

**Mactaquac Aquatic Ecosystem Study**

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# **Environmental Considerations for Large Dam Removals**

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*Intended Use and Technical Limitations of the Environmental Considerations for Large Dam Removals:* The sole purpose of this report is to: 1) summarize known physical and ecological effects of large dam removals as described in published literature; and 2) to inform the Mactaquac Project team, as an initial step in the process, of the identified potential issues in support of their engineering design and environmental impact assessment and associated processes. The information contained herein does not necessarily represent the opinion of the CRI.

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# 1. Introduction

## 1.1 Background

The Mactaquac Generating Station (MGS) is Canada's 25<sup>th</sup> largest constructed river barrier by reservoir volume (Hanneman 2010) and was built in the late 1960s approximately 20km upstream of Fredericton, New Brunswick (NB) on the Saint John River (SJR). Alkali-aggregate reactions are causing expansion and subsequent weakening of the concrete used to build the concrete portions of the MGS. As a result, the MGS is projected to reach the end of its service life by 2030; for this reason, the concrete structures must be replaced or removed (New Brunswick Power Corporation 2014). New Brunswick Power's Mactaquac Project includes three end-of-life options for the MGS:

- Option 1—Repowering: Refurbish the MGS by constructing a new powerhouse, spillway, and other components, followed by the removal of the existing concrete structures;
- Option 2—Retain the headpond (no power generation): Build a new concrete spillway and maintain the dam as a water control structure without power generation, followed by the removal of the existing concrete structures; or
- Option 3—River restoration: Remove the MGS and enable the river to return to a free-flowing state (New Brunswick Power Corporation 2014).

## 1.2 Objective

Dam removal is a burgeoning science that has made great advancements in the past 20 years but inconsistent and/or limited monitoring programs has left many unanswered questions. Most of the dams that have been removed to date were relatively small and there is uncertainty about whether the physical and ecological effects of removing larger structures will be similar. The objective of this *Physical and Ecological Effects of Large Dam Removal* report was therefore to summarize the present state of the science of large dam removal as a starting point to understanding the potential environmental effects that Option 3 (river restoration by dam removal) may have on the SJR. It is important to note that this report precedes the completion of related science research and environmental evaluation of the proposed Mactaquac Project options; that specific qualities of the environments of the Mactaquac Project (other than height of the MGS) were not taken into account in the scope of this report; and therefore that inclusion of information in this report does not imply its relevance to the Mactaquac Project.

## 1.3 Scope

This report is limited to the physical and ecological effects of dam removal, though it is acknowledged that there are social, economic and other effects of dams and their removal. In this regard, the report focuses on the effects to aquatic ecosystems, including riparian, floodplain, and upland terrestrial habitats where they contribute to the connectivity of the landscape/ecosystem ecology (e.g., delivery of materials to aquatic ecosystem). To facilitate an understanding of the known effects of dam removal, the report includes a basic review of the effects that large dams have had on river ecosystems.

The International Commission on Large Dams (ICOLD) define large dams as those greater than 15 m in height. Since the number of large dams that have been removed is few, with even fewer having rigorous study of physical and ecological effects of removal, discretion has been used to expand this definition and some dams that were less than 15 m in height were included in the review.

## **2. Large Dams**

Canada has 1,166 large dams (ICOLD 2011), and it is estimated that there are over 10,000 small dams in Canada, though data on the number of small dams is incomplete (CDA 2003, Environment Canada 2004). A period of rapid dam building occurred in the United States from 1950 through 1980 and many dams are coming to the end of their life expectancy (Graf 1999). In fact, the American Society of Civil Engineers (2013) indicates that 70% of dams in the United States will be over 50 years old by 2020.

The life expectancy of an aging dam varies greatly. Hydro-electrical components have a 20-50 year life-expectancy, while the actual lifespan of structural components may vary with chemical (e.g. alkali-aggregate reactivity), physical and mechanical (e.g., thawing-freezing and drying-wetting cycles), and biological (e.g., plant growth in cracks) processes which occur on a case-specific basis (Wieland 2010). Thus, as dams age they typically require increasing amounts of maintenance to continue safe and efficient operations. Over time dam removal may become a viable and attractive option from economic, social, and ecological viewpoints. A variety of case-specific circumstances may result in the consideration of dam removal, including safety, obsolescence, economics of maintenance, siltation of the reservoir, and the ongoing ecological impacts of the dam. In the United States, aging infrastructure has invigorated support for dam removal over the past 20 years. While the effects of damming a river have been well described, particularly for large dams (Petts 1984, Collier et al. 1996, Poff et al. 1997, Rosenberg et al. 1997, Nilsson and Berggren 2000), the effects of dam removal are less well established (Bednarek 2001), particularly for large dams. However, significant advances in the collective understanding of the effects of dam removal and the recovery of river ecosystems after dam removal are accumulating.

### **2.1 The effects of large dams**

It is fundamental to understand the environmental effects of damming rivers because the potential negative environmental effects associated with a dam removal are likely a result of damming the river in the first place. In addition, a basic understanding of the environmental effects that large dams have on river ecosystems is necessary context for dam removal to be considered a river restoration strategy. Water development – largely dams and diversions – impacted a third of the threatened or endangered species in the United States, more than any other resource related activity (Losos et al. 1995). This section gives a brief description of some of the environmental effects dams have on river ecosystems.

Essentially, large dams are used to store water and/or raise water levels. Run-off is stored for controlled release during other times of the year while raising water levels is useful for navigation, increasing hydraulic head for power generation, recreational impoundments, and water diversion (McCully 1996). At the most fundamental level, dams alter water and sediment flux, temperature regimes, and habitat connectivity. Typical large dams result in the establishment of a reservoir with a long hydraulic residence

time (“HRT”; change from lotic to lentic habitat), greater alteration of temperature regimes, more severe impaired connectivity, lower peak flows, impaired sediment flux, and isolation of the downstream channel from riparian habitat (Bednarek 2001).

### **2.1.1 Reservoir**

The damming of a river necessarily floods a formerly terrestrial area, creating the lentic habitat of a reservoir from the lotic habitat of a river resulting in a number of physical and biogeochemical changes. Immediately, species requiring flowing water are displaced and those preferring lentic water may proliferate depending on habitat suitability (Baxter 1977). The decomposition of flooded terrestrial vegetation depletes oxygen in the hypolimnion and releases greenhouse gases (carbon dioxide and methane) to the atmosphere (Baxter 1977, Rosenberg et al. 1997). Mercury previously deposited on terrestrial soils becomes methylated, increasing its biological availability and allowing it to bioaccumulate in the food web (Rosenberg et al. 1997). The increased HRT combined with upstream nutrient loading and nutrient release from flooded vegetation often stimulates algal blooms that contribute to an anoxic hypolimnion (Baxter 1977). In some cases, anoxia in the hypolimnion creates reducing conditions that allow the release of additional nutrients (phosphorus) as well as sulfide, ferrous, and manganous ions from the sediments and further impacts water quality (Baxter 1977). In addition, reservoirs affect the downstream transport of nutrients, affecting downstream productivity. Reservoirs releasing water from the surface typically act as nutrient traps, limiting downstream nutrient levels, while hypolimnetic releases can increase downstream nutrient levels (Robert and Walker 1990). If dam operation includes reservoir drawdown, the littoral zone and the terrestrial and aquatic biota that rely on it are severely impacted (Baxter 1977).

### **2.1.2 Connectivity**

Impaired upstream-downstream habitat connectivity is a central feature of dams impacting anadromous, catadromous, and potamodromous species (Drinkwater and Frank 1994, Pringle et al. 2000). A barrier to migration interrupts life cycle completion and may ultimately result in local species extirpation (Pringle et al. 2000, Bunn and Arthington 2002). In other cases gene flow is disrupted, creating genetic isolation among populations and a potential loss of genetic diversity (Nielsen et al. 1997, Neraas and Spruell 2001).

A variety of structures (fishways) have been used to assist upstream fish passage at large dams including mechanized fish lifts, rock ramp fishways, natural channels, and a variety of fish ladders (EPRI 2002). These structures are often designed for specific species of commercial value (e.g., salmonids) and are somewhat successful in passing target species (Schilt 2007). However, salmonid fishways are wholly inadequate for fish community passage (Mallen-Cooper and Brand 2007), though multispecies approaches are becoming more common (Thiem et al. 2013).

Downstream fish passage occurs over the spillway, passing through turbines, or diversion to a fish bypass or collection area via behavioural guidance devices, physical barriers, collection systems, and diversion systems (EPRI 2002). The turbines are typically the greatest threat to downstream passage and can cause fish injury or mortality through extreme pressure changes, turbulence, shear, strike, and grinding (Cada 2001, Schilt 2007). Spillways are considered to be a more benign passage route but high

dissolved gas levels, pressure change, fall height, and turbulence remain potentially fatal issues (Schilt 2007).

Delayed migration at dams disrupts migration timing that is evolved to coincide with optimal environmental (e.g., temperature) or physiological conditions (e.g., smoltification; Schilt 2007). In the upstream direction, flow through the fishway is a fraction of flow through the turbines and fish tend to have difficulty finding the fishway entrance (Schilt 2007). In some cases, a fish might successfully pass the dam only to fallback below the dam (Boggs et al. 2004). In the downstream direction, low flow in the reservoir delays migration and injury or disorientation can occur during passage (Schilt 2007). Avian and fish predators often congregate at dams to feed on delayed or disoriented fish that are concentrated by fish passage structures, further increasing mortality among migratory fish (Schilt 2007).

### **2.1.3 Flow regime**

Dam operation creates a variety of flow scenarios in the downstream environment, however, a decrease in the magnitude and frequency of peak flows downstream is a unifying theme. The result is a decrease in lateral connectivity with the floodplain which has consequences for river geomorphology, sediment and nutrient delivery, propagule dispersal, fish habitat, and biodiversity (Junk et al. 1989, Graf 2006). While dams result in an overall flattening of the annual hydrograph, at shorter timescales water levels may fluctuate dramatically as hydroelectric dams respond to peak electricity demands (Petts 1984). The rapid rising and falling river stage associated with hydroelectric dams drives changes in wetted area and hyporheic exchange (Sawyer et al. 2009), affecting riparian vegetation (Nilsson and Berggren 2000), invertebrates and fish (Cushman 1985), fish spawning (Curry et al. 1994), and food web dynamics (Power et al. 1996). The effect is a shift toward generalist species that can tolerate rapid changes in flow and habitat availability with an overall decrease in biological diversity and change in productivity and energy flow (Poff et al. 1997). At the other end of the spectrum, some dams create artificially stable flows that promote prolific macrophyte growth and increased invertebrate abundance albeit at lower diversity compared to unregulated reaches (Bunn and Arthington 2002).

Large dams operated in northern climates store water during the summer months and release it during the winter when hydroelectric demands peak. On the Peace River, dam operation causes a complete reversal of the annual hydrograph with annual low flows occurring during June and peak flows occurring during December (Prowse et al. 2002). Increased flows during winter modifies the extent and duration of ice cover as well as the integrity of ice cover with concomitant impacts on minimum and maximum water levels (Environment Canada 2004).

### **2.1.4 Temperature**

Reservoirs that are deep and have long HRT tend to take on many characteristics of a large lake and can thermally stratify with warmer water floating over denser cool water on the bottom. Downstream environments are subjected to changes in water temperature modifying biogeochemical and metabolic rates (Poff and Hart 2002). Since reservoirs store heat, they buffer diurnal temperature change and delay timing of seasonal water temperature changes compared to unregulated rivers. Thus, regulated rivers experience spring and early summer water temperatures that may be cooler and late summer and fall water temperatures that may be warmer compared to unregulated rivers (Olden and Naiman 2010).



Although temperature effects are greatest immediately downstream of the dam, altered temperature conditions can persist over 100 km downstream depending on temperature differences, discharge volume, and downstream tributary inputs (Preece and Jones 2002, Bartholow et al. 2004).

Dam design and operation can have an effect of the magnitude and direction of downstream temperature changes (Olden and Naiman 2010). For example, summer operation of a bottom draw dam releases cold water from the hypolimnion that may negatively impact warm water species (Clarkson and Childs 2000) while warm water flowing from a top draw dam may negatively impact cold water species (Lessard and Hayes 2003). During winter operation in northern climates, the reverse is true with warmer water being discharged from a bottom draw dam which may affect the ice regime downstream of the dam (Environment Canada 2004). Affects to the ice regime include ice formation, ice cover growth and progression, ice cover breakup, and the movement and jamming of ice (White and Moore 2002). Typically, the presence of a dam reduces the probability or severity of ice jamming downstream of the dam (White and Moore 2002). The natural ice regime affects river hydrology, geomorphology, and water quality which affect river ecology (Prowse 2001a, 2001b). Some dams have a selective withdrawal system that allows the dam operator some control over the temperature of water released to the downstream environment.

### **2.1.5 Sediment flux**

Reduced flows in the reservoir impairs sediment flux with large and coarse sediments accumulating at the upstream end of the reservoir and where tributaries enter along the length of the reservoir (Kondolf 1997, Abraham et al. 1999). Finer sediments may be transported further, but in many large reservoirs sediment retention is 99% complete and clear water nearly devoid of suspended material is transported downstream (Williams and Wolman 1984). Clear water has a greater erosive capacity and can cause channel incision immediately downstream of the dam as the water equilibrates its sediment carrying capacity (Petts 1984, Kondolf 1997). Farther downstream, reduced flows may cause the river bed to aggrade where tributaries deliver sediments because peak flows no longer have the capacity to flush incoming sediments downstream (Collier et al. 1996). Dams on coastal rivers may alter salinity, nutrient supply, and primary productivity in estuaries and coastal waters in addition to impairing the sediment supply to estuaries and beaches resulting in erosion and habitat loss (Rozenfurt and Haydock 1993).

## **3. Short term effects of dam removal**

The short term environmental effects of dam removal are generally related to the movement of sediments that have accumulated in the reservoir, though the rate of reservoir drawdown can also have effects (Bednarek 2001). The known physical and ecological effects, their duration and potential mitigation options are discussed in this section.

### **3.1 Reservoir drawdown**

Reservoir drawdown can cause a variety of negative effects including loss of habitat, mobilization of sediments, fish stranding, and dissolved-gas supersaturation. In the case of dam removals, the effect of reservoir drawdown on habitat loss is a long-term effect and will be discussed in Section 4. The

mobilization of sediments associated with reservoir drawdown is a major concern and is treated separately in Section 3.3.

### **3.1.1 Physical effect**

#### **3.1.1.1 Fish stranding**

Reservoir drawdown leaves a potentially large area of previously flooded habitat, bare. While the loss of this habitat is a long-term effect, for mobile species (e.g., fish), drawdown is a one-time, short-term effect. As the reservoir is drawn down, isolated pools may become disconnected from the main channel forming barriers to movement, stranding fish, and eventually drying out.

#### **3.1.1.2 Dissolved-gas supersaturation**

Hydroelectric dams have long been known to cause dissolved-gas supersaturation if the velocity and pressure created by releasing impounded water is too great (Weitkamp and Katz 1980). Thus, reservoir drawdown associated with dam removal has the potential to cause dissolved-gas supersaturation in the downstream environment (Bednarek 2001). To our knowledge, dissolved-gas supersaturation has not been considered or measured during a dam removal project, possibly because few large dams have been removed to date. However, experimental reservoir drawdown at two Snake River dams increased dissolved-gas supersaturation downstream of the dam (Wik 1995). In this case, dissolved-gas supersaturation increased from background levels of 100-104% to 135% (U.S. Army Corps of Engineers 1993).

#### **3.1.1.3 Air quality**

During the Condit Dam removal, a variety of activities and conditions were expected to impact air quality through dust emission including demolition, road traffic, and wind erosion of sediments exposed after drawdown of Northwestern Lake. Although the scope of this report is on the aquatic environment, it is worth noting that during the summer after dam removal, local homeowners complained that significant quantities of blowing dust were being generated by wind eroding newly exposed and graded reservoir sediments (PacifiCorp Energy 2012a). It is possible that wind-moved sediment could re-enter the aquatic environment.

### **3.1.1 Ecological effect**

#### **3.1.1.1 Fish stranding**

Organisms stranded during reservoir drawdown would be expected to perish without intervention. At the Lower Granite and Little Goose dams on the Snake River, fish and benthic organisms were stranded in isolated pools that eventually desiccated (Wik 1995). In the planning stage of dam removal on the Elwha River, fish stranding during reservoir drawdown was identified as a concern. Specifically, planners were concerned that bull trout (*Salvelinus confluentus*) would be stranded in dewatered channels of the delta at the head of Lake Mills and Lake Aldwell (Crain and Brenkman 2011). Information regarding effectiveness was not located.

### **3.1.1.1 Dissolved-gas supersaturation**

The potential for injury or death resulting from dissolved-gas supersaturation increases as total gas concentrations exceed 110% (Weitkamp and Katz 1980) and thus negative effects of dissolved-gas supersaturation on the downstream biota would be expected. To our knowledge, the effect of dissolved-gas supersaturation on downstream organisms has not been studied for a dam removal project.

### **3.1.2 Duration of effect**

The duration of any effect associated with reservoir drawdown is directly related to the volume and inflow of the reservoir and rate of discharge over the dam. The reservoir at the Lower Granite dam was lowered 11.13 m at a rate of 0.61 m per day (Wik 1995) and, thus, the effect of dissolved-gas supersaturation is expected to have lasted about 18 days though complete dewatering would have taken longer. Dewatering of Lake Aldwell and Lake Mills on the Elwha River was staged over three years (Czuba et al. 2011). In contrast, the reservoir behind Marmot Dam was dewatered in eight hours (Major et al. 2012) and Northwestern Lake was dewatered in just over an hour when the Condit Dam was breached (PacifiCorp Energy 2012b).

### **3.1.3 Mitigation**

#### **3.1.3.1 Fish stranding**

Fish stranded by reservoir drawdown can be rescued using a variety of techniques including dip-nets, electrofishing, and seining as well as reconnection using heavy machinery (Crain and Brenkman 2011). However, the instability of exposed reservoir sediments can hamper rescue efforts (Wik 1995, Crain and Brenkman 2011).

For the Elwha River dam removals, plans were made for the capture and upstream relocation of 100 bull trout of various age classes from Mills Lake prior to reservoir drawdown. In addition, during reservoir drawdown, the delta areas of Lake Mills and Lake Aldwell would be monitored two days per week for stranding or barriers to migration. Stranded fish would be rescued and relocated, though unsafe working conditions during the first few months of drawdown were identified as a complicating factor. The use of heavy machinery to remove barriers would be considered as needed (Crain and Brenkman 2011).

#### **3.1.3.2 Dissolved-gas supersaturation**

Dissolved-gas supersaturation during reservoir drawdown has not been widely considered for dam removal schemes. The propensity for dissolved-gas supersaturation to occur during spilling operations at dams is well known, though dam design heavily influences supersaturation potential (Weitkamp and Katz 1980, Ruffing 1996). Modeling the effect of dam-specific conditions may lead to operational strategies that limit dissolved-gas supersaturation (Politano et al. 2012).

#### **3.1.3.3 Air quality**

During the Condit Dam removal, a variety of dust limitation and control measures were prescribed for demolition, road, and traffic/transport activities (PacifiCorp Energy 2009). These are standard measures associated with many construction activities not specific to dam removal and are not described further.

However, after Condit Dam was removed, blowing dust eroded from exposed reservoir sediments was impacting air quality for local homeowners. Initially, silt fence was placed perpendicular to prevailing winds but dust emissions from the site persisted. An application of tackifier and mulch successfully suppressed the erosion of exposed sediments and effectively eliminated the blowing dust problem (PacifiCorp Energy 2012a).

## **3.2 Contaminated sediment**

### **3.2.1 Physical effect**

#### ***3.2.1.1 Downstream transport of contaminated sediments***

The Fort Edward Dam removal on the Hudson River is an example of the negative effects that can occur if contaminated sediments are released to the downstream environment. When the Fort Edward Dam was removed in 1973, the amount of sediment that had accumulated behind the dam was vastly underestimated and the effect of dam removal on sediments upstream of the dam was not considered (American Rivers et al. 1999). In fact, over 227,000 kg of polychlorinated biphenyls (PCBs) from industrial activity had been released to the Hudson River over 30 years, much of it accumulating in the sediments upstream of Fort Edward Dam (U.S. EPA 1981). When the dam was removed, contaminated sediments were eroded from the reservoir and distributed downstream, impacting water quality, aquatic habitat, navigation, and public health (American Rivers et al. 1999). Contaminated sediments were transported downstream as far as New York Harbour but were concentrated within 40 km of Fort Edward where sediment concentrations ranged from 5 to 1,000 mg/kg on the depositional shore (U.S. EPA 1981). In addition, when the Fort Edward Dam was removed, contaminated sediments upstream of the dam were exposed requiring stabilization and containment (U.S. EPA 1981). The redistribution of PCB contaminated sediments in the Hudson River has required a massive remediation effort that is ongoing, 40 years after the breaching of the Fort Edward Dam (U.S. EPA 2014).

Mining and milling operations in the headwaters of the Clark Fork River basin released arsenic and a variety of unrecovered metals to local creeks. During high flows and flood events, sediments contaminated with arsenic and metals were transported downstream to the Clark Fork River and deposited behind the Milltown Dam. Since the mid-1980s, sediment deposition and scour in Milltown reservoir has been in dynamic equilibrium (sediment is scoured from the reservoir at the same rate as it is deposited), though deposition and scour processes occur over multiple years during low and high flows, respectively (U.S. EPA 2004). Arsenic and copper concentrations in water downstream of the dam increased during high flow events and were correlated with higher suspended sediment concentrations, consistent with the transport of contaminated sediment from upstream of the reservoir or scoured from the reservoir itself (U.S. EPA 2004). In 2004, the decision was made to remove 1.7 million m<sup>3</sup> of the most contaminated of an estimated 5 million m<sup>3</sup> of sediments contained behind the dam. During the construction and remedial action phases of the dam removal, temporary surface water standards and more stringent warning level standards were set for arsenic, metals of concern, and suspended sediments (estimated by turbidity). The warning level standards were exceeded multiple times during construction and remedial activities (May 2006 through December 2010), though many of these were related to turbid conditions upstream of the reservoir. High suspended sediment levels downstream of

the reservoir were related to construction activities when the dam was breached in 2008 and these conditions persisted for a period of approximately 3 months (Pacific Western Technologies 2011). During this time total copper concentrations (un-filtered recoverable copper) were elevated compared to concentrations upstream of the dam, but dissolved arsenic concentrations only exceeded the warning level (8 µg/L) three times and the temporary standard (10 µg/L) once (Pacific Western Technologies 2011, Sando et al. 2014). No other metals exceeded temporary or warning level standards (Pacific Western Technologies 2011). By 2010, total copper and suspended sediment concentrations were only slightly higher than those measured in 1996, indicating that the contaminated sediments stored in Milltown reservoir had already been flushed downstream of the monitoring station (~4.5 km downstream of the former dam site; Sando et al. 2014). Water quality data from the end of 2010 indicated a small increase in dissolved arsenic concentration downstream of the former Milltown Dam, which may be related to increased contributions from groundwater with elevated arsenic concentrations (Sando et al. 2014).

The elevated suspended sediment levels measured during construction and remedial activities associated with the removal of the Milltown Dam indicate that contaminated reservoir sediments were being transported downstream. During permanent reservoir drawdown, prior to breaching the dam, annual sediment loads attributed to remedial activities were an estimated 25,900 tons in 2006 and 130,000 tons in 2007 compared with pre-remediation conditions from 1985 to 2005 that saw an average annual accumulation of 5,750 tons in the Milltown Reservoir. In 2008, the post-breach annual sediment load was an estimated 391,000 tons and in 2009 the annual sediment load was 76,200 tons (Sando and Lambing 2011).

Suspended sediment and bed sediment data were used to track the source and sequence of sediment release after breaching the Milltown Dam in 2008. Elevated bed sediment metals concentrations were measured up to 254 km downstream of the dam, with concentrations decreasing exponentially with downstream distance (Garcia 2012). During low flows that occurred after dam breach, heavily contaminated fine-grained sediments were flushed downstream, but, two months later, mixed grain sediments from further up the reservoir were flushed downstream during high flows, and these less contaminated sediments diluted the more contaminated fine grain sediments that had been deposited previously (Garcia 2012). Within 5 months bed sediment metals concentrations had returned to near pre-breach conditions (Garcia 2012). Overall, downstream aggradation was limited to bars and side channels in a multi-thread reach 21 to 25 km downstream of the dam though significant infilling of substrata interstices may have a lasting ecological effect (Wilcox 2010).

### **3.2.1.2 Groundwater**

A major effect of the contaminated sediments in Milltown Reservoir was that arsenic was leaching into the local aquifer under Milltown, contaminating the public drinking water supply, and creating a public health issue (U.S. EPA 2004). Factors affecting the pathway and transport of arsenic from reservoir sediments to the aquifer include hydraulic head, aquifer constraint, and sediment grain size, thickness, arsenic concentrations, and redox conditions (U.S. EPA 2004). Drawdown of the Milltown Reservoir, excavation of the most contaminated sediments, and removal of the Milltown Dam and powerhouse was expected to not only stop further contamination of the local aquifer, but also remediate arsenic

contamination in the local aquifer through groundwater dilution (U.S. EPA 2004). Ten years was the targeted timeline for monitoring wells to achieve the compliance standard for arsenic concentrations (10 µg/L), though no estimate of the time required to achieve background conditions was given. Preliminary results indicate that groundwater remediation is underway with arsenic concentrations lower than compliance standards in test wells with temporally decreasing arsenic concentrations being measured in many test wells (Pacific Western Technologies 2011).

### **3.2.2 Ecological effect**

#### ***3.2.2.1 Downstream transport of contaminated sediments***

Although PCBs were most likely affecting the biota upstream and downstream of the Fort Edward Dam prior to its removal, the effect would have been local and largely contained. With the removal of the Fort Edward Dam, PCB contaminated sediments were distributed further downstream creating a much larger area of impact. Although there were no baseline data collected prior to dam removal, research conducted after dam removal found that PCBs bioaccumulated in benthic invertebrates and fish, and that concentrations were the highest ever documented in fish at the time (Nadeau and Davis 1976, Brown et al. 1985). The concentration of PCBs in fish tissue prompted a ban on fishing in 1976 on the Upper Hudson River and a ban on most commercial fisheries in the Lower Hudson River (U.S. EPA 2012), including the 40 million dollar striped bass fishery (American Rivers et al. 1999). An ecological risk assessment has determined that PCB concentrations in the Upper Hudson River remain a threat to fish communities as well as birds and mammals foraging in and around the Hudson River (TAMS Consultants Inc. and Menzie-Cura & Associates Inc. 2000).

Prior to the remedial activities in Milltown Reservoir and the removal of Milltown Dam, elevated metals concentrations associated with high flow events, ice jamming or scouring events, or reservoir drawdown resulted in fish mortality downstream of the Milltown Dam (Clark and Schmetterling 2013). Fish cage experiments completed during remedial activities in years 2006 through 2008 indicated significant fish mortality below Milltown Dam compared with the control stations (Schmetterling and Clark 2011). Caged fish mortality was attributed to elevated stream temperatures, low stream flow, and elevated sediment loads downstream of dam removal activities (Pacific Western Technologies 2011). Caged fish surveys from 2009 through 2011 indicate no effect of dam removal (Clark and Schmetterling 2013). Electrofishing and telemetry data indicate that fish populations experienced higher mortality during remedial activities in 2006 and 2007. In 2008, there was evidence of fish migration out of the Milltown Dam area but not fish mortality (Pacific Western Technologies 2011, Schmetterling and Clark 2011). Low fish numbers persisted in 2009 but the 2010 and 2011 monitoring years saw fish return to the Milltown Dam area (Pacific Western Technologies 2011, Schmetterling and Clark 2011). Overall, activities associated with removal of Milltown Dam negatively affected fish populations, however, recovery is occurring and the net effect of dam removal and remediation on fish populations is expected to be positive (Pacific Western Technologies 2011, Clark and Schmetterling 2013).

Benthic macroinvertebrate surveys immediately downstream of the Milltown Dam indicate a small decrease in biotic integrity during remedial activities from 2006 to 2008, however, the community rebounded to an unimpaired condition in 2009 (Pacific Western Technologies 2011). Thus, the biotic

integrity of the macroinvertebrate community was not drastically affected by dam removal operations and mobilization of contaminated sediments (Pacific Western Technologies 2011).

### **3.2.3 Duration of effect**

#### ***3.2.3.1 Downstream transport of contaminated sediments***

In the case of the Fort Edward Dam, where there was no attempt to remediate PCB contaminated sediments prior to dam removal, the effect is ongoing and predicted to remain if no attempt at remediation occurs (TAMS Consultants Inc. and Menzie-Cura & Associates Inc. 2000). PCBs do not readily biodegrade and, in the Hudson River, have an estimated half-life approaching 6 years making them persistent in the environment (Brown et al. 1985).

Milltown Dam and sediment removal activities were conducted from 2006 through 2008 when the dam was breached. A small increase in contaminated sediment loading during 2006 was associated with reservoir drawdown and contaminated sediment removal (Sando and Lambing 2011). Sediment loading associated with remedial activities increased through 2007 and peaked in 2008 during high flows, two months after the Milltown Dam was breached, though the greatest effect on water quality occurred when the dam was breached (Sando and Lambing 2011, Garcia 2012, Sando et al. 2014). Sediment transport and water quality had returned to pre-dam removal levels by 2010 (Sando et al. 2014).

### **3.2.4 Mitigation**

#### ***3.2.4.1 Downstream transport of contaminated sediments***

Reservoirs containing significant quantities of contaminated sediments may require mitigation prior to dam removal to avoid the environmental, social, and economic costs associated with sediment release to the downstream environment during and after dam removal (e.g., Fort Edward Dam). Mitigation options fall into two categories: removal or stabilization. The removal of contaminated sediments is accomplished using mechanical or hydraulic techniques followed by transport and disposal at an appropriate facility. The sediment removal approach tends to be the most costly depending on the amount of contaminated sediment being removed, the distance to an appropriate disposal site, and whether the disposal site needs to be lined (Bureau of Reclamation 2006).

Sediment stabilization options include engineering a channel through or around the reservoir sediments. Constructing a channel through sediments requires a relatively small volume of sediment to be moved and it could be deposited in the floodplain. Diverting the channel around the contaminated sediments is an attractive option, but is dependent on the availability of an appropriate route. Risks associated with sediment stabilization include the potential for bank or floodplain erosion during peak flows. Vegetation can be planted to limit surface erosion but bank stabilization structures would be needed at channel and floodplain edges and may require ongoing maintenance to ensure sediment stability. Tributary inflows to the reservoir complicate the design because flooding from tributaries can damage main channel protection and mobilize significant amounts of sediment if tributary channels are not constructed. Typically, sediment stabilization is less costly than removal, however, costs increase rapidly if the constructed channel extends a significant distance up the reservoir (Bureau of Reclamation 2006).

At the Milltown Dam, the most contaminated sediments were fine-grained and located close to the dam with less contaminated mixed grain sediments located further upstream (U.S. EPA 2004, Garcia 2012). The project plan included re-aligning the channel through the lower reservoir with a constructed floodplain to improve stability. In addition, a bypass channel was constructed and sheet piling used to reduce erosion of the most contaminated sediments during reservoir drawdown, dam and powerhouse removal, and contaminated sediment removal (U.S. EPA 2004). The timing of construction activities was critical in limiting impacts to local aquatic resources and activities were largely scheduled based on the annual hydrograph (U.S. EPA 2004).

#### **3.2.4.2 Revegetation**

Restoration of riparian and newly created floodplain associated with the Milltown Dam removal included a variety of grading, bioengineering, planting, and vegetation salvage approaches to re-establish vegetation (Geum 2012). A monitoring program that included short-term (0-15 years) vegetation performance criteria was used to assess whether revegetation was meeting project goals. Vegetation surveys in 2012 indicated that the vegetation cover of the engineered flood plain one year after revegetation was 36%, lower than the performance criterion of 90%. However, plant species richness was high, noxious plant richness and cover low, and desirable species accounted for 58% of total cover. Overall, the site is trending toward meeting project goals and objectives (Geum and University of Montana 2013).

### **3.3 Sediment release**

#### **3.3.1 Physical effect**

Sediment accumulates behind dams because water velocities decrease in the reservoir causing suspended particles to settle – a process referred to as “sedimentation”. The accumulation of sediment behind dams is often a problem affecting the safety, storage capacity, and utility of dams (Bureau of Reclamation 2006). From an environmental evaluation perspective, one of the most important issues for dam removal is the transport and fate of the accumulated sediments when the reservoir is drawn down and a natural flow regime re-established. During transport, the water quality may be adversely affected such that it becomes directly harmful to fish and other aquatic organisms. The rate of sediment movement and sedimentation (i.e., sediment flux) and the resulting redistribution of sediment after transport (i.e., sediment fate) may result in changes to the fluvial morphology upstream and downstream of the removed dam and longer-term indirect effects to fish and other aquatic organisms (e.g., change in fish habitat quality). It is important to note that sediment transport can occur both in suspension, as “suspended sediment”, and along the bottom as “bedload”.

##### **3.3.1.1 Water quality**

The sediment that had accumulated in Northwestern Lake, behind Condit Dam on the White Salmon River, WA was flushed downstream in a rapid, controlled release through a tunnel excavated, and blasted, at the base of the dam. The lake and much of the accumulated sediments were expected to be flushed downstream within 6 hours (PacifiCorp Energy 2011a). During this period, massive amounts of sediment were expected to be mobilized and suspended sediment concentrations were projected to be 100,000 mg/L to 250,000 mg/L and turbidity 50,000 to 127,000 nephelometric turbidity units (NTU);



Ecology 2007). Elevated turbidity was expected to persist downstream to Bonneville Pool and the mouth of the Columbia River due to the transport of clay particles which were expected to remain suspended (Washington State Department of Ecology 2007). In addition, naturally elevated mercury concentrations in Northwestern Lake sediments would enter the water column after dam breaching. It was estimated that mercury concentrations in White Salmon River would exceed Washington's acute and chronic water quality criteria for 20 and 49 days, respectively (Washington State Department of Ecology 2010). In the Columbia River, the chronic water quality criterion for mercury was expected to be exceeded for 17 days in Bonneville Pool (35 km downstream) and for 7 days downstream of Bonneville Pool (Washington State Department of Ecology 2010).

When Condit Dam was breached on October 26, 2011, it took under 90 minutes to drain the reservoir with an estimated 10% of sediments contained in the reservoir being flushed downstream in a slurry that was up to one third sediment and two thirds water (PacifiCorp Energy 2012b, Wilcox et al. 2014). At peak discharge, suspended sediment concentrations were 3000 mg/L, but increased to 850,000 mg/L 53 minutes later and maintained levels exceeding 100,000 mg/L for at least 6 hours after breaching the dam (Wilcox et al. 2014). The day after breaching, sediment concentrations were 50,000 mg/L and a week after breaching, concentrations were 10,000 to 25,000 mg/L. Suspended sediment concentrations were less than 500 mg/L one month after breaching the dam, though concentrations increased during higher flows (Wilcox et al. 2014). Suspended sediments, as measured by turbidity, continued to decrease during 2012 with the exception of planned sediment disposal events and rainfall/runoff events (PacifiCorp Energy 2012a). By 2013, turbidity measurements downstream of the former dam site were similar to upstream except during high flow conditions when turbidity increased modestly (PacifiCorp Energy 2013).

Marmot Dam on the Sandy River was breached on October 19, 2007. Construction crews initiated the breaching by creating a notch in the temporary coffer dam that held back water in the reservoir. Elevated suspended sediment levels measured 400 m downstream of the dam peaked at 49,000 mg/L, with the initial pulse having a higher proportion of silt and clay, largely derived from the coffer dam. Within an hour the suspended sediment load was 80% sand. After the initial 2 hours, suspended sediment concentrations declined to less than 10,000 mg/L and after 24 hours concentrations were less than 1,000 mg/L, similar to levels measured 10 km upstream of the dam. Elevated suspended sediment concentrations persisted downstream at least 18 km. Suspended sediment concentrations continued to be elevated immediately downstream of the dam for two months after breaching, and after this time water ran clear between high flow events. Seven months after breaching, suspended sediments were similar at sites upstream and downstream of the former dam site, even during high flow events (Major et al. 2012).

The Elwha and Glines Canyon dams on the Elwha River were removed in stages, over 3 years (2011 to 2014). However, at the time of publishing this report, only the first two years of data on the Elwha and Glines Canyon dam removal were available. Thus, the predicted response of reservoir sediments to dam removal activities and the actual response of reservoir sediments to the first two years of removal activities will be discussed. The incremental removal of Elwha and Glines Canyon dams was planned over 3 years, to allow the massive amount of sediment stored in Lake Aldwell and Lake Mills (19

million m<sup>3</sup> combined) to be released gradually. Lake Aldwell contained 3 million m<sup>3</sup> ( $\pm$  0.8 million m<sup>3</sup>) of sediment, of which 67% was fine grained (silt and clay) and Lake Mills contained 15.6 million m<sup>3</sup> ( $\pm$  2.7 million m<sup>3</sup>) of sediment, of which 48% was fine grained. Approximately one half to two thirds of these fine grained sediments were expected to be transported downstream, much of it as suspended sediment, impacting downstream water quality during and after dam removal (Czuba et al. 2011). Suspended sediment concentrations downstream of the Glines Canyon Dam were expected to exceed 10,000 mg/L for several weeks each year, though concentrations were expected to be lower (500 mg/L) at the river mouth (Konrad 2009). Suspended sediment concentrations upstream of Lake Mills are typically 1.6 mg/L (median flow of 34 m<sup>3</sup>/s) and during higher flows, 121 mg/L (101 m<sup>3</sup>/s which is exceeded only 5% of the time; Czuba et al. 2011). Suspended sediment concentrations were projected to return to background levels within 4 years of dam removal, though high flows could periodically mobilize additional sediments resulting in elevated concentrations beyond this time frame (Konrad 2009).

During the first two years of dam removal on the Elwha River, Elwha dam was removed completely and Glines Canyon Dam was reduced to 16 m, 25% of its original height (64 m; East et al. 2015). Suspended sediment concentrations were estimated based on turbidity and though there was considerable uncertainty in the model, broad trends were well defined. Suspended sediment concentrations measured in the lower river were driven by reservoir drawdown (sediment availability) and discharge (sediment transport capacity; Magirl et al. 2014). Notable increases in suspended sediment concentrations occurred when the Elwha dam was completely removed in April 2012 (8 months into removal activities) and when Lake Mills reservoir had been filled in by the prograding delta that reached the partially removed Glines Canyon Dam in October 2012 (Curran et al. 2014, Magirl et al. 2014). For 8 months after beginning removal activities, suspended sediment concentrations were generally less than 500 mg/L until Elwha Dam was completely removed, after which concentrations exceeded 500 mg/L periodically during high flow events in the spring of 2012. Through the summer of 2012, suspended sediment concentrations were less than 500 mg/L and were often less than 100 mg/L in late summer and early fall. Overall, suspended sediment concentrations exceeded 500 mg/L 17% of the time and 1000 mg/L 9% of the time during the first year of dam removal (Magirl et al. 2014). The second year of dam removal brought sustained elevated suspended sediment concentrations. In late 2012, high flows elevated suspended sediment concentrations in the lower river to 1,000 mg/L, often to 5,000 mg/L, and occasionally to 10,000 mg/L (Magirl et al. 2014). A brief low flow period in February allowed a decrease in suspended sediment concentrations, but high rainfall and snow melt during spring 2013 caused concentrations above 1,000 mg/L from March through June 2012. Clear water conditions coincided with low summer flows in August and September 2013 with suspended sediment concentrations often less than 100 mg/L (Curran et al. 2014). During year two of dam removal, suspended sediment concentrations exceeded 500 mg/L 74% of the time, 1000 mg/L 59% of the time, and 10,000 mg/L less than 1% of the time (Magirl et al. 2014).

Prior to dam removal on the Elwha River, turbidity in groundwater close to the river increased during high flows when turbidity in the river increased. Thus, it was expected that turbidity in groundwater downstream of the Glines Canyon and Elwha dams would be elevated during and shortly after dam

removal activities on the Elwha River. Sediments from Lake Mills and Lake Aldwell also contain elevated iron and manganese pore water concentrations. This could cause elevated iron and manganese concentrations in both river water and groundwater, though it is likely that iron and manganese will oxidize and be transported downstream as particulate matter (National Park Service 2005).

The chemistry of phosphorus in Lake Mills sediments and its potential behaviour under dam removal conditions was examined experimentally prior to dam removal on the Elwha River (Cavaliere and Homann 2012). Phosphorus concentrations were higher in fine-grained sediments compared to coarse sediments, though sediment-phosphorus concentrations were low compared to eutrophic ecosystems. During dam removal, downstream soluble reactive phosphorus concentrations were expected to double due to the removal of the dams as a sediment and nutrient trap (Duda et al. 2010), and the suspension of reservoir sediments (Cavaliere and Homann 2012). However, absolute soluble reactive phosphorus concentrations were expected to remain low, 2 to 5 µg/L (Cavaliere and Homann 2012).

### **3.3.1.2 Sediment transport**

The Condit Dam removal used a rapid sediment release strategy which resulted in 20% of sediments stored in Northwestern Lake being transported downstream in the first 24 hours (Wilcox et al. 2014). Within the first few minutes of breaching there was little erosion of reservoir slopes because these were vegetated and had been exposed repeatedly during previous fluctuations of reservoir level. However, once un-vegetated, fine-grained, and saturated sediment was exposed, slope failure progressed rapidly through the lower portion of the reservoir delivering massive amounts of sediments and woody debris downstream. Bedrock underlying the original river channel was exposed within an hour of breaching the dam and channel incision through the sediments continued rapidly upstream. Within 5 hours, significant erosion of sediments had occurred at least 1 km upstream and after 24 hours, 20% of stored sediment (360,000 m<sup>3</sup>) had been transported downstream (Wilcox et al. 2014). Erosion remained high over the next two weeks but slowed after reaching almost 3 km upstream of the dam. An estimated 1.1 million m<sup>3</sup> of sediment had been eroded 8 weeks after the dam was breached with an estimated additional 0.2 million m<sup>3</sup> of sediment being eroded over the next 6 months indicating substantial stabilization of remaining reservoir sediments (Wilcox et al. 2014). The high proportion fine grained sediments (35% silt and clay, 60% sand, 5% gravel) was a key factor in the mass movement of Northwestern Lake sediments (Wilcox et al. 2014).

Breaching Condit Dam resulted in a massive flux of water and sediment from Northwestern Lake. Much of the sediment was transported in suspension as indicated by extremely high suspended sediment concentrations measured in the hours and days after breaching. However, deposits of sand at channel margins was evident indicating that a proportion of mobilized sediments were deposited along White Salmon River, though little channel aggradation was evident in the first hours after breaching (Wilcox et al. 2014). In the days and weeks following the breaching of Condit dam, bedload transport downstream of the dam was more evident and channel aggradation occurred with over a meter of material being deposited within the river. After 11 days, sediment supply from the reservoir was decreasing and net erosion of aggraded material commenced; in three days the river incised 0.8 m to a new stable stage. Pools present prior to breaching were filled with gravel and sand and gravel bars had formed on either side of the channel (Wilcox et al. 2014). Sediment transported further downstream accumulated at the

mouth the White Salmon River and formed a sandbar at the confluence with the Columbia River (PacifiCorp Energy 2012b).

Major et al's (2012) detailed description of the breaching Marmot Dam is paraphrased as follows:

*Similar to the Condit Dam, a rapid release approach was used to breach the temporary coffer dam that was used during the removal of Marmot Dam on the Sandy River, though differences in the breaching technique and stored sediment composition made for a somewhat less spectacular sediment release (Lovett 2014). Compared to Condit Dam, the sediments impounded by Marmot Dam were coarse, consisting of equal parts sand and gravel, effectively slowing the rate of erosion. Within 20 minutes of the coffer dam being notched, a small channel had formed creating 2 to 3 m steps on the face of the dam that moved upstream at rate of meters per minute. After 45 minutes, the steps joined to create a 2 m tall knickpoint at the dam crest that began migrating rapidly upstream. The incision of the dam crest released the small volume of water held in the sediment filled reservoir, increasing flow and rapidly incising an 8 m deep channel through the coffer dam that began eroding laterally 1 hour after the initial notching. The knickpoint continued its migration and within 15 hours had travelled 400 m upstream, leaving a channel 40 m wide in lower 300 m of the reservoir. After 60 hours, discharge waned and rapid erosion ceased, though in this time 125,000 m<sup>3</sup> of sediment had eroded from the reservoir, about 17% of the total impounded volume. After 3 days, the knickpoint resembled a 1 m tall riffle that migrated upstream about 10 m each day for the first month, though erosion slowed progressively and migration was only a few meters each day 7 to 12 months after breaching. After 2 months 40% of the impounded sediments had been eroded, and subsequently, major erosion only occurring during high flows. Two years after breaching Marmot Dam, 425,000 m<sup>3</sup> of sediment (60% of the total) had eroded from the reservoir.*

After breaching the coffer dam on Sandy River, sediments eroded from the reservoir were largely deposited in a 1.3 km sediment wedge that was 4 m thick immediately downstream of the former Marmot Dam. This sediment wedge accounted for approximately 50% of the sediments eroded from the reservoir within the first 60 hours. A month after breaching, minor sediment deposition was measured 9 km downstream of the former dam site, and no deposition was measured 18 km downstream. During the year after breaching, an additional 45,000 m<sup>3</sup> of sediment accumulated in the wedge downstream of the dam and the remaining 330,000 m<sup>3</sup> that was eroded during this time passed downstream. Two years after breaching, the sediment supply from the reservoir declined and the wedge below the former dam began to incise signalling a change to net downstream sediment transport in this reach. Up to 50% of the sediments eroded during the first 60 hours were not accounted for and presumed to be located within a poorly accessible gorge located 2 km downstream of the dam site and stretching for 7 km. Indeed, surveys in the gorge 18 months after breaching revealed that 2 to 3 m deep pools in this section had largely filled with sand and gravel. Although, sediment transported downstream of the gorge had increased after breaching the dam, no substantial aggradation was measured downstream of the gorge (Major et al. 2012).

The Elwha Dam was removed incrementally from September 2011 to April 2012 while the larger Glines Canyon Dam was removed incrementally from September 2011 to September 2014. The incremental strategy of dam removal was employed due to the massive volume of sediments impounded in Lake Aldwell and Lake Mills. Drawdown of the reservoirs was to occur in 2 to 6 m stages, not exceeding 0.46 m per day in order to mobilize as much erodible material as possible while preventing landslides. A reservoir drawdown experiment indicated that incremental drawdown would allow unvegetated fine-grained sediment to become suspended, though low flows in the reservoir section would allow settling with a low proportion of fine-grained sediments being transported out of the reservoir during early stages of removal. Coarse sediments would be transported as bedload and accumulate at the upstream end of the reservoir resulting in a prograding delta. During periods between drawdown, stable reservoir elevation would allow lateral erosion of upstream sediments through channel migration. Prior to drawdown, a pilot channel was placed in the center of the Mills Lake delta and vegetation removed to ensure that channel formation did not favour one side of the canyon because that had the potential to decrease the total amount of sediment mobilized and leave a high sediment terrace on the opposite canyon wall that would be vulnerable to landslides over time. As dam removal progressed and reservoir elevation lowered, the capacity for the reservoir to retain suspended sediments would decrease resulting in progressively higher suspended sediment loads transported downstream. At the same time, the prograding delta would eventually reach the lowered dam structure, effectively filling the reservoir and eliminating sediment and water storage capacity. Thus, subsequent dam removal increments would release sediment pulses of both fine-grained sediments (suspended) and coarse sediments (as bedload) to the downstream environment (Czuba et al. 2011).

Preliminary results on the response of reservoir sediment to incremental dam removal and reservoir drawdown indicate that pre-dam removal predictions of the progression of sediment erosion and transport were accurate in many respects (Magirl et al. 2014, Randle et al. 2015). However, during the first two years of dam removal the total amount of sediments mobilized from the reservoirs were less than expected. In this time, only 23% of Lake Aldwell sediments were transported downstream ( $1.12 \pm 0.07$  million  $m^3$ ), 83% of that during year one (Randle et al. 2015). Upstream, 37% of Lake Mills sediments were transported downstream ( $5.95 \pm 0.12$  million  $m^3$ ) during the first two years of dam removal, though only 3% was transported during year one because of sediment retention within the reservoir (Randle et al. 2015). However, Elwha River sediment loads downstream of the dams were 3 and 20 times the mean annual sediment load during year one and two of dam removal, respectively (Magirl et al. 2014). The majority of sediment transported downstream of Lake Mills was coarse grained and occurred as bedload after the prograding delta reached the dam (Magirl et al. 2014, Randle et al. 2015). Upstream and within the reservoir, channel width during drawdown was influenced by sediment grain size; areas with coarse, non-cohesive sediments had wider channels than areas with fine-grained cohesive sediments, though discharge influenced channel width as well (Randle et al. 2015). Erosion of the upper delta led to aggradation in the delta and prodelta where water velocities slowed. During periods of stable reservoir elevation, aggradation affected channel slope and caused channel confinement as it moved upstream stimulating lateral erosion (Randle et al. 2015). The erosion of fine-grained sediments from Lake Aldwell and Lake Mills was lower than expected because their greater resistance to erosion stranded these sediments in terraces, effectively limiting sediment availability.

Similarly, erosion of coarse grained sediments from Lake Mills was greater than expected because there was little resistance to lateral erosion of these sediments resulting in greater channel migration and greater overall erosion of the valley (Randle et al. 2015).

During the first year of dam removal on the Elwha River, limited (<0.1 m) channel aggradation occurred between Glines Canyon Dam and Elwha Dam. In the lower Elwha River channel aggradation was typically limited to about 0.1 m of silt and sand which accumulated within interstitial spaces of pre-removal cobble, though these fine-grained sediments were eroded rather quickly (Draut and Ritchie 2013). Locally, aggradation in the low river was greater with up to 0.5 m of sediments deposited in large pools and in floodplain channels behind areas of obstruction (e.g., log jams; East et al. 2015). During the second year of dam removal, sediments in Lake Mills overtopped Glines Canyon Dam increasing downstream sediment transport and leading to substantial channel aggradation between the dams and in the lower river (Magirl et al. 2014, East et al. 2015). Within in a month, aggradation from 0.5 to 1.5 m had occurred throughout the 14 km reach between the dams though much of this material was moved downstream in subsequent months with channel beds having a net aggradation of 0.5 m or less and in some cases returning to their previous elevation (East et al. 2015). In the lower river, there was evidence of channel aggradation within two weeks of sediment overtopping Glines Canyon Dam, though some reaches did not accumulate any sediment and served to transport sediment farther downstream throughout the study (East et al. 2015). Channel beds aggraded 0.6 m over 5 months after which they were relatively stable for an additional 5 months, though river kilometer 1 aggraded an average of 1 m and increases of 1.6 m were measured near the former Elwha Dam and 2.0 m near the river mouth. Greater aggradation rates in the lower river was a response to decreased slope due to aggradation of the river into the Strait of Juan de Fuca (East et al. 2015). Deposition of new sediments also occurred in floodplain channels with an average increase in channel elevation of 0.5 m. The influx of large amounts of sand and gravel during year two of dam removal caused an almost universal fining of channel-bed grain size downstream of Lake Mills. During this time, river morphology changed from riffle-pool to braided channel because of new sediment accumulating in pools and the formation of bars. By the end of the second year of dam removal, some aggraded channel beds had begun incising (East et al. 2015). Despite the extensive channel aggradation observed downstream of Glines Canyon Dam, only 10% of released reservoir sediments remained in the river and floodplain channels with the rest being transported to, or past the river mouth (East et al. 2015, Warrick et al. 2015). An estimated 2.5 million m<sup>3</sup> of eroded reservoir sediment accumulated in the delta at the mouth of the Elwha River, extending the delta 200 m seaward (Gelfenbaum et al. 2015). Therefore, much of the eroded reservoir sediments (55%) was deposited further offshore in the Strait of Juan de Fuca (Gelfenbaum et al. 2015, Warrick et al. 2015). The effects of removing the final 16 m of Glines Canyon Dam (removed from September 2013 to September 2014) are likely substantial, though documentation of this has not been released. A further 14 million tons of sediment (133% of that already released) is stored behind the remaining structure, 40% of it being fine-grained (Warrick et al. 2015).

### 3.3.2 Ecological effect

#### 3.3.2.1 *Water quality*

The elevated mercury levels associated with the release of sediments contained behind Condit Dam were expected to represent acute and chronic effects for aquatic organisms. However, the effect of the sediment flux itself (see Section 3.3.2.2) was predicted to be so harmful to aquatic biota, the elevated mercury levels were deemed to represent no additional impact (Washington State Department of Ecology 2010).

Elevated suspended sediment concentrations associated with Marmot Dam removal construction activities and breaching was expected to have a negative effect on fish populations downstream of the dam. According to empirical equations on the effect of suspended sediments (Newcombe and Jensen 1996), suspended sediment concentrations measured after breaching were elevated for a duration that would certainly cause sub-lethal effects and was at the threshold for lethal (and para-lethal) effects for juvenile and adult salmonids immediately downstream of Marmot Dam. Sub-lethal effects likely continued for about 3 months after breaching. Sub-lethal effects include physiological stress, reduction in feeding rate and success, impaired homing ability, and habitat degradation, and para-lethal and lethal effects include reduced growth rate, delayed hatching, reduced fish density, and possible direct mortality of sensitive individuals (Newcombe and Jensen 1996).

According to Newcombe and Jensen's (1996) empirical equations, adult and juvenile salmonids downstream of the Elwha River dams would have been periodically exposed to lethal and para-lethal effects during the highest turbidity periods which occurred during active reservoir drawdown and high flows. An extended period of elevated turbidity occurred beginning in the second year of dam removal that would have exerted sub-lethal effects that extended for 8 months (Magirl et al. 2014). The final environmental impact statement also recognized that fish, their eggs, and other aquatic life downstream of the Glines Canyon and Elwha dams would likely suffer high mortality and the fish restoration plan assumed that all fish in the Elwha River downstream of Glines Canyon Dam would be killed (National Park Service 1995, Ward et al. 2008). Elevated turbidity levels associated with removal of the Glines Canyon and Elwha dams on the Elwha River were also expected to reduce immigration and disrupt homing of anadromous salmon, effectively increasing stray rates and prolonging recovery of fish populations (National Park Service 2005, Pess et al. 2008). Despite this, it is expected that salmonids will establish populations between and upstream of the former Elwha River dam sites within 1 to 5 years of dam removal (Pess et al. 2008). Elevated turbidity downstream of the Glines Canyon and Elwha dams during and for a short period after dam removal are expected to decrease periphyton and benthic invertebrate abundance and diversity, though aggradation and shifting substrates are likely to have a similar effect (Morley et al. 2008). However, soluble reactive phosphorus concentrations were also expected to increase and may stimulate periphyton growth, offsetting some of these effects (Cavaliere and Homann 2012).

#### 3.3.2.2 *Sediment flux*

The rapid sediment release strategy used for the Condit Dam removal is possibly the most destructive method of dam removal for downstream habitats. The White Salmon River downstream of the dam

experienced short periods of hyperconcentrated flows similar to mass wasting events such as those occurring during landslides or volcanic eruptions (Wilcox et al. 2014). In addition, the river bed was blanketed with over 1 m of sand. The *Condit Dam Final Environmental Impact Statement* indicated that all fish and aquatic macroinvertebrate would likely be killed or displaced and that macroinvertebrate populations would take years to re-establish (Washington State Department of Ecology 2007). In addition, all aquatic organisms residing in Northwestern Lake when Condit Dam was breached were expected to be killed, though some fish may be flushed down to the Columbia River. Conditions downstream of the dam were expected to be unsuitable for fish for three months after breaching and fish migration would be impaired for six months. Any anadromous fish in the river and downstream of the dam at the time of breaching was expected to be killed and any redds (nests) of successful spawners were expected to be destroyed, effectively eliminating a year-class of fry. An estimated 2,289 fall chinook salmon (*Oncorhynchus tshawytscha*) escaped into White Salmon River in 2011, 77% of which were not translocated, indicating a high proportion of the fall chinook salmon run lost to dam removal operations (Engle et al. 2013). The potential for sediment and large woody debris to create fish passage issues upstream and downstream of the dam was recognized.

However, fish returned to the White Salmon River within 3 weeks of breaching the dam and 5,500 chinook, from three populations, were counted 2 years later (Lovett 2014, Washington Department of Fish and Wildlife 2014). In 2012, 215 fall chinook salmon carcasses were recovered from the lower 12 km of White Salmon River. Most fall chinook salmon spawning (as indicated by red observation) was located downstream of the Condit Dam site with spawning occurring in the former reservoir area and as far as 7 km upstream of the former dam site (Engle et al. 2013). In 2013, 42% of spring chinook redds were counted upstream of the former dam site compared to 1 to 1.5% of fall chinook redds (Washington Department of Fish and Wildlife 2014). Creel surveys conducted in August 2012 indicated that anglers were catching steelhead upstream of the former dam site and one Endangered Species Act-listed bulltrout was angled (Engle et al. 2013). Steelhead spawner and red surveys conducted in 2012 did not detect either fish or redds, but redds along with live fish and carcasses were detected in tributaries upstream of the former dam site in 2013 and 2014 (Yakama Nation Fisheries Program 2014). In fall 2014, coho were observed in tributaries upstream of the former Marmot Dam for the first time in 100 years (Brady Allen, personal communication, January 26, 2015).

After breaching Condit Dam, approximately 331,000 m<sup>3</sup> of sediment accumulated in the Columbia River at its confluence with White Salmon River (PacifiCorp Energy 2012b). It was expected that macroinvertebrates in this area would be buried and fish temporarily displaced due to the associated sediment plume (Washington State Department of Ecology 2007). Larval lamprey occupancy in lower White Salmon River was marginally lower in 2012 compared to 2011 surveys conducted prior to breaching, and this may be related to loss or change in habitat associated with sediment release (Jolley et al. 2013). However, the deposition of fine-grained sediments and formation of a delta at the mouth of White Salmon River may have provided habitat for the lamprey that were detected there in 2012, but not prior to dam removal (Jolley et al. 2013).

The removal of Marmot Dam and breaching of the temporary coffer dam released a 5 to 10 year supply of sediment into the Sandy River, and formed a sediment wedge that was 4 m thick immediately



downstream of the dam which tapered considerably within the first 2 km (Major et al. 2012). Channel aggradation immediately downstream of the dam was likely to have highly negative effects on any redds in the reach. Short term negative effects on macroinvertebrate community density and diversity were also predicted due to elevated suspended sediment concentrations and sediment deposition associated with dam removal (Stewart and Grant 2005). However, despite the highly aggraded channel, a relatively diverse macroinvertebrate community was found in this section two years after breaching. Channel aggradation was expected to smother habitat and create shifting channel morphology that was expected to eliminate salmonid habitat value. Further downstream, pool habitat was expected to aggrade affecting rearing habitat quality (Portland General Electric Company 2003). Riparian vegetation colonizing sand and gravel bars located 9, 12.5, and 28 km below the former dam site changed little over 2 years following dam removal.

Although the number of spring chinook salmon redds observed upstream of Marmot Dam in the year it was removed was low (271), the 5 year average number of redds upstream of Marmot Dam has increased from 424 before the dam was removed (2002-2006) to 1,079 after dam removal (2008-2012; Schroeder et al. 2013). This trend is more striking considering that the 5 year average for spring chinook returns for the whole of Sandy River was 8,379 before the dam was removed (2002-2006) and 5,030 after dam removal (2008-2012), indicating an increase in habitat usage upstream of the former dam site (Schroeder et al. 2013, Joint Columbia River Management Staff 2014). However, local management goals are to protect wild spring chinook genetics and prior to the 2007 dam removal, wild fish passed over Marmot Dam and hatchery fish were removed. With the removal of the dam, this management tool was no longer available and hatchery fish were able to spawn with wild fish in greater numbers, potentially impacting the genetics of the wild population. From 2002 to 2007, a mean 89% of spring chinook spawning upstream were wild compared to 40% from 2008 to 2011 (Schroeder et al. 2013). In response, Oregon Department of Fish and Wildlife installed weirs to remove hatchery fish upstream of the Marmot Dam site and as a result the target 90% wild fish spawning upstream of the dam has been met in recent years (Luke Whitman, personal communication, January 28, 2015). Since Marmot Dam was removed, mean coho salmon (*Oncorhynchus kisutch*) estimates were greater (1,500 from 2008 to 2013) compared to pre-removal estimates (944 from 2002 to 2006) (Oregon Department of Fish and Wildlife 2014). However, this difference is not large compared to the uncertainty of the estimates (target = 30%) and the data is from a base monitoring program and was not designed to detect an effect of dam removal (Mark Lewis, personal communication, February 2, 2015). It is premature to conclude that the removal of Marmot Dam has had an effect on local populations, but it certainly appears that habitat use upstream of the Marmot Dam site has increased, though not necessarily by wild populations. An important lesson from the Marmot Dam removal is that if project objectives include detecting whether there was an effect on anadromous fish populations, a specific study is necessary, particularly in the case of dams that had fish passage structures.

Downstream of Glines Canyon Dam on the Elwha River, interstitial spaces filled with silt during the first year of dam removal with expected negative effects on biota (Draut and Ritchie 2013). Fining of sediments decreases sediment pore space limiting the flow of oxygenated water with potentially negative consequences for incubating fish eggs and benthic invertebrates. Indeed, a decline in spawning

habitat quality was predicted prior to dam removal due to deposition of reservoir sediments and channel destabilization (Ward et al. 2008, Pess et al. 2008). In the second year of dam removal, large quantities of gravel covered channel beds armored with cobble resulting in an overall increase in available spawning habitat (East et al. 2015), though autumn high flows are expected to periodically incise or aggrade the channel bed, causing changes in channel-bed elevation and particle-size distribution, negatively affecting the chinook salmon that are incubating in redds at that time (Konrad 2009). Sediment deposition in floodplain channels represents a decrease in juvenile salmonid habitat, though this may be somewhat offset by braiding and increased complexity in the main channel (East et al. 2015).

The release of sediment stored in Lake Mills and Lake Aldwell was expected to affect vegetation on the lower Elwha River and Estuary through geomorphic changes caused by altered sediment and large woody debris supply. These changes could affect estuarine shorelines, tidal connections, and water levels influencing hydraulic connectivity and salinity resulting in changing vegetation communities (Gelfenbaum et al. 2011). In between the former dams and on the lower Elwha River, seedlings have already been observed on newly created bars (East et al. 2015). In the Elwha River Delta and nearshore areas the sediment release caused a fining of substrates, which could decrease the amount of kelp in the area because they require a relatively stable substratum for growth and reproduction (Gelfenbaum et al. 2011, 2015). Plants that propagate vegetatively may be resistant to sedimentation and substrate fining (Gelfenbaum et al. 2011). The turbid conditions associated with elevated suspended sediment concentrations in the river plume increased light attenuation and may have affected plant photosynthesis with a subsequent decline in densities (Besso 2014). The deposition of sediment in the Elwha Delta was up to 7.5 m thick 2 years into dam removal and is expected to cause a decline in species richness and change the community composition of plants and invertebrates in the delta and nearshore areas (Gelfenbaum et al. 2011, 2015). Bedrock/boulder reefs supported the most diverse community in nearshore areas prior to dam removal and were expected to be the most impacted by sediment release because of their susceptibility to burial by newly delivered sediment (Gelfenbaum et al. 2011).

### **3.3.3 Duration of effect**

#### ***3.3.3.1 Sediment flux***

The strategy for the Condit Dam removal was to flush the majority of sediments contained in Northwestern Lake downstream in a short time period, recognizing that the effect on downstream biotic communities would be acute, but that re-establishment processes could begin as quickly as possible. After the initial mass sediment transport associated with breaching the dam, high levels of sediment transport continued for two months and after six months, the rate of sediment transport from Northwestern Lake had slowed significantly (Wilcox et al. 2014). Sediment transport was sporadically high, and sometimes artificially high due to planned sediment disposal associated with grading activities, for 12 months post removal. Later, only high flow events caused turbidity levels to be greater than those measured upstream of the sediment deposits, indicating that the remaining sediments had largely stabilized (PacifiCorp Energy 2011b, 2012a, 2013).

The strategy for the Marmot Dam removal was also to flush the impounded sediments downstream rapidly. After 2 months, 40% of the impounded sediments had been flushed downstream after which substantial erosion occurred only during high flows. A year after breaching the dam, sediment deposited in a wedge immediately downstream of the former dam site began to erode signaling that the sediment supply from the reservoir had decreased and downstream channel aggradation had ceased. Two years after breaching the dam, the aggraded channel immediately downstream of the dam was largely stable with the exception of during high flows (Major et al. 2012).

The large volume of sediments impounded by Glines Canyon and Elwha dams on the Elwha River necessitated an approach where the dams were removed incrementally with a staged drawdown of Lake Mills and Lake Aldwell. This approach was expected to limit the maximum suspended sediment concentrations and avoid catastrophic conditions and changes in the downstream environment. Active dam removal activities were carried out over 3 years, though it is expected that elevated sediment flux will continue for years after removal. Sediment loads causing major adverse effects were expected to occur periodically for 3 to 5 years post dam removal, though sediment transport conditions during low and minimum flow conditions were expected to recover quickly (National Park Service 2005, Konrad 2009). Generally, suspended sediment concentrations were expected to approach background conditions 4 to 7 years after dam removal (Czuba et al. 2011). However, no explicit predictions were made about how long elevated sediment concentrations would occur during high flows and Konrad (2009) recognized that remaining reservoir deposits could become a chronic source of suspended sediment to the lower Elwha River.

### **3.3.4 Mitigation**

#### **3.3.4.1 Closure of fish passage structures**

As part of the Marmot Dam removal, the fish ladder was decommissioned 3 months prior to breaching, effectively eliminating fish passage while the remaining structure was removed. Alternative fish passage was provided using a Denil fish ladder and a trap with attraction flow piped from above the dam. A barrier was installed downstream of the dam to limit migration beyond the temporary trap. Fish in the trap were trucked and released upstream of the dam. The trap was operated until the coffer dam was breached (Portland General Electric Company 2003).

#### **3.3.4.2 Sediment flux**

Prior to breaching the Condit Dam, the area immediately downstream was dewatered to allow equipment access to the dam to support removal efforts. The following species were caught during the fish salvage operation: sculpin (*Cottus sp.*; 87), rainbow trout (*O. mykiss*; 81), lamprey (ammocetes; 53), spring chinook salmon (26 adults and 1 smolt), and steelhead (*O. mykiss*; 20) for a total of 268 fish transported 40 m downstream. In the case of the Condit Dam removal, no effort was made to mitigate the initial flux of sediment from Northwestern Lake, though any barriers to fish passage that formed as a result were to be removed as needed. In addition, a minimum of 500 fall chinook salmon were to be captured and transported upstream of Condit Dam to avoid the expected loss of the entire spawning population and seed available spawning habitat upstream of the dam (PacifiCorp Energy 2011c). Overall, 679 fall chinook salmon and 10 pink salmon (*Oncorhynchus gorbuscha*) were transplanted

above the dam prior to breaching (Engle et al. 2013). River reaches upstream of the dam were surveyed for redds prior to dam removal with 191 being counted, though an estimated 45 of these were lost to reservoir erosion after breaching (Engle et al. 2013). Once Northwestern Lake had been drawn down, a plan was in place to grade and stabilize the remaining sediments and prepare the area for revegetation and the establishment of wetlands (PacifiCorp Energy 2011b). Stabilization and grading of exposed reservoir sediments was done by considering slope and the location and depth of bedrock that would constrain river migration and slope erosion. Slopes steeper than 30 degrees were assessed for stability and if necessary stabilized using heavy equipment, hydraulic excavation, blasting, or revegetation (PacifiCorp Energy 2011b).

Prior to breaching Marmot Dam, the potential for fish passage issues caused by channel aggradation from the release of the sediments impounded behind Marmot Dam was recognized. A fish passage monitoring plan was developed whereby the channel-bed complexity would be used as an indicator of future risk of fish passage issues. The concept was that a channel with low complexity (wide and shallow) would have a higher potential to impede fish passage compared to a complex channel (channel with defined bars and thalweg). Sites were selected for monitoring before and after dam removal and the standard deviation of bed elevation was the metric used to describe channel complexity (Stillwater Sciences 2002). Fish passage issues did not occur because, despite significant aggradation, the channel maintained a well-defined thalweg (Cui and Wooster 2014).

During dam removal on the Elwha River, all in-river fisheries would be stopped for a minimum 5 years and removal operations were curtailed periodically creating 'fish windows'. Fish windows were used to improve water quality conditions and were timed to occur when salmon were migrating or spawning (Ward et al. 2008). While three fish windows were planned annually, sustained high suspended sediment concentrations during year two of dam removal prompted cessation of further removal of Glines Canyon Dam for 10 months (Ward et al. 2008, Magirl et al. 2014).

Although natural recolonization of the Elwha River with native fish was the preferred restoration option, it was recognized that alternative strategies were likely needed to establish self-sustaining populations for some species. Prior to dam removal, some species were extirpated from the system, existed at a very low abundance, or natural stocks were intermingled with hatchery stocks. In addition, fish downstream of Glines Canyon Dam were expected to experience high mortality during dam removal due to elevated suspended sediment concentrations (Ward et al. 2008). Each species' pre-removal status and life history was used to create a restoration strategy that included natural recolonization and a combination of hatchery rearing and planting of eyed eggs, release of fry, pre-smolts, age-0, age-1, and age-2 smolts, and the creation of a reserve population using similar approaches in an adjacent watershed. At the same time, non-native hatchery fish production were to be reduced to limit the risk of genetic introgression to native fish (Ward et al. 2008). Accelerating the recovery of appropriate habitat and habitat-forming processes was identified as essential for fish recovery and was a high priority for the Elwha River. These efforts focussed on floodplains and included reforestation and restoring lateral connectivity to foster the natural input of large woody debris and instream flows. Throughout the recovery process (17 years including pre-removal, removal, and post-removal periods) a monitoring program will be used to assess the success of restoration activities allowing for adaptive

management of the recovery effort. Using this strategy, significant progress toward fish population restoration was expected within 20 to 30 years (Ward et al. 2008).

Municipal and industrial water intakes were located downstream of the Glines Canyon and Elwha dams on the Elwha River. It was expected that high suspended sediment concentrations associated with the erosion of reservoir sediments during and for a period after dam removal would impact water quality to these end users. To ensure acceptable water quality throughout the project, a water treatment plant was built with the capacity to process high turbidity water in addition to having a water intake weir that would exclude sediment (National Park Service 2005).

### **3.3.4.3    *Revegetation***

Revegetation of exposed sediments after removing Condit Dam was planned to provide additional slope stability and erosion control, and to establish appropriate upland and riparian biotic communities to an early successional stage within three to five years. Similarly, plans to revegetate temporary access roads and other disturbed areas were also made (PacifiCorp Energy 2011d, 2011e). Prior to breaching Condit Dam, existing vegetation communities in the area were surveyed to establish reference conditions by which subsequent revegetation efforts could be assessed. Approximately 11 hectares of exposed reservoir sediments were expected to require revegetation (PacifiCorp Energy 2011e). Sediments exposed after breaching the Condit Dam were assessed for fertility and stabilized or graded in preparation for revegetation (PacifiCorp Energy 2011b). To speed the natural revegetation process, a combination of seeding and bare-root tree plantings were planned. A revegetation seed mix was used to initially establish cover and reduce erosion in both riparian and upland areas. 300 tree saplings of four species appropriate for either riparian or upland areas were planted per acre and each sapling protected from browse and competition from weeds. In addition, locally sourced live willow stakes were planted at the high water mark on suitable river banks to encourage bank stabilization. A key objective of the revegetation strategy was preventing the establishment of new noxious weeds and limiting the occurrence of noxious weeds to levels no greater than reference conditions on nearby properties. Integrated Pest Management (IPM) and Best Management Practices (BMPs) were used to limit the emergence and spread of noxious weeds. Revegetated areas were monitored for three years to assess whether vegetation communities were trending toward project goals or whether intervention (additional plantings) would be required (PacifiCorp Energy 2011e).

After decommissioning Marmot Dam, a plan was in place to revegetate disturbed areas as well as re-contour and revegetate newly created riparian habitat associated with reservoir drawdown. Revegetation of native species was completed through a combination of natural recruitment from the existing seed bank, seeding, and planting of rooted shrubs and trees. The upland seed mix was selected for local conditions with hydroseeding being the preferred application method. Target planting densities for rooted trees and shrubs was 375 per acre. Riparian areas were expected to be unstable for a number of years and revegetation would be delayed until stream banks stabilized. To reduce the establishment of non-native species and noxious weeds, weed free seed mixes were used as part of a prevention and control strategy. In this case, the project proponent indicated that some noxious weeds were already widespread in the region and that control would be nearly impossible. The proponent proposed to prioritize control activities according to species present as a new population or an outlier to

a prior infestation, or if the species outcompetes or invades otherwise undisturbed native communities (FERC 2005). Revegetation of the Marmot Dam site was successful with no erosion problems, greater than 90% vegetation cover, and less than 20% non-native invasive species (Portland General Electric Company 2012). Newly created stream banks exposed during reservoir dewatering had stabilized within 2 years of dam removal and revegetation had begun naturally. In year 5 post-dam removal, stream banks were 90% revegetated to the high water mark through natural recruitment and no active revegetation was necessary. Though some noxious weeds were present, particularly in steep and inaccessible areas, noxious weed control treatments were successful in limiting the spread of existing weed populations and the establishment of new weed populations (Portland General Electric Company 2012).

The removal of Glines Canyon and Elwha dams on the Elwha River is expected to expose over 320 ha of sediments. Natural primary succession was expected to be slow due low nutrient levels, moisture availability, high wind erosion and distance of some areas from intact forests which otherwise would provide seeds, spores, and organic matter that would speed succession. After reservoir drawdown, sediments were left in terraces up to 7 m thick with steep slopes (Chenoweth 2014, Randle et al. 2015). Initially, these sediment terraces would be expected to erode, though the long-term objective was the establishment of native forest and stabilization. Thus active revegetation that included seeding (7,000 pounds) and planting 400,000 seedlings, trees, and live stakes was to occur during the 6 years after dam removal activities began (Chenoweth et al. 2011, Calimpong 2014, Chenoweth 2014). Seeding and planting was largely experimental in the first years of dam removal in an effort to identify the native species that had the greatest success in specific locations and soil types (Chenoweth et al. 2011, Chenoweth 2014). During the initial drawdown of Lake Mills and Lake Aldwell prior to dam removal, willow and cottonseeds delivered from upstream tributaries and main stem were deposited in bands on newly exposed sediments. A year into dam removal, dense bands of willow and cottonwood seedlings had grown where the seeds were deposited (Chenoweth 2014). Initial revegetation of actively seeded sites was successful and a study of 5 woody plants indicated that survivorship was typically greater than 90% (Calimpong 2014, Chenoweth 2014). Exposed reservoir sediments were expected to be susceptible to invasive species colonization. To combat their establishment, invasive species were surveyed and controlled, before (starting 9 years before dam removal), during, and after dam removal activities (Chenoweth et al. 2011).

#### **3.3.4.4 Wetlands**

Existing wetlands were delineated prior to breaching the Condit Dam to support the project goal of no net wetland loss (PacifiCorp Energy 2011e). It was determined that 1.9 hectares of primarily low function wetlands would be impacted by the drawdown of Northwestern Lake. The draining of Northwestern Lake was expected to expose numerous streams and seeps where wetlands would be allowed to establish naturally and no net loss of wetlands was expected. In addition, the initial sediment pulse and restoration of natural sediment flux was expected to create additional wetlands downstream of the dam, particularly at the mouth of White Salmon River (PacifiCorp Energy 2011e). In the event that a net loss of wetlands exists 5 years after breaching Condit Dam, wetlands will be created actively in the vicinity of the project area (PacifiCorp Energy 2011e).

## 4. Long term effects of dam removal

### 4.1 Loss of reservoir

#### 4.1.1 Physical effect

Removal of a dam necessarily results in a loss of much of the aquatic habitat previously occupied by the reservoir. A portion of the reservoir changes from lentic to lotic habitat with a river channel establishing itself within the former river valley, though not necessarily within its original, pre-dam channel (Macbroom 2011). The loss of a reservoir created by a large dam differs from those created by a small dam because large dams inundate upland soils that previously would never be flooded where small dams often only flood riparian areas (Lafrenz et al. 2013). In addition, a lowering of the water table is expected, the extent of which is dependent on local geology and surface water elevation changes (Berthelote 2013). After dam removal, new soil terraces may exist if erosion via channel incision and migration was incomplete. These terraces consist of previously well-drained soils that were inundated for a significant period of time and are covered by sediment deposited in the reservoir. Exposed sediments have a variety of grain sizes and chemical composition which varies within (Cavaliere and Homann 2012) and between reservoirs (Orr and Stanley 2006).

Three years after the removal of Marmot Dam, Lafrenz et al. (2013) studied the development of exposed reservoir soils by comparing their ripeness, a measure of soil development, to unaffected soils downstream of the former reservoir. Exposed reservoir sediments had physically ripened (dehydrated) to the depth of buried soil and had chemically ripened (changed from reducing to oxidizing conditions) to a depth of 16 cm. There was little evidence of biological ripening (equilibrium of soil nitrogen).

Prior to dam removal on the Elwha River, an experiment using Lake Mills sediments documented soil changes in small vegetation plots. Over 20 months, dewatered sediments had shifted from anaerobic to aerobic, had increased pH, improved soil aggregate stability, and had lower organic carbon and nitrogen levels, even in unvegetated plots (Cook et al. 2011). These changes appear to indicate a relatively rapid physical and chemical ripening with some biological ripening occurring.

On the Clark Fork River, dewatering of the Milltown Reservoir prior to dam removal was expected to lower the water table affecting some shallow groundwater supply wells (Berthelote et al. 2007). In addition, changes to the water table altered the groundwater flow direction (Pacific Western Technologies 2011).

#### 4.1.2 Ecological effect

The loss of a reservoir reduces or eliminates the amount of available or appropriate habitat for a variety of species. In some cases these species could be valued for recreational purposes as in the case of some fish (Baxter 1977) or for their regional importance as in the case of trumpeter swans' winter use of reservoirs on the Elwha River (National Park Service 1995). Conversely, the loss of a reservoir creates additional habitat for river species not to mention the potential watershed scale benefits of reconnecting upstream-downstream habitats (Hart et al. 2002). In the case of large dams, a significant portion of the river valley may be returned to a terrestrial ecosystem benefiting wildlife, particularly in

northern regions where river valleys may be the most productive landscape type (Nilsson and Berggren 2000).

Riparian areas existing along former reservoir margins would be exposed to upland conditions after reservoir drawdown and thus vegetation may be impacted or even killed if they are sensitive to changes in water table elevation (Shafroth et al. 2002a, Auble et al. 2007). The exposure of previously inundated soils and sediments after reservoir dewatering creates an abundance of new habitat for vegetation to colonize, though the fertility of exposed soils may be reduced (Orr and Stanley 2006, Cook et al. 2011, Michel et al. 2011, Lafrenz et al. 2013). Succession of newly exposed sediments would be expected to begin with weedy species with effective dispersal, high seed production, and high growth rates (Shafroth et al. 2002a). For tall dams, it is expected that much of the exposed area will be largely disconnected from the river and will become upland habitat (Auble et al. 2007). The initial exposure of wet sediments will favour the establishment of wetland species but rapid turnover of dominant species occurs as sediments become progressively drier after dewatering (Auble et al. 2007). In some cases establishment of invasive species may inhibit colonization by native species and succession of riparian areas (Shafroth et al. 2002a).

In Wisconsin, Orr and Stanley (2006) completed riparian vegetation surveys in 13 former reservoirs where dam removal had occurred from 1 to 47 years ago. Vegetation cover was high in all cases indicating that colonization was rapid. Species diversity and the number of trees increased with time since dam removal, though there was no relationship between time since removal and the abundance of other growth forms. In addition, an introduced grass (*Phalaris arundinacea*) dominated vegetation cover at a number of sites and there was no relationship between time since removal and the number and frequency of introduced species. These results indicate that beyond rapid revegetation and a general increase in species diversity and the abundance of trees, site specific conditions (e.g., soil fertility) control succession of riparian areas in dewatered reservoirs (Orr and Stanley 2006). Schmitz et al. (2009) studied a dam failure and a dam removal and found that the long term response of downstream vegetation was most heavily influenced by valley and reach geomorphology, though they indicate that the magnitude and timing of flood events, including that associated with breaching, was also important.

### **4.1.3 Mitigation**

In areas of where the risk of invasive species colonization is high, active revegetation is often used to guard against establishment of invasive, non-native, or undesirable species. Additional benefits of active revegetation include faster recovery and decreased erosion. However, active revegetation has been unsuccessful in some cases due to incomplete consideration of plant specific germination and seedling growth requirements, and grazing pressure (Shafroth et al. 2002a). Shafroth et al. (2002) discussed the potential to encourage valued plant species colonization in former reservoir riparian areas by manipulating the timing and pattern of reservoir drawdown to match germination and seedling requirements.

Prior to dam removal on the Elwha River, a variety of experiments were completed to assess the suitability of Lake Mills sediment for native plant growth. Cavaliere and Homann 's (2012) analysis of Lake Mills sediment-phosphorus chemistry indicated that sediment-phosphorus concentrations were



adequate to support long term riparian and mature forest phosphorus availability. However, Chenoweth et al. (2011) indicated that Lake Mills sediments had low overall fertility (primary nutrient concentrations) compared to forest soils in the area. Seed dispersal studies indicated that seed rain on minimally vegetated gravel bars was low compared to regional values (Michel et al. 2011) and that the viability of seeds trapped in Lake Mills sediments was low (Brown and Chenoweth 2008). A study investigating the role of birds in seed dispersal following dam removal indicated that birds can disperse almost 60% of native woody species and that most seeds are deposited on logs (as indicated by scat; McLaughlin 2013). These results imply that birds could be recruited to disperse seeds following dam removal through the placement of large woody debris (McLaughlin 2013). In greenhouse experiments, fine-grained reservoir sediments from Lake Mills reduced the germination and growth of four native species compared to typical alluvial sediments while a common invasive species was unaffected by the reservoir sediments (Michel et al. 2011). Small scale plot experiments examined the effect of mycorrhizal fungi spores and mulch on plant growth (Cook et al. 2011). After 20 months, even plants that grew in uninoculated plots had formed mycorrhizal associations indicating that natural colonization of mycorrhiza may be adequate, at least at small scales. The mulch treatment contained significantly greater vegetation cover and more naturally colonizing native species, likely due to greater moisture retention (Cook et al. 2011). Overall, the characterization of Lake Mills sediments, the seed dispersal conditions of Elwha River area, and the germination and growth of native and invasive seeds on Lake Mills sediments indicated that in addition to natural recolonization, some active revegetation and invasive species control would be necessary to ensure the successful revegetation of Lake Mills and Lake Aldwell reservoir sediments. These studies also served to inform active revegetation through identification of species and strategies that are likely to be successful in the conditions expected after dam removal.

The drawdown of the Milltown Reservoir and removal of Milltown Dam was expected to lower the water table in the Milltown area, potentially affecting the production of local water wells (Berthelote et al. 2007). This effect was modelled prior to drawdown and predictions were made about which wells would be affected (Berthelote and Woessner 2008). This allowed proactive mitigation with 101 wells receiving preemptive action (i.e., new deeper wells and lowering of pumps) to ensure water supply was not disrupted by groundwater level changes associated with reservoir drawdown (Berthelote and Woessner 2008).

## **4.2 Longitudinal Connectivity**

### **4.2.1 Physical effect**

The upstream-downstream connectivity of a river is fundamental to watershed functioning. It allows the unimpeded flow of water and transport of sediments which, along with valley characteristics (e.g., slope or confinement), define the downstream geomorphology of the river, riparian area, and floodplain. The removal a dam immediately restores connectivity to the river, restoring the natural water flow regime, though the river must digest impounded sediments before the natural sediment transport regime is restored. Many rivers a dammed multiple times such that the removal of a single dam may not restore connectivity to the entire upstream watershed.

### 4.2.2 Ecological effect

Dam removal restores the longitudinal connectivity required for fish (and some macroinvertebrates) to migrate and optimally complete their life histories. Dam removal can reestablish hydrochorous seed dispersal (dispersed by water) and gene flow between upstream and downstream populations. In some cases dams have been used as a management tool to restrict upstream access to undesirable species or populations.

One of the greatest effects of removing a dam is opening up fish habitat for migratory fish, particularly if the dam being removed didn't have effective fish passage facilities. Removal of Condit Dam on the White Salmon River and Elwha and Glines Canyon dams on the Elwha River had blocked anadromous fish from upper river and tributary reaches for almost 100 years. Within the first years of dam removal, anadromous fish have been observed upstream of former dam sites and spawning nests (redds) have been observed (Engle et al. 2013, Yakama Nation Fisheries Program 2014). Using a spawner-recruit model, Elwha River chinook, coho, chum, pink, and steelhead stocks were predicted to recover within 10 to 25 years depending on the species (Ward et al. 2008). No estimates were made for sockeye, bull trout, and cutthroat trout due to limited data (Ward et al. 2008). Rainbow trout and bull trout populations located above the Elwha River dams were also expected to resume downstream migrations and anadromous life histories upon dam removal (Brenkman et al. 2008b).

On the Sandy River, a fish passage structure at Marmot Dam had been used as a management tool to enumerate migrating salmon and restrict hatchery raised fish (chinook salmon) from access to the upper river in order to maintain the genetic diversity of the wild population. When Marmot Dam was removed, the habitat use (as indicated by the number of observed redds) increased upstream of the former dam, though it was believed that was largely an effect of more hatchery fish spawning naturally and potentially interbreeding with the wild population (Schroeder et al. 2013, Joint Columbia River Management Staff 2014). This necessitated the installation of weirs upstream of the former dam to restrict hatchery fish from accessing spawning tributaries (Luke Whitman, personal communication, January 28, 2015). On the Elwha River, genetic sampling prior to dam removal established the baseline genetic diversity of remnant wild populations, hatchery fish, and populations from adjacent streams that may be a source of recolonizing fish (Winans et al. 2008). These data will be used to monitor the genetics of fish during recolonization (Mchenry and Pess 2008, Winans et al. 2008).

On the Kennebec River, removal of the Edwards dam allowed Atlantic salmon (*Salmo salar*) and gaspereau (collectively alewife, *Alosa pseudoharengus*; and blueback herring, *Alosa aestivalis*), American Shad (*Alosa sapidissima*), Atlantic sturgeon (*Acipenser oxyrinchus*), shortnosed sturgeon (*Acipenser brevirostrum*), striped bass, American eel (*Anguilla rostrata*), and sea lamprey unfettered access to 27 km of main stem habitat for the first time in 160 years (Main State Planning Office 2008).

The reintroduction of anadromous fish after dam removal has a number of potential effects on species interactions. For example, dam removal on the Elwha River will expose rainbow and bull trout populations to competition from anadromous species for the first time in almost 100 years (Brenkman et al. 2008b). Brenkman et al. (2008) predicted that the location of greatest competition would be in the middle reaches of the river because the greatest overlap of species occurs there, lower reaches would

not be fully recolonized by upstream species, availability of refugia, and barriers limiting the upstream migration of some colonizing species as well as resident species. In addition, recolonization of the upper river could expose resident and anadromous fish populations to pathogens with uncertain results (Brenkman et al. 2008a).

The reintroduction and expected recovery of anadromous fish returns to rivers that were previously dammed is expected to reestablish the supply of marine derived nutrients (MDN) to aquatic and riparian habitats, particularly for dams that didn't have fish passage structures (Gregory et al. 2002, Duda et al. 2010). Marine derived nutrients from anadromous fish can increase periphyton production (Wipfli et al. 1998), macroinvertebrate growth rates and biomass (Minakawa et al. 2002), and fish growth rates (Bilby et al. 1996). Predators, scavengers, and flooding also move salmon carcasses and their nutrients, thereby fertilizing riparian and upland terrestrial habitats (Ben-David et al. 1998). On the Elwha River, projected salmon population recovery could increase annual nitrogen and phosphorus nutrient loading by 1,275–10,900 kg and 210–1,350 kg, respectively (Duda et al. 2010). The distribution of recolonizing fish and thus the marine nutrients they deliver are expected to vary spatially (e.g., differences in migration capability among species) and temporally (e.g., differences in run timing; Brenkman et al. 2008, Duda et al. 2011b).

Brown and Chenoweth (2008) found that Glines Canyon Dam on the Elwha River reduced the abundance of hydrochorous seed transported downstream by 90% with an 84% decrease in species richness. Removal of the Elwha River dams could increase riparian downstream species diversity. However, the seeds of invasive plants were found in Lake Mills samples indicating that dam removal could increase dispersal of invasive species (Brown and Chenoweth 2008).

No more turbine mortality, fall back, delayed migration (upstream/downstream)

Juvenile spring Chinook salmon bypass mortality at Marmot dam estimated to be 27% (Portland General Electric Company 2003).

## **4.3 Flow regime**

### **4.3.1 Physical effect**

Dam type and operation strongly influence how dam removal would affect flow regime downstream of the dam site (Poff and Hart 2002). Dams operated as run of the river have a smaller influence on downstream flow regime compared to storage dams. However, there remains a flattening of the hydrograph (lower peaks and higher lows) even with run of the river dams (Graf 2006). For example, the Elwha and Glines Canyon dams on the Elwha River have been largely operated as run of the river dams since 1975 with reservoir inflows equaling dam discharge and downstream river flows approximating a natural river ecosystem. Even so, Duda et al. (2011) indicated that low and moderate daily flows would remain similar post-dam removal but peak flows (100 year flood) could increase by as much as 10 to 15%.

Increases in flow were expected to occur after the removal of Marmot Dam because water would no longer be diverted from Sandy River for hydroelectric production in a neighbouring watershed.

However, since high flows were spilled over Marmot Dam and no water was diverted when flows were less than minimum requirements, the greatest effect would be during intermediate flows (Portland General Electric Company 2003).

Increases and changes in the timing of peak flows after the removal of a dam operated to store water (for controlled release at a later time) would be expected to be much greater (Hart and Poff 2002). To our knowledge, no dam with a large water storage capacity has undergone removal. However, the effect is obvious. In the case of the Colorado River at the Glen Canyon Dam peak flows are, by law, not to exceed 700 m<sup>3</sup>/s; a fraction of annual peak flows that were historically greater than 2400 m<sup>3</sup>/s (Collier et al. 1996). For 29 dams studied by Williams and Wolman (1984) annual peak flows were a mean 39% lower than pre-dam conditions. For 36 large American dams (reservoir capacity a minimum 1.2 km<sup>3</sup> of water), Graf (2006) found that downstream annual peak flows were reduced by a mean 67%.

Overall, the effect of dam removal results in an increase in peak flows and flow variability over annual and decadal time-scales and in the case of dams providing peak electricity demands, stabilization of flow over hourly and daily time-scales (Williams and Wolman 1984). In cases where the river is affected by a single dam, dam removal will re-establish a natural flow regime immediately. However, in many cases rivers are dammed multiple times (Graf 1999) or flows are affected by water diversions or withdrawals causing an incomplete return to natural flows.

The long-term geomorphic response of individual river ecosystems to a restored natural flow regime after dam removal is difficult to predict precisely. While a few large dams have now been removed (Marmot, Condit, Elwha, Glines Canyon), a steady state within each ecosystem is not likely to have been reached. Graf (2006) described the downstream geomorphic response of rivers to hydrologic changes associated with the construction and operation of 36 large dams. Compared to unregulated reaches, the elevated low flows associated with large dam operation resulted in larger low flow channels. Decreased high flows caused high flow channels to be 50% smaller and numerically reduced by 77%. The effect of these hydrologic and geomorphic changes on floodplains is an overall simplification of the landscape with 72% less active floodplain area and 37% less complexity compared with unregulated rivers. The removal of a large dam would immediately restore a natural flow regime (assuming the reach is not impacted by another structure upstream) with its associated higher peak flows and typically lower low flows. The larger low flow channels created during the period of regulation would be expected to partially limit the advance of peak flow waters to high water channels and the floodplain. However, the reestablishment of sediment supply would be expected to aggrade low flow channels and build low bars, which should effectively raise the river water level, allowing water to advance into high water channels and the floodplain during peak flows. In this way, the reestablishment of a natural flow regime along with restored sediment supply would be expected to increase the active area and complexity of river channels and floodplains indicating the restoration of geomorphology downstream of a large dam.

### **4.3.2 Ecological effect**

No large dam with a large water storage capacity and large effect on flow regime has been removed. Thus, it is unclear what the long term ecological effects of large dam removal and natural flow restoration will be on the downstream community (Stanley and Doyle 2003). However, the current

understanding of the effect of hydrology on aquatic communities indicates that the restoration of a natural flow regime would tend to restore natural aquatic communities (Poff et al. 1997). Similarly, the hydrological effects on downstream geomorphology indicate that a natural flow regime with its greater variation in flows will result in greater channel and floodplain complexity and connectivity (Graf 2006). The ecological effect of shifting from a regulated river with low disturbance on seasonal, annual, and decadal timescales and potentially very high disturbance on daily timescales to an unregulated, intermediately disturbed environment would theoretically be a switch to a more diverse ecosystem as predicted by the intermediate disturbance hypothesis (Connell et al. 1978, Ward and Stanford 1983). Moreover, the greater channel and floodplain complexity implies a greater habitat heterogeneity which is also predicted to increase diversity (MacArthur and MacArthur 2014). Finally, the restoration of temporal patterns in flow regime benefits native species with life histories that depend on a varied flow regime and similarly, limit the success and proliferation of invasive species (Bunn and Arthington 2002).

These theoretical conclusions appear to be borne out, generally, for fish (Guégan et al. 1998), macroinvertebrates (Townsend et al. 1997), riparian vegetation (Nilsson and Svedmark 2002), and river ecosystems as a whole (Junk et al. 1989). Moreover, evidence of community simplification associated specifically with flow regulation (Power et al. 1996, Hughes and Parmalee 1999, Jansson et al. 2000, Bunn and Arthington 2002) bolsters the hypothesis that restoring natural flows would lead to greater community diversity.

Alteration of the natural flow regime by large dams causes the simplification of downstream river reaches impacting aquatic organisms, vegetation communities, and wildlife (Bunn and Arthington 2002, Graf 2006). If reestablishment of a natural flow regime (along with sediment supply) can allow the geomorphological recovery of channels, floodplains, and riparian areas, the negative effects on aquatic organisms, vegetation, and wildlife would be expected to be reversed (though see Hilderbrand et al. 2005) or at least alleviated. What remains unclear is the effect that a prolonged period of flow regulation will have on the return of downstream ecosystems to pre-dam forms and processes. For example, Shafroth et al. (2002b) hypothesized that the observed increase in density of woody vegetation in riparian areas associated with decreased flooding could hamper recovery if natural flows were restored because denser riparian vegetation would be more resistant to erosion, decrease flow velocities, and increase sedimentation potentially reducing the effects of future disturbance. Thus, it is unclear whether biotic communities would eventually be restored to pre-dam condition or if an alternative stable state (Scheffer et al. 2001, Beisner et al. 2003) would develop, or another intermediately degraded form would persist without further intervention (Hilderbrand et al. 2005).

### **4.3.1 Mitigation**

Mitigation for changes in flow regime are highly case specific. Increased peak flows along with restored sediment supply and associated channel aggradation (see Section 4.3) after the removal of a large dam is likely to increase flood frequency, extent, and duration. This, coupled with the general trend of infrastructure encroaching on floodplains during periods of flow regulation due to the abatement of flooding, leads to conflicts between the objectives of river restoration and protection of property. This could lead to private (or public) actions to control flooding (e.g., levees) or channel migration (e.g., riprap) which could compromise or otherwise limit the extent of the overall restoration.

## 4.4 Temperature

### 4.4.1 Physical effect

The restoration of upstream-downstream connectivity and a natural flow regime have an effect on river water temperature in the downstream environment that can vary between seasons and because of case specific circumstances (Olden and Naiman 2010). Case specific circumstances include former reservoir depth and water retention time, and regional climate (i.e., did the former reservoir thermally stratify?), and dam operation (top, bottom, or selective withdrawal). Thus, all directions of temperature change could theoretically occur after dam removal. However, we are only aware of top draw dams that have been removed and there is general consensus about the response of the downstream environment in these cases. Typically, studies focus on summertime temperatures as they affect fish species both in terms of temperature-range and dissolved oxygen requirements. The removal of a top draw dam is typically expected to cause a decrease in summertime water temperatures due to the increased minimum, intermediate, or peak flows that occur after dam removal (see Section 4.1) and because water is no longer slowed while moving through the reservoir where it would be exposed to heating.

The removal of Condit Dam on the White Salmon River was expected to cause a small decrease in downstream water temperatures due to a decreased water retention time (exposure to heating; Washington State Department of Ecology 2007). Similarly, the removal of Marmot Dam on the Sandy River was expected to improve downstream water quality through increased flows (water would no longer be diverted from Sandy River), and decreased water retention time that would result in lower summer water temperatures and higher dissolved oxygen levels (Portland General Electric Company 2003). Based on monitoring small dam removal, Stewart and Grant (2005) predicted that coarse sediment deposition downstream of Marmot Dam may also increase hyporheic exchange, thereby buffering temperature changes during the summer months and improving temperature and dissolved oxygen conditions.

On the Elwha River, removal of the Glines Canyon and Elwha dams was expected to result in a 2 to 4 °C decrease in summer water temperature downstream of the former dams because warmer surface water would no longer be discharged over the dam (National Park Service 1995).

A modelling study of the temperature effects of a series of 4 large dams on the Klamath River indicated the dams caused a small increase in mean temperatures on an annual basis, though day-to-day temperature variability and maximum annual temperatures were much reduced. The most intriguing finding of this study was the 18 day delay in seasonal temperature progression compared to unregulated conditions (Bartholow et al. 2004).

The removal of a bottom draw dam would be expected to cause an increase in downstream summer water temperatures because cooler water is drawn from the hypolimnion (Olden and Naiman 2010). In Australia, bottom draw dams in the Murray-Darling Basin cause downstream temperature decreases that range from 5 to 12 °C immediately below the dam with depressed water temperatures persisting for hundreds of kilometers downstream (Preece and Jones 2002, Olden and Naiman 2010).

The removal of a dam with selective withdrawal may not cause any downstream temperature change because water can be withdrawn from multiple depths allowing active management of downstream water temperatures. At the Hungry Horse Dam in Montana, near 4 °C water was released to the Flathead River year round causing river water temperatures to be up to 8 °C cooler than normal that were detectable 64 km downstream (Marotz et al. 1994). The dam was retrofitted with a selective withdrawal system in 1994 to reduce temperature effects on downstream biota, particularly bulltrout and cutthroat trout (Marotz et al. 1994, Vermeyen 2006). Target temperatures ( $\pm 2$  °C) were generally attained during selective withdrawal operations. Temperatures remained higher than normal during winter operation when the selective withdrawal system was not used and lower than normal during the first two weeks of June when reservoir waters were too cold to achieve target downstream temperatures (Vermeyen 2006). In addition, downstream water temperatures were managed to be slightly cooler than normal during the warmest summer months to improve trout growth conditions (Vermeyen 2006). The improvement of downstream temperature conditions was expected to increase fish growth by a factor of 2 to 5 (Marotz et al. 1994).

#### **4.4.2 Ecological effect**

On the White Salmon River, removal of the Condit dam was expected to lower summer river water temperatures that was expected to be a direct positive for salmonids returning to White Salmon River as well as providing thermal refuge from the Columbia River for other salmonid populations, including sockeye, coho, steelhead, bull trout, and cutthroat trout (Washington State Department of Ecology 2007).

Prior to dam removal, summer water temperature downstream of Marmot Dam exceeded the Oregon temperature standard for salmonid rearing (Portland General Electric Company 2003). In addition, there were concerns that warmer water temperatures were a thermal barrier to salmonid migration where the Sandy River enters the cooler Bull Run River. The removal of Marmot Dam was expected to improve temperature and dissolved oxygen conditions for salmonids, though no estimate of the magnitude of temperature change was given or whether the hypothesized thermal barrier would be eliminated (Portland General Electric Company 2003).

On the Elwha River, warmer temperatures downstream of the Glines Canyon and Elwha dams were linked to the occurrence of *Dermocystidium salmonis*, a disease that had caused increased mortality in chinook salmon. Lower river water temperatures were also expected to increase reproductive success of salmonids downstream of the dams (National Park Service 1995).

On the Klamath River, a modelling study estimated an 18 day delay in natural temperature progression exposes migrating and spawning fish to much higher temperatures during the fall season and cooler temperatures during spring and early summer which could affect rearing juvenile growth rates (Bartholow et al. 2004). However, peak run timing of Klamath River fall chinook occur about 2 weeks later compared to pre-dam conditions indicating that some fish species change their behavior due to river regulation with a variety of potential consequences. Overall, the temperature effects of a series of dams on the Klamath River may negatively affect adult salmon migration and spawning and may have a positive effect on juvenile rearing and migration (Bartholow et al. 2004).

## 4.5 Sediment transport

### 4.5.1 Physical effect

The initial sediment pulse after dam removal shifts the downstream environment from a sediment starved state (pre-dam), to sediment transport capacity limited state. General characteristics and response of this change include elevated suspended sediment concentrations, reduced channel armouring, aggradation in some areas, a general fining of substrates, creation of new bars, decreased river bed stability, and increased channel mobility in unconstrained valleys (Major et al. 2012, Magirl et al. 2014, Wilcox et al. 2014, East et al. 2015). As reservoir sediments are flushed through the downstream environment, the river begins to shift to a sediment supply limited state and quasi-equilibrium state achieved (Pizzuto 2002, Konrad 2009). The time required for downstream geomorphology to reach a quasi-equilibrium state after dam removal is not well defined but is typically and vaguely estimated as several years to decades (Pizzuto 2002).

The transport of sediment should be broken down to coarse (sand, gravel, and cobble) and fine (silt and clay) sediments because these fractions have different functional roles in the river ecosystem. Coarse sediments are considered to be a structural element because they are required to form and maintain the channel bed, bars, riffles, and often banks (Kondolf et al. 2014). Fine sediments are much easier for rivers to transport and they affect turbidity, and nutrient and contaminant transport (Kondolf et al. 2014), though anthropogenically elevated levels often result in habitat degradation (Henley et al. 2000). The importance of re-establishing the supply of large woody debris to downstream environments must also be recognized (Abbe and Montgomery 1996). The differing behaviour of coarse and fine grained sediments suggests that the amount and composition of the sediment load coupled with the modifying effect of large woody debris will have a strong effect on downstream geomorphic response as has been observed with small dam removal (Kibler et al. 2011).

Since, few large dams have been removed, and where they have been, study efforts focus on short term effects, the range of downstream geomorphological response to re-established sediment supply is unclear. The effect of sediment impoundment by large dams on downstream geomorphology is relatively well described (Collier et al. 1996, Kondolf 1997, Graf 2006). The release of 'clear water' from the reservoir causes channel incision with coarse and fine grained sediments being transported downstream without being replaced (Kondolf 1997). The result is an armoured channel no longer has the required sediments to build and maintain a natural river geomorphology (e.g., gravel beds, bars, islands, and floodplain). For example, Graf's (2006) study of 36 large dams found that islands downstream of large dams were 75% smaller because of sediment starvation, compared with unregulated reaches. Similarly, the lack of sediment to form and maintain channel bars resulted in 68% fewer low bars which were 52% smaller, and 52% fewer high bars compared to unregulated reaches. Initially, dam removal would reverse this situation by resetting the downstream environment with a large sediment pulse that is expected to increase channel mobility and instability to a level greater than pre-dam (Major et al. 2012, Wilcox et al. 2014, East et al. 2015). Ostensibly, the restoration of natural flow and sediment transport regimes should return geomorphological processes and functioning in the downstream environment to pre-dam conditions in the long term. Limits to restoration of



geomorphological processes and functioning include, but are not limited to, channel stabilization (riprapping) and flood control structures (levees). During the life of the dam upland vegetation may establish in floodplain and riparian areas with an unknown effect on restoration trajectory after dam removal (Shafroth et al. 2002b).

On the Elwha River, the expected and observed aggradation of channels downstream of the Glines Canyon and Elwha dams increased river water levels which were expected to increase the water table elevation by 0.6 to 1.5 m (National Park Service 2005, East et al. 2015). Elevation of the water table was expected to render some residents' septic systems ineffective (National Park Service 2005). In addition, increased river water levels were expected to increase flood frequency, potentially contaminating wells that had been drilled in the floodplain (National Park Service 2005).

Coastal areas deprived of sediments due to river impoundment may have retreating river deltas and eroding beaches and shorelines as was the case of the Elwha River Delta (Warrick et al. 2009). Restoring sediment transport may help restore coastal ecosystems, or at least halt erosion. On the Elwha River, the delivery of large volumes of sediment to the river mouth caused the river channel to change from a single thread to multiple channels. Two years into dam removal on the Elwha River, the Elwha Delta progressed 200 m seaward, though it remains to be seen whether these geomorphological changes will persist (Gelfenbaum et al. 2015).

#### **4.5.2 Ecological effect**

Long-term studies of the ecological response of river ecosystems to restored sediment supply after the removal of a large dam have not been conducted, or if underway have not been completed. In addition, the complexity of ecosystem response to large dam removal is expected to increase with river size making predictions of specific changes increasingly difficult. Combined with the restoration of natural flows, restored sediment supply is expected to restore the physical processes that build and maintain river ecosystems (Pizzuto 2002). Since local species are adapted to the natural flow and sediment transport regime, and the habitat that these processes help to create and maintain, it is expected that biotic communities will respond positively to dam removal. For example, the resupply of spawning sized gravel has been identified as an important process for the maintenance of fish spawning areas (Power et al. 1996). Channel aggradation is expected to raise the water table and increase the frequency of overbank flows increasing water availability and sediment delivery to riparian areas. On the Elwha River, restoring sediment transport was expected to deliver gravel and large woody debris downstream of the dams, increasing habitat complexity and floodplain connection that would increase fish habitat availability (Gelfenbaum et al. 2011). Preliminary results indicate that habitat complexity in the Elwha River and estuary has increased, though its stability and ecological effect have yet to be determined (East et al. 2015, Gelfenbaum et al. 2015).

#### **4.5.3 Mitigation**

Since the sediment supply to the lower Elwha River had been effectively nil since dam commissioning, the lower river bed had become degraded, reducing flood risk for some areas over time (National Park Service 2005). Dam removal was expected, and did, cause aggradation of the river bed in the lower river, effectively increasing river water levels and decreasing the effectiveness of existing flood control

structures (National Park Service 2005, East et al. 2015). To mitigate this effect, an existing flood control structure was raised, strengthened, and extended to protect infrastructure at either existing or greater than flood protection levels, though it was recognized that this action would represent a minor adverse impact to the floodplain over the long term (National Park Service 2005). In addition, increased river water levels were expected to increase the elevation of the water table, disrupting septic systems in the area (National Park Service 2005), and in this regard, a community wastewater treatment facility was built (Nicole 2012).

## **4.6 Ice Regime**

### **4.6.1 Physical effect**

In northern climates, dam removal is expected to restore the natural ice regime during winter, though differences in local conditions will create different ice effects. The natural ice regime has major effects on winter flows, inundation of backwaters or riparian areas, dissolved oxygen concentrations, sediment transport/deposition, and river geomorphology. In this context, these subjects are beyond the scope of this report, but see Prowse (2001b) for a review. Here we present a few case studies where the effect of dam removal on the ice regime was studied either before or after dam removal.

Where dams are located at a steep section, frazzle ice is expected to form and accumulate downstream resulting in thicker ice cover that may cause ice jams. In addition, ice jams that may have formed at the upstream end of the reservoir may be transported downstream and form at a different location. Ice movement and jamming in the former reservoir would be expected to increase scour and mobilize large amounts of sediment (White and Moore 2002), at least in the short-term.

On the Israel River near Lancaster, NH, four small dams were damaged during high flows and were effectively eliminated by 1950. During winter, the dams no longer retained frazzle ice that was transported downstream where it accumulated near Lancaster creating ice thicknesses up to 2.1 m. During ice break-up, the upstream dams no longer retained broken ice that flowed downstream and accumulated against the thick ice located near Lancaster where ice jams were initiated, spread upstream, and often caused flooding. In the 70 years prior to dam failure, Lancaster experienced 2 floods attributed to ice jamming, and in 30 years after dam failure Lancaster experienced 15 floods attributed to ice jamming (White and Moore 2002).

Prior to removing the Edwards Dam on the Kennebec River near Augusta, ME, ice jams occurred approximately 4 km downstream and occasionally caused flooding. After dam removal, ice jams formed during freeze-up 150 m downstream of the former dam site, an area that remained ice-free during dam operation. Ice jams forming in this new location have not caused major flooding (White and Moore 2002).

In preparation for removal of the Ballville Dam on the Sandusky River, OH, the effects of dam removal on ice jamming was modelled to assess whether Freemont, located downstream of the dam, was likely to experience increased flooding. The results indicated that Freemont would experience increases in the

frequency and severity of flooding if the Ballville Dam were removed without mitigation (Carr et al. 2011).

#### **4.6.2 Ecological effect**

The ecological effect of changes to the ice regime after dam removal has received little attention in the growing dam removal movement. Where the ice regime has been considered, the focus has been on changes to ice jamming and subsequent flooding (White and Moore 2002, Carr et al. 2011) or bridge scour (Tuthill et al. 2009). These studies make no mention of the ecological effects that a return to a natural ice regime may have after dam removal. The seasonal river-ice regime consists of freeze-up, main winter, and break-up and can exert considerable influence over aquatic organisms, particularly the distribution and movements of fish and benthic invertebrates and the distribution of aquatic and riparian vegetation (Prowse 2001a). Seasonal evolution of river-ice, particularly break-up and ice jamming, represent disturbance to aquatic and riparian habitat and the organisms that use them. Prowse (2001a) reasoned that if flow regulation eliminated dynamic breakups (high level of disturbance) and only thermal breakups (low level of disturbance) occurred that biotic diversity would decrease. In this case, dam removal should restore the natural ice regime and the resulting increase in ice related disturbance would be expected to increase biotic diversity.

#### **4.6.3 Mitigation**

In some cases, the change to a natural ice regime has caused increases in ice jam associated flooding or increased risk to infrastructure (e.g., bridges). The return to a natural ice regime has been mitigated through building ice control structures. On the Israel River, a 2.7 m weir with four sluice ways was constructed to control ice. This structure was not sufficient to reduce the frequency of ice jamming downstream, though it reduced ice jam severity and the severity of associated flooding (White and Moore 2002). On the Sandusky River, an ice control structure upstream of the Ballville Dam would reduce the increase in flood stage associated with ice jamming caused by dam removal to 15 cm for the 5 year return period event compared with a 1.5 to 3.4 m increase in flood stage if the dam were removed without an ice control structure (Carr et al. 2011).

### **5. Removal technique and schedule**

The removal technique and the removal schedule are critical determinants of the effect of dam removal because they largely control the erosion of sediment from the reservoir. The removal technique for dams that impound contaminated sediments must focus on limiting the downstream transport of those sediments. In the case of Milltown Dam on the Clark Fork River, this was accomplished by excavating the most contaminated sediments from the reservoir and disposing of them offsite. Other contaminated sediments were stabilized in the floodplain. Some lesser contaminated sediments were allowed to be eroded from the reservoir and transported downstream. An important consideration when stabilizing contaminated sediments in the floodplain is the geomorphic variability of the river and the probability that the river will erode structures meant to stabilize contaminated sediments. A historical analysis indicated that the Clark Fork River had a geomorphology that was transitional between wandering channel and braided channel. Despite this, a single thread meandering channel was constructed (with

contaminated sediments in the floodplain) on the Clark Fork River. Soon after project completion, the constructed channel was damaged by flooding, illustrating the perils of constructing a channel that does not fit within the historical geomorphic variability of the river (Woelfle-Erskine et al. 2012).

The Condit and Marmot dam removals were both planned to rapidly release the sediments retained by each dam. However, the release of water and sediments through a hole in the base of the Condit dam resulted in a much more rapid release of water and sediments compared to the Marmot Dam removal where the top of an earthen coffer dam was notched and the released water eroded the structure allowing more water and sediments to be transported downstream. These approaches are used for their cost effectiveness (no cost of sediment removal) with a trade-off between greater initial disturbance associated with the sediment release and the time until the recovery process can begin.

The incremental removal approach is practical for large dams impounding volumes of water and sediment that would be unsafe to release all at once. Essentially, incremental removal controls the release of water and sediments to a level that is acceptable for stakeholders and avoids the total destruction of downstream biotic communities. In fact, dam removal could be drawn out over a long period such that there were few downstream effects, but for large dams that would likely take an extraordinarily long time over which biotic communities would not get the benefits of a free flowing river. Thus, there remains a trade-off between the magnitude of downstream disturbance and the time to restore the connectivity of the river. The Elwha and Glines Canyon dams on the Elwha River were removed concurrently and incrementally over three years. The removal schedule included time periods when dam removal activities ceased, allowing for lower suspended sediment concentrations during ecologically important events such as fish migration and spawning (Ward et al. 2008). The removal schedule was also adaptively managed. For example, dam removal ceased when a downstream water treatment plant required repairs.

The timing of dam removal can be an important consideration that affects how erosion of the reservoir may proceed as well as the physical and ecological effects downstream. Condit Dam was removed in during the fall after chinook, coho, and steelhead runs in the Columbia River were finished (Washington State Department of Ecology 2010). This timing also allowed the translocation of chinook spawners upstream of Condit Dam and thus avoiding the loss of an entire year class of fish (PacifiCorp Energy 2011c). Marmot Dam was removed in the fall after the chinook migration and breaching occurred during a rainstorm to ensure flows were high enough to mobilize reservoir sediments and decrease the risk of creating fish passage problems (Major et al. 2012). The timing of construction activities during the removal of Milltown Dam and the associated contaminated sediments was cited as a major factor in limiting adverse ecological effects (U.S. EPA 2004).

## **6. Specific Considerations for the Mactaquac Project**

### **6.1 Scale**

The scale of the dam to be removed, reservoir to be drained, and the river on which it is removed is important because a river's ecological response becomes progressively more difficult to predict as the

size of the dam and the size of the river increases (Graf 2003). The largest dam removal project in the world was completed in 2014 on the Elwha River, though physical and biological monitoring continues (Warrick et al. 2015). The project included the removal of the 33 m Elwha dam and the 64 m Glines Canyon Dam which impounded a combined 21 million m<sup>3</sup> of sediment (Warrick et al. 2015). While the 55 m MGS is not as tall as Glines Canyon Dam, it impounds approximately 25 times the water and its headpond covers an area 50 times greater than Mills Lake, the reservoir impounded by Glines Canyon Dam. Moreover, the SJR has a mean discharge that is 15 times that of the Elwha River. Although estimates of the impounded sediment volume are still forthcoming, overall, if MGS is removed it would be the world's largest dam removal project to date.

## 6.2 Location

The characteristics of the region (e.g., climate) and watershed (e.g., geology, morphology) are equally as important as the scale of the dam. There are many lessons to be learned from previous dam removals, however, these lessons must be examined in the context of the local and regional location in which they were learned. Thus, lessons from other dam removal projects are not necessarily transferrable (at least not directly) to the SJR. This section notes some of the key local conditions that may affect a dam removal scenario on the SJR and compares and contrasts how these conditions may differ from other dam removal projects.

Although MGS has a large storage capacity (estimated as 1.3 km<sup>3</sup>), precipitation and run-off in the region is high such that the residence time is relatively short (approximately 3 weeks on average) compared to large dams located in arid regions (see Graf 2006). A historical analysis of SJR flow indicated that the construction of MGS did not cause a substantive change in peak flows, and no additional change in low flows occurred because low flows had already been affected by the prior construction of the Beechwood Dam upstream (Kidd et al. 2010). However, rapid changes in water levels are known to occur downstream of MGS due to fluctuating regional energy demands and it is possible that these changes stress aquatic biota through changes in habitat availability and temperature effects, ultimately altering biodiversity and biological functions (Kidd et al. 2010). In addition, the sediment load of Atlantic Canadian rivers such as the SJR is low compared to western regions (Stichling 1974) due to the relatively shallower gradient, more mature surficial geology, and established vegetation. Also, in the SJR system the Beechwood Dam retains much of the mainstem SJR bedload. As such, the downstream geomorphological impacts of MGS are expected to be less severe because the dam has a limited control over downstream flows and sediment supply which has allowed the lower river to maintain much of its pre-dam characteristics.

## 6.3 Fish passage

Fish passage is often cited as a major reason for dam removal. A summary of potential fish passage considerations is presented in the Fish Passage in Large Rivers: A Literature Review report (Linnansaari et al. 2015a). Specific considerations for the Mactaquac Project, resulting from an expert workshop, are provided in the Proceedings of the Fish Passage Expert Workshop: Global Views and Preliminary Considerations for Mactaquac (Linnansaari et al. 2015b).

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