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Forecasting Environmental Responses to Restoration of Rivers Used as Log Floatways: An Interdisciplinary Challenge

Christer Nilsson,[1]* Fabio Lepori,[2] Björn Malmqvist,[2] Erik Törnlund,[1] Niclas Hjerdt,[1] James M. Helfield,[1] Daniel Palm,[3] Johan Östergren,[3] Roland Jansson,[1] Eva Brännäs,[3] and Hans Lundqvist[3]

[1]Landscape Ecology Group, Department of Ecology and Environmental Science, Umeå University, SE-901 87 Umeå, Sweden; [2]Stream Ecology Group, Department of Ecology and Environmental Science, Umeå University, SE-901 87 Umeå, Sweden; [3]Department of Aquaculture, Swedish University of Agricultural Sciences, SE-901 83 Umeå, Sweden

ABSTRACT

Log floating in the 19th to mid 20th centuries has profoundly changed the environmental conditions in many northern river systems of the world. Regulation of flow by dams, straightening and narrowing of channels by various piers and wing dams, and homogenization of bed structure are some of the major impacts. As a result, the conditions for many riverine organisms have been altered. Removing physical constructions and returning boulders to the channels can potentially restore conditions for these organisms. Here we describe the history of log driving, review its impact on physical and biological conditions and processes, and predict the responses to restoration. Reviewing the literature on comparable restoration efforts and building upon this knowledge, using boreal Swedish rivers as an example, we address the last point. We hypothesize that restoration measures will make rivers wider and more sinuous, and provide rougher bottoms, thus improving land-water interactions and increasing the retention capacity of water, sediment, organic matter and nutrients. The geomorphic and hydraulic/hydrologic alterations are supposed to favor production, diversity, migration and reproduction of riparian and aquatic organisms. The response rates are likely to vary according to the types of processes and organisms. Some habitat components, such as beds of very large boulders and bedrock outcrops, and availability of sediment and large woody debris are believed to be extremely difficult to restore. Monitoring and evaluation at several scales are needed to test our predictions.

Key words: channelization; fish; floatways; forecasting; invertebrate; log floating; recovery; restoration; retention; riparian ecosystems; river; Sweden; transport history.

INTRODUCTION

In recent years, increasing insights into the values of healthy running waters have stimulated a large number of restoration attempts (Maddock 1999; Ward and others 2001; Roni and others 2002). Restoration has been defined as the ‘‘return of an ecosystem to a close approximation of its conditions prior to disturbance’’ (US National Research Council, in Bradshaw 2002). However, increasing realization that full recovery is often difficult to

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*Corresponding author; e-mail: christer.nilsson@emg.umu.se

attain has prompted more pragmatic views, such as that of the Society for Ecological Restoration Science (SER 2002), which emphasizes ''the process of assisting the recovery'' of a previously stressed system over the achievement of truly natural conditions. It is in this second sense, often referred to as 'rehabilitation', that the term restoration is used in this work. In any case, the success of restoration rests upon the assumption that if the ecosystem stressors are removed, the biota will recover.

The restoration of rivers faces many challenges. First, because many rivers have been impacted by humans for centuries it is often impossible to know in what ways, and to what degree, a particular site or entire river has been affected. Spontaneous recovery processes often lead to situations that might seem natural but differ considerably from the pristine states because of the legacies of historic land use (Wohl 2001). Historic knowledge is therefore a prerequisite for providing information on the full extent of human impacts. Second, even if natural states were known, rivers might be beyond recovery. For example, the channel or the watershed might have changed too much, or appropriate or affordable restoration techniques might not be available. Third, even if the target state for restoration and the methods of reaching this exist, it is impossible to know in advance whether the environmental recovery will meet the objectives. The presently limited knowledge about environmental responses to restoration depends on the facts that many processes take a long time to recover and that restoration is a fairly recent activity that has not yet exhibited its full potential.

During the last few decades, progress in the field of river ecology has accelerated through the integration of different spatial scales, from microhabitat to basin (Cooper and others 1998), and different disciplines such as history, economics, hydrology, geomorphology and ecology (Benda and others 2002; Nilsson and others 2003). Although there are methods to evaluate the environmental responses to river restoration projects (for example, Palmer and others 1997; Downs and Kondolf 2002; Jungwirth and others 2002), their development is in its infancy. Because most restoration attempts are so recent, forecasting the impacts of planned and ongoing restoration work is difficult. However, because science has an important role to fulfill in the societal decision-making process it is important to face this challenge (Clark and others 2001).

Timber-floating represents an industry that was once well-developed in several European countries, in the United States, in Canada, and in parts of Asia (Williams 1989; Turnock 1991; Hollister-Short 1994; Agnoletti 1995; Törnlund and Östlund 2000). In Germany, timber floating probably occurred during Roman Rule, which extended until the mid fifth century (Scheifele 1999). Log floating faced basically the same challenges in different regions, and many of its effects on the environment were comparable. For example, during the 19th century, log driving was similar between the U.S. and Sweden, using similar floatway structures, techniques and tools for logging and timber floating (Rector 1953; Wood 1971; Smith 1972; Törnlund and Östlund 2000).

Here, we attempt to predict the environmental responses to restoration of boreal Swedish rivers that have been modified for log driving over many decades. Predicting the responses of complex environments that are restored after human disturbances that have lasted over historic timescales requires multidisciplinary efforts. The necessary first step involves studying the history of log floating and associated activities and the effects on river geomorphology. When and how were the environments affected, and what were the purposes and results of the disturbances? These questions are difficult to answer, in part because nobody living today has seen the river reaches in their predevelopment states. However, log driving and the development of floatways have been well documented, and historic archives provide excellent opportunities to identify environmental impacts and the historic transformation of the river landscape (Törnlund and Östlund 2002). Analysis of the log-driving impacts and the pre-driving state will also pinpoint which reaches cannot be completely restored for technical reasons (for example, dynamited areas), and other kinds of land uses (for example, hydroelectric dams) that will interfere with the recovery processes. Understanding the link between history and the abiotic environment is therefore a prerequisite for successful restoration, and for evaluating its environmental effects, including those on aquatic and riparian organisms. In the present paper, we follow this rationale by first analyzing the history of timber floating in Sweden. We then review its effects on riverine habitats and their biota and predict the likely long-term effects on the removal of floatway structures.

HISTORY OF TIMBER FLOATING IN SWEDEN

The development of the export-oriented forest industry, that is, sawmills and later pulpmills, played an important role in the industrialization of Sweden at the end of the 19th century (Schön 2000). Increasing international demand for sawed wood and square timber pushed the logging frontier from the southern coastal areas further inland and northward on the Scandinavian peninsula toward unexploited forest, especially old growth pine forests. Originating in southern Norway at the beginning of the 19th century, the logging frontier reached northern Sweden in the middle of the century; at the end of the same century Finland and Russia were also affected (Björklund 1984; Östlund 1993).

In northern Sweden, all major forestry-related industry such as sawmills and pulpmills were situated along the coast, from which products were shipped to the international market (Östlund and others 1997). Because most logging took place in the inland regions, rivers were crucial for transporting the logs to the coast. From the 1850s, therefore, the expansion of logging was accompanied by an increase in the number of rivers used as floatways. In the beginning of the 20th century dams and floatable channels were developed within the existing floatway network. In the 1920s, the average distance for lumber transport by horse was 4.2 km to the nearest floatway, and 22.2 km to the nearest railroad (Hellstrand 1980). By the early 1900s, Sweden had a dense network of log floatways, affecting more than 30,000 km of the major rivers and tributaries (Figure 1), and in Finland the floatways comprised about 40,000 km (Törnlund and Östlund 2000). In fact, very few river stretches outside the alpine areas remained unused by log drivers. By that time, timber floating was the dominant human impact on rivers. From the mid 1900s on, when numerous, large dams were built on Swedish rivers, the conditions for timber floating changed. However, on many rivers, these two activities took place simultaneously during several decades. After the 1950s, the road network was developed, and transport by trucks became the predominant mode of log transportation leading to the gradual abandonment of floatways (Törnlund 2002; Törnlund and Östlund 2002). Since 1991, timber floating has no longer been carried out in Swedish rivers.

Several factors made Sweden especially well suited for log floating, compared to other countries in Northern Europe. For example, in contrast to Russian rivers, Swedish rivers generally run from north to south, which means that the spring ice breakup starts in the lower reaches of the rivers and proceeds upstream. By the time log floating started each spring most of the flow path was cleared from ice. Timber-floating operations required the control of channel morphology and water flow. For instance, waterfalls and torrential rapids in large rivers made timber floating extremely difficult. Also, large boulders in the middle of the channel could cause logjams that were difficult to eliminate (Figure 2a). Therefore, boulders, rocky outcrops and large woody debris (LWD) were removed and the flow was regulated, leading to a number of important changes to the physical river environment (Table 1). Originally, timber floating was carried out only during high flows, especially in small rivers with floods of short duration (Figure 2b). Various floatway structures were later developed to compensate for the shortage of water after flooding (Figure 3). Dams (Figure 3a) and

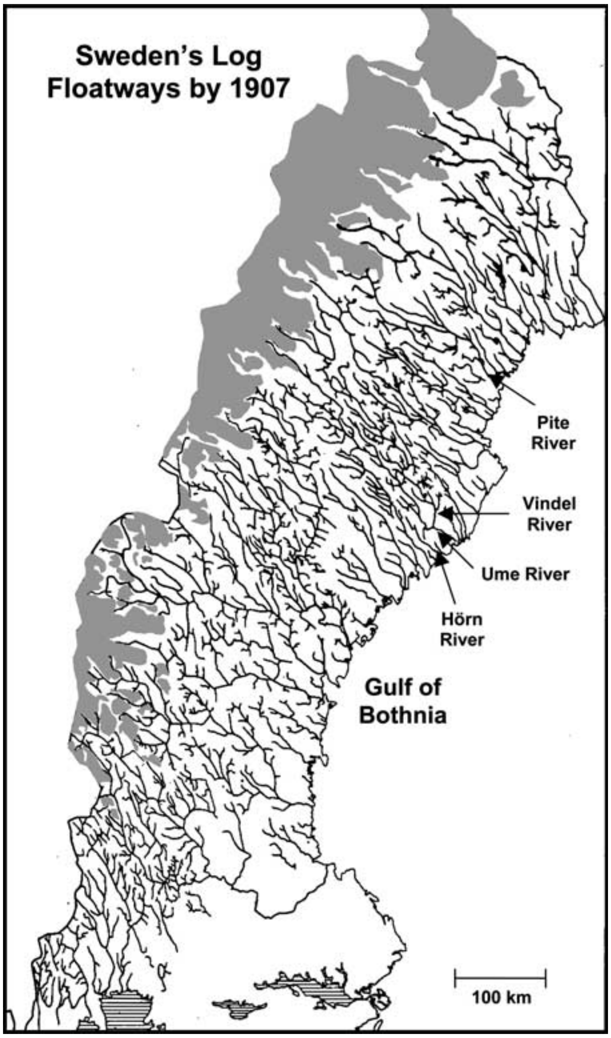

Figure 1. Map showing the network of log floatways used in northern and central Sweden (in northern Europe) by 1907 (river stretches not used for floating are not depicted). The *hatched areas* denote alpine regions without conifer log production (adopted from Andersson 1907). *Arrows* indicate the location of rivers mentioned in the text.

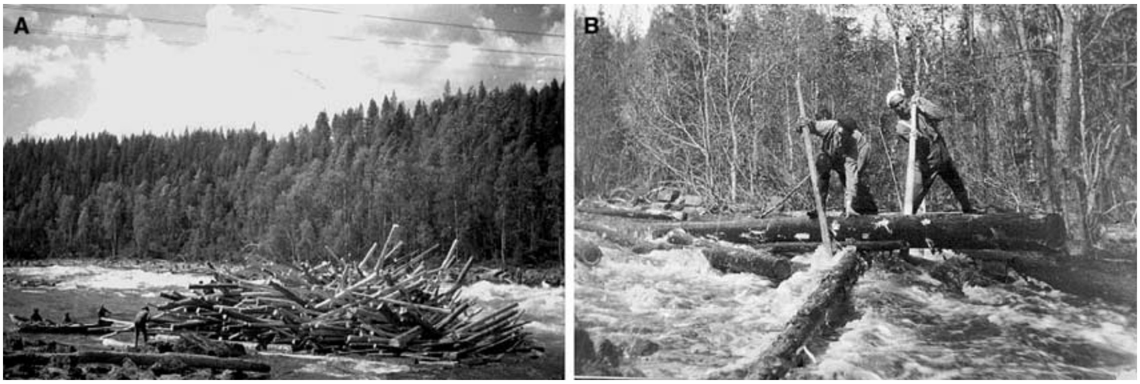

Figure 2. There were different opportunities for log driving in large and small rivers. **a** in large rivers, the log-driving season was long because of long-lasting floods, and large quantities of timber could be transported. The photograph shows log drivers in the Ume River in July 1957, working to remove a logjam. **b** log driving in small rivers relied on a short spring flood that was sometimes extended by means of reservoirs. This photograph shows log driving in a small tributary of the Ume River, northern Sweden, in spring 1922. Photographs by courtesy of Västerbotten's Museum, Umeå.

various kinds of stone piers (Figures 3b, d, e) were built to regulate the flow to extend the floating season, to cut off eddies and side channels, thus preventing logs from stranding or jamming, and to force flow to the central parts of the river channel. Flumes (Figure 3c) or canals were used where the river channel was too steep, too rocky, or too sinuous to allow log driving. As a result of the different constructions, more timber could be floated in a shorter time.

Originally, floatway structures were made by hand using basic tools, whereas black powder [a mixture of potassium nitrate (saltpeter), sulfur, and charcoal] and stone burning were common means of removing bedrock and large boulders from stream channels. Dynamite replaced black powder in the late 1900s. Stone burning refers to heating stones with fire and then cooling them down with water to make them crack. Simple hoisting cranes were also used in both stream clearing and construction works; in the beginning of the 20th century motorized cranes were introduced (Figure 4). New technology after World War II included the use of bulldozers, which was considered fast and convenient for clearing stream channels, and for building dikes to regulate water flow (Törnlund 1999, 2002; Törnlund and Östlund 2002). Construction techniques and the types of structures built also changed over time. During the second half of the 19th century, floatway structures were mainly made of wood and naturally shaped stones, but in the beginning of the 20th century stone piers made of cut stone became more common (Figure 3d).

Most floatway structures were built in rapids, where steep and shallow sections presented the most serious obstacles to log drivers. For instance, in the Pite River in northern Sweden, about 350 km (87%) of the 402 km long main channel was used as a log floatway (see Figure 1). Floatway constructions were built along the entire stretch. A couple of lakes excluded, no stretches longer than about 10 km were without constructions (Sundborg and others 1980). The constructions spanned a total length of 58 km, including piers comprising at least 417,600 m^3 of stone. In addition, about 970 km long reaches of the tributaries of the Pite River were used for log driving (J. Isaksson and others, unpublished data, Figure 1). Some rivers have several generations of constructions, exhibiting a successive development of the floatway. For example, Figure 5 shows how a river stretch was first constricted by means of simple box booms, later blasted and straightened, and in a final step even further compressed using additional stone piers. In such cases, it is difficult to imagine what the pristine state could have looked like. Logjams (such as the one in Figure 2a) were often removed by explosives, which caused considerable damage to channel morphology and biota.

EFFECTS OF LOG FLOATING ON RIVER ECOLOGY

The channelization of rivers for log floating differs from most other types of channel dredging and realignment projects in that it generally only affects high gradient sections of the rivers (such as rapids or riffles) where logs tended to jam up; tranquil and sluggish reaches were not affected to the same extent. Thus, the more serious effects of floatway constructions on river systems are 'patchy', al-

Table 1. An Overview of Timber Floating Operations and the Physical Effects on Rivers and Riparian Forests

Operation	Purpose	Floatway construction activity	Physical effect
Channelization	To straighten and reduce the retention capacity of the stream channel	1. Installation of wing dams and stone piers so as to line banks and cut off backwaters and side channels (Figure 3b) 2. Installation of wing dams and stone piers as deflectors to funnel streamflow and raise water levels in rocky rapids (Figures 3d and e) 3. Dredging and removal of substrates by bulldozers, cranes and explosives (Figure 4) 4. Construction of flumes to bypass steep and/or complex reaches (Figure 3c)	1,4. Reduced sinuosity and flow path complexity 1,2. Increased flow velocity in main channel 2,3. Reduced ranges of water depth 2,3. Reduced channel roughness 1,4. Dewatering of backwaters, side channels and bypassed reaches 1. Reduced bank length and riparian ecotone 1. Less frequent, more intense flooding 1,4. Reduced hyporheic exchange
Damming and impoundment	Reservoir storage to control water flow	5. Installation of splash dams (Figure 3a)	5. Creation of reservoirs and ponds 5. Inundation of riparian areas above dams 1–5. Alteration of the timing, magnitude and duration of peak flows

Numbers in the physical effect column indicate the floatway construction activity responsible for each effect

though they might have cumulative effects on the entire river. Rapid flow reaches often support diverse patches of riparian vegetation (Nilsson and others 2002), and dense communities of macroinvertebrates (Zhang and others 1998). Channelization of such areas has reduced land-water interactions, increased current velocities and the erosion of gravel and fine sediments, and simplified the river ecosystem.

Natural rivers tend to equilibrate and reach a state of 'grade' where material transport rates are essentially equal to supply rates at different points along the channel (Mackin 1948). As rapids and fast-flowing river reaches became channelized and dredged, river forms that had evolved over thousands of years changed abruptly. Geomorphic effects of river channelization include a reduced channel roughness, shortened flow path distances and steepened slopes (Talbot and Lapointe 2002). Such changes cause the channel to adjust to a new equilibrium, or sometimes a state of disequilibrium (Brookes 1988; Knighton 1998; Surian and Rinaldi 2003). Straightened reaches usually experience an increase in shear stress and the capacity to transport bed material, leading to erosion and degradation of the channel bed, destruction of pool-riffle variability, and increased sedimentation and flooding downstream (Yearke 1971; Brookes 1985; Wyzga 2001; but see Petts 1984; Dietrich and others 1989; Talbot and Lapointe 2002). Although much of the channel work was conducted in rapids, most pristine rapids in the Swedish rivers were relatively gently-sloping with plenty of fine material available in sections with less turbulent flow, such as in backwaters and downstream of boulders.

Channelization reduced the natural diversity of velocity and substrate patterns, which altered habitat conditions for aquatic organisms. Although a few studies have addressed the general ecological effects of such alterations during timber floating (for example, Karau 1975), most studies have focused on fish. For example, the channelization of rivers during the log-driving period destroyed spawning and rearing habitats for fish (Newbury and Gaboury 1988; Jutila and others 1998; Scruton and others 1998). In some cases, fish movement was also affected, for example when log accumulations covered entire river reaches. Byström (1868) questioned the timber transport in rivers after observing decreasing abundances of Atlantic salmon (*Salmo salar* L.) in the Ume River in northern Sweden during the middle 19th century. As seen from a recon-

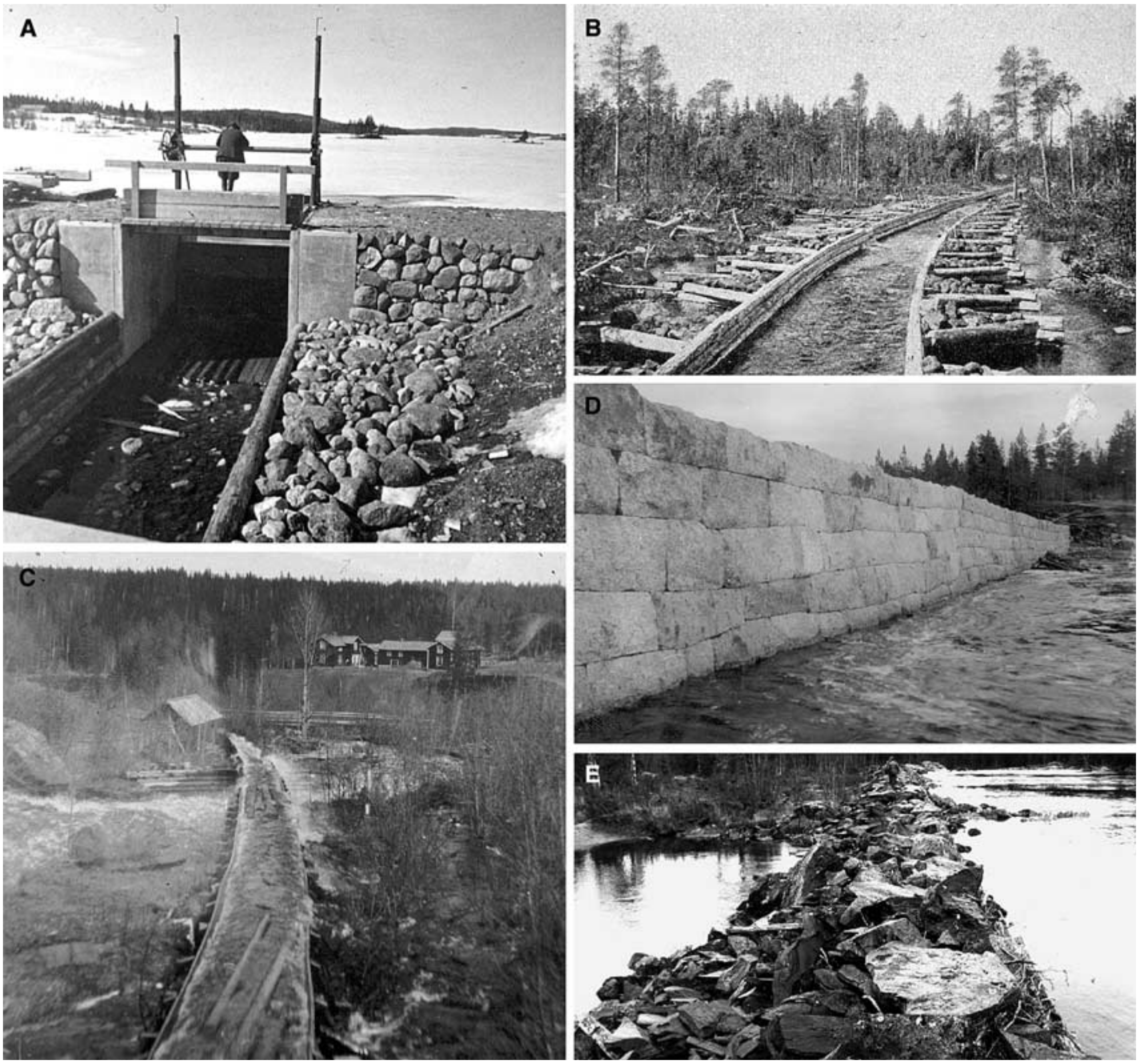

Figure 3. Examples of log-driving constructions from northern Sweden. **a** a splash dam, the Hörn River in 1950, **b** cribbed wing dams completely channelizing a small stream, **c** a flume, the Kul River, a tributary of the Vindel River, **d** a well sculptured stone pier, the Ume River in 1926, **e** stone pier of blast stones, the upper Vindel River (note people in the upper middle for scale). Photographs **a**, **c–e** by courtesy of Västerbotten's Museum, Umeå, photograph **b** from Ekman (1922), with permission.

struction of the total catches of this species in the Gulf of Bothnia during the last two centuries (McKinnell and Karlström 1999), catches varied considerably during the 19th century and then decreased drastically during the 20th century until stocking of fish was introduced (Figure 6a). A part of this reduction may have been caused by log driving, which processed large timber volumes and intensified floatway developments in the channels during this period (Figures 6b and c). Other possible causes include hydropower development, which began in many large Swedish rivers during the late 1800s, and the fact that many fishing rights were bought out during the same time, leading to a reduced fishing effort.

During the log-driving era, many riparian forests were cut, but even intact riparian vegetation functions might have been hampered by the channelization work. Floatway structures reduce the total length of riparian corridor per unit stream area, thereby limiting land-water interactions. Hence, channelization eliminated natural channel migration and the creation of new sediment surfaces suitable for plant recruitment (Goodwin and others 1997). Given the low sediment content and that channelized reaches are often situated in fast-flowing reaches in which current velocity increased further following channelization, the abundance of aquatic plants is generally low (C. Nilsson, personal observations).

Figure 4. Motorized crane clearing the stream channel and building a stone pier in the Ume River (1930s). Photograph courtesy of Västerbotten's Museum, Umeå.

Channelization also affects macroinvertebrate communities by reducing the retentive capacity of allochthonous detritus. In Finland, the channelization of streams for timber floating involved the removal of retentive structures (for example, debris dams, back-waters, boulders), which reduced retention efficiency of detritus to one order of magnitude below natural references (Muotka and Laasonen 2002). As a result, channelized streams had among the lowest standing stocks of detritus reported in the literature; moreover, leaf litter was rapidly processed and occurred in the channel only for a short period following leaf abscission in autumn (Haapala and Muotka 1998). The reduction in detritus availability following channelization likely had substantial ecological effects. Not only is allochthonous detritus the main source of energy to aquatic communities in forested streams (Vannote and others 1980; Naiman and others 1987), but also its availability limits the secondary production of macroinvertebrates, particularly that of leaf-shredding species (Richardson 1991, Dobson and Hildrew 1992). The limiting effects of detritus might be particularly acute in boreal systems, where inputs are naturally low as deciduous trees are sparser and smaller than in southern temperate regions, and the dominant species (gray alder, *Alnus incana* and birch, *Betula* spp.) have leaves that are processed quickly in water (Malmqvist and Oberle 1995; Haapala and others 2001). Even in pristine streams, such conditions could limit food to shredding macroinvertebrates before the emergence in late spring, forcing populations through resource bottlenecks (Haapala and Muotka 1998). In agreement with the hypothesis of severe resource limitation, channelized (low-order) streams in Finland held substantially fewer detritivorous macroinvertebrates per reach than natural reference sites (Muotka and Laasonen 2002; Muotka and others 2002).

RESPONSES TO RESTORATION

By removing artificial floatway structures and reconditioning channel geomorphology and hydraulics, ecological processes are likely to change towards more pristine conditions. For example, most of the effects listed in Table 1 can probably be reduced or eliminated. Recent restoration projects in boreal rivers include removal of splash dams, wing dams and piers, return of boulders to the river channel, and creation of nursery habitats as the main measures (C. Nilsson and others, personal observations). Hereafter we review the expected environmental responses to such restoration.

Reshaping riverbeds to states that resemble their pre-channelization state will likely induce a wide array of physical, chemical and biological responses. These responses are predicted to occur on different temporal scales, from short-term responses that immediately follow the restoration work to long-term, more gradual responses. In the short term, the removal of log-floating constructions is hypothesized to drastically alter the distribution of energy within the river as channel dimensions change abruptly. We predict that the river will immediately begin to adjust morphologically to the new conditions, redistributing sediments according to the new hydraulic pattern and

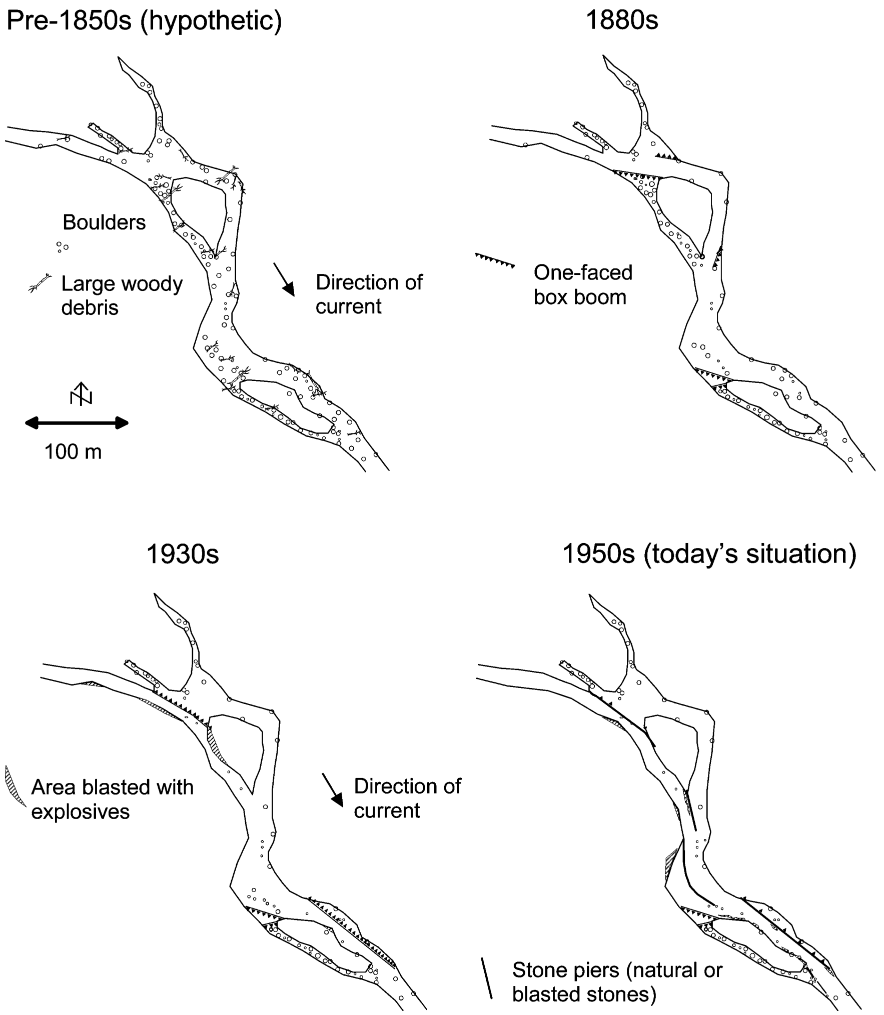

Figure 5. Maps showing the successive development of a floatway from the mid 1800s to the mid 1900s. The example is from the Bergvatten Rapids in the Bjur River, a tributary of the Vindel River in northern Sweden. Note how the originally diverse stream channel was successively straightened and narrowed.

flow regime. Over time, channel morphology is supposed to reach a new equilibrium where the river is neither degrading nor aggrading, but the time required to reach this new equilibrium might vary considerably from place to place.

Petts (1980) proposed that the total ecological response of a river to channel changes could be conceptualized as a hierarchy of responses. Primary impacts affect the transfer of energy into and within the river. Secondary impacts are the channel changes and changes of the floodplain dynamics and primary production arising from local effects of the first-order impacts. Subsequent effects on benthic invertebrates, fish, and floodplain fauna result from a combination of all the previous impacts as well as from biotic interactions between populations. The complete adjustment of biological populations must therefore be preceded by the adjustment of abiotic factors, which can take a considerable time, even hundreds of years (Brookes 1994).

Geomorphic and Hydraulic Responses

Any attempt to recreate a self-sustaining natural system requires an understanding of the geomorphic processes that characterize a certain river and how these processes can be supported by present (or future) flow regimes (Clarke and others 2003). This section summarizes both local and cumulative responses and their linkages.

Sediment transport will probably respond quickly to restoration work. Fine to medium-sized sediments can be expected to become deposited in the previously confined and eroded channel (Brookes 1994); this redistribution will likely evolve over time and be most efficient during high flows. Both individual sediment grains and larger-scale channel irregularities such as channel curvature, promote hydraulic complexity (Dietrich 1987). For instance, a comparison of channelized and natural stream reaches has shown that straight channels exhibit homogeneous, one-dimensional (longitudinal)

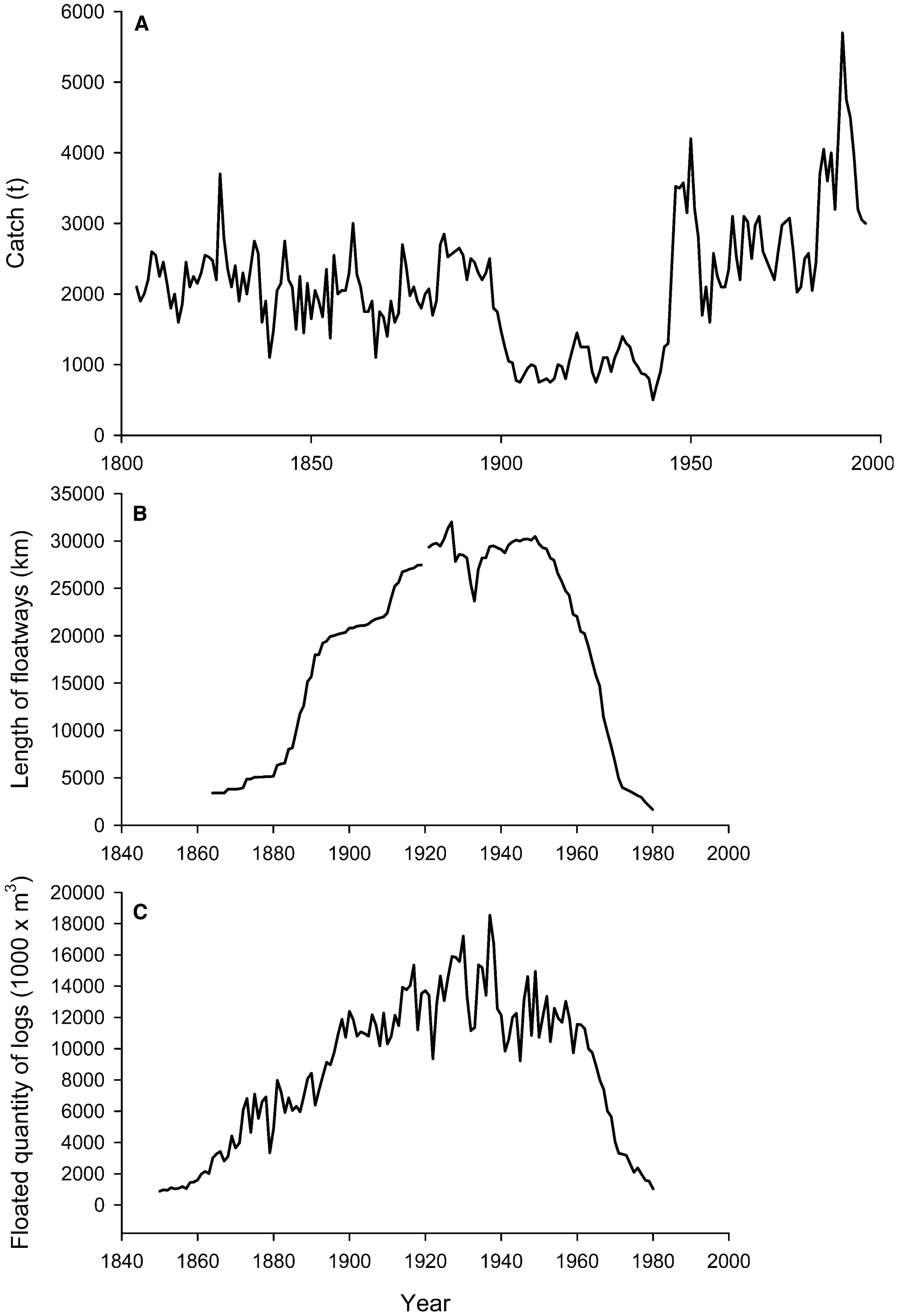

Figure 6. A possible relationship between timber floating and salmon catches in Sweden. **a** A reconstruction of the variation in all-nation catches (in metric tons) of Baltic salmon over the last 200 years. The figures are based on statistics from the largest Swedish fisheries (modified from McKinnell and Karlström 1999). Relationship between **b** the length of log floatways in operation, and **c** the amount of floated timber in Sweden 1850—1980 (from Törnlund 2002).

flow characteristics whereas naturally curved channels generate three-dimensional flow with large cross-channel and vertical components (Rhoads and others 2003). Complex hydraulic characteristics support a broader range of habitats (Kemp and others 1999). A complex flow structure may be important for providing good habitat for certain communities of insects and fish (Schlosser 1985; Rhoads and others 2003).

The hyporheic zone is a subsurface flow path along which water 'recently' from the channel will mix with subsurface water 'soon' to return to the channel (Harvey and Bencala 1993). Studies have shown that hyporheic exchange is enhanced by an increase in flow complexity, for example, by placing obstacles in the flow (Hutchinson and Webster 1998) and by introducing pool-riffle sequences (Wörman and others 2002; Kasahara and Wondzell 2003). Similarly, increased sinuosity and channel avulsion may alter spatial patterns or rates of hyporheic downwelling and upwelling (Malcolm and others 2003). From these findings we predict

Table 2. Hypothesized Environmental Responses to River Restoration

Restoration method	Geomorphic response	Hydraulic/hydrologic response	Ecological response
1. Return of boulders to river channel (S,R)	<10 years	<10 years	Riparian
2. Removal of stone piers and wing dams (S,R)	1–3. Increased channel area (R)	1–3. Increased water depth and reduced water velocity (R,W)	<10 years
3. Reconnection of cut-off side channels (R)	3. Increased sinuosity of river channel (R)	1–3. Reduced water depth in cases of extreme increases in channel area (R)	1–3. Increased habitat complexity (S,R)
	2. Increased riparian connectivity (longer land-water ecotone) (S,R)	1–3. Increased total flow resistance and water volume (increased retention time of water) (R,W)	2,3. Increased dispersal of plant propagules from riparian areas (S,R,W)
	2. Increased amount of bare soil in reconnected riparian areas (S)	2,3. Increased variation in hyporheic flow pathways (S,R)	>10 years
		1–3. Reduced temporal variability of water levels in river channel (R,W)	2,3. Adjustment of plant zonation and species composition (S,R)
	>10 years	1–3. Increased frequency and duration of flooding of reconnected riparian zones (S,R)	1–3. Increased primary production (S,R)
	1–3. Increased roughness of river channel (R)		1–3. Increased concentrations of foliar nutrients (S,R)
	1,2. Increased sediment and nutrient availability in riparian and channel areas (S,R,W)		1–3. Increased export of litter and LWD to river channel (S,R,W)
			Aquatic
			<10 years
			1–3. Increased habitat complexity (S,R)
			1–3. Improved migration and spawning success of fish (especially salmonids) (R,W)
			1–3. Increased retention of detritus (R,W)
			1–3. Increased abundance of filamentous algae (S,R)
			1–3. Increased life-form diversity of macroinvertebrates (R,W)
			1–3. Improved overwintering conditions for fish (R,W)
			>10 years
			2,3. Adjustment of species composition (S,R)
			1–3. Increased life-form diversity of aquatic plants (R,W)

1–3. Increased species diversity (R,W)
1–3. Increased production and biomass of aquatic plants, macroinvertebrates and fish (R,W)
1–3. Improved resilience (because of more refugia) following large floods (R,W)

Although the three restoration methods imply different means of action, the expected responses are similar. Connections between restoration methods and response types are indicated by numbers 1–3. The spatial scale is defined by site (S), reach (R) and watershed (W). Two scales indicating predicted time needs for recovery to states close to pre-channelization states are given. In addition, some ecological response variables may suffer from temporary disturbances immediately following restoration. LWD = Large Woody Debris.

that hyporheic exchange will increase following restoration, leading to longer residence times of minerals, nutrients and particulate organic matter in restored reaches. Because hyporheic flow paths are important for providing oxygen and nutrients to bed sediments (Hancock 2002; Malcolm and others 2003), such effects would be highly beneficial to river organisms.

The cumulative effect of adding boulders to the channel bed is that channel roughness will increase. We can therefore expect an increase in reach flow resistance, that is, the loss of energy caused by interactions between the flow and the channel boundary. Resistance is influenced by roughness elements at all scales (Leopold and others 1964) and varies with flow (Curran and Wohl 2003). Coarser scale roughness elements, such as undulating bed forms, step-pool and pool-riffle sequences, dominate total flow resistance during bankfull flow conditions when individual bed particles are submerged (Millar 1999; MacFarlane and Wohl 2003). It has been argued that large-scale heterogeneity, such as the irregularity in channel width, sinuosity, bifurcations, and so on, constitutes a higher level of flow resistance in river networks (Li and others 1992; Bathurst 1993; Yen 2002), and thus provides means for dissipating energy. We therefore predict that restoration measures such as the reopening of side channels will mainly increase flow resistance during high flow conditions.

The predicted increase in flow resistance following restoration implies that mean flow velocity will decrease in restored reaches (cf. Laasonen and others 1998). This implies that the volume of water held in a channel reach at any given time will increase following restoration, leading to channel widening or increased frequency of overbank flow. Water depth could either increase or decrease depending on the relative changes in channel width and volume, but Tikkanen and others (1994) reported higher water levels after restoration.

As a net effect of the restoration of a large number of channelized reaches, the flood hydrographs will be dampened and lengthened overall because side channels, riparian areas and channel banks will act as buffer zones to retain flood waves and later release water back into the channel during recession periods. Natural channels allow flood waves to spread out laterally and dissipate their energy across the floodplain instead of confining water to the channel where it can build up momentum in the longitudinal direction. In this process, floodplain vegetation generates additional flow resistance and acts as an energy loss mechanism to slow the flow (Helmiö 2002). Theoretical analyses by Knight and Shiono (1996) showed that the speed of a flood wave peaks at 0.67–1 × bankfull depth, and is drastically reduced when the floodplain is inundated. We can expect the floodplains in restored reaches to be flooded more frequently, and the cumulative effect of this would be a reduction in peak flow.

Three factors make it difficult to restore the original channel morphology of rivers subject to log-floating activities. First, many bedrock outcrops and large boulders formerly situated in the channel have been removed with the use of explosives, and are difficult to replace. Second, increased current velocities and transport capacity after channelization have flushed out large volumes of fine sediments from the reaches. Last, intensive logging of the riparian forest inhibits the recruitment of LWD to the channel.

In summary, we suggest that channel area, sinuosity and roughness, water depth and velocity, total flow resistance, hyporheic flow pathways, and water-level fluctuations be included in subsequent validation of the geomorphic and hydrologic/hydraulic changes of the river channel following restoration (Table 2).

Responses of Riparian Organisms

Studies of responses to restoration of rivers used for log driving have overlooked the riparian organisms, but general predictions can be made based on experiences from other types of riparian restoration (for example, Goodwin and others 1997; Middleton 1999; Stromberg 2001; Rood and others 2003). First, restoration is likely to provide greater habitat for establishment of riparian plants. Along channelized reaches, the relocation of stone piers and boulders from the riparian zone to the channel exposed fine-grained soils for plant colonization (J. M. Helfield, R. Jansson, and C. Nilsson, personal observations). Over longer periods of time, changes in flood regimes are expected to enhance riparian productivity and diversity. In most ecosystems, species richness tends to be greatest at intermediate levels of disturbance frequency (for example, Connell 1978; Huston 1994). Studies of riparian ecosystems have demonstrated that diversity is controlled to a large extent by flood frequency, with the most species rich communities occurring at sites with intermediate levels of flood frequency and high levels of spatial variation of flood frequency (Pollock and others 1998). Frequent, low-intensity floods limit competitive exclusion by dominant species and create open patches for colonization by opportunistic species (Nilsson and Grelsson 1990; Auble and Scott 1998). Local-scale flooding and sediment deposition also contribute nutrients to riparian soils and facilitate the dispersion of riparian propagules (Nilsson and others 1991; Andersson and others 2000). In contrast, infrequent floods of high intensity or duration may denude large areas of riparian vegetation by dislodging or burying plants (Bendix 1999), or by creating anaerobic soil conditions (Blom and Voesenek 1996; Friedman and Auble 1999). Because floatway structures shield riparian plant communities from all but the most infrequent, catastrophic floods, their removal is expected to result in more frequent but less intense disturbance and correspondingly enhanced species richness. Because riparian areas are among the most species-rich habitats in the boreal and temperate zones (Naiman and others 1993; Nilsson and Jansson 1995), restoration may therefore have important implications for regional biodiversity.

At the interface between terrestrial and aquatic ecosystems, riparian zones exert an important influence on the physical and ecological characteristics of rivers (Gregory and others 1991; Naiman and Décamps 1997; Naiman and others 2000). Because floatway structures form barriers between rivers and riparian forests, their removal is expected to increase the effective length of riparian ecotone and enhance riparian functions affecting instream habitat. For example, shading and allochthonous organic matter inputs will likely be enhanced in areas where riparian trees have colonized banks formerly occupied by boulders or retaining walls. Similarly, recruitment of LWD may be enhanced following restoration due to increased frequencies of fluvial disturbances such as channel avulsion and anastomosis. Over longer periods of time, riparian influences on stream ecosystems are expected to increase with increases in riparian productivity and dispersion of riparian propagules. In some cases, floatway structures themselves may provide some beneficial functions. For example, retaining walls may provide some bank stabilization and control of erosion and siltation. However, any loss of this function following restoration may be offset by increases in riparian productivity and root density.

The removal of floatway structures is also expected to restore connectivity between the river and its hyporheic zone. Patterns of hyporheic exchange are influenced by changes in channel constraint (Stanford and Ward 1993; Fernald and others 2001), sinuosity (Vervier and others 1993; Wondzell and Swanson 1996) and streambed topography (Harvey and Bencala 1993; Hill and others 1998; Kasahara and Wondzell 2003). To the extent that riparian plants and instream algal communities acquire nutrients from hyporheic water (see Harner and Stanford 2003), these changes may affect primary production within and adjacent to restored streams. Increased nutrient availability in riparian soils or hyporheic flows contributes to faster growth of riparian trees and results in increased concentrations of foliar nutrients in riparian plants (Harner and Stanford 2003). Plants with higher concentrations of foliar nitrogen are generally more nutritious and palatable to terrestrial herbivores such as moose (*Alces alces* L.) and hare (*Lepus* spp.), and are browsed preferentially as a result (Bryant 1987; Pastor and others 1988). By increasing nutrient availability and changing the spatial distributions of riparian plants, restoration might therefore affect patterns of browsing, which in turn affects nutrient cycling, successional processes and plant species composition (McInnes and others 1992; Kielland and Bryant 1998; Suominen and others 1999).

In summary, we suggest that riparian connectivity (a longer riparian ecotone), nutrient availability, amount of bare soil, primary production, plant zonation patterns, and export of litter and plant propagules be included in future assessment of riparian changes following restoration (Table 2).

Responses of Aquatic Organisms

Plants. The distribution and abundance of aquatic plants are largely controlled by light, water chemistry, flow parameters, and substrate composition (Chambers 1987; Chambers and Kalff 1987; Madsen and others 2001), of which at least the latter two are expected to change following restoration. Species richness and abundance of aquatic plants are highest along sheltered and tranquil reaches with silty or muddy soils, whereas turbulent rapids with boulder bottoms are more or less devoid of aquatic vascular plants but may have bryophytes and algae (Erixon 1981; Nilsson 1987; Chambers and others 1991). Tranquil reaches with fine-grained bottoms may have well-developed helophytes (that is, emergent aquatics) in shallow water with elodeids (that is, long-shoot plants) in deeper waters, but their abundance decreases as flow velocities increase. Nymphaeid vegetation (submerged plants with floating leaves) is generally scarce in boreal streams and rivers, but may occur, for example, in sheltered bays or lagoons. Large water-level fluctuations and ice disturbance favor isoetids (that is, rosette forming aquatics) on fine-grained sediments (Renman 1989; Nilsson 1999).

The physical disturbance exerted during restoration is likely to temporarily reduce the abundance and species richness of aquatic plants. Given that most aquatic plant species reproduce vegetatively, they are likely to recolonize within a few years. Restoration is expected to permanently inundate some areas that have been periodically or permanently laid dry by log driving structures, allowing aquatic species to recolonize, given that a source pool of propagules is available, and that flow disturbance is not too strong. In some situations the construction of deflectors has created permanently inundated areas protected from flood disturbance where aquatic plants have established. In such areas current velocity and water-level fluctuations will likely increase following restoration, which may lead to local losses of helophytes, elodeids and nymphaeids, whereas isoetids may be favored. In channelized reaches, the predicted lower current velocities and larger wetted channel area following restoration may allow helophytes and some elodeids to increase in abundance provided new sediments will also be deposited.

Macroinvertebrates. Most research on the effects of restoration on macroinvertebrates has emphasized the impact on the retentive capacity of allochthonous detritus and the consequences for macroinvertebrate production (for example, Haapala and Muotka 1998; Laasonen and others 1998; Muotka and others 2002). The restoration in Finland has shown that replacement of rocks in the streams was effective in enhancing channel retentiveness (Muotka and others 2002), although the recovery was partial in the short term due to the dislodgement of aquatic mosses—a key retentive feature—caused by the restoration work (Muotka and Laasonen 2002). Through the reinforcement of the aquatic-terrestrial linkage, restoration was expected to increase the production of detritivorous macroinvertebrates, with effects potentially reverberating bottom-up to higher trophic levels, including invertebrate predators and fish (Wallace and others 1997). Supporting these predictions, the density of detritivores increased considerably in Finnish streams during the years following restoration, paralleling an increase in detritus standing stock (Muotka and others 2002). In the restored streams, the longer riparian ecotone should also increase litter input per unit of channel area in the short term. In the long term, more litter might also be present due to the expected increase in production by the riparian vegetation.

In general, increases in litter input and retentiveness are expected to affect the production of benthic assemblages in tributaries, but not necessarily in the main rivers, where allochthonous detritus is secondary relative to other energetic sources, particularly primary production (Vannote and others 1980). A specific restoration action that will probably enhance invertebrate densities in both tributaries and large rivers would be the removal of stone piers along the margins. Through the expansion of the wetted channel area, macroinvertebrate habitat and net macroinvertebrate production might increase per unit of channel length.

Although an increase in basal resources (detritus) is expected to enhance benthic densities, other outcomes of the restoration might have an opposite effect on biomass. Most fish common to boreal streams (for example, Atlantic salmon, brown trout—*Salmo trutta* L., European grayling—*Thymallus thymallus* L., and bullhead—*Cottus gobio* L.) feed largely on aquatic invertebrates and populations of these fish species are expected to increase following restoration. Although empirical studies on the impact of fish on macroinvertebrates have yielded equivocal results (review by Arnekleiv and Raddum 2001), limiting effects by predation on benthic densities have repeatedly been experimentally demonstrated (Dahl 1998). Restoration could therefore increase the top-down control of macroinvertebrates by fish predation, counteracting to some extent the expected increase in density. There is, however, little evidence that increasing salmonid abundance in rivers (for example, by stocking) could reduce macroinvertebrate densities (Arnekleiv and Raddum 2001). Nevertheless, parallel increases in benthic-feeding fish (for example, bullhead) might have greater impact, due to the postulated higher dependency of these fish on aquatic invertebrates (Dahl and Greenberg 1996).

Compared to functional changes, the potential effects of restoration on the diversity and composition of benthic assemblages are more difficult to predict. Changes in taxonomic richness appear uncertain. Consistent with other work on the impact of forestry practices on benthic assemblages (for example, Carlson and others 1990; Stone and Wallace 1998), Liljaniemi and others (2002) found no differences in richness between Finnish streams channelized for timber floating and pristine Russian streams in the same watershed. Still, habitat characteristics such as fine detritus and LWD were significantly lower in the impacted streams (Liljaniemi and others 2002). By contrast, Laasonen and others (1998) and Muotka and others (2002) suggested that the structural simplification of the habitat produced a loss of diversity in channelized Finnish streams. Indeed, these authors found that species richness increased following restoration in the systems they examined (Muotka and others 2002).

Changes in assemblage composition are likely following restoration. First, Müller (1962) showed that channelization of boreal rivers for timber floating favored macroinvertebrates with a preference for fast currents (for example, blackfly larvae), and at the same time disfavored others with affinity for slow currents. Restoration is expected to cause a faunal change in the opposite direction. Second, the enhancement of litter inputs and retentiveness (see above) are expected to favor macroinvertebrates involved in the detrital food chain—especially shredders and collectors—to a larger extent than other functional groups (Muotka and others 2002). Third, the return of boulders to the stream channel might favor those aquatic insects requiring rocks protruding from the water during particular life stages, such as *Baetis* mayflies, whose females use such rocks for oviposition (Peckarsky and others 2000). Finally, restoration might favor macroinvertebrates associated with LWD, a large and heterogeneous group (>60 species in Central Europe, Hoffmann and Hering 2000), encompassing taxa ranging from those that require LWD as habitat to those that are obligate xylophagous (that is, wood eating). It seems likely that such invertebrates would have restricted distribution in the 'almost debris free' (Muotka and others 2002) channelized boreal streams. However, in contrast to North America (for example, Anderson and others 1978), the abundance and distribution of these invertebrates are poorly described in boreal regions, as in general in Europe (Hoffmann and Hering 2000).

By reopening cut-off side-channels and replacing rocks in the stream (thus enhancing hyporheic exchange), restoration should increase the availability of potential refugia. This might be particularly true in low-gradient watercourses, where the predominant fine sediment (for example, sand, gravel) offers few refuge opportunities to macroinvertebrates during high discharges (Borchardt 1993; Hax and Golladay 1998).

Fish. Until recently, reductions of natural fish populations caused by habitat degradation have not been regarded as reasons to restore river watersheds and improve fish habitat in Scandinavia (McKinnell 1998). Instead, remedial policies in such situations have focused on stocking new fish, that is, symptoms have been addressed instead of causes. In the

Gulf of Bothnia and its surrounding rivers, for example, large-scale compensatory production of reared smolts has long been considered an important salmon conservation policy (Lindroth 1963). Few, if any, long-term management programs have been developed for other affected fish species.

Today, there are more diverse views on how running waters should be managed and habitat restoration has become a common practice. The effects of restoration of channel heterogeneity have mainly been evaluated for salmonids, such as Atlantic salmon and brown trout (for example, Näslund 1989; de Jong and others 1997; Linløkken 1997; Scruton and others 1998). Although these studies involved different methods and comprised relatively short reaches, often within the same river, they generally show that fish abundance increased following restoration. The responses of northern pike (*Esox lucius* L.), burbot (*Lota lota* L.) and Eurasian perch (*Perca fluviatilis* L.), the main predators on juvenile salmonids, are unknown. However, it is reasonable to assume that these species similarly will benefit from increased availability of suitable habitats and an increased abundance of prey.

The reduced current velocities that are predicted to result from restoration will probably favor anadromous and resident trout, Eurasian minnow (*Phoxinus phoxinus* L.), Eurasian perch and northern pike, all of which prefer slower flows. A reduced flow might also be advantageous for the European grayling which generally spawn in tranquil reaches with a mixture of sand and pebbles (Sempeski and Gaudin 1995), and the whitefish (*Coregonus* spp.), whose egg survival depends on flow refuges close to the bottom because they cannot withstand high current velocities (Lindroth 1957; Freeberg and others 1990). However, flow reduction could increase sedimentation in areas with slow currents, thus increasing the rate of egg mortality for whitefish and lamprey (*Lampetra* spp.) and stimulating the drift of lamprey larvae, therefore increasing the risk of predation. The effect of slower water flow on Atlantic salmon is more difficult to predict. In the main channels of northern Swedish rivers, salmon are often found in deeper water and at higher current velocities than trout (Heggenes and Saltveit 1990). It is unclear whether this is an effect of different habitat preferences between the two species or an effect of interspecific competition, as juvenile Atlantic salmon are considered to be competitively inferior to brown trout when the two species coexist (Lindroth 1955).

Increased hyporheic exchange due to restoration might also enhance habitat quality for salmonid fishes. For example, downwelling areas are preferred spawning habitat for many populations, as downwelling enhances intragravel flow, provides dissolved oxygen (DO) for incubating embryos, and removes metabolic byproducts from redds (Vaux 1968; Curry and Noakes 1995; Baxter and Hauer 2000). Subsurface upwelling is also associated with spawning habitat for various salmonid species (for example, Zorbidi 1988; Lorenz and Eiler 1989; Leman 1993). Although subsurface waters typically have lower concentrations of DO than surface waters, upwelling provides relatively constant intragravel flow and DO delivery (Lorenz and Eiler 1989; Leman 1993). The hydraulic action of upwelling also results in a loose and unconsolidated spawning substrate (Lorenz and Eiler 1989), which may be essential for fry emergence in some fish populations (Bams 1969). Because subsurface temperatures are generally more stable than those of surface waters, upwelling provides a relatively constant incubation temperature, thereby protecting overwintering embryos from freezing and reducing the effects of environmental fluctuations on emergence timing (Zorbidi 1988). Upwelling also provides thermal refugia for adults and free-swimming fry in winter (Craig and Poulin 1975; Cunjak and Power 1986) and in summer (Gibson 1966; Nielson and others 1994), so that habitat for all life history stages is enhanced in reaches with extensive hyporheic exchange (Baxter and Hauer 2000). Reduced habitat complexity is considered to be the main factor for the population declines of Atlantic salmon and brown trout in freshwater habitats (Chapman and Knudsen 1980). Restoration of tributaries that provide important nursery habitats for many species is thus important for the total fish production in the main river (Crisp 2000; Bagliniere and Maisse 2002).

Pristine rivers in Sweden presented relatively few constraints to fish migration. In floated rivers, the main obstacles were splash dams. For example, in the Ume River system, there were 375 splash dams by 1932 (Törnlund 2002). Removal of splash dams and reconnection of cut off side channels stemming from the timber floating period will increase the stream quality for migrating species both within the freshwater system and species with anadromous life histories, such as Atlantic salmon and sea-running brown trout. Reconnected side channels would also serve as alternative passages for migrating juveniles and adults during high flows or as bypass channels for upstream migrating adults when obstacles such as high falls and rapids occur in the main channel. As a result of increased connectivity, historically important spawning and nursery areas may become used again, favoring fish production. Re-opened side-channels and backwaters also provide suitable refuge habitat for juvenile fish sensitive to high flows (Heggenes 1988; Moore and Gregory 1988; Meyer and Griffith 1997).

Increased variation in water depth allows fish to move easily, thus facilitating their use of high-production feeding areas during summer, while in winter, deeper pools will act as refuges from ice formation and prevent bottom freezing. In northern Scandinavia, winter mortality is assumed to control the population density of fish, especially in years with low flows (Quinn and Peterson 1996; Mäki-Petäys and others 2000). In waters without deep pools, fish have to emigrate to find suitable winter habitats or they will be trapped and die. The replacement of boulders into the river channel and the concomitant increase in habitat and hydraulic complexity probably will favor local fish diversity, size and species composition because of increased winter survival caused by the increased availability of shelters and flow refuges.

In summary, we suggest that the following variables be included in monitoring programs; for aquatic plants: abundance, species richness, and life form; for macroinvertebrates: detritus mass and processing, production, species richness and evenness, trophic guild structure and life form; and for fish: migration success, spawning success, overwintering conditions, species richness, and production (Table 2).

DISCUSSION AND CONCLUSIONS

There is much to gain from restoration of former floatways. Table 2 summarizes the changes that are predicted in response to the return of boulders into the channels, the removal of stone piers and wing dams, and the reconnection of cut-off side channels. As suggested above, we expect most ecosystem components to be affected and responses to overlap between restoration methods. The major hypothesized changes are that the sinuosity and roughness of the channel will increase, that land-water interactions will be enhanced, that sediment deposition and nutrient availability will increase, that water flow will slow down and depth will increase (except in cases where the channel has been made much wider), that hyporheic flows will increase, and that the water level will become more stable although reconnected riparian areas will experience increases in flooding frequency. These hypothesized changes are believed to rapidly prompt the recovery of habitat complexity, migration, spawning, and overwintering success of fish, detritus retention, and dispersal of plant propagules from riparian areas. In a somewhat longer perspective we suggest that the primary productivity of riparian areas will increase, as well as the export of litter and LWD from the riparian zone to the river channel. We further propose that the plant zonation and the species composition will adjust to the new conditions, that diversity of aquatic plants and macroinvertebrates will increase, and that the production of macroinvertebrates and fish will increase. Although restoration actions will take place on specific sites or reaches, where floatway constructions are located, the spatial scales of abiotic and biotic responses are likely to vary from the site to the reach to the watershed levels (Table 2).

There are several experimental designs available for monitoring and evaluating river restoration projects (for example, Barmuta 2002; Downs and Kondolf 2002; Jungwirth and others 2002). These include studies both before and after restoration, and both at restored and control sites. Pre-restoration monitoring helps to define the project design and is useful for identifying deficits by comparison with target conditions. Pre-defined targets provide baseline data for evaluating restoration success or failure. Post-project evaluation gives the opportunity to judge the effectiveness of the restoration project, and provides a basis for modifying the designs of future projects (that is, adaptive management, for example, Walters 1986). Unexpected responses to restoration measures potentially provide important lessons about the functioning of river systems (Downs and Kondolf 2002). Thus, the assessments face three potential problems: (1) identification of target conditions, (2) choice of an appropriate set of indicators for evaluating success, and (3) choice of the appropriate time scale for the evaluation, which depends on the indicator chosen.

(1) Most people would anticipate the target states for restoration as being similar to the pristine (late pre-floating) situation and make comparisons with rivers that were not used for timber floating and thus not impacted. However, such examples are rare and not necessarily representative. Even if they were, some floatway reaches have been so strongly impacted that natural or near-natural conditions cannot be recreated. For example, in cases where only small, dynamited rock pieces are available in the piers, channels with large (natural) boulders cannot be (re)created unless such boulders are brought from upland areas. Whatever strategy is chosen, knowledge about the pre-floating state is useful. The inclusion of historic infor-

mation has proven to be an invaluable means for reducing uncertainties about the pre-floating state by detailing the modifications to the rivers caused by log driving. Many log-driving installations are obvious, but there are surprisingly many river reaches that appear virtually natural, although they have been considerably modified (cf. Wohl 2001). This applies especially to reaches where the major impact is that boulders and bedrock outcrops have been removed from the channel but no constructions have been built.

(2) In choosing appropriate indicator variables, important levels of ecological organization, relevant landscape elements, and appropriate scales should be represented (Jungwirth and others 2002). The time scale for recovery and the magnitude of the response vary between organisms and processes, and may also vary between different parts of a river (Uehlinger 2000). Therefore, an important question to be addressed is whether any particular group of organisms, process, or river reach could serve as an indicator for the responses of the entire river ecosystem, or at least major parts of it.

(3) Post-project monitoring needs to be long enough to allow evaluation of whether or not the system has reached dynamic equilibrium. Both the spatial extent of the restoration and the scale of response are important to consider for evaluating effect sizes. Whereas allochthonous litter inputs may increase soon after restoration, changes in nutrient status, productivity, species composition, and LWD production will likely develop over several years or decades. For example, although filamentous algae may recover within months (Shannon and others 2001; Ledger and Hildrew 2001), the ability of the riparian zone to supply LWD to the river channel may require several centuries for complete recovery. One reason is that Scots pine (*Pinus sylvestris* L.), the most important producer of sustainable LWD in the Nordic countries, is a slow-growing tree; another reason is that relationships between tree height and channel width may have changed following the removal of LWD and subsequent log driving, and can be difficult to restore (for example, Brooks and others 2003). Moreover, a wide range of stochastic events (for instances, climatic change, extreme weather events) may influence riparian forest succession, making it difficult to predict the long-term results of restoration.

As mentioned, the last two points are interdependent. For example, research in Finland showed that the response of macroinvertebrates to restoration varied depending on community traits and the time scale considered. In the short-term, macroinvertebrate abundance declined due to the disturbance created by the restoration work, but returned to pre-restoration levels within days to months depending on taxon and disturbance intensity (Tikkanen and others 1994; Laasonen and others 1998). The subsequent recovery occurred over several years, and varied in rate between the functional and the taxonomic characteristics of the assemblage. Whereas detritivores reached and even exceeded control densities within 8 years from the restoration, suggesting a relatively fast increase in invertebrate production in response to the enhancement of base resources (detritus), parallel changes in taxonomic composition and diversity were more gradual (Muotka and others 2002). Macroinvertebrate composition in the restored streams changed inconsistently, and remained considerably different from that of natural streams even more than a decade after the restoration work (Laasonen and others 1998; Muotka and others 2002). In other words, the case studies in Finland showed that restoration was effective in strengthening the energetic link between terrestrial and aquatic systems, but effects on biodiversity remain poorly substantiated so far. Whatever set of targets is chosen, hopefully it will be possible within 5–10 years to see what sort of ‘recovery trajectory’ a river is on, and to learn whether the methods applied fulfill the goals.

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