Low Flows, Instream Flow Needs and Fish Ecology in Small Streams

Michael J. Bradford and John S. Heinonen

Abstract: Low flows in Canadian streams and rivers can occur in both summer and winter and can be stressful for fish and other aquatic biota. Low flows can cause a reduction in habitat availability, food production, and water quality and can accentuate the effects of river ice during the winter. Human demands for out-of-channel use of water during low flow periods have resulted in the development of a suite of tools for determining the instream flows needed to maintain desired ecological attributes (most often fish populations) of the stream. This paper reviews the impact of low flows on aquatic resources in small streams as well as instream flow methods and the empirical support for them and concludes that there remains substantial uncertainty in the prediction of impacts of flow reductions or diversions. Some of this uncertainty is due to a lack of understanding of the relationship between flow and fish populations, but much is probably due to site- and time-specific variation in how stream biota responds to habitat changes. A risk-based approach, that explicitly acknowledges the uncertainty in both the hydrology and biology, is needed for decision-making in water management.

Résumé : Les basses eaux dans les rivières et cours d'eau du Canada peuvent se produire tant en été qu'en hiver et peuvent causer du stress aux poisons et au biote aquatique. Les basses eaux peuvent aussi entraîner une diminution de la disponibilité de l'habitat, de la production alimentaire et de la qualité de l'eau et peuvent accentuer les effets de la glace fluviale au cours de l'hiver. Les demandes humaines face à l'utilisation hors du chenal de l'eau pendant les périodes d'étiage ont donné lieu à la création d'une série d'outils servant à déterminer les débits réservés nécessaires au maintien des attributs écologiques voulus du cours d'eau (le plus souvent les populations de poissons). La présente communication examine l'incidence des basses eaux sur les ressources aquatiques dans les petits cours d'eau ainsi que les méthodes de calcul du débit réservé et le soutien empirique à leur égard, pour arriver à la conclusion qu'il persiste encore une incertitude considérable en ce qui a trait à la prédiction des impacts des réductions ou des déviations des débits. Cette incertitude est attribuable en partie à un manque de compréhension de la relation qui existe entre le débit et les populations halieutiques, mais elle est probablement attribuable en grande partie à une variation en fonction du site ou de la période dans la manière dont le biote des cours

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d'eau réagit aux modifications de l'habitat. Une approche fondée sur le risque, qui reconnaît explicitement l'incertitude en fait d'hydrologie et de biologie, s'avère nécessaire à la prise de décisions en matière de gestion de l'eau.

Introduction

Low flows in streams and rivers have long been recognized as drivers for aquatic and riparian ecosystems. Low flows are defined as those typical during a prolonged dry period (Smakhtin, 2001), or more precisely in the Canadian context, those that occur during periods without significant rainfall or snowmelt input. During low flows most stream habitat types are reduced in extent and changes in water quality can occur, which can be stressful for fish and other biota (IFC, 2002). In summer, periods of low flow are also those when human demands for water for agricultural or domestic needs are greatest. In winter, electricity generation by run-of-the-river hydroelectric projects creates demands for the diversion of flows during this low-flow period. Consequently, much of the focus of water management has been on finding a balance between instream and out-of-channel needs during the low-flow periods.

In Canada there is tremendous diversity in low flow hydrologic regimes (Burn *et al.,* this issue). This diversity needs to be considered when assessing the effects of low flows on aquatic biota, and when determining instream flow needs during the low flow period. The daily flows in 2005 for three small (mean annual discharge [MAD] 0.6 to 0.9 m^3/s) streams illustrate this diversity (Figure 1). For Vaseux Creek, located in the dry southern Interior of British Columbia, there was a single interval of high flow caused by spring rains and snowmelt, and flows remained low for the rest of the year. Despite being in an arid area, baseflow is the largest of the three examples at approximately 15% of MAD, probably because of the relatively large catchment (110 km^2) supplying groundwater to the channel. Carnation Creek, located on the west coast of Vancouver Island, has a small catchment (10.1 km^2), and flows are dominated by rain from Pacific storms throughout the year. Even in the winter months flows can quickly recede to base levels, which are 2-3% of MAD. The third example is provided by Catamaran Brook in central New Brunswick (basin area 28.7 km^2), where peak flows result from both spring snowmelt and early winter rains. Low flows occur during both late summer and under ice in winter (Cunjak, 1995). The baseflow in Catamaran Brook is intermediate at 6-7% of MAD. This hydrologic diversity, combined with variation in catchment geomorphology, vegetation, climate and species life histories will influence the way low flows affect stream communities.

Streams with drainage areas less than 100 km^2 are often the subject of significant flow alterations from developments and extractive out-of-stream water uses. Despite being the most numerous, watersheds under 100 km^2 in area comprise only about 15% of the basins gauged in Canada (Water Survey of Canada, HYDAT database, 2007). The absence of gauging records for hydrologic analyses and the low number of similarly sized basins for regionally-based analyses pose problems for assessing the hydrologic regime and development proposals. Synthesized hydrographs, even when bolstered by a short gauging record, are undermined by uncertainty, the level of which is seldom quantified or presented. Projected future hydrographs have the added uncertainty of climate change (Whitfield *et al.,* 2002), which is important since many developments may have impacts that extend well into the future.

The purpose of this paper is threefold. First, we briefly review the effects of low flows in small streams on fishes (and aquatic invertebrates, their main food source) during both summer and winter. Next, we summarize the methods for determining instream flow needs during low flow periods and the empirical evidence for their effectiveness. Finally, we highlight the significance of uncertainty in both the prediction of low flows in small basins and the biological impacts of altering those flows.

Effects of Low Flows on Aquatic Life

Open Water Season

In most of Canada, flows recede after the spring freshet and low flows can persist through most of the summer and fall (Figure 1). As flows decline, the wetted width, depth and mean velocity will all decrease in accordance with power functions defined by the hydraulic geometry of the channel; decreases in velocity are usually the greatest (Kraft, 1972; Park, 1977; Dewson *et al.,* 2007). The reduction in flow can cause water temperatures to

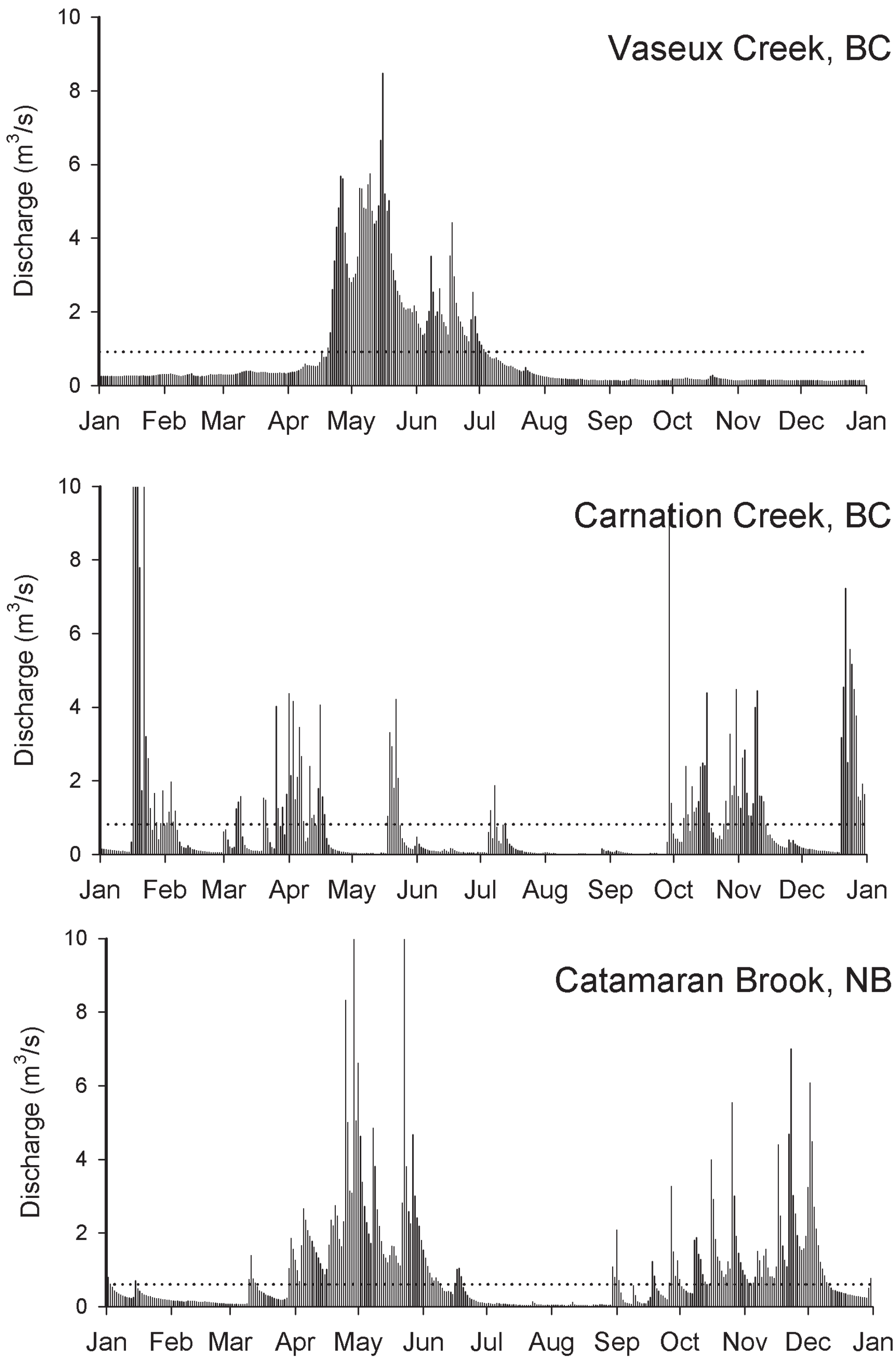

Figure 1. Daily discharge in 2005 for three streams with a mean annual discharge <1 m³/s (indicated by the dashed line), located in three contrasting climatic regions of Canada. For Carnation Creek, peak discharges reached 13.5 m³/s and for Catamaran Brook a peak flow of 11.7 m³/s was measured in May; both are off the scale of the vertical axes. Data from the Water Survey of Canada.

rise in summer, and possibly decline in cool seasons, and the concentration of some solutes can be elevated as the influence of groundwater inputs increases relative to surface water (Feller and Kimmins, 1979; Caruso, 2001; Malcolm *et al.,* 2004). These conditions are usually conducive to high biological productivity during the summer months. However, in streams with excessive nutrient levels (e.g., from agricultural runoff) aquatic macrophytes can proliferate during periods of extended low flows, which can affect habitat conditions for fish and other biota (Clausen *et al.,* 2004).

Fish production can be tightly coupled to the production of aquatic invertebrates (Huryn, 1996) which, along with falling terrestrial insects, are the primary food source for most stream-dwelling fish. Riffles are generally thought to be the prime areas for the production of aquatic invertebrates as many species prefer to live among the coarser substrates of faster flowing habitats feeding on drifting particulate matter. Riffle habitats are most adversely affected by low flows, especially when they are accentuated by water withdrawals. Because of the linkage between riffles, invertebrates and fish production, a number of studies have manipulated flows or examined the effects of drought on invertebrate abundance and diversity. A wide range of outcomes have been documented, as highlighted by a number of recent studies that report the results of short-term reductions in flows during the summer baseflow period. In a Wisconsin stream, Wills *et al.* (2006) found a reduction in invertebrate densities when streamflows were reduced by 90% from the average summer flow, but not at a 50% reduction in flow. In contrast, benthic invertebrate densities *increased* in three New Zealand streams after a >90% reduction in summer flows (Dewson *et al.,* 2007), although the absolute abundance of invertebrates may be lower because of the 25% reduction in wetted area resulting from the lower discharge. Harvey *et al.* (2006) noted a decrease in the delivery of drifting organisms to pools where rainbow trout were found after an 80% reduction in a small California stream. Suren and Jowett (2006) found little impact of prolonged low flow periods on invertebrate abundance or diversity and suggested that floods, rather than low flow periods, were more important in shaping these communities. Variation in the responses of invertebrates to flow reductions has been observed in other studies (e.g., Wood *et al.,* 2000). Because the hypothesized relation between wetted area and invertebrate abundance in riffles (as a food source for fish) is often used as the rationale for instream flow methodologies, it would be profitable to subject the existing information to a formal meta-analysis to assess the state of the evidence for this relation, similar to the analysis of Haxton and Findlay (2008), who focused on the effects of hydropeaking on invertebrates.

Low summer flows can affect fish populations in a variety of ways. For species preferring riffles or other shallow areas, the physical attributes of their habitats (depth and velocity) are impacted more rapidly by decreasing flows than pool habitats that are less affected by flow changes. Studies have confirmed that species that are riffle or fluvial specialists are more likely to be affected by changes in low flows than species that use slower microhabitats or are habitat generalists (Jowett *et al.,* 2005; Freeman and Marcinek, 2006; Lamouroux *et al.,* 2006; Haxton and Findlay, 2008). For example, the abundance or survival of pool dwelling brook trout was insensitive to reductions of up to 75 to 90% of summer flows in small streams (Kraft, 1972; Nuhfer and Baker, 2004; Harvey *et al.,* 2006). Eventually low flow will also affect pool dwelling species: a drought that reduced flows to 4% of normal summer flows was found to affect the abundance of brook trout in the Appalachian Mountains (Hakala and Hartman, 2004).

Lower flows can reduce the delivery of invertebrates to rearing fish causing a decrease in growth opportunities. Lower fish growth in years of low flows has been observed (Havey and Davis, 1970; Deegan *et al.,* 1999; Harvey *et al.,* 2006; Sotiropoulos *et al.,* 2006). The relation between discharge and fish growth is a trade-off between the increase in invertebrate abundance with higher flows, and a decrease in foraging efficiency and higher energetic costs associated with higher velocities. These factors have been successfully modelled (Hughes and Dill, 1990; Nislow *et al.,* 2004), enabling habitat to be evaluated with respect to the potential rate of net energy gain (defined as the energy consumed less the costs of metabolism, foraging and swimming) by a fish occupying it, possibly subject to modification due to the risk of predation (Rosenfeld, 2003).

There are other mechanisms by which low flows can affect stream fish populations. While many fish migrations are prompted by high flow events that are not the subject of this review, more routine movements among habitats can be restricted if low flows result in the significant shallowing of riffle areas. Changes in water temperature can also affect the competitive balance among native fish species (Reeves *et al.,* 1987). Low flows have been demonstrated to favour exotic species over native fishes, if the latter are adapted to higher flows and greater flow variability (Marchetti and Moyle, 2001). However, most instream flow assessments are conducted on single target species and interactions among species are not often considered.

Ice Covered Season

For most of Canada, winter is a period during which flows are low and small streams become covered with ice. In northern regions, flows can be nonexistent in small streams when the freezing of the soil from the surface extends down to the permafrost layer, preventing shallow groundwater from reaching the stream channel (Woo, 1986). Across the southern part of the country the severity of the winter low-flow period depends on the climate (Figure 1).

For fall spawning species and in particular many salmonids, eggs and larvae (alevins) remain in spawning gravels through the winter, to emerge the following spring. As flows recede through the winter, eggs and larvae experience much lower flows in mid-winter compared to the time of spawning. In extreme cases river stages can recede sufficiently to dewater spawning areas. Although eggs can survive in moist air, alevins are not as resilient and mortality can result within a few hours. Alevins do have the ability to move downwards within spawning gravels to avoid desiccation (Dill, 1969) but successful fry emergence also requires sufficient flow so that alevins can access areas of free flowing water.

Hyporheic flows deliver important oxygenated water to redds and alevins and remove metabolic waste and carbon dioxide. Lower surface velocities associated with receding flows can result in lower hyporheic flow and an increased presence of deeper groundwater in spawning beds (Wickett, 1954). The lower dissolved oxygen levels of the deeper groundwater (Malcolm *et al.,* 2004) can cause mortality of incubating eggs and larvae.

Low winter flows and cold air temperatures can cause the freezing of interstitial water in spawning beds or the restriction of subsurface flows due to the freezing of surface ice on the stream bottom. Direct freezing is a cause of mortality in spawning redds, especially in years of low snowfall, since snow can serve as an effective insulator (Reiser and Wesche, 1979).

Winter is considered a critical stage for stream-dwelling fishes as the combination of low water temperatures, low flows and ice presents substantial challenges to their survival (Cunjak *et al.,* 1998). Prior to winter, fish attempt to build an energy reserve in the form of stored lipids; thereafter they use a combination of stored energy and a reduced rate of feeding to balance their energy needs. In many cases fish are nocturnal in winter, emerging from the substrate or cover at dusk to feed (Cunjak, 1996). This behaviour is thought to be a predator avoidance mechanism as vertebrate predators have been observed to be active in winter, and the escape responses of fish are likely reduced by lower water temperatures. The density of invertebrate food sources is also reduced in winter (Martin *et al.,* 2001) and is affected by ice and freezing conditions, which may affect their availability.

Winter habitat use varies among species. Juvenile salmonids may occupy shallow habitats, as they do in summer months, but will make use of coarser substrate elements to find daytime shelter in the streambed (Cunjak, 1996). Many species will move to specific overwintering locations, which are typically deeper water areas that are less vulnerable to the effects of declining flows and ice conditions (Dare *et al.,* 2002). Along the Pacific coast winter flows vary between storm induced spates and base flows (i.e., Carnation Creek, Figure 1), and species such as coho salmon will make use of off-channel habitats that are sheltered from flow extremes (Bustard and Narver, 1975).

In general, reduced flows in winter months tend to exacerbate the factors that make this a stressful period for stream fishes. Notably, reduced flows will reduce the size of the wetted channel and the amount of habitat for riffle-dwelling species. Pools and other slack waters will be less affected, although their connectivity might be reduced by limited water depths or flows in intervening riffle areas. Low flows can increase the incidence of frazil or anchor ice as more of the streamflow is exposed to cold air, especially over riffle areas where cold air is entrained by turbulence. Fish tend to avoid frazil ice, as ice crystals can damage gill tissues; a number of studies have documented fish movements during frazil ice episodes (Brown, 1999; Simkins *et al.,* 2000). The subsequent deposition of frazil ice generally occludes habitat from use, although Roussel *et al.* (2004) document Atlantic salmon hiding under anchor ice formations associated with large boulders. Frazil ice can also fill pools, forcing fish to move from these habitats (Komadina-Douthwright *et al.,* 1997). Anchor ice has also been implicated in a winter-long decline in invertebrate abundance (Martin *et al.,* 2001).

Although likely less of an issue in small streams, low flows under winter ice can also affect water quality, and in particular dissolved oxygen concentrations. Low dissolved oxygen levels are often observed as a result of natural and anthropogenic biological oxygen demand, and the increased significance of oxygen-poor groundwater during winter (Whitfield and McNaughton, 1986). These factors are likely to increase under situations of reduced surface flows (Prowse, 2001).

Empirical support for the linkage between flows and fish production is available in the form of positive correlations between overwinter survival and discharge. Winter discharge during the egg/alevin incubation period and survival were related for Atlantic salmon in Quebec, Newfoundland and New Brunswick (Chadwick, 1982; Frenette *et al.,* 1984; Gibson and Myers, 1988; Cunjak *et al.,* 1998) and sockeye salmon in Kamchatka (Selifonov, 1987) supporting the hypothesis that diminished low flows can increase egg or alevin mortality due to desiccation or freezing. Positive correlations between winter flows and the survival or abundance of juvenile fish have also been identified (White *et al.,* 1976; Cunjak *et al.,* 1998; Hvidsten, 1993; Mitro *et al.,* 2003). However, the winter can be a very dynamic period and other factors such as air temperatures, precipitation patterns, and ice events (e.g., ice jams) will also affect survival and may obscure relations between base flow levels and fish population responses (Cunjak *et al.,* 1998).

Finally, overwinter survival of fish is often a function of the size and lipid content of fish at the onset of winter (Biro *et al.,* 2004). If low flows in summer affect fish growth and condition adversely (Havey and Davis, 1970; Harvey *et al.,* 2006), a negative synergism could result for fish experiencing the combination of low flows in both the summer and the subsequent winter.

Low Flows and Instream Flow Needs

Summer

The period of low flows is usually when the greatest competition for water exists between instream flow needs and out-of-channel users. During the summer, irrigation and domestic water supply needs peak when flows in streams and rivers are low. During winter, electricity demands place a premium on the diversion of flows for power generation purposes. Consequently, most instream flow methodologies were developed for the determination of the so-called "minimum flows" for the low flow periods. In recent years there has been an increasing recognition of the role of seasonal and interannual variation in flows in structuring river and riparian communities, and most modern instream flow prescriptions take into account the major features of the natural flow regime (Poff *et al.,* 1997). In practical terms this usually means ensuring there are appropriately timed high flow periods, as would occur in an unimpacted stream (IFC, 2002). These high flow events are important for the maintenance of geomorphological processes, the flushing of fine sediments, and ensuring connectivity between the main channel and the floodplain. High flows are often integral to biological processes such as riparian recruitment and cueing fish migrations (Poff *et al.,* 1997). The following discussion, however, will focus on the low flow periods.

The determination of instream flow needs during the low flow period depends on the ecological objectives for the stream. These can vary from legal or policy requirements such as Canada's no net loss policy for fish habitat (DFO, 1986), a desire to maximize the production of game or commercially valuable fish, or to provide conditions to benefit native fish communities (IFC, 2002). Riparian and recreational values can also be important. In general, most methods have focused on providing suitable or optimal conditions for game fish species, though the movement to the consideration of the whole aquatic system is under way (Anderson *et al.,* 2006).

Instream flow methods vary with respect to their information requirements and their aims. The simplest and easiest to implement, especially where flow data are limited, are the *standard setting methods* based on the mean annual discharge (MAD) of which the Tennant (1976) method and its variants are the most widely used. These methods are derived from the well-known curvilinear relations between discharge and the width, depth and velocity of the stream (i.e., river hydraulic characteristics). Minimum flows are set at the point at which stream characteristics (usually width and depth) begin to change rapidly as discharge decreases. In the case of streams in the American west, Tennant (1976) concluded, largely from observation, that flows of 10% of MAD would be the minimum to sustain life and that 30% of MAD would result in "good" conditions.

A number of studies have demonstrated that the use of a flow standard based on MAD can have different outcomes depending on the size and nature of the stream. The goal of these methods is to maintain sufficient wetted width and flow to sustain food production and habitat characteristics. However, O'Shea (1995) and Rosenfeld *et al.* (2007) found that a greater proportion of MAD than suggested by Tennant is needed to meet these goals in smaller streams. They also found that Tennant's method tends to overestimate flow needs in larger streams. Standard setting methods are sensitive to the hydrologic regime under consideration. MAD is a function of precipitation in the catchment, which is often concentrated in a portion of the year, and in wetter regions flow during the base flow period will tend to be a lower fraction of MAD than in more arid areas (see Figure 1). Tennant's method was developed for the interior of the western United States and may need to be recalibrated for other climatic regions that have distinctly different flow regimes than those used in its development.

Tennant's method was based on alluvial channels in relatively gentle terrain, and its relevance to small, steep streams that are often used for small-scale hydro or water storage projects in montane regions is unclear. Reid (2005) found that wetted width was insensitive to changes in discharge across a wide range of flows in small mountain streams in coastal British Columbia. The average hydraulic geometry exponent for width, b, was 0.2. These streams had extremely rough channels and higher volumes were accommodated by an increase in velocity as the resistance to flow decreased significantly at larger flows. Whether the flows suggested by Tennant (1976) or Rosenfeld *et al.* (2007) are adequate to support fish in these streams is not known.

A slightly more complex means of determining instream flows during the low flow period are the *historic flow methods* that use historical flow information and set flows that are within the range of the observed flows. The implicit biological assumption here is that stream biota will have adapted to commonly experienced flows and should not be adversely affected by flows that are within the range of those that occur naturally. No specific target species or ecological process is identified. A commonly used instream flow recommendation is the median (Q_{50}) monthly flow observed in the late summer; other variants are detailed in IFC (2002). A shortcoming of these methods is that there is a greater requirement for daily flow information: Caissie *et al.* (2007) show that coefficients of variation (CV) of 3-6% are associated with estimates of Q_{50} with 10 years of gauging data. For ungauged basins, or in cases where only a year or two of data are available, the uncertainty would be considerably greater. Although these methods prescribe discharges within the historical range of low flows, they may still impact fish populations if aquatic productivity covaries with flows within the natural range (Havey and Davis, 1970, Nislow *et al.*, 2004). If flows above Q_{50} are licensed for abstraction the benefits that would be associated with years of higher than average flows would be eliminated, resulting in a loss in long-term productivity.

Hydraulic methods are site-specific and take into account the shape of the stream channel to derive flows that maintain aspects of the river's characteristics thought necessary to achieve the management objectives. The stream width or wetted perimeter required to maintain riffles to support invertebrate production is the most commonly used metric (Jowett, 1997). Flow recommendations are usually made on the basis of breakpoints or the point of maximum curvature in the relation between stream width and discharge, usually measured or modelled in riffle habitats. The identification of breakpoints can be difficult (Gippel and Stewardson, 1998); a more robust procedure may be to set an arbitrary threshold such as maintaining a certain proportion of the original wetted width (IFC, 2002).

In contrast to the previous methods, *habitat preference methods* explicitly attempt to estimate the relations between flow and the specific biological requirements of target species. Most commonly, this is the amount of physical habitat (defined as preferred depth, velocity and sometimes substrate) available for specific species and life stages at various flows (Jowett, 1997). The PHabSim software was the first to take this approach and used a river simulation model and fish habitat preference information to calculate the "weighted useable area" (WUA) for each species and life stage as a function of discharge (IFC, 2002). Improvements to the hydraulic models (e.g., River2D, Ghanem *et al.,* 1996) and the ways that the biological data are used have occurred over the past 25 years. As currently practiced, it is the most complex and expensive instream flow methodology in use.

While widely used, PHabSim and its variants have been equally widely criticized. Some of the issues surrounding the modelling of flows have been largely overcome with technological advances; the most significant unresolved issues surround the assumptions underlying the biological components of the approach. The development, use and transferability of the habitat suitability curves have been extensively debated, and recent studies have shown that habitat preferences of the fish (depth and velocity) may in fact vary with flow itself, invalidating a key assumption of the approach (Holm *et al.,* 2001). The more detailed modelling of growth potential discussed earlier is likely a more realistic approach to habitat selection for drift-feeding fishes, but well beyond the likely means for instream flow analysis in most streams and rivers. The critical assumption that there is a positive correlation between the model output, WUA and the abundance of the target fish species has not been supported by empirical studies (Scott and Shirvell, 1987; Rosenfeld, 2003). There are many factors not related to hydraulic habitat preferences that could lead to a failure of fish populations to respond to changes in WUA (Sabaton *et al.,* 2004). Finally, and perhaps most significantly, the recent trend to broad-based instream flow assessments that incorporate a wide variety of taxa and ecological processes, as well as an emphasis on the role of the natural flow regime, may not warrant the effort required to model physical habitat for a few target species and life stages as practiced by PHabSim (Anderson *et al.,* 2006).

Despite the proliferation of methods for determining instream flow needs, and their ongoing use, there continues to be a lack of empirical data for the efficacy of these tools and the decisions that are made with them. This lack of evidence compelled Castleberry *et al.* (1996) to declare that "no scientifically defensible method exists for defining the instream flows needed to protect particular species of fish or aquatic ecosystems"; a similar opinion was expressed a decade earlier by Larkin (1984). Such evidence will consist of before-after monitoring of streams that have had an instream flow prescription applied to them to evaluate the response of the target biota relative to predictions made from instream flow methods. Sutherland *et al.* (2004) call for the use of an evidence-based approach for environmental management that is similar to that now used in the medical sciences, so that decisions are made on the basis of experience rather than accepted practice. Unfortunately, such empirical examples are remarkably few given the time and effort expended trying to predict these effects with the various instream flow models and other approaches. The absence of an extensive body of knowledge probably stems from the very significant investments in monitoring that are required to detect changes in stream biota, and in particular fish abundance, that might occur as a result of a change in habitat conditions (Bradford *et al.,* 2005).

A few studies have experimentally reduced flows during the summer low-flow period in small streams and monitored the short-term response of biota to the change. The results of those studies are fairly consistent in revealing little change to invertebrate or fish populations with the diversion of 50-75% of the summer low flows, which leave approximately 10-20% of MAD in the channel (Kraft, 1972; Rimmer, 1985; Nuhfer and Baker, 2004; Wills *et al.,* 2006; Dewson *et al.,* 2007). Decreases in abundance or production were observed when most (>75%) of the summer flow was diverted (usually leaving <10% MAD residual flow). The absence of a response in the fish populations were attributed to the relatively small changes in wetted width with flow, and the preference of some of the target species for pool habitats, which are little affected by flow reductions (Kraft, 1972). Nuhfer and Baker (2004) and Wills *et al.* (2006) note little correspondence between the predictions made by the PHabSim modelling tools and the observed responses.

There is a small body of published case histories where instream flows have been increased below diversion projects as a river restoration strategy and the response of fish and other biota has been monitored. Examples from smaller streams are summarized in Table 1. In most cases, these have involved increases from very low existing flows, ranging from <1 to 3% of MAD to flows ranging from 4-18% of MAD. Increases in fish or invertebrate abundance were sometimes found, however, in other cases, factors other

Table 1. Case studies of small (MAD <10 m³/s) streams where the minimum instream flow was increased and the response of biota was monitored. Flows are indicated as m³/s, and as a percentage of MAD where possible. Examples in larger rivers are provided by Jowett and Biggs (2006) and Lamouroux *et al.* (2006).

Stream	MAD	Flow Before	Flow After	Response
Barrows Stream[1]	N/A	0.02	0.10	40% increase in age 0+ salmon abundance, increased growth
Candover Brook[2]	0.7	≈0.2 (28%)	≈0.4 (57%)	No change in trout abundance or survival
Douglas Creek[3]	0.89	0.03 (3.2%)	0.16 (18%)	4-5 fold increase in trout abundance
Roizonne[4]	2.4	0.07 (2.5%)	0.28 (12.5%)	Decrease in trout abundance
Lignon[4]	2.9	0.08 (2.5%)	0.35 (12.5%)	3-fold increase in trout abundance
Aude[4]	7.3	0.20 (2.5%)	0.63 (12.5%)	No change in trout abundance
Moawhango[5]	9.6	0.06 (0.6%)	0.6 (6.3%)	Restoration of invertebrate populations

[1]Havey (1974); [2]Soloman and Paterson 1980; [3]Wolff *et al.* (1990); [4]Sabaton *et al.* (2004); [5]Jowett and Biggs (2006).

than the minimum flow were felt to be controlling fish abundance, which prevented the newly created habitats from being fully utilized (Sabaton *et al.,* 2004).

Other examples include cases where base or minimum flows have been increased below facilities with daily flow fluctuations (hydropeaking) resulting in an increase in quantity and diversity of fishes, probably as a consequence of the increased stabilization of habitats (Travnichek *et al.,* 1995). Baran *et al.* (1995) found a positive correlation between the amount of residual flow released below hydro facilities and brown trout abundance in a suite of French rivers. Correlations between annual variations in instream flows and fish populations within regulated rivers are often more difficult to detect as there can be other biotic and abiotic factors that vary from year to year that will also affect growth, survival and abundance (Nehring, 1979).

Two observations emerge from the summary of case histories. First, the categorization of low summer flows on fish and other aquatic biota by Tennant (1976) may be inconsistent with observations made elsewhere. While Tennant considered flow from 10-30% of MAD "fair or degrading" the flow reduction experiments do not show evidence of reductions in fish populations at these flows (see also Baran *et al.,* 1995). This may be a result of the species or stream types in those studies, and this generalization may not apply to all situations. It is important to note that no water quality issues were identified in the case studies, nor were interactions among species considered; these will certainly be concerns in some areas.

Secondly, in the cases of flow augmentation, there was diversity in the responses of the target fish populations highlighting the comments made by many authors that physical habitat conditions during low flow periods may not always be the key factor limiting abundance. In these cases either habitat modelling or subjective evaluation (e.g., Tennant's method) would suggest the increase in flows should have had a significant positive effect on habitat availability and consequently on fish abundance. However, there was a range of responses to the flow change highlighting the importance of site-specific factors in determining the outcome of an instream flow change.

Winter

The prediction of the effects of water withdrawals during low flow periods in winter is problematic. While standard setting and hydrological methods can and do provide instream flow recommendations for the winter months (Tennant, 1976; Caissie and El-Jabi, 1995; Hubert *et al.,* 1997) their efficacy in protecting aquatic biota is largely unknown. Standard habitat modelling can be used, but it is also unclear whether the abundance of appropriate physical habitat (depth, velocity, substrate) in winter is limiting fish populations relative to other factors such as ice formation. Reductions in flow in the winter months have also been speculated to increase the likelihood of frazil and anchor ice formation, especially below dams, where ice-free conditions and shallow, turbulent flows can result in supercooled water in periods of low air temperatures (Prowse, 2001). The development of models that incorporate winter habitat use by fish, as well as the complexities of ice formation and the effects of ice on flows, is under way (Alfredsen and Tesaker, 2002), but because of their complexity the models are unlikely to find routine use in small stream assessments.

Part of the challenge for winter instream flow assessments is that the life history and behaviour of the target fish species and the local habitat and climate conditions can interact to result in a wide range of possible impacts of flow reductions on fish. In regions of Canada with mild winters, flow reductions may be more benign because food requirements of fish are generally low and low flows may tend to increase the amount of low velocity habitats used by nocturnally active fish (Allen, 2000). However, in colder regions reduced winter flows and ice formation may restrict the availability of this habitat (Cunjak, 1996). Mitro *et al.* (2003) demonstrated that low winter flows restricted access to bank habitat that was preferred by rainbow trout, resulting in a strong correlation between overwinter survival and flow. In other situations the use of pool habitats may buffer fish from low flows (Dare *et al.,* 2002) although the accumulation of frazil ice from riffles can be significant (Simkins *et al.,* 2000). Because the mechanisms by which winter flows and conditions affect fish performance and survival are still being described, it is unlikely that a standardized instream flow assessment methodology for the winter period will be developed in the near future. Nonetheless, the prevalence of positive correlations between winter base flows and fish production cited earlier suggests there are significant risks to aquatic biota from water withdrawals during winter (Cunjak, 1996).

Dealing with Uncertainty

The prospect of increasing human demands for water during low flow periods and the associated changes in hydrologic regimes means that the determination of instream flow needs during the low flow periods will continue to be a critical component of water management strategies. Factors such as climate change will further complicate this process in the future. The uncertainty in both the characterization of instream flow "needs", as well as the variability in the response of aquatic biota and fish populations to those flows, requires that risk assessment and risk management procedures be used for formulating management advice. Instream flow assessments are often characterized in terms of flow "requirements" or thresholds that may result in a sense of certainty that isn't supported by the available evidence. Accumulation of experience from ongoing and proposed monitoring programs at sites across Canada where flow changes are proposed, or have been enacted, would help to better describe the uncertainty that surrounds predictions of the effects of water withdrawals during the low flow periods.

The presence of uncertainty is not restricted to the response of aquatic biota. In small ungauged basins, the prediction of flows that form the basis for development proposals and most instream flow methods is often far from accurate. Uncertainty analysis is not standard practice in hydrologic modelling and it is common to present results without uncertainty bounds to decision-makers (Pappenberger and Beven, 2006). It has been strongly suggested that statistical confidence levels should be attached to even the most complex scientific predictions (Giles, 2002). It seems evident that a set of guidelines for uncertainty analysis, associated with hydrologic predictions in ungauged basins, would be beneficial to both practitioners and decision-makers. Pappenberger and Beven (2006) suggest that a Code of Practice is needed to formalize guidance on methods and applications of uncertainty analysis. With information about the uncertainty associated with hydrologic estimates, decision-makers would be in a better position to evaluate the strengths and weaknesses of development proposals from both engineering and ecological perspectives.

The current state of knowledge about the effects of alterations of low flows on fish and other aquatic resources can be summarized by Figure 2, which is a generalization of a figure produced by Healey (1998). This diagram suggests that risk generally increases as low or base flows are reduced, but other than at the two endpoints, there is considerable uncertainty in the biological responses for a given hydrological change. Part of the risk is due to our inability to predict both the hydrology and biological responses with models and other tools, and part is due to the effect unknowable future events (both physical and biological) have on the

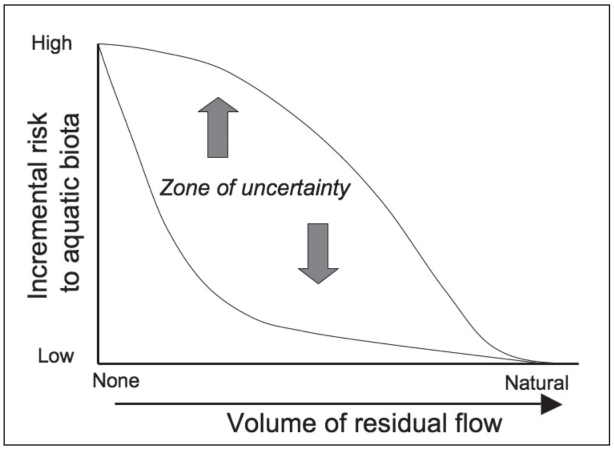

Figure 2. Conceptual relation between the risk to aquatic biota and the residual flow during the low flow period for a hypothetical diversion project, adapted from Healey (1998). Risk decreases as the flow remaining in the channel increases, but there is a large "zone of uncertainty" representing the combined effects of a lack of knowledge of the system (which might be reducible through monitoring and research) and unpredictable natural variability in the response of biota to the change in flow.

outcome (Healey, 1998). Although it may be possible to improve the predictions made by the models over time, there will continue to be considerable unpredictability in the response of stream biota to changes in the flow regime that cannot be reduced by the development of new models or tools. The admission of this uncertainty into the decision-making process will change the way both users of out-of-stream water and regulators approach instream flow determinations, as it will require an explicit consideration of risk and risk tolerances in the context of trade-offs between the multiple uses for water.

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