a fifth order river, which supports an abundant salmonid community which is dominated by Mountain Whitefish (*Prosopium williamsoni*) and Rainbow Trout (*Oncorhynchus Mykiss*). Throughout the early 1900's, large sections of the river's main stem were subject to stream channelization, which has significantly altered the instream habitat compliment of these reaches. In non-channelized reaches, the

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Crowsnest River features meandering geomorphology which has produced heterogeneous

habitat composed of shallow riffles, deep runs and very large, deep pools. In channelized

reaches, habitat is quite homogeneous and is characterized mostly by shallow runs with

some riffles and deep runs, and pools are, in most instances, non-existent. The contrasting

habitat of these reaches provide an ideal opportunity to compare the efficacy of two

sampling techniques (single pass electrofishing and snorkel survey) across reaches

featuring either complex (non-channelized) or simple (channelized) habitat.

Figure 3-1. Map of Study Reach Locations.

3.2.2 Site Selection

Through a collaborative effort to assess the salmonid population of the Crowsnest River,

the Alberta Conservation Association carried out single pass electrofishing along 18

randomly distributed reaches of the Crowsnest River above Lundbreck Falls, an

impassable barrier to fish movement (Blackburn 2010). Of these reaches, seven were

channelized, three were non-channelized and were upstream of all channelization

impacts, and the remaining eight were non-channelized but downstream of channelization

impacts.

Table 3-1. Name and Description of Sample Reaches Selected for Snorkel Survey and Single Pass Electrofishing.

	Site Name	Habitat	Sampling Method
Complex Habitat Comparison #1	SS1	Complex	Snorkel Survey
	EF1	Complex	Single Pass Electrofishing
Complex Habitat Comparison #2	SS2	Complex	Snorkel Survey
	EF2	Complex	Single Pass Electrofishing
Simple Habitat Comparison #1	SS3	Simple	Snorkel Survey
	EF3	Simple	Single Pass Electrofishing

Of these reaches, careful consideration had to be given to site selection for snorkel surveys, due to comparability issues among sites, as well as to address various safety concerns. In total, three reaches were selected (two complex habitat, one simple habitat, see table 3-1), which were located in sections of the river in which it would be logistically possible to perform underwater observations. These snorkel survey reaches were then shifted either slightly upstream or slightly downstream in a fashion which allowed us to continuously sample a comparatively sized reach while avoiding potential risks to observer safety. The largest distance that a snorkel survey reach was shifted was less than 50m, a distance which is well within the potential daily movements of *P. williamsoni* and *O. Mykiss*. In light of this, I do not expect that these methods would introduce any additional bias which would not have already occurred had the exact same reaches been sampled with both methods, but on different days. A map of study reach locations is available in figure 3-1.

3.2.3 Single Pass Electrofishing

A Smith-Root LR-6 tote-barge electrofisher was used with a four man crew working in a downstream progression (Blackburn 2010). Captured fish were kept in live wells until measurements were taken, which included species, total length (mm) and weight (g).

3.2.4 Characterization of Depth Habitat

Depth habitat for all three snorkel survey reaches were mapped from visual observations. Instream area was divided into distinct units of habitat (patches) based on four sub-categories of depth (<20 cm, 20-59 cm, 60-100 cm, >100cm). These data were then integrated into a spatially referenced GIS (Geographic Information System) to calculate the proportion as well as the total cumulative area of each sub-category of depth per reach.

3.2.5 Snorkel Survey

Snorkel surveys were performed by a single diver, accompanied by one on shore data recorder. The methods used in this study were designed to reduce the probability of double counts while retaining a definitive sample area. After completing several trial runs, a repeatable transect line in downstream progression was developed and mapped in a GIS. I applied this transect with the depth patch data acquired from the habitat component of the study to determine which patches could be adequately observed from the transect line. Those patches which met the criteria were included in the sample area, those that did not were excluded (Figure 3-2).
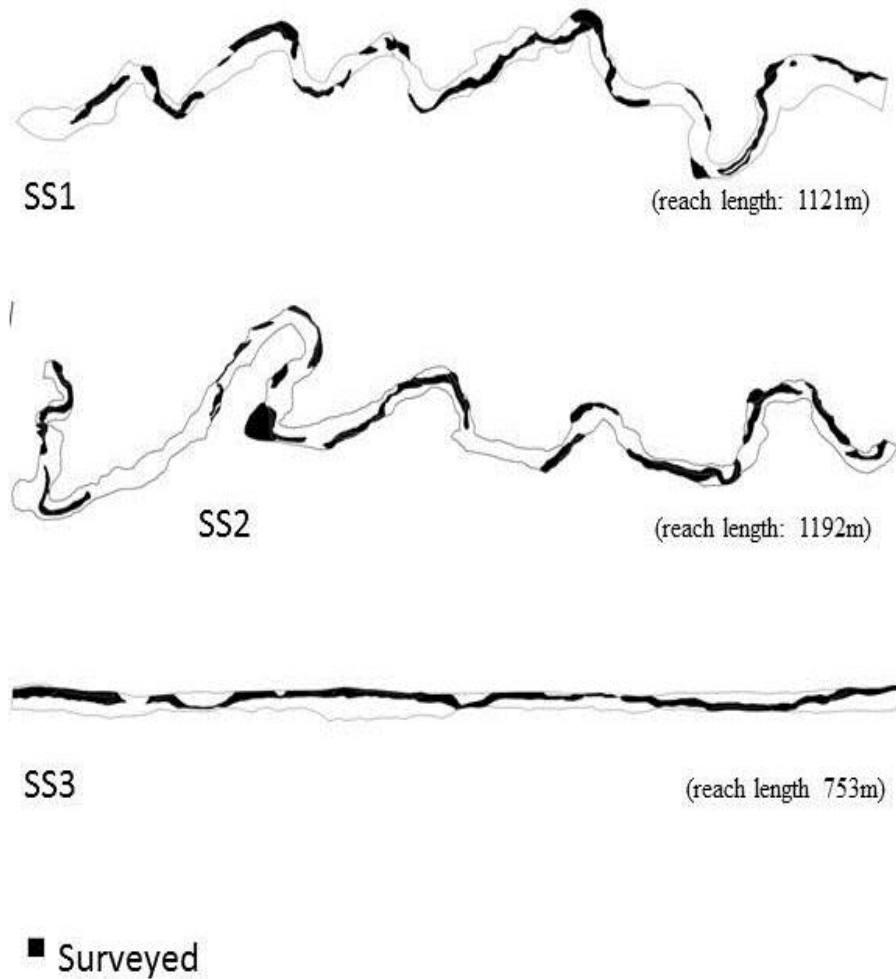
Figure 3-2. Map of Snorkel Survey Routes.

In late July, August, and early September a total of three underwater observations were

made for each of the three reaches, between 10:00 and 16:00 hr when visibility was high.

The observer worked in a downstream progression, recording species, size class and

location on a large slate attached to their arm, stopping at pre-determined points spaced

approximately 50 m apart to relay information to the on shore data recorder and exchange

slates. Prior to observations, wooden planks in the shape of fish, of known sizes, were

observed at multiple distances to calibrate visual observations of fish lengths made in the

field. Any fish whose behavior appeared to be affected by the presence of the observer

were recorded for abundance estimates, but were not included in the analysis of habitat use. Abundance was calculated as the mean value of fish/100 m^2 for each sub-category of depth and for each size class of fish from the three observational passes for each reach.

To ensure that abundance estimates were comparable across both methods, I first expanded the abundance values for each sub-category of depth to reflect the total area of each respective depth per each (multiplying abundance estimates for each depth category by the total area available of that depth in that reach, divided by the area of that depth that was actually sampled). Once an abundance estimate was available for each category of depth which reflected the total instream habitat area of the reach, I took the sum of these values and divided it by the total instream area of that reach to produce a value of fish/100m^2.

3.2.6 Analysis

Chi square contingency table analysis was used to determine if the frequency distribution of size classes acquired from sampling techniques was significantly different between underwater observations and single pass electrofishing values. These analyses were also performed on both complex and simple habitat reaches, to determine if complex habitat, such as deep pools, had any effect on this relationship.

We used one-way t-tests to determine if any statistically significant differences existed in the abundance estimates of distinct size classes across snorkel surveys and single pass electrofishing for each pair of sample reaches. I then compared these results across the complex and simple habitat reaches, to make inference as to how the effect of complex habitat might affect this relationship.

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Abundance counts obtained from underwater observations were used to estimate salmonid biomass for channelized and unchannelized reaches (GANDA 2008). Because individual lengths were not recorded, the mean size for a category was applied to all fish within the size class (i.e., for the size class 150-300mm, all fish were treated as 225mm). Lengths were converted to weights(Blackburn 2010) (Figure 3-3A,3-3B) and combined with abundance estimates, biomass was estimated.

Figure 3-3A. *O. mykiss* **Weight to Length**

log weight = -5.036607 + 3.0229763 x log length

Figure 3-3B. *P. williamsoni* **Weight to Length**

log weight = -5.665674 + 3.2639904 x log length

Biomass (g/m^2) was estimated and compared (t-test) for each study reach

3.3 Results

3.3.1 Characterization of Depth Profile of Study Sites

Both reaches characterized by complex habitat (SS1, SS2) featured a greater proportion of habitat deeper than 100 cm than the reach with simple habitat (SS3), with these deep patches accounting for 20 % and 17 % of the total area in reaches SS1 and SS2, respectively, but amounting to only 9 % of the total area in SS3. With respect to habitat heterogeneity, both SS1 and SS2 featured relatively evenly distributed proportions of depths, with the most abundant of depth class accounting for only 34 % of the total area of habitat in SS1 and only 32 % of the total area of habitat in SS1. Reach SS3, however, featured a much more homogeneous compliment of depths, being mostly dominated by glide-like depths of 60-100 cm which accounted for 48 % of the total area of habitat. A

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comparison of the total area of each depth class per sample reach across all sample
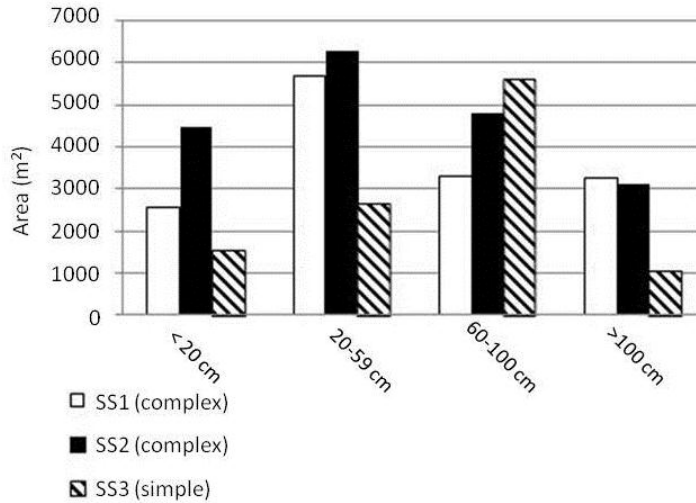
reaches is available in Figure 3-4.

Figure 3-4. Total Area of Various Depths per Sampling Reach.

3.3.2 Snorkel Survey

In total, 537 salmonids were observed in three underwater passes through each of the

study reaches. Total salmonid abundance estimates were significantly higher in the

complex reaches SS1 and SS2, compared to estimates from SS3 (t-test, p=0.0006 and

p<0.0001, respectively). In both complex habitat reaches, the most abundant size-class

observed was 150-300 mm, followed by the 300-450 mm. In the simple habitat reach, the

most abundant size-class observed were salmonids measuring 150-300 mm. Also of note,

no salmonids measuring >450 mm length were observed on any occasion in the reach

featuring simple habitat length (Table 3-2).

3.3.3 Single Pass Electrofishing

In total, 280 fish were captured upon completing a single pass of electrofishing along

each of the three sample reaches. When comparing abundance estimates across complex

and simple habitat reaches, single pass electrofishing estimated a greater abundance of

total salmonids in the simple habitat reach EF3 (1.66 fish/100 m^2) compared to

abundance estimates from the two complex habitat reaches EF1 (0.98 fish/100 m^2) and

EF2 (0.70 fish/100 m^2). In complex habitat reaches, the most abundant size-class

captured was 0-150 mm at EF1, and 150-300 mm at EF2. In the reach featuring simple

habitat, the most abundant size-class captured was 0-150 mm, accounting for

approximately 50 % of all salmonids captured (Table 3-2).

Table 3-2. Abundance Estimates of Snorkel Surveys and Single Pass Electrofishing across Sample Reaches.

| Site Name	Sampling Method	Size Class			
		0-150 mm	150-300 mm	300-450 mm	>450 mm
SS1 (complex)	Snorkel Survey	0.116 ± .019	0.490 ± .005*	0.433 ± .061*	0.091 ± .018*
EF1 (complex)	Electrofishing	0.551*	0.261	0.159	0.019
SS2 (complex)	Snorkel Survey	0.203 ± .054	0.734 ± .066*	0.249 ± .023*	0.056 ± .016*
EF2 (complex)	Electrofishing	0.213	0.331	0.145	0
SS3 (simple)	Snorkel Survey	0.179 ± .059	0.283 ± .061	0.050 ± .008	0.0 ± .00
EF3 (simple)	Electrofishing	0.745*	0.667*	0.214*	0

Average abundance (fish/100m^2) ± 1 s.e.
*Indicates significant difference among sampling methods

3.3.4 Comparison of Methods

When examining relative length class frequencies of salmonids acquired from the two

sampling techniques, chi-square analysis revealed significant differences across both

pairs of complex habitat reaches, SS1 vs EF1 (p=1.5^{-120}) and SS2 vs EF2 (p=4.2^{-11}) (Figure 3-5). The largest differences on these frequencies which were consistent across both pair wise reach comparisons appeared to be in the 0-150 mm and >450 mm size classes.

When I examined length class frequencies acquired from snorkel surveys and single pass electrofishing across the simple habitat pair wise comparison (SS3 vs EF3), however, the difference between techniques was not significant (Fisher's exact, p=.3415) (Figure 3-5).
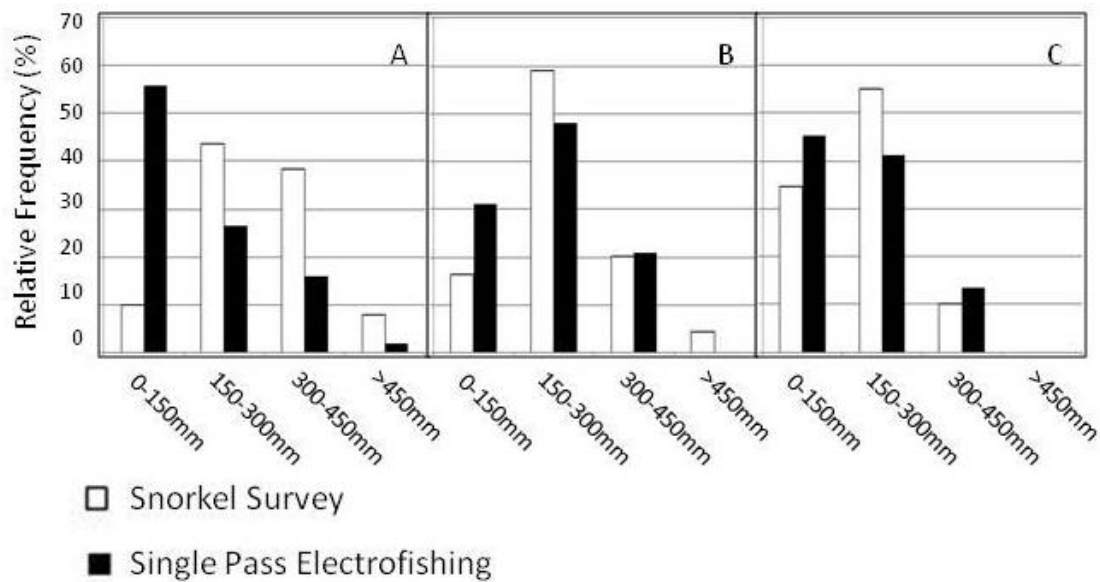
A- Comparison of complex habitat reaches SS1 and EF1
B-Comparison of complex habitat reaches SS2 and EF2
C-Comparison of simple habitat reaches SS3 and EF3

Figure 3-5. Length Class Frequencies from Snorkel Surveys and Single Pass Electrofishing across Sample Reaches.

One-way t-tests revealed a significant difference in the abundance estimates of 0-150 mm sized salmonids across the two sampling techniques for the pair wise comparison of complex habitat reaches SS1 vs EF1 (p=0.0072). This was the only size class of salmonids for which single pass electrofishing revealed higher abundances then snorkel

surveys in this complex habitat reach comparison. For the 150-300 mm, 300-450 mm and >450 mm size classes, snorkel surveys revealed greater abundances of salmonids, however, in no circumstance were these differences statistically significant (p=0.1462, p=0.1535, p=0.1794, respectively) (Figure 3-6).
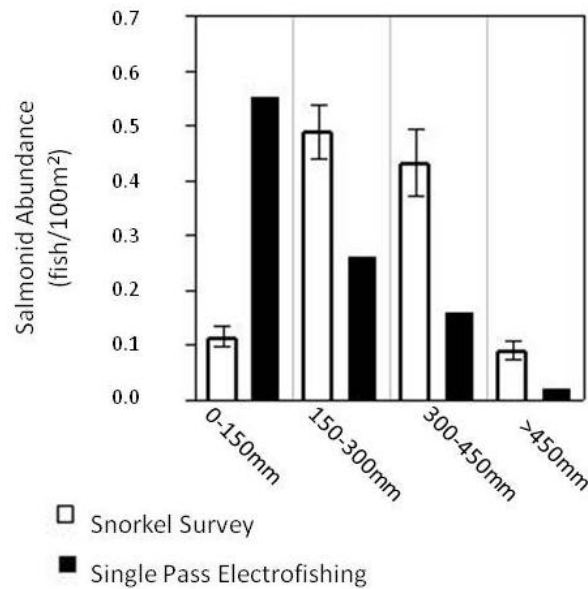
Snorkel survey abundance estimates depicted with bars representing 1 s.e.

Figure 3-6. Abundance Estimates by Size Class from Snorkel Surveys and Single Pass Electrofishing in Complex Habitat Reaches SS1 and EF1.

For the second comparison of abundance values across sampling methods in complex habitat reaches (SS2 vs EF2), snorkel surveys once more revealed greater abundances of salmonid size classes 150-300 mm, 300-450 mm and >450 mm then what was observed from single pass electrofishing. However, similar to my first comparison, no differences were statistically significant with 95 % confidence (one-way t-test, p=0.0920, p=0.1472, p=0.2217, respectively). Also similar to my other comparison of techniques in complex habitat reaches, single pass electrofishing obtained higher abundance estimates for only

one size class, 0-150 mm, however in this instance the difference in values was only
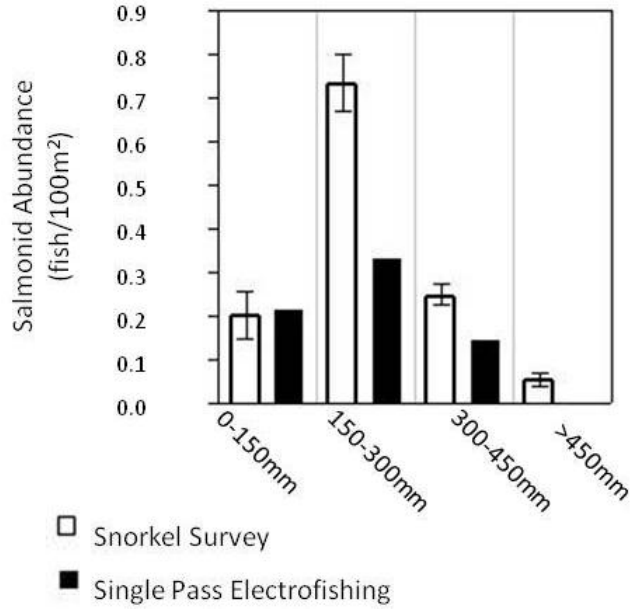
marginal (Figure 3-7).

Snorkel survey abundance estimates depicted with bars representing 1 s.e.

Figure 3-7. Abundance Estimates by Size Class from Snorkel Surveys and Single Pass Electrofishing in Complex Habitat Reaches SS2 and EF2.

Snorkel survey abundance estimates depicted with bars representing 1 s.e.

Figure 3-8. Abundance Estimates by Size Class from Snorkel Surveys and Single Pass Electrofishing in Simple Habitat Reaches SS3 and EF3.
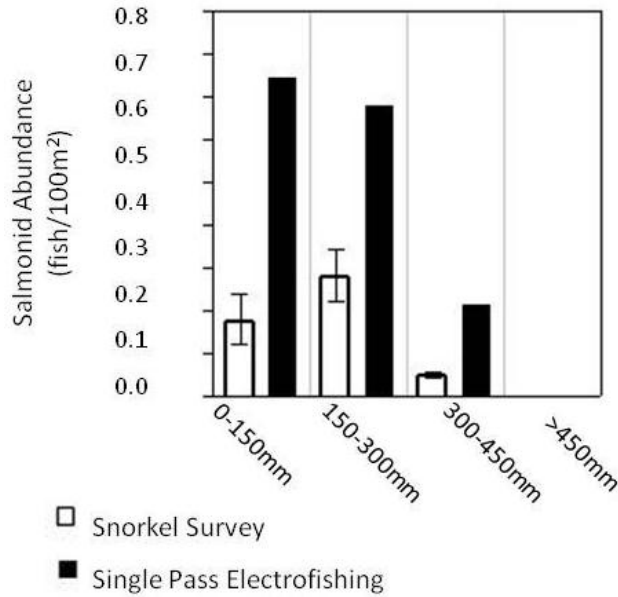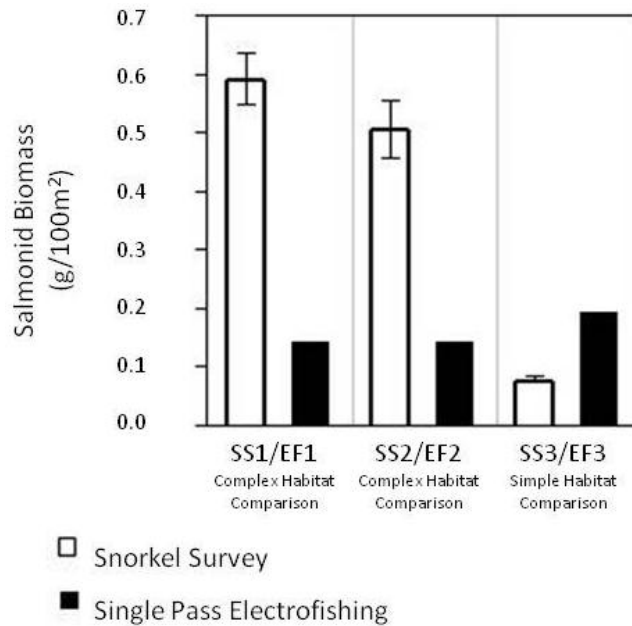
When comparing abundance estimates across the two sampling methods in reaches featuring only simple habitat (SS3 vs EF3), one-way t-tests revealed significantly higher abundance estimates from electrofishing, compared to observations from snorkel surveys, for both the 0-150mm (p=.0407) and 300-450mm (p=.0098) size classes. Single pass electrofishing also obtained greater abundance estimates for the 150-300 mm size class, however this difference was not significant at the 95 % level of confidence (one-way t-test, p=0.0846). No salmonids greater than 450 mm were observed from either sampling method within reaches SS3 or EF3 (Figure 3-8).

Snorkel survey abundance estimates depicted with bars representing 1 s.e.

Figure 3-9. Biomass Estimates from Snorkel Surveys and Single Pass Electrofishing across all Site Comparisons.

In both of the complex habitat site comparisons, snorkel surveys produced greater biomass estimates in comparison to electrofishing estimates. For the first comparison (SS1/EF1), this difference was significant at the 95 % level of confidence (one-way t-test, p=0.0347), while for the second, the difference was significant only at the 90 % level of

confidence (one-way t-test, p=0.0629) (Figure 3-9). The comparison of methods in reaches with simple habitat, however, exhibited the opposite trend, with the electrofishing method producing significantly higher biomass estimates (one-way t-tests, p=0.0092) (Figure 3-9).

3.4 Discussion

With this study I have demonstrated that in a fifth order river featuring complex habitat, which compromises the efficacy of electrofishing, snorkel surveys should be a beneficial complimentary method in assessing abundance and length class frequency, and certainly biomass estimates for salmonid populations. However, in reaches which feature monotonous habitat devoid of complex habitat features, such as deep pools and undercut banks, snorkel surveys may provide little to no benefit as a complimentary sampling method to single pass electrofishing.

On average, single pass electrofishing estimated 58 % greater abundance of salmonids between 0-150 mm in complex habitat reaches. This difference is most likely due to the habitat in which fish belonging to this size class most frequently inhabit. In the Crowsnest River, juvenile salmonids (<300 mm) occupy depths shallower than 60cm more frequently than adults (Lennox, unpublished data). Habitats which are shallow are ideal for electrofishing gear and crews, however, as the depth of habitat decreases it becomes inherently more difficult for snorkelling. Large interstitial spaces in the substrate where juvenile salmonids are most commonly found, especially in cold water streams (Hillman et al. 1992, Doloff et al. 1996), are difficult to observe in shallow depths. Similarly, Joyce and Hubert (2003) reported that snorkelling frequently failed to

observe individuals smaller than 300 mm of two coldwater salmonid species due to their common occurrence among interstitial and/or macrophyte cover (Joyce and Hubert 2003).

For each of the remaining size classes of salmonids (150-300 mm, 300-450 mm and >450 mm) observed in complex habitat reaches, abundance estimates from snorkel surveys were greater than those acquired from single pass electrofishing. This was somewhat surprising, as many previous studies report that snorkel surveys produce estimates which account for only 66-75 % of estimates acquired through electrofishing efforts (Thurow and Schill 1996, Mullner et al. 1998, Wildman and Neumann 2003). The fact that I observed greater salmonid abundance with snorkel surveys then with electrofishing in this study is most likely attributable to the habitat characteristics of the complex habitat study reaches, most notably the frequency of deep, large surface area pools. These forms of habitat make sampling via electrofishing quite difficult. Research by Gardiner (1984) showed in that in habitats deeper than 1m, electrofishing surveys become much less accurate in estimating fish abundance (Gardiner 1984). If this theory holds, then it would seem logical that in a coldwater salmonid community, any inefficiency in sampling large, deep volume pools would likely most greatly affect the estimation of the abundance of large individuals (>450mm) which most commonly occupy such habitat. This is what was found by Joyce and Hubert (2003), who reported a decline in the observed abundance of large coldwater salmonids (>450 mm) via electrofishing, in relation to estimates from snorkel surveys, compared to three smaller size classes (Joyce and Hubert 2003).

It remains to be explained why, however, unlike Joyce and Hubert, snorkel surveys in this study were more efficient in estimating the two middle size classes (150-300 mm and 300-450 mm) in the same complex reaches. I propose that this can be explained by the schooling behavior of *P. williamsoni*. Unlike many other cold water salmonid species which will actively defend optimal habitat from other individuals (Johnsson et al. 2004), *P. williamsoni* exhibit schooling behavior and therefore often will congregate among other *P. williamsoni* individuals of all size classes in deep, large volume pools (Davies and Thompson 1976, McPhail and Troffe 1988). It is then likely that in a *P. williamsoni* community, such as that of the Crowsnest River, the inhibited performance of electrofishing in deep habitats would underestimate all size classes present, and not be limited to only the largest size class.

With respect to reaches with simple habitat, and in the absence of deep water habitats, electrofishing methods were un-hindered and as such produced higher abundance estimates than snorkel surveys. Electrofishing estimates of fish abundance for size classes 0-150 mm and 300-450 mm were significantly greater with 95 % confidence, and greater with 90% confidence for the 150-300 mm size class. During low fall flows, habitat >1 m in depth accounted for less than 10 % total area in the simple habitat reaches, and these patches were always <4 m wide, posing little to no hindrance for electrofishing crews. In contrast, the shallow nature of these reaches likely reduced the efficiency of the underwater observational method due limited visibility. Also contributing to the reduction in the efficiency of snorkel surveys was the higher relative abundance of smaller, juvenile salmonids in these impacted reaches (Lennox, unpublished data). Previous research has shown that snorkel surveys are more likely to underestimate the

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abundance of smaller salmonids, compared to larger individuals, due to their cryptic nature and tendency to be found occupying interstitial spaces in shallow, marginal habitats (Cunjak et al. 1988). In communities which feature a greater density of juvenile salmonids and a high percentage of shallow habitat, underwater observational surveys may be at a greater disadvantage when estimating salmonid abundance, certainly when compared to electrofishing estimates.

Fish biomass seemed to best demonstrate the effect of underestimating the larger adults of a population. In both comparisons of complex habitat reaches, snorkel surveys produced greater total abundance estimates than did electrofishing, however, these differences were not significant. This was partly due to the fact that while snorkel surveys performed much better at estimating abundances of large fish in these reaches, electrofishing seemed to perform better at estimating the smaller size classes, which in terms of abundance make up a greater percentage of all individuals. When I transform abundance values to biomass, however, less influence is given to smaller individuals due to their much smaller mass, and much more influence is given to larger individuals, revealing difference in values obtained across methods which are statistically significant.

With this work, I have demonstrated that in deep water habitats, snorkel surveys offer a distinct advantage over electrofishing in the assessment of salmonid abundance, however, are perhaps disadvantageous in shallow, marginal habitat, and therefore should be applied with caution. In resource limited studies, snorkel surveys can and should be used in combination with electrofishing to provide a more accurate assessment of salmonid community abundance than would be available from electrofishing alone.

Conclusions

Channelization is a prevalent form of anthropogenic disturbance affecting our natural lotic ecosystems, and those aquatic communities within. With this research, I have demonstrated that stream channelization has imposed significant alterations to stream habitat of reaches of the Crowsnest River, and that these alterations have led to a statistically significant decline in abundance and biomass of *O. mykiss* and *P. williamsoni*, as well as a significant decline in their food source. I attribute this decline in abundance and biomass directly to a loss of habitat, based on in field observations of habitat use, rather than assumptions based on habitat preference literature, which I feel provides added strength to my findings.

A major factor in the decline of salmonids in channelized reaches was a significant reduction in deep water habitat, which I suggest is of greater consequence to *P. williamsoni* populations compared to those of other cold water salmonids, attributed to their schooling nature. As a result of this unique life history trait, the effects of a loss of deep habitats are not limited to only the adults, but also the juveniles of the population. As such, efforts to characterize the impacts of channelization on *P. williamsoni* populations based on previous cold water salmonid studies, would likely underestimate impacts on the population.

The results of this study should be considered with respect to future management of the Crowsnest River. To my knowledge, I am the first to demonstrate the deleterious effects of channelization on the salmonid community of the Crowsnest River, which is currently impacted along approximately 40 % of its length above the impassable Lundbreck Falls.

Furthermore, if I am correct in my hypothesis that the length of the impacted section is inhibiting colonization of upstream habitats from downstream reaches, it follows that channelization is adversely affecting even non-channelized reaches. In light of this, mitigation efforts such as the proposed stream re-activation project would not only benefit the biota of the Crowsnest River by increasing the availability of optimal habitats, but would also increase connectivity to additional optimal habitats upstream, and for this reason I am an advocate of the project's execution.

 I have also demonstrated that snorkel surveys offer a distinct advantage over electrofishing surveys in the assessment of salmonid abundance in deep water habitats which frequently occur in large, unaffected streams. This advantage, I feel, may even be more pronounced in studies which incorporate biomass as a community response metric. As such, I feel that the use of snorkel surveys in this research proved to be beneficial. However, snorkel surveys are perhaps disadvantageous in the assessment of shallow, marginal habitat, and therefore should be applied with caution. In light of this, I concede that the use of electrofishing in channelized reaches may have produced a more accurate depiction of the salmonid community, although, in the interest of applying only one sampling method across all sites for comparative purposes, I feel I was better served performing snorkel surveys. In resource limited studies, snorkel surveys can and should be used in combination with electrofishing to provide a more accurate assessment of salmonid community abundance than would be available from electrofishing alone.

For future studies which may be interested in applying these two methods in tandem, I would suggest that in order to ensure the most accurate results possible, methodologies should focus on applying these methods in a manner which capitalizes on the strengths of

each method. One way to do so would be to use electrofishing in the shallow habitats of streams and focus snorkel efforts on deep pools in which electrofishing crews are less accurate. This could be done by first applying a single pass of electrofishing in which all fish captured are removed and kept in live-wells on shore, while outlining a pre-determined buffer zone around deep water habitats which would remain undisturbed by electrofishing crews to limit sampling-induced movement of any fish occupying those habitats. This would then be followed by a single pass of underwater observation in those deep water habitats which were excluded from electrofishing. Doing so would provide an accurate community assessment fish within all habitats present in the stream, while at the same time limiting the occurrence of recounts.

While I feel that the application of such a sampling methodology would produce a more accurate assessment of salmonid abundance than either snorkel surveys or electrofishing would produce alone, it is should be noted that it is unlikely that estimates acquired from such tandem methodologies would produce as accurate abundance estimates if sufficient resources would be available to perform boat electrofishing, or other methods which are able to perform electrofishing in such a way which is not compromised by the presence of deep water habitat. However, employing such techniques is often very costly, requires unique equipment, and in many cases may simply not be feasible due to other logistical reasons, such as on a fifth order river like the Crowsnest where navigating a motorized boat is not possible. In conclusion, I propose that snorkel surveys are a cost effective solution to improving the accuracy of electrofishing abundance estimates in streams featuring deep water habitat, and where limited resources may otherwise compromise the accuracy of salmonid community assessments.

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