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RESTORATION OF STREAM HABITAT FOR FISH USING IN-STREAM STRUCTURES

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ABSTRACT

Restoration of fish habitat by manipulating channel morphology is a common practice in western North America. Widespread damage to streams has been caused by logging, by erosion from destabilised land, by direct removal of wood from the streams, and by the use of stream channels for log transport. Most restoration techniques focus on providing pool habitat and low-velocity refuges through placement of large wood or boulders in the low flow stream channel.

In many instances, placement of structures has not been particularly well planned, or its effect on fish numbers evaluated. Effort has been wasted in placement of inappropriate structures, and because structures have been built without consideration of the interaction of species' habitat requirements within a given stream gradient or hydrologic regime. Successful habitat restoration must take into account the limits imposed by natural stream width and gradient, beyond which enhancement cannot hope to improve, and the known habitat requirements of the species to be enhanced.

The habitat requirements of many native freshwater fish species in New Zealand are insufficiently understood to permit habitat restoration with guaranteed success. In addition, restoration of rearing habitat may be of little value in the case of migratory species, for which the real limitation could be survival at sea or at some point downstream in the migratory route that may limit access. The best options to restore or protect in-stream habitat are 1) re-establishment of natural processes that provide structure to stream channels, e.g., planting woody vegetation in the margins, and 2) prevention of disturbance to riparian zones with existing woody vegetation.

INTRODUCTION

Habitat diversity influences the structure and composition of stream fish communities (Gormann and Karr 1978, Schlosser 1982, Angermeier and Karr 1984). More diverse habitat conditions support a greater range of species and age classes than do simple habitats. Habitat diversity can also mediate biotic interactions such as competition (Kalleberg 1958, Hartman 1965) and predation (Crowder and Cooper 1982; Schlosser 1982).

Simplification of stream habitat can influence the structure and composition of the fish community. Physical consequences of simplification include a decrease in the range and variety of hydraulic conditions (Kaufmann 1987) and reductions in structural elements (Bisson *et al.* 1987), frequency of habitat units, and substrate types (Sullivan *et al.* 1987). The response of individual species to such changes will depend on the extent to which the required habitat features and processes such as predation and competition are altered.

Habitat simplification generally results in a decrease in diversity of the fish community (Angermeier and Karr 1984, Li *et al.* 1987, Rutherford *et al.* 1987, Reeves *et al.* 1993). Some species may decline in numbers and others increase in response to habitat changes. As a rule, species with the greatest tolerances to environmental extremes are better able to maintain themselves than are species with smaller tolerances (Rutherford *et al.* 1987).

Agencies and organisations responsible for managing fish habitat have used in-stream structures to restore habitat. Such techniques have been used for more than 50 years in North America (Reeves *et al.* 1991). In-stream structures have not been used extensively in New Zealand, but with fish habitat degraded in many streams by past or current land-use management practices, it is relevant to ask if habitat restoration could be accomplished through the use of structures here. The objectives of this paper are to: 1) review the use of in-stream structures for restoring habitat for stream fishes in western North America, and 2) assess the applicability of using structures for restoring habitat for stream fishes in New Zealand.

RESTORING STREAM HABITAT IN WESTERN NORTH AMERICA

Types of Structures and Use

In the past twenty years, a diverse array of in-stream structures have been used to restore or modify fish habitat in streams throughout the Pacific coast of North America. In the vast majority of programmes designed to restore habitat altered as a result of land-use management activities, structures are the core components. These programmes have focused primarily on a number of species and races of anadromous salmonids (*Oncorhynchus* spp.). Other projects involving in-stream structures have been used in attempts to increase numbers or biomass of fish in streams where some element of habitat limits production. No exact accounting of the costs of such undertakings is available, but it is estimated that hundreds of thousands of dollars have been spent on the design, construction, and maintenance of in-stream structures during this time period.

These structures varied from single logs or boulders, which were placed in various positions within the channel, to gabion, log, and rock weirs, and complex combinations of materials. The three predominant systems of providing channel structure and in-stream cover are 1) bank placements, particularly at the outsides of bends; 2) weirs across the channel to create dammed pools upstream of the weir, and plunge pools below the weir; and 3) mid-channel placements of material, to create scour holes and velocity refuges.

Materials for these structures are found either in the vicinity of where work is done or are brought to the site from outside areas. Several different types of material are in common use for creating structures. Large logs may be transported to the stream, or sometimes trees are felled into the stream directly from the riparian zone. Boulders can also be used, usually from quarries, or alternatively gabion baskets can be filled with quarry rock or alluvial gravels from the channel. Floods can scour around objects in stream channels, undermining and transporting them downstream. Wood, because of its low density compared to rock, is the material most susceptible to movement. To improve stability and prevent transport, structures are often anchored into position, either by excavation or using stainless steel wire

Figure 1 **Trees, stumps, and logs cabled into position are used for bank protection and to provide habitat for salmonid fish (from Bisson *et al.* 1987).**

cable. Logs can be cabled to boulders, and trees felled into the stream from the riparian zone can be cabled to their stumps.

The various types of structures, common materials used, position in the channel, and requirement for anchoring are summarised in Table 1. Reeves *et al.* (1991) describe the more commonly used structures. Other types have been designed and constructed by fish biologists but have not been formally described. The primary reason for this lack of documentation is that the structures generally failed to perform or create the desired habitat features. It is unfortunate that there is a reluctance to report failures with the same enthusiasm as successes, because much could be learned from failures (Reeves *et al.* 1991).

Table 1 Types, channel positions, materials used, and requirements for anchoring of structures used for restoration of in-stream habitat for fish.

Position of structure	**Material**		
	Wood	**Boulder**	**Gabion baskets**
Bank (usually on the outside of bends in large streams or rivers)	Trees, stumps, or logs cabled into position (Fig. 1)	Wing deflectors or jetties	Wing deflectors or jetties (Fig. 2)
		Rock revetment	Gabion revetment – rectangular basket or mattress form
Weir (across the channel in small streams)	Log weir – V-weir, or perpendicular to flow	Rock weir – angled or perpendicular to flow, or V-weir	Gabion basket weir – V-weir (Fig. 3A) or perpendicular to flow (Fig. 3B)
Mid-channel	Logs, stumps, or trees – usually cabled to boulders	Large single boulders, or clusters	Not generally used

Figure 3 Weirs made of gabion baskets create scour pools and trap spawning gravel for salmonids. (A) Plan view of V-weirs, which concentrate flow in the middle of the stream, and (B) side view of a weir perpendicular to the stream flow (from Reeves *et al.* 1991).

There has been a progression in the complexity of structures designed and built during this period. Hall and Baker (1982) and Reeves *et al.* (1991) reviewed this development history. Initial efforts were relatively simple, often patterned after structures used in the Rocky Mountains or mid-western portions of the United States. Many of these initial designs were unsuccessful because streams in the Pacific Northwest were much more dynamic than

Figure 2 Gabion baskets are used as jetties to reduce bank erosion and provide cover for salmonid fish (from Reeves and Roelofs 1982).

Figure 4 The seasonal increase in mean density of juvenile coho salmon was greater in experimental reaches (with added channel structures) than in control reaches (with no added structures) in Southbay Dump Creek, a stream on the Queen Charlotte Islands, British Columbia (from Tripp 1986).

streams in other regions. Most in-stream structures have been designed for use and constructed in 2nd-5th order streams (Strahler 1957). Streams in which such structures have been used are generally between 1 to 5% in gradient with hydrographs dominated by either winter rain or spring snow melt. As a consequence, structures used in other areas have been modified or new structures developed that are better suited to regional conditions.

Results of Use of In-stream Structures

Results from habitat restoration and modification programmes have been mixed. Some programmes have apparently achieved their purpose of increasing the number of juvenile, smolt, and/or spawning adult anadromous salmonids (e.g., House and Boehne 1985, 1986, House *et al.* 1989, Crispin *et al.* 1993). In contrast, Frissell and Nawa (1992) found that the vast majority of structures that they examined failed to create or improve the habitat conditions for which they were intended. They found that some even had negative impacts on habitat. Reeves *et al.* (1990) found that, in a long-term evaluation of an intensive habitat modification project, desired habitat conditions (i.e. increased area of pools and number of pieces of large wood) were achieved within five years of the project completion. However, there was no discernable increase in numbers of juvenile, smolt, or adult steelhead trout (*Oncorhynchus mykiss*) and coho salmon (*O. kisutch*).

Some restoration programmes have resulted in the increase of one species or life-history stage at the apparent expense of another. Anderson (1984) found that creating habitat for coho salmon resulted in decline in available habitat for other species, particularly age 0 steelhead trout. In another case, pools were created by removing boulders from the stream margin and forming a berm across the stream (Everest *et al.* 1985). This resulted in the reduction of winter habitat for all species in the stream.

Several programmes using a variety of types of in-stream structures have been conducted in eastern North America for resident salmonids (rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), and brook char (*Salvelinus fontinalis*)). Results reported tend to be more favourable than those found for anadromous fish (see examples in Reeves *et al.* 1991). A number of projects resulted in increases in numbers and biomass of fish populations. It is not clear why there is this difference in the response of fish populations in the two regions. Two factors that are obviously important are differences in the life-histories of the species and differences in the physical characteristics of the streams.

Ideally, the probable success of structures should be known in advance of their construction. Too often, however, the effects on the fish populations have not been evaluated. Evaluation should be planned at installation, and the cost should be included. A well-planned evaluation of channel restoration was carried out in Southbay Dump Creek in the Queen Charlotte Islands, British Columbia (Tripp 1986). Logging was carried out in an area of steep hillslopes and high rainfall. Debris torrents following logging removed structure from stream channels, limiting overwinter survival and smolt production of coho salmon. Wood was placed in the channel in 1982 to provide structure and fish cover. Restored experimental sections had an overwinter (1982-1983) survival rate of juvenile coho salmon four times greater than the control (unrestored) sections (Fig. 4, Tripp 1986). Survival over the second winter (1983-1984), however, was lower because of a summer flood that filled the pools originally created by the habitat enhancements with gravel.

Demonstration of the benefits of enhancement of stream channels to fish can be elusive. The success of selected structures (gabion V-weirs and placed boulders) in enhancing salmonid populations was evaluated in an Oregon Coast Range stream (House and Boehne 1985, 1986). Densities of coho salmon fry, cutthroat trout parr (*Oncorhynchus clarki*), and sculpins were increased by the gabions, whereas boulder placements most increased densities of steelhead parr (House and Boehne 1985). However, an increase in the number of spawning steelhead caused by a barrier upstream of the enhanced reach complicated evaluation of the worth of structures for trout fry. A carefully-planned evaluation of habitat enhancement in Quartz Creek, Oregon Cascades, has so far failed to show conclusive benefit to salmonid populations (Gregory and Wildman 1993).

Cost/benefit Analyses

Restoration of stream channels using structures can be costly. For the commercially valuable Pacific salmon species, costs and benefits can be calculated (Tripp 1986). Each structure in the South Dump Bay evaluation cost $CAN46-109 to build, totalling $CAN4,500 total for 500 m stream reach. Each adult coho salmon was assumed to have a net wholesale value in 1982 dollars $CAN15.48. The accumulated benefits in 10 years were estimated as $CAN8,500, and in 20 years as $CAN11,700, assuming all sites were successful in enhancing winter survival, yielding cost/benefit ratios of 1.5:1 to 3:1, depending on habitat. Such a financial cost/benefit analysis is clearly not possible for species without commercial value.

Inadvertent costs of habitat restoration are frequently ignored. Placing pieces of large wood in channels creates an additional hazard during floods, as floating timber from habitat enhancement structures torn apart during floods can damage bridges and streamside property (Rothacher and Glazebrook 1968). As most pieces of wood placed in stream channels as part of structures are generally well labelled, tracing the originators is not difficult. The restoration of channel structure in Quartz Creek used both free and anchored wood, and managers required that a large trash rack be constructed downstream of the restored reach to prevent any added wood from floating downstream.

Lessons

There is a growing consensus among researchers and managers that in-stream structures have not been the panacea that they were originally expected to be. Some people contend that the primary reason for this is that in most instances the structure or structures were not appropriate for the particular channel conditions (or types), that they were poorly located within the channel, or a combination of these (e.g. Rosgen and Fittante 1986).

The more commonly held view is that in-stream structures have not been put into the appropriate ecological context. In-stream structures were the sole component of almost all restoration programmes for stream habitat. In most cases, the habitat has been moderately to severely altered, often as a consequence of a land-management activity or a suite of activities (see Hicks *et al.* (1991) for a comprehensive review of the impact of land-management activities on stream habitat in the Pacific Northwest). The structures have frequently been put into place without regard to whether the causes of degradation have been corrected. For example, there are many instances where structures have been placed in streams to accumulate and stabilise gravel for spawning. Often such efforts have failed because they have not been accompanied by parallel efforts to halt the causes of habitat degradation occurring from problems outside that channel. Continued inputs of fine sediments or excess large materials can cause structures to fail to achieve their desired results.

Other factors have also necessitated a watershed-level approach to habitat restoration. One has been the high cost of construction and maintenance of in-stream structures and the relatively short projected life-span (i.e. 15-20 years) of many types of structures. Another is the recognition that only a small fraction of the length of stream requiring restoration can be treated with in-stream structures. Sedell and Beschta (1991) estimated that only 10% of the stream length in the Pacific Northwest could be treated with in-stream structures. Budget constraints and limitations imposed by access and geologic conditions, among other things, make it impractical to use in-stream structures in the remaining areas. Thus, a more comprehensive approach is needed to restore fish habitat.

This more comprehensive approach to habitat restoration has required a broader perspective and understanding of watersheds. All parts of a watershed are not similar ecologically (Naiman *et al.* 1992). For example, streams or stream reaches that are confined by valley walls, and that generally have high gradients, are ecologically different from streams that are less constrained, and that usually have lower gradients. The former are primarily areas of material transport. The latter are sites of material storage and processing, and they are also sites of more extensive hyporheic zones. Biologically, these less constrained reaches have the greatest potential for production and diversity of organisms, including fish. They are also sites that are most sensitive to disturbance, because of their depositional nature. This type of knowledge can be used to determine the location and type of structures that may be deployed within various locations of a watershed.

Even within these large-scale spatial features, there are locations that may be more ecologically important than others. Naiman *et al.* (1992) refer to these as "ecological nodes". These include features such as tributary junctions where material, such as wood and gravel, accumulate. Locating structures at such points may improve their success.

Lateral habitats, which include stream margins and off-channel (floodplain) areas, have received much attention in terms of habitat restoration efforts lately. These are important as habitat for recently emerged fish (Moore and Gregory 1988) and winter habitat for a suite of fish species and age classes (Hartman and Brown 1987). Crispin *et al.* (1993) reported large increases in numbers of juveniles and smolts following development of off-channel areas in a coastal Oregon stream. Habitat improvement of the main-channel that decreases the quantity and quality of edge habitat may reduce fry populations and decrease the carrying capacity of over-wintering habitat for all age-classes (Everest *et al.* 1985).

Figure 5 Examples of limiting factor "bottlenecks" that occur (A) during the winter, just before salmon smolts migrate to the ocean, and (B) during summer, early in the life of young salmon. (C) Attempts to increase fish abundance, e.g., by augmenting the food supply in summer, before a limiting factor such as overwintering acts on the population, usually fail (Mason 1976, from Reeves *et al.* 1991).

The impact of habitat restoration efforts must be considered in the context of the fish community and not just a single species or age-class. Most habitat restoration programmes, and particularly earlier ones, were directed at the habitat of a single species or age-class. As a consequence, some species increased but others declined (e.g. Anderson 1984, Everest *et al.* 1985). Many projects are now attempting to create a variety of habitat conditions that will potentially benefit all fish species and ages.

The success of any habitat restoration or modification effort is dependent to a large extent on the identification of the factor or factors that currently limit production. This requires knowledge about habitat needs of individual species through all life-history stages, which may change seasonally (Bjornn and Reiser 1991). Failure to correctly identify the limiting factor may result in an increase in fish at one stage or season that is lost when a "bottleneck" occurs later. A crude, but useful, example of limiting factors is shown in Figure 5. The choice of type, nature, and position of the structure is usually made having regard for the characteristics of the stream reach to be restored, and the target fish species and life stage. For anadromous fish, enhancement of summer habitat would be of little use if, in fact, winter rearing habitat was the bottleneck to production (Fig. 5A). On the other hand, enhancement of breeding success and winter and spring survival is of little use if low flow summer habitat is the factor limiting output to the ocean (Fig. 5B). In an experiment that tested this theory, smolts were fed through summer, and this resulted in 6-7 times more fish in streams in summer than without feeding. Output to the ocean in the following spring, however, was the same for both fed and unfed fish populations, indicating that overwinter habitat was in fact limiting (Mason 1976, in Reeves *et al.* 1991, Fig. 5C). Bisson (in press) presents a very thorough discussion of approaches to determining limiting factors. Reeves *et al.* (1989) developed a dichotomous key for identifying factors limiting coho salmon production that has been fairly successful. However, such formal procedures are limited to this one attempt at present.

In the rush to initiate habitat restoration programmes, management agencies created what are now seen as unrealistic expectations from in-stream structures. These agencies projected that there would be immediate physical and biological responses to such programmes. In many instances these expectations have not been met. Now the public, funding providers, and politicians are asking "Why?". In hindsight, it is apparent that habitat restoration is more complicated and that biological responses to physical changes take longer than originally thought.

Perhaps the clearest lesson that can be learnt from the experience with the use of in-stream structures on the Pacific coast of North America is that watershed protection is the most successful method of habitat rehabilitation (Reeves *et al.* 1991). It is very costly to repair damage once it has occurred, and some damage may not be reversible. Mimicking the complexity of natural conditions is also difficult (Kauffman 1987). In addition to the physical attributes of in-stream habitat, the processes and functions that create and maintain the habitat are altered by the effects of land-management processes (Naiman *et al.* 1992). The prudent policy therefore, both ecologically and economically, is to prevent damage to streams and the aquatic ecosystem, rather than attempting to repair or mitigate it afterwards.

New Approaches

As a result of such experience, in-stream structures are now viewed differently in western North America. They are considered a part of more comprehensive programmes designed to restore not only the habitat but also ecological functions and processes (e.g., Forest Ecosystem Management Assessment Team 1993, Chapter V). In such schemes, structures are viewed as a catalyst that may facilitate the recovery of habitat in the short term, while other components of the restoration are being carried out. Other components may include establishment of riparian and hillslope vegetation, removal of roads, and stabilisation of erosion-prone areas. The latter component will initiate the longer term recovery of the ecosystem. Sedell and Beschta (1991) present a strong argument in favour of this more holistic approach, particularly with regard to the need to restore riparian vegetation. Such programmes are designed to develop a diverse array of habitat conditions and types throughout a watershed, which benefits the fish community rather than a single species or age class.

RESTORING STREAM HABITAT IN NEW ZEALAND

Stream Environments in New Zealand

We contend that much that is applicable to New Zealand can be learned from the use of structures to restore streams in western North America. Though the fish communities are not identical, some fish species native to the Pacific coast of North America also occur in streams in New Zealand (rainbow trout, quinnat salmon (*Oncorhynchus tshawytscha*), and sockeye salmon (*O. nerka*)). In addition, the rainfall-dominated hydrologic regime and extent of steeplands are similar on the Pacific coast of North America and in New Zealand. Lastly, anthropogenic disturbances to the stream environment in New Zealand are in many respects similar to those in western North America, i.e., removal of primeval native forest. One major difference, however, is that native forest in New Zealand has normally been replaced with pasture grasses, and maintained in that state for pastoral farming, whereas forest tree species are usually replanted in North America, for the purpose of cyclic timber harvest. Native forest once covered 78% of New Zealand, but now covers only 23%; most native forest was removed from New Zealand between 1840 and 1910 (McDowall 1990). Most of the remaining native forest is on steep land; lowland forests have all but disappeared.

The effect of forest removal on New Zealand streams has not been documented in the majority of cases. A few studies in New Zealand (Mosley 1981, Evans *et al.* 1993), plus the similarity of the nature of the physical disturbances in western North America and New Zealand, lead us to speculate that widespread forest removal has resulted in dramatic habitat simplification in the majority of streams. This is usually combined with increases in water temperature fluctuation, light availability, sediment input, peak flows, and reductions in base flows and bank stability (Hicks *et al.* 1991).

Figure 6 (A) Abundance of coarse woody debris (CWD, mean density + 1 SE) and (B) pool frequency (mean number of pools per 100 m of stream + 1 SE) in native and pine forests of different ages in Otago and Southland (from Evans *et al.* 1993).

Figure 7 The effect of woody debris on channel morphology and salmonid populations in two streams with similar salmonid biomass in Washington State, northwestern U.S.A. Beaver Creek, with abundant woody debris, had a much more diverse salmonid community than did Thrash Creek, which had little woody debris (from Hicks *et al.* 1991).

Faunal Changes

The effects of habitat changes on New Zealand's stream fauna is a matter of even more speculation than the nature of the physical changes themselves. Populations of most fish species appear to have declined since the arrival of Europeans, and one species has become extinct (the grayling, *Prototroctes oxyrhynchus*). However, commercial exploitation of native fish and introduction of alien species occurred at the same time as native forest removal (McDowall 1990). Thus it is not possible to conclude that habitat changes have been solely responsible for the apparent declines in New Zealand's stream fish populations. Nevertheless, large galaxiids in streams today are strongly associated with native forest (e.g. Main *et al.* 1985, Taylor and Main 1987, Taylor 1988, Hanchet 1990); their almost complete absence from pasture streams without wood makes us suspect that the effects of forest removal have been severe for some native species.

One of the main contributions of forest to stream channels is large woody debris. Wood provides natural structure in stream channels, forming pools at high flows as water scours around individual pieces or clusters of large woody debris. Wood is also a major factor causing pool formation in New Zealand streams (Evans *et al.* 1993). Native forest >120 years old provides larger and more stable wood than does young forest (either native or exotic) and controls pool formation (Fig. 6). Woody debris forming cover in streams and low water temperatures are key features of the habitat of large galaxiids (Hanchet 1990). Wood removal without other habitat modifications has been demonstrated to reduce the suitability of stream habitat for salmonids (Fig. 7). Riparian zones are the major source of wood for stream channels in western North America, and removal of streamside trees reduces the rate of entry of wood to the stream (Lienkaemper and Swanson 1987). Thus the simplest management option to retain or produce wood for channel structure is to maintain or plant woody vegetation in the riparian zone.

Figure 8 The relationship between stream gradient and (A) net pool depth and (B) surface area of pools in mass-wasted stream reaches in the Queen Charlotte Islands, British Columbia. Mass-wasted reaches have experienced debris torrents as a result of hillside erosion, wheras non mass-wasted reaches have not (from Tripp and Poulin 1986).

Sediment input to streams is generally low in catchments with undisturbed forest vegetation, but is variable from year to year (Swanston 1991). Removal of forest vegetation increases erosion, and entry of sediment to streams. Both coarse and fine sediment that has entered stream channels by erosion will cease to be transported at some point downstream of entry (Swanston 1991). Coarse material, resulting from erosion (mass wasting), and deposited in the stream channel, can overwhelm the transport capacity of a stream, reducing depth and surface area of pools (Figure 8A and B), and in extreme cases, causing the stream to flow below bed level. Increased sediment resulting from vegetation removal can reduce fish cover (e.g., reduction in crevice habitat and large woody debris Murphy and Hall (1981), Andrus *et al.* (1988)) and fill the spaces in salmonid spawning gravels (Hartman *et al.* 1987).

Habitat is important for fish survival, but so is fish food. Therefore, habitat restoration or protection measures should ideally enhance suitability of habitat for prey species as well as for the fish themselves. Many prey species are aquatic macroinvertebrates, and these may be smothered by the addition of fine sediment to streams (Hartman *et al.* 1987). In New Zealand, increased fine sediment can reduce the abundance of sensitive taxa such as stonefly larvae and *Ameletopsis* mayfly larvae, and increase the abundance of sediment-tolerant taxa such as chironomids and *Deleatidium* mayfly larvae (Graynoth 1979). Another study, however, found that *Deleatidium*, along with Chironomidae, *Austrosimulium*, some Trichoptera, and the amphipod *Paracalliope fluviatilis* were all sensitive to increased amounts of fine sediment (Ryder 1989). Sediment deposited in interstitial spaces in the bed was particularly harmful. Oligochaetes, elmid larvae, and the gastropod *Potamopyrgus antipodarum* were either unaffected or increased in number with increased fine sediment. Deposits of fine sediment can coat stone surfaces, reducing food supply and eliminating attachment points for some macroinvertebrates. Suspended fine sediment can clog the food-filtering or trapping apparatus of stream insects (Ryan 1991). Sand-sized sediment can also overwhelm a channel's transport capacity, filling in pools and interstices, as demonstrated in the response of stream channels to logging in the granitic regions of Idaho (Platts and Megahan 1975). In New Zealand, even unpaved forest roads contribute significantly to the sediment load (Fahey and Coker 1992). Another contribution of forests to stream fish is food. Terrestrial invertebrates are important prey for some New Zealand stream fish (Main and Lyon 1988), and removal of overhanging forest vegetation may reduce inputs of terrestrial invertebrates.

Habitat degradation causes reductions in diversity of fish species and age classes. Gross productivity, however, can increase. Canopy removal that often accompanies disturbance to stream channels allows more light to reach the bed, and higher water temperatures than in undisturbed streams, thereby increasing both primary and secondary production (Hicks *et al.* 1991). This may be the reason that stream reaches in pasture immediately below forested sections have higher diversity and biomass of fish than either forested reaches or pasture reaches well away from forest (Hanchet 1990). Fish in stream reaches immediately downstream of forests possibly have the best combination of wood for stream structure from upstream, and elevated light and water temperatures.

Knowledge of Habitat Requirements

Figure 9 Sustained swimming velocities for shortfinned eel elvers (*Anguilla australis*), inanga (*Galaxias maculatus*), banded kokopu (*G. fasciatus*), common smelt (*Retropinna retropinna*), grey mullet (*Mugil cephalus*), and the common bully (*Gobiomorphus cotidianus*) (from Mitchell 1989).

Most stream restoration techniques aim to provide pool habitat and low-velocity refuges for fish through placement of large wood or boulders in the low flow stream channel. In-stream structures can also be used to trap gravel in gravel-poor stream reaches, and this may improve spawning conditions for salmonids. In general, our ability to restore habitat for native fish and invertebrate species is hampered by our lack of knowledge of life histories and habitat requirements of many species. For some species, however, habitat requirements are moderately well known.

Microhabitat preferences have been estimated for benthic invertebrates such as the mayflies *Deleatidium* spp., *Coloburiscus humeralis*, and *Nesameletus* spp., the stonefly *Zelandoperla* spp., the caddisflies *Pycnocentrodes* spp., *Olinga feredayi*, and Hydrobiosidae, the dipteran *Aphrophila neozelandica*, and the gastropod mollusc *Potamopyrgus antipodarum* (Jowett *et al.* 1991). Size, sex, and morphotype of *Deleatidium* nymphs influences their microhabitat distribution, with larger nymphs occurring more commonly in fast water velocities than smaller nymphs (Collier 1994). Depth and velocity preferences of several species of the dipteran family Chironomidae have also been described (Collier 1993).

Habitat requirements of introduced fish, such as the family Salmonidae, are relatively well known if we assume that the criteria developed overseas apply equally well to the same species, e.g., rainbow trout, in New Zealand (Bovee 1978, Raleigh *et al.* 1984, Waite and Barnhart 1992). One problem is knowing which habitat criteria to apply in New Zealand. Published depth and velocity requirements for brown trout, for instance, vary considerably.

Hayes and Jowett (in press) have investigated this problem, and conclude that different sizes of fish and differences in habitat availability among studies are a large part of the reason for the apparent variability in habitat requirements.

Habitat requirements of native fish species remain relatively unknown. Even basic life history information for some species, such as spawning in the giant and shortjawed kokopu (*Galaxias argenteus* and *G. postvectis*), are unknown. A major advance in knowledge of native fish habitat requirements is in preparation, using data collected from a recent survey of native fish at the "100 river" sites (Biggs *et al*. 1990).

Swimming performances have been documented for a few diadromous native species (shortfinned eel elvers, *Anguilla australis*; inanga, *Galaxias maculatus*; banded kokopu, *G. fasciatus*; common smelt, *Retropinna retropinna*; grey mullet, *Mugil cephalus*; and the common bully, *Gobiomorphus cotidianus*; Fig. 9). Flume data combined for shortfinned eel elvers, inanga, banded kokopu, common bully, and common smelt suggest that these species can sustain a swimming velocity of 0.26 m s^{-1} for 30 m (115 s), and a velocity of 0.34 m s^{-1} for only 5 m (15 s) (Mitchell 1989). Grey mullet have even poorer sustained swimming performance. These measurements have implications for design of structures, and for velocities that diadromous fish might encounter during upstream migration.

Figure 11 The association of longfinned eel biomass with stream channel morphology. (A) Amount of pool habitat is inversely related to channel gradient. (B) Increases in stream channel gradient are associated with reduced eel biomass, whereas (C) increased amounts of pool habitat are associated with increased eel biomass (after Chisnall and Hicks 1993).

Figure 10 Habitat-suitability curves for water depth, water velocity, and substrate size for four native fish and two introduced fish in the Rakaia River. FL is fork length. Substrate codes: 1, mud; 2, silt; 3, sand; 4, gravel; 5, small cobbles; 6, large cobbles; 7, boulders; 8, bedrock (from Glova and Duncan 1985).

Depth, velocity, and substrate preferences for native fish in the Rakaia River, developed into probability-of-use curves, have been documented for torrentfish (*Cheimarrichthys fosteri*), bluegilled bullies (*Gobiomorphus hubbsi*), common bully, and longfinned eels (A. australis) (Davis *et al.* 1983, Glova and Duncan 1985, Rowe 1991; Fig. 10). Some authors have suggested that pH may influence fish distribution in a broad way (McDowall and Eldon 1980, Main *et al.* 1985, Taylor and Main 1987, Taylor 1988, Rowe 1991). Koaro (*Galaxias brevipinnis*) appear to avoid acid waters (pH <6.6), whereas banded and shortjawed kokopu appear to occupy waters of low pH (<6.4). These data should be interpreted with caution, as the cause of the distribution may be some factor associated with pH, and not the pH itself.

A number of studies that were not designed specifically to look at habitat requirements do in fact contain useful information about habitat requirements. For instance, gradient determines the natural occurrence of pools (Fig. 11A); thus creation and maintenance of pools in high-gradient reaches, or riffles in low-gradient reaches, will be difficult, if not impossible. The proportion of a stream channel as pools controls the distribution of some fish species, e.g., longfinned eels (Fig. 11B and 11C), so habitat for pool-dwelling species cannot be easily maintained in high gradient stream reaches.

In much the same way that the majority of stream-dwelling salmonids in western North America require access to and from the sea, so a large proportion of New Zealand's native fish species are diadromous, and similarly require access to the ocean, or at least to harbours and estuaries (McDowall 1990). This means that for about 70% of New Zealand's freshwater fish species, habitat conditions, competition, and predation must be considered throughout their migratory routes, as well as in their adult habitats. The majority of diadromous species (torrentfish, bullies, and galaxiids, comprising 10 species) are swept or migrate downstream as juveniles, grow for about 6 months in the sea, return to freshwater as juveniles, and migrate upstream to rearing and adult habitats.

In view of the relatively complex and generally poorly known life histories of New Zealand's native fish, and the small amount of knowledge of habitat requirements, reliable analysis of the relative importance of factors limiting production is impossible. The assumption underlying all habitat restoration procedures is that the quality of in-stream habitat is a factor limiting one or more vital stages of growth or survival of fish (Bisson in press). For instance, improvement of local habitat conditions through the use of structures may be of little benefit to a species if ocean survival, or predation along its migratory route, are limiting factors. Added to this problem is the limited commercial importance of New Zealand's native freshwater fish. McDowall (1990) makes a powerful statement about the need to develop a conservation ethic for native species for their own sake, but realistically the costs of rehabilitating large amounts of native fish habitat would be prohibitive, even if we knew how to do it correctly.

Conclusions

Managers of fish habitat in New Zealand can learn several important lessons from the habitat restoration programmes that have been conducted in western North America during the last 20 years. Firstly, a community and seasonal-level approach is required in the development of programmes involving the use of in-stream structures. Managers should be aware of the potential implications of habitat developed for one species on other fish species and age classes. Also, knowledge of seasonal habitat requirements is needed so that habitat restoration does not result in reduced quantity and quality of habitat for other species at different times of the year.

Secondly, in-stream structures need to be part of a comprehensive watershed restoration programme, not only to restore habitat, but also to restore the ecological processes and functions that create and maintain habitat in the long term. In-stream structures by themselves have had limited success. However, if designed, constructed, and maintained properly, in-stream structures can create habitat in the short term; over the long term, work done within all parts of a watershed, riparian and hillslope, will be necessary to restore habitat in streams. This requires that managers have a good understanding of watersheds, the different components of the watershed, and how these components operate and interact. This more comprehensive and long-term approach is likely to have benefits for all organisms, even those whose habitat requirements are not known exactly.

Thirdly, agencies and organisations should be sure that the public and funding sources are aware that restoration may not produce immediate results. Presenting realistic expectations will increase the probability of support and ensure that these programmes are assets rather than liabilities for those involved.

Figure 12 A comparison between the standard metabolic rate (SMR), expressed as oxygen consumption rate, of fishes that have evolved in different latitudinal (climatic) regions. Note that fishes typical of cold waters have higher SMRs than fishes typical of warmer waters (McFarland *et al.* 1979).

Fourthly, if programmes using in-stream structures are initiated in New Zealand, they should include accompanying evaluation programmes. This does not mean that every programme should be evaluated. However, at least initially, appropriate evaluations should be done for selected programmes to ensure that the desired physical and biological results are achieved. Positive and negative results should be presented. This type of information will be invaluable in the evolution of habitat restoration programmes. The lack of it has almost certainly hindered restoration programmes in western North America.

Finally, and probably most importantly, habitat restoration should not be considered a substitute for habitat protection. Restoration has proven in some instances to be cost-effective. However, preventing the initial degradation is wiser ecologically and more economical than repairing it. There is no guarantee that restoration will work, and in some cases the damage simply cannot be reversed.

FUTURE DIRECTIONS

Several key issues seem to indicate relevant ways to pursue future research into habitat restoration in New Zealand. The most obvious barrier to determining appropriate habitat restoration techniques for stream fish in New Zealand is the lack of knowledge of detailed life histories and seasonal habitat requirements of native fish species. In addition, the habitat requirements of salmonid species, developed overseas, require validation in New Zealand. The apparent requirement of some native fish species for cool temperatures also requires investigation. Are large galaxiids cool stenotherms, and thus restricted to a narrow range of cool temperatures, like salmonids? Studies relating standard metabolism to temperature would answer this question (Fig. 12). Finally, what is the requirement for riparian vegetation, for both temperature control and terrestrial food supply, of stream fish in New Zealand? Can stream shading offset the heating effect caused by flow reduction, and what density of shading is required to provide a given temperature reduction (e.g., Cooke *et al.* 1992)? We have the technology to answer many of these questions. It remains to provide the resources and skilled researchers to apply the technology to the problems of stream habitat restoration.

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