# REVEGETATION OF POST-DAM-REMOVAL RIPARIAN SEDIMENTS IN THE LOWER ELWHA RIVER, WA

by

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ABSTRACT

Revegetation of Post-Dam-Removal Riparian Sediments in the Lower Elwha River, WA

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With the deconstruction of Glines Canyon Dam nearly complete, Olympic National Park's Elwha Revegetation Crew has implemented the second season of a 7-year native riparian plant restoration effort. A critical component of restoring the lower Elwha River ecosystem is the establishment of late seral riparian forests to provide stream shading, woody debris, nutrient inputs and erosion control. As a restoration strategy, installing woody plants may prevent invasive weed colonization, facilitate native plant recruitment, and provide seed sources beyond intact forest edge. The greatest challenge facing native plant establishment in the dewatered Lake Mills reservoir is the survival of vegetation in post-dam-removal sediments. Fine sediments (alluvial silt and clay) lack porosity, are subject to inundation and desiccation during wet and dry periods, and create hypoxic growth conditions. Coarse sediments (gravel, cobbles, sand) are highly porous and prone to desiccation and wind erosion. Determining which plant species can survive in specific sediment textures may provide guidelines for successful plant establishment as Elwha River post-dam-removal plant restoration progresses.

This study focuses on the survival and performance of six native woody plant species in three post-dam-removal sediment textures on the dewatered Lake Mills reservoir during the initial 2012 revegetation efforts. Selected species included ocean spray (*Holodiscus discolor*), Nootka rose (*Rosa nutkana*), thimbleberry (*Rubus parviflorus*), western redcedar (*Thuja plicata*), black cottonwood (*Populus balsamifera* ssp. *trichocarpa*), and Douglas-fir (*Pseudotsuga menziesii*). Total individuals surveyed equaled 860 (n=80-180 samples per species). Sediment moisture, particle size and nutrient content were also assessed. By late September plant mortality reached a total of 66, with 82% of dead individuals being Douglas-fir. The lowest mortality (<1%) occurred for black cottonwood. Proportionally, plant mortality was highest in the coarse sediment. These and similar data collected over the next 5 years may be useful in determining woody plant species selection in other post-dam-removal restoration efforts.

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# I. Ecology of Dam Removal

## 1.1 Impacts of Dams and Benefits of Dam Removal

### *Impacts of Dams*

In the United States and worldwide, dams influence the majority of river systems. Since the founding of the U.S., more than 965,000 of over 5.6 million kilometers (km) of U.S. rivers have been impounded, with only 2% of U.S. rivers remaining uninfluenced by dams (Grant and Parks, 2009). Put in perspective, roughly 80,000 dams have been constructed in the U.S., nearly the equivalent of building one dam per day from the time of U.S. establishment to the early 21$^{st}$ century (Grant and Parks, 2009). Worldwide, virtually 80% of the upper one third of all large river discharge has been at least partially obstructed in order to meet human needs (Bednarek, 2001).

Overwhelmingly, the influences of dams on river systems are negative. Dams impact riverine ecosystems in a variety of ways: serving as barriers to anadromous fish passage, impeding seed delivery to downstream riparian reaches, influencing channel dynamics and natural flood events, and limiting freshwater nutrient inputs to estuaries and other marine systems (Brown and Chenoweth, 2008; Downs et al., 2009; Nilsson et al., 2010). In some cases, hydropower benefits of dams which reduce human dependence on polluting energy sources are thought to outweigh negative influences of dams. But in many cases, dam construction may produce high ecological, economic and social costs without the intended benefits coming to fruition. Water shortages, for example, are becoming more prevalent and widespread despite the potential for reservoirs to store water resources and avoid such shortages (Stanley and Doyle, 2003). Sedimentation of dam reservoirs poses eventual public safety hazards in the absence of costly dredging.

Additionally, $CO_2$ emissions produced from decay of this sediment and organic matter trapped behind the dams potentially contribute to climate change (Parekh, 2004). From a cost-benefit standpoint, negative ecological impacts of dams increasingly outweigh societal gains when compared in the development of science and policy supporting dam removals.

Dams alter river flow, which can impact aquatic and riparian biodiversity. A decrease in periodic and seasonal floods and increased periods of low flows can reduce aquatic faunal diversity through changes in temperature or dissolved oxygen (DO) (Bednarek, 2001). As a result, impounded rivers may eventually support only animals suited to constant flows and reduced flood conditions, while organisms poorly adapted to increased temperature and low DO may become locally extinct. Riparian floral diversity may also be negatively influenced by altered flow regimes, as downstream transport of seeds and the stems of resprouting woody plants is hindered (Vesk et al., 2006; Brown and Chenoweth, 2008). Riparian plant species composition can then become fragmented, with greater similarity of riparian plant communities at sites among impoundments than those of reaches in-between (Jansson et al., 2000). Dams and impoundments can decrease river velocity, reducing the downstream delivery of floating diaspores (seeds, fruits or spores), which may instead sink to the river bottom or become beached (Jansson et al., 2000). In contrast, diaspores in unimpounded rivers can pass freely and rapidly downstream in conjunction with seasonal floods and high flows, and subsequently riparian plant communities vary more gradually in species composition downstream compared to dammed reaches (Jansson et al., 2000). Dammed rivers may also undergo hourly, daily or weekly flows which are far more severe than those of free-flowing rivers, depending on water energy needs or other water resource consumption patterns (Bednarek, 2001).

In coastal regions such as the Pacific Northwest, river impoundments have far-reaching effects on both riverine and marine productivity. Retention of sediment and woody debris (WD) behind dams leads to loss of freshwater nutrient delivery to estuaries and limits marine productivity (Bednarek, 2001). Reciprocally, dams prevent tidal surges in coastal rivers from travelling significant distances upstream, and limit the degree of cyclical flooding, both of which are mechanisms for upstream and downstream migration of anadromous fish and shrimp (Bednarek, 2001). Loss of riverine sediment and WD delivery to coastal regions depletes resources key to the building and maintaining of nearshore habitat (Shaffer et al., 2008).

The decline of fish populations common in regulated rivers can have a cascading effect on riparian corridors. Dams are a primary contributor to the reduction of salmon populations in the U.S., along with fishing, habitat alteration, fish hatcheries, climate change and the presence of invasive species (Pess et al., 2008). Dams negatively influence anadromous fish by blocking passage and altering the sediment and flood regimes of rivers. Over time, this flow alteration leads to the decline of sustaining food chains in streams (Bednarek et al., 2001). The decline of anadromous fish populations can adversely influence terrestrial wildlife directly, as with a loss of protein and high calorie resources, or indirectly through the absence of fish carcasses, which fertilize riparian plants (Helfield and Naiman, 2002; Sager-Fradkin, 2008).

### 1.2 Implications of Dam Removal

A common focal point for the ecological benefits of dam removal, in some regions of the northern hemisphere, is the return of native salmon runs. In Pacific Northwest riverine ecosystems, restoring anadromous fish populations often means restoring riparian habitat for a variety of plants and animals, supporting diverse aquatic wildlife, and restoring salmon-centered indigenous cultures (Valadez, 2002; Doyle et al., 2005). While salmon recovery indeed provides ecological benefits, the roles of aquatic insects and bottom-dwelling aquatic organisms (zoobenthos) in supporting fish and other higher-trophic aquatic organisms is of equal significance (Morley et al., 2008). A shift in aquatic insect and substrate-dwelling organisms can mean a shift in the higher-trophic organisms a river ecosystem can support (Morley et al., 2008). Due to their short generation time, invertebrate populations can recover from disturbance more rapidly than other life forms such as riparian plant communities, in which different successional pathways of undetermined temporal scales may result in variability of species diversity over the first decade or so (Doyle et al., 2005). Concurrently, the sources of available carbon needed to support aquatic invertebrates depend on detrital inputs from riparian plant communities, nutrient delivery from downstream marine systems, and algal communities within the river. These interdependent relationships illuminate the complexity of post-dam-removal recovery of a river ecosystem; a multitude of factors beyond the removal of fish passage barriers are at play (Casper et al., 2006). Hence, in riverine restoration planning, it is critical to addresses entire ecosystems rather than a single species such as anadromous fish (Doyle et al., 2002, 2005).

Degree of flow in a riverine ecosystem can influence food webs and ecosystem processes such as litter decomposition rates by modifying macroinvertebrate communities. Casper et al. (2006) studied zoobenthos and changes in trophic food webs, via isotopic signatures, following the Edwards Dam removal on the Kennebec River, Maine. Overall, at three locations along the river, a greater density increase of zoobenthos was observed in the site closest to the former dam. Diversity was variable, but few differences between pre-and post-dam-removal species richness or evenness were apparent. A more apparent change took place in post-dam-removal insect community composition, with the addition of 8 genera to the existing population consisting of mayflies, caddisflies, oligochaetes, and a variety of predatory and non-predatory chironomid midges. The difference between zoobenthic communities in restored and free-flowing sites was less distinct over time while stable isotope signatures changed, indicating that the feeding tactics and quantities of aquatic organic matter were altered (Casper et al., 2006). Muehlbauer et al. (2009) conducted a study in which leaf litter bags were strategically placed above and below Fossil Creek dam, near Strawberry, AZ, and measured decomposition rates and presence of macroinvertebrates and fungal decomposers prior to and following the decommissioning of the dam. They found that, while litter decomposition rates did not vary above or below the dam prior to dam removal, the macroinvertebrate and fungal decomposer communities increased below the dam post-removal. Restoration of full water flow appeared, in this case, to positively impact decomposer populations (Muehlbauer et al., 2009).

While dam removal is generally expected to benefit aquatic systems in the long-term, this restoration approach is not devoid of potential ecological detriments, particularly in the short-term. Water quality may be negatively affected by high concentrations of total suspended solids (TSS) during initial post-dam-removal high-flow events (Downs et al., 2009). Fine sediment released and deposited on river bed surfaces may reduce conductivity and water infiltration, and increase the downstream flooding and burial hazards of salmonid and macroinvertebrate habitat by sediment deposition. In urban, agricultural or other developed river basins, buildup of toxins in reservoir sediment may pose a health hazard to humans and wildlife with their release (Stanley and Doyle, 2003; Wildman and MacBroom, 2005). Additionally, the sediment exposed following dam removal can serve as a vector for invasive weed colonization, providing seed sources for other locations along riparian corridors (Orr and Stanley, 2006). A lack of complete, pre-and post-dam-removal monitoring datasets in conjunction with the small dam removals which have occurred over the past decades only increases these concerns when addressing the removal of large dams (Downs et al., 2009; Muehlbauer et al., 2009). Of the 500 dam removal projects which have taken place in the U.S., few have received pre-and post-dam-removal monitoring to measure ecosystem changes over time (Duda et al., 2008; Woodward et al., 2008).

***Sediment***

A key aspect of dam removal recovery centers around geomorphic impacts, in terms of sediment storage. Thus, a key component of post-dam-removal river ecosystem restoration is an understanding of the direction, distance and speed of sediment transport, to control release rates and minimize harm to aquatic organisms (Doyle et al., 2002). Over time, dams result in upstream sediment storage, while sediment delivery and accumulation downstream is diminished (Bednarek, 2001; Shaffer et al., 2008). It is generally expected that dam removal results in a reversal of this trend, based on the erosion of upstream sediment and aggradation of downstream reaches from fluvial transport (Konrad, 2009). The overall geomorphic response of a riverine system to dam removal depends on: 1) the level of sediment which has accumulated in the reservoir; 2) the flow and ability of the river to transport this sediment; 3) rates of sediment erosion, driven by high flows during wet seasonal periods; and 4) response of channel morphology downstream, which may determine the overall geomorphic response of the system to dam removal (Konrad, 2009). River systems with higher rates of discharge and/or higher slope generally move sediment loads more rapidly than rivers with lower discharges and slopes. Additionally, the texture of impounded sediment influences removal rates, with fine, silt-dominated sediment delivery occurring more rapidly than that of coarse sediment (Shafroth et al., 2002; Mussman et al., 2008). Based on the limited dam removal ecosystem recovery data available, geomorphic adjustments tend to occur within the first 1 to 5 years (Doyle et al., 2005).

### 1.3 Decommissioning Dams: A New Approach to Management and Restoration of Rivers

Due to a greater recognition of the ecological impacts of dams, more progressive regulations for dam maintenance and operation are in effect. Non-federal hydroelectric projects require licensing under the Federal Power Act (FPA, 16 U.S.C. 791-828c as amended; Chapter 285, June 10, 1920; 41 Stat. 1063). Testament to a changing public perception of the role of dams, the Federal Energy Regulatory Commission (FERC) modified its relicensing process for existing, non-federal dams in the early 1990s. For the first time, general consensus supported the decommissioning of operating dams for ecological restoration alone, not simply due to public safety concerns (Gowan et al., 2006). Greater priority has been assigned to weighing the benefits of the restoration of impounded rivers in relation to the utility of maintaining existing dams. Rather than basing renewal chiefly on the provision of public service, the environmental impacts of these dams are now investigated when considering the extension of the 30-to-50-year licenses issued to private and public power producers. Currently, factors such as increased minimum flow, added or enhanced fish passages, allowance of periodic high flows, and adjusted flow regulation to accommodate riparian environments are included in the criteria for dam relicensing (Bednarek, 2001).

Contemporary public views on river impoundments with the passage of time have placed aging dams under increasing scrutiny. Since the late 1990s, the licenses of hundreds of hydroelectric dams have expired; the majority of these dams were constructed in the 1950s and 1960s, and are subject to relicensing investigations. If dams fail to meet specific safety and ecological provisions, they are decommissioned (Grant and Parks, 2009). An estimated 85% of aging dams in the U.S. will reach the end of their operational lives by the year 2020 (Doyle et al., 2003). As with other aging structures, public safety concerns play a role in determining whether older dams remain operable. In 2002, the Federal Emergency Management Agency (FEMA) categorized 9,200 U.S. dams as “high hazard” due to structural instability and risk to downstream human developments in the event of a dam break (Downs et al., 2009). In such cases, the utility of river restoration may clearly outweigh the benefits of the prolonged and potentially hazardous operation of outmoded dams.

### 1.4 Dam Removal Case Studies in the United States

Previous dam removal operations, if monitored and documented, may provide insight to the rates of recovery and cost-benefit analysis entailed post-dam-removal river restoration efforts. Critical data include timeframe of sediment release and erosion, altered channel dynamics, riparian vegetation recovery, water quality improvement, and eventual improvement of fish populations once passage is restored (Bernhardt et al., 2005). Generally, the goals of post-dam-removal river restoration include restoration of fish passage, improvement of water quality, management of riparian zones and enhancement of aquatic habitat. Timing of drawdown and dam deconstruction in relation to high and low flows, the degree to which a river is regulated (i.e., number of obstructions), and the size of dams all play a role in the amount of time required for ecosystem recovery (Bednarek, 2001).

***Marmot Dam, Sandy River, OR***

The 2007 Marmot Dam removal on the Sandy River, Oregon provides an example of an ecologically successful dam decommissioning. The former 15-m-tall dam lay 43 km above the confluence of the Sandy and Columbia Rivers, and its deconstruction created the largest post-dam-removal sediment release up to that time (Grant, 2000; Podolak and Pittman, 2010). A cofferdam, protecting the Marmot Dam removal construction site, was breached in 2008 (Stillwater Sciences Technical Memorandum, 2009). The 90 km Sandy River drains a 1300 $km^2$ basin in the southwestern Cascade Range, near the base of Mt. Hood (Podolak and Pittman., 2010). Since the construction of Marmot Dam in 1913, sediment had accumulated to its crest height, composed of approximately 750,000 $m^3$ of cobble, and gravel sediment overlying a finer sand layer (Stewart and Grant, 2005; Downs et al., 2009). The sand and gravel, originating from volcanic debris on Mt. Hood, were promptly released into the river following dam removal. Both headward and lateral erosion removed the sand and gravel which had accumulated for nearly a century, following a swift alteration of the river's profile, termed a "knickpoint" (Downs et al., 2009; Major et al., 2008). In less than three days following the breaching, the Sandy River channel morphology 2 km downstream from the former dam changed from single-thread to multiple-channel, and the channel bed had aggraded nearly 4 m (Major et al., 2008).

Sandy River sediment removal occurred at a rate more rapid than projected, with 15% of all stored sediment transferred within the first 48 hours (Major et al., 2010). By late winter of 2008, approximately half of the sediment had been redistributed over the first 3.2 km downstream of the dam. New braided channels, bars and riffles had formed, flooding was not extreme (only 3 to 4 times that of average summer low flows), and a greater variety of salmon habitat was soon apparent (Grant and Parks, 2009). One year following dam removal, the topographic complexity of the Sandy River was comparable to pre-dam conditions in at least one site, the primary difference being higher channel bed elevation in the immediate vicinity of the former dam and a temporary decrease in channel complexity following the coffer dam breaching (Stillwater Sciences, 2009). Post-dam-removal channel bed recovery of the Sandy River was projected to be at least 10 years, but serious concerns of fish impediment were alleviated within days of dam removal (Downs et al., 2009). Relicensing Marmot Dam would have cost $20 million, including the enhancement measures which would have been required by FERC, with additional costs of upgrading the aging dam structures; in contrast, removal of the dam cost $17 million, an investment which restored 43 km of riverine habitat (Grant and Parks, 2009; Kober, http://www.water. ca.gov/ fishpassage/docs /dams /dams.pdf).

### *Grangeville and Lewiston Dams, Clearwater River, ID*

The removal of the Grangeville and Lewiston Dams on the Clearwater River, Idaho produced successful river ecosystem recovery prior to downstream redamming. The dams, 17 and 14 m tall, respectively, were constructed in 1903 and 1927. These impoundments impeded salmon and steelhead runs, adversely impacting the 1855 fishing and treaty rights of the Nez Perce tribe. The 1963 and 1973 removals of these dams initially improved salmon runs, freeing 68 km of river and hundreds of km of tributaries. The silt released during dam removal had little adverse impact (American Rivers, 1999). In fact, the removal of the Lewiston Dam was the first occasion in which the Army Corp of Engineers cooperatively decommissioned a federally licensed dam for the purpose of river restoration (American Rivers, 1999). Unfortunately  the 1970s construction of the Dworshak Dam on the north fork Clearwater River, along with the installation of 4 federally-operated dams on the lower Snake River, all but negated the benefits of the earlier dam removals and led to a drastic decline in salmon populations in the Snake River (American Rivers, 1999).

### *Other U.S. Dam Removals*

As indicated in the previous case studies, dam removal can improve the long-term health of riverine ecosystems. Some dam removal efforts, however, have produced negative results, particularly in areas of urban or agricultural development. The two-stage removal of the Fort Edwards Dam on New York's Hudson River resulted in the release of toxins contained in accumulated sediment (Stanley and Doyle, 2003). The initial and partial removal in 1973 released petroleum wastes and polychlorinated biphenyls (PCBs) downstream and required expensive cleanup efforts (Stanley and Doyle, 2003). Released sediment restricted the flow of the river, blocking access both for navigation and sewage waste distribution downstream and created additional health hazards (Stanley and Doyle, 2003). Following the second and final stage of the Fort Edwards dam removal in 1991, PCB concentrations in striped bass proved to be twice that of previous tests (Stanley and Doyle, 2003; American Rivers, 1999). A similar problem arose with the removal of the Anaconda and Union City dams on the Naugatuck River, Connecticut. Polyaromatic Hydrocarbons (PAHs) were discovered in post-dam-removal sediments, and had to be removed to prevent further downstream contamination (Wildman and MacBroom, 2005).

Since the 1960s, the state of Wisconsin has undertaken numerous small dam removals. Initial dam removals were carried out due to failing structural integrity. Beginning in the 1990s, dam decommissioning occurred increasingly for the purpose of ecological restoration, with mixed success. Following the 1988 removal of the Woolen Mills Dam on Wisconsin's Milwaukee River, river flushing and accompanying fine sediment removal downstream took place over a 6 month time period, and an overall improvement in passage was seen for fish and other aquatic organisms (Bednarek, 2001). Invasive common carp populations diminished due to faster-flowing waters, while native smallmouth bass populations increased (Bednarek, 2001). In contrast, removal of the Fulton Dam on Wisconsin's Yahara River led to the replacement of native cattail and sedge with non-native wet meadow grasses, and a decline in muskrat and duck populations resulting from the loss of the reservoir. Displacement of desirable flora and fauna must be carefully considered in the planning of dam removals (Bednarek, 2001).

Few studies have followed the long-term recovery of drained reservoirs, particularly in relation to riparian vegetation following dam removal. However, limited data are available from vegetation surveys conducted in thirteen post-dam-removal sites in Wisconsin (Orr and Stanley, 2006). Vegetation surveys depicted grasses and forbs as the dominant riparian plants within the first few years following dam removal, coupled with trees in later years. Aquatic and semiaquatic plants, such as rushes, sedges and reeds showed negative trends over time. Invasive plants also colonized the exposed sediment, persisting throughout the study, with the exception of sites dominated by shrubs and a stinging nettle (*Urtica dioica*) (Orr and Stanley, 2006). Reed canary grass (*Phalaris arundinacea*), a highly invasive rhizomatous grass introduced from Asia, became established despite an apparent lack of local seed source. Other than a general trend of increasing tree frequencies as time following dam-removal elapsed, plant species richness and species composition proved variable during the 2-to-50-year post-dam-removal time period, illuminating the unpredictability of riparian plant succession following a major substrate-altering disturbance (Orr and Koenig, 2006).

With little known about post-dam-removal plant succession or success of artificial plant regeneration on drained reservoir sediments, performance of the Elwha Revegetation Crew plantings must be documented throughout the project. The intent of this study is to provide potential expected outcomes when selected woody plant species are installed in specific dewatered reservoir sediment types. This small body of research may contribute to the limited knowledge base of revegetated reservoir sediments post-dam-removal, particularly where active plant restoration is to take place.

## II. Elwha River Ecosystem and Dams

### 2.1 Geography and Vegetation

The Elwha River, 72 km in length, lies 10 km west of the city of Port Angeles on the northern central Olympic Peninsula of Washington State. The Elwha River basin covers 830 $km^2$ and forms 161 km of tributaries (Acker et al., 2008; Brenkman et al., 2008). The headwaters of the Elwha River are situated at the base of Mount Barnes in the Bailey Range of the Olympic Mountains. The mouth of the river enters the Strait of Juan de Fuca, a marine passage connecting the Pacific Ocean with the waters of Puget Sound and British Columbia (Duda et al., 2008; Brenkman et al., 2008). Elevations in the watershed range from sea level to 1372 m at the headwaters (Duda et al., 2008; Brenkman et al., 2008). The Elwha River basin typically experiences warm, arid summers and cool, wet winters. Total annual precipitation averages just over 143 cm, with an average total snowfall of 37 cm and an average annual temperature range of 4° to 14° C (Western Regional Climate Center, http://www.wrcc.dri. edu/cgi-bin/cliMAIN.pl?waelwha).

Soil types in the Elwha River basin are classified as Haplumbrepts, typical of temperate to warm regions, of the order inceptisol, with parent materials of post-Pleistocene origin (Jackson and Kimerling, 2003; Chernicoff and Venkatakrishnan, 1995). Much of the Elwha River basin, Olympic Mountain Range and Strait of Juan de Fuca geology and topography were formed by seafloor scraping of the Pacific and North American plates, and glacial incision. The Juan de Fuca shoreline was formed by the southern advance of the Cordilleran ice sheet approximately 16,000 years ago (Warrick et al., 2009; Shaffer et al., 2008).

The Elwha River basin is comprised of many Olympic Peninsula vegetation zone classifications. Lower elevations fall within the western hemlock (*Tsuga heterophylla*) zone, generally dominated by Douglas-fir (*Pseudotsuga menziesii*) and co-dominated by western hemlock, with western redcedar (*Thuja plicata*) in moister sites (Henderson et al., 1989; Duda et al., 2008). Drier areas around Lake Mills and the Lillian River tributary consist of vegetation classes associated with the Douglas-fir zone. Higher elevations lie in the subalpine fir (*Abies lasiocarpa*) zone on the drier eastern side, and the mountain hemlock (*Tsuga mertensiana*) zone on the wetter western side, within the Cascade Subalpine Forest Zone Complex. Floodplains, river terraces and valley bottoms consist of red alder, black cottonwood, grand fir (*Abies grandis*), and bigleaf maple (*Acer macrophyllum*) associations, with canopy dominance of the different species varying (Duda et al., 2008; Jackson and Kimerling, 2003).

### 2.2 History and Features of the Glines Canyon and Elwha Dams

The Glines Canyon and Elwha hydroelectric dams were constructed in the early 20$^{th}$ century. The former Elwha Dam, 33 m in height and located 7.9 km upstream from the mouth of the Elwha River, was built in 1913, forming the Lake Aldwell reservoir; the Glines Canyon Dam, 64 m in height and located 21.6 km upstream, was completed in 1927, forming the Lake Mills reservoir. The dams, designed to power a lumber mill and enhance economic development on the North Olympic Peninsula, were not built to accommodate salmon passage (Duda et al., 2008). These dams were classified as high head (>30 m) storage dams, one of the two major structural dam categorizations (Poff and Hart, 2002; Woodward et al., 2008). Storage dams bear large hydraulic heads, afford storage volume, long hydraulic residence time (HRT), and control the rate of water release. The other major dam classification, “run of the river” dams, usually pertain to smaller dams in which hydraulic head and storage volume are minimal, HRT is short, and control of water release is minor to nonexistent (Poff and Hart, 2002). The Lake Aldwell reservoir was 1.08 $km^2$ in size, and Lake Mills was 1.68 $km^2$.  The two reservoirs together once flooded more than 9 km of riverine habitat. To-date, the Glines Canyon Dam is the tallest dam to be removed in U.S history (Duda et al., 2008). Of the post-dam-removal Lake Mills dewatered reservoir surfaces exposed, 30% were projected to be floodplain, and 40% were projected to be upland terraces consisting of abandoned residual sediment, with the remainder forming steep valley walls (Duda et al., 2008).

Approximately 21.7 million $m^3$ (28 million $yd^3$) of sediment were retained by the Glines Canyon Dam during its operation (ONP, 2013). Upland dewatered reservoir landform soils were covered by 8 cm to 6 m of sediment; the floodplains as much as 60 m (2011 ONP soil probe surveys). Post-dam-removal substrates retained on the benches, upland terraces and valley walls in the former Lake Mills reservoir consist of fine, alluvial silt and clay sediments, coarse sand and gravel sediments, and fine-coarse sediment mixtures (Winter and Crain, 2008). Restriction of sediment transport following the Glines Canyon Dam construction led to the formation of a delta at the head of Lake Mills up to 924 m in length, 24 m thick, and supplying much of the coarse gravel, sand and fine sand substrate remaining on the dewatered reservoir in 2013 (Winter and Crain, 2008).

Removal of the Glines Canyon and Elwha dams was initially set in motion by the Elwha River Ecosystem and Fisheries Restoration Act (PL 102-495), mandated by Congress in 1992, which required the full restoration of the Elwha River ecosystem and its native anadromous fish (Gowan et al, 2006). Environmental Impact Statements (EIS) completed in 1995 by the National Park Service, U.S. Fish and Wildlife Service, Bureau of Reclamation, Bureau of Indian Affairs, and Lower Elwha Klallam Tribe determined that restoration of this ecosystem could be achieved only through the removal of both dams (Duda et al., 2008). Removal of only one dam would mean maintaining an upstream or downstream obstruction to fish passage and sediment retention, thus negating full restoration of river continuity and overall restoration progression (Bednarek, 2001). It was determined that removal of both dams would result in ecosystem recovery sufficient to justify the expense of removal and restoration efforts, based on studies of the efficacy of different fish passage facilities, prior dam removals, and earlier dam-related Elwha River fisheries and wildlife conservation efforts (Winter and Crain, 2008).

The 1913 Elwha Dam construction occurred prior to the formulation of the Federal Power Act (FPA), hence the owners of the dam were not obligated to obtain a license. The Glines Canyon Dam, completed in 1927, was granted a 50-year operating license (1926). Following an investigation of unlicensed non-federal hydroelectric projects by FERC, beginning in the 1960s, a combined license (No. 588) was issued for both dams in 1979, upon determination that the two dams were “hydraulically, electrically, and operationally interconnected” (Winter and Crain, 2008). By the 1980s the matter of relicensing the Elwha River dams, never officially licensed to begin with, became a source of public debate. That the Lake Mills reservoir lay within Olympic National Park further fueled this controversy, where the Glines Canyon Dam was concerned (Duda et al., 2008). Given that dam re-licensing fell under the regulatory umbrella of the National Environmental Policy Act (NEPA), FERC facilitated an EIS procedure which required applicants to gather new environmental data, to be considered along with data contributed by various federal agencies, when constructing a cost-benefit analysis of keeping versus removing dams. This was a precursor to the 1992 congressional Elwha River Ecosystem and Fisheries Restoration Act (EREFRA). Prior to the passage of the EREFRA, no hydroelectric dam removal in the U.S. had been based centrally on fish and wildlife benefits (Winter and Crain, 2008). Thus began the legacy of the largest dam removal restoration attempted in U.S. history prior to 2013 (Woodward et al., 2008).

### 2.3 Environmental Impacts of the Elwha River Dams

As is often the case with large dams, the Glines Canyon and Elwha dams negatively impacted the health of the lower Elwha River, the river basin, and nearshore habitat. For nearly 100 years the dams obstructed access to over 113 km of high-quality anadromous fish habitat. Loss of access to spawning grounds dramatically reduced salmon runs, which in turn altered the foraging patterns of Elwha River terrestrial mammals and birds. Reduced activity of mammals along the Elwha River corridor has affected the distribution of nutrients and seeds through zoochory (Duda et al., 2008). Hydrochorous seed transport has also been hindered, preventing the downstream colonization of resilient, diverse riparian vegetation and altering plant species composition (Brown and Chenoweth, 2008; Duda et al., 2008). Additionally, the dams have prevented the delivery of sediment and woody debris, crucial sources of marine nutrients and beach-building materials, to the Elwha estuary, delta, and nearshore habitat in the Strait of Juan de Fuca (Shaffer et al., 2008).

### *The Elwha Estuary, Delta and Nearshore Habitat*

A common tendency of dams, particularly storage dams, is the retention of sediment and woody debris which would otherwise be distributed throughout the river basin. During their existence, the Elwha River dams and reservoirs retained sediment and large woody debris (LWD) critical to building shoreline beaches and bluffs. From the years 1939-2006, shoreline erosion around Port Angeles, WA was measured at approximately 0.6 m per year, or 24,000 $m^3$ (Warrick et al., 2009). Historically, the Elwha River delivered roughly 160,000 $m^3$ of fine and coarse sediment per year to the mouth of the river. This delivery of sediment and wood provided nutrients responsible for estuarine and marine productivity, as well as contributing beach-building materials to the approximately 21 km of shoreline between Freshwater Bay and Ediz Hook, bordering the city of Port Angeles. Installation of shoreline bulkheads during the 1950s compounded the problem of beach erosion, obstructing sediment input from eroding coastal bluffs. The formerly mixed-grain-size nearshore habitat and beach texture of the Strait of Juan de Fuca was replaced by cobble-and-boulder-dominated substrate. Estuarine habitat near the Elwha River mouth declined from 0.4 $km^2$ to 0.12 $km^2$ (Shaffer et al., 2008). Estuarine decline is of particular concern, as the Elwha River estuary was documented as having the greatest plant species diversity recorded in Pacific Northwest coastal wetland surveys (Shafroth et al., 2009). Additionally, riverine fine sediment deposition is critical to the sustenance of coastal marshes in a future of climate change and sea level rise (Poff and Hart, 2002; Stoeker Ecological, 2011).

The nearshore habitat near the mouth of the Elwha River historically supported a variety of marine wildlife. Riverine nutrient input from the Elwha River to the Strait of

Juan de Fuca, in conjunction with deep marine water, powerful winds, and strong currents conducive to mixing, maintain well-mixed, cool and nutrient-rich conditions in the Strait. Loss of sediment delivery has led to a decline of wetlands and associated shellfish, clam, eelgrass and kelp populations (Shaffer et al., 2008). Estuarine and nearshore habitat between Ediz Hook in Port Angeles and Cape Flattery, Neah Bay are known to be critical early saltwater rearing zones for Chinook salmon (Shaffer et al., 2008). Kelp forests make up 40% of the Strait of Juan de Fuca shoreline, and consist of giant kelp (*Macrocystis integrifolia*), bull kelp (*Nereocystis luetkeana*), and understory kelp (*Pterygophora californica*), while eelgrass (*Zostera marina*) forms 20% of the shoreline. Modification of low-tide beaches from mixed-grain to coarse-grained textures has resulted in unsuitable shellfish habitat. Removal of the Elwha River dams will likely benefit the estuarine and shoreline habitat near the river mouth and delta, with the return of riverine nutrients and debris. However, much of this land lies outside the boundary of Olympic National Park. Full restoration of this marine nearshore habitat must be a collaborative effort among Tribal, private, city, and state landowners, to fully restore the severely altered shoreline (Shaffer et al., 2008).

### *Dam Impacts on Salmon*

The presence of the Elwha River dams has had adverse effects on salmon diversity and overall salmon populations. Since the construction of the dams, native salmon runs have declined by 90% in the Elwha watershed (Pess et al., 2008). The majority of the Elwha River lay above the dams, unavailable for use as spawning habitat. Historically, annual returns of anadromous salmonid populations were estimated at 380,000 to 500,000. Average numbers between 1990 and 2000 were less than 5000 returning salmon, with ranges of <5000 to 19,800 per year (Pess et al., 2008). Prior to damming, the Elwha River was home to all anadromous salmonids native to the Pacific Northwest: coho (*Oncorhynchus kisutch*), Puget Sound Chinook (*Oncorhynchus tshawytscha*), sockeye (*Oncorhynchus nerka*), pink (*Oncorhynchus gorbuscha*), Strait of Juan de Fuca/Hood Canal summer chum (*Oncorhynchus keta*), steelhead trout (*Oncorhynchus mykiss*), and cutthroat trout (*Oncorhynchus clarkii*). Potamodromous bull trout (*Salvelinus confluentus*) and rainbow trout (*Oncorhynchus mykiss*) have been confined to areas above the dams, when historically they migrated within the freshwater reaches above, between and below both dams (Brenkman et al., 2008).

***Salmon-Dependent Wildlife***

The Elwha River dams, in addition to impacting the life cycles of aquatic organisms, have impacted migratory patterns of terrestrial organisms, among them the North American black bear (*Ursus americanus*) (Sager-Fradkin et al., 2008). The black bear, prevalent in the wilderness of Olympic National Park, plays an important role in nutrient distribution between marine and terrestrial environments in the Elwha and other river basins. Typically, bears feeding on healthy riverine salmon populations provide crucial nutrients to riparian and inland forests through fish consumption, defecation and transportation of salmon carcasses inland (Helfield and Naiman, 2002; Sager-Fradkin et al., 2008). Bears preparing for hibernation require fatty food sources such as salmon, but lacking significant fish populations will substitute this food for more consistently abundant calorie sources, such as huckleberries (*Vaccinium* sp.) or whitebark pine seed (*Pinus albicaulis*) where available at higher elevations (Smith et al., 2008). Currently, resident black bears in the Olympics spend the fall period leading up to hibernation in higher elevations feeding primarily on huckleberry fruits. In particular, female bears linger at higher elevations throughout most of the spring, summer and fall seasons leading up to hibernation, which draws male bears up in elevation during the early summer mating season (Sager-Fradkin et al., 2008). Historical reports indicate that black bears in western Washington would more typically feed on salmon prior to hibernation. A decline in salmon runs following the installation of the Elwha and Glines Canyon dams may have altered the movement and feeding patterns of Olympic black bears (Sager-Fradkin et al., 2008). As a component of a 5-year pre-dam-removal bear tracking study conducted by ONP, U.S. Geological Survey, and the University of Idaho Department of Fish and Wildlife Resources, salmon carcasses were placed in riparian areas of the Olympics near the time of hibernation, and were readily eaten by black bears (Sager-Fradkin et al., 2008). The removal of the Elwha River dams may eventually result in the movement of black bears back to the river if salmon populations are restored (Sager-Fradkin et al., 2008).

## III. Projected Outcomes of the Elwha River Dam Removal

Few examples of post-dam-removal ecosystem recovery following the removal of large dams are available to aid in predicting the long-term outcomes of the Elwha River dam removals. The best current example would be a sediment transport model, designed for the Elwha River, which predicted increasing sediment stability over time, with occasional flooding disturbance disrupting that stability, following dam removal (Konrad, 2009). Original Lake Mills sediment load estimates amounted to 20.4 million $m^3$ of sediment (or 204 million $km^3$), and were later determined to be nearly 21.4 give or take 3 million $m^3$ (approximately 214 million $km^3$) (USGS sediment team, 2012). While the majority of U.S. rivers are impounded by dams, most are small dams on small-to-midsize rivers, and dam removal efforts thus far have been no exception (Doyle et al., 2005). The enormity of the Elwha River dam removal and restoration has been compared to the recovery of the North and South Fork Toutle River basins, following the 1980 eruption of Mount Saint Helens, WA (Bednarek, 2001). This eruption, produced by highly explosive pyroclastic lava, released nearly 2.3 billion $m^3$ of sediment into the North Fork Toutle River basin, and over 38.2 million $m^3$ into the South Fork (Bednarek, 2001). These massive mudslides resulted in the burial of 90% of salmon habitat, and virtually all riparian vegetation. Despite these unfavorable odds, salmon populations began to recover a mere 3 months following the volcanic sediment delivery (Bednarek, 2001).

### 3.1 Restoration: Research Needs and Site Conditions

The removal of the Elwha River dams has resulted in the large-scale exposure of sediment, much of it well beyond the range of seed rain from the intact forest edge

(Michel et al., 2011). Restoration measures are in effect to create river-sustaining riparian forest habitat, prevent weed colonization, support riparian wildlife and to aid in eventual soil development (Bradshaw, 1982; Chenoweth et al., 2011). Since the fall of 2011, Olympic National Park's Elwha Revegetation Crew has been planting upland terraces and benches of the dewatered Lake Mills reservoir to meet such needs. All plants installed are propagated from seed collected within the lower Elwha River basin and processed at the Matt Albright Native Plant Center near Agnew, WA, $4^{th}$ Corner Nursery (Bellingham, WA), and Corvalis Plant Materials Center (Salem, OR); many of the conifer seedlings are propagated by Silvaseed Nursery in Roy, WA.

Revegetation of the exposed sediments in the Lake Mills and Aldwell reservoirs may trigger more rapid riparian plant recovery and development of late-seral forests (Chenoweth et al., 2011). The roots of installed plants may provide a carbon source for microbes participatory in soil formation, fragment compacted sediment through the formation of root pathways, and eventually develop macropores which aid in fluid transport when roots penetrate soil (Angers and Caron, 1998). Enhancement of riparian forest development can also improve stream ecosystem health. Riparian forests provide shade critical to stream temperature regulation and large woody debris (LWD) input, which increases channel complexity beneficial to spawning fish (Collins and Montogmery, 2002).

Plants installed at Lake Mills consisted of potted and bare-root specimens, planted according to three different planting prescriptions (**Table 1,** Section IV). The timely establishment of woody riparian vegetation, in particular, is a goal of the ONP and the

Lower Elwha Klallam Tribe (LEKT) restoration plans. Woody plants provide substantial shading competition to non-native plants in newly exposed ground, provide a sheltered growth medium and microclimate for volunteer and planted seedlings, and provide seed and detritus to areas far from seed sources (Castro et al., 2004; Chenoweth et al., 2011).

A particularly challenging aspect of post-dam-removal revegetation in the dewatered Lake Mills and Lake Aldwell reservoirs is the survival of plants in residual sediments. Post-dam-removal sediment in Lake Mills includes fine, silt-and-clay-sized particles, 50-67% of which is predicted to erode and wash downstream over the several-year period following dam removal, with the remainder consisting of coarse sand, gravel and cobble (Mussman et al., 2008; Czuba et al., 2011). Fine sediment, composed of alluvial silt and clay, is low in nutrients, lacks porosity and hydraulic conductivity, and creates hypoxic growing conditions for riparian plants. Approximately 20% of the fine sediment consists of clay, with the remaining ~80% consisting of silt, typical of Lake Mills sediment (Mussman et al., 2008; Warrick et al., 2009; Czuba et al., 2011). Coarse sediments, found primarily on the reservoir delta and future valley walls, are predominantly sand and highly porous, erosive, nutrient-poor gravel possessing little moisture retention properties. The erosive nature of coarse substrates coupled with windy site conditions at the former Lake Mills poses the risk of plant root exposure (Chenoweth et al., 2011).

Over time, with the establishment of vegetation in post-dam-removal sediment, organic matter can contribute greatly to mineralization of the substrate (Berendse, 1998). Vegetative litter created by isopod defoliation can harbor litter-decomposing microbes, which in turn aid in soil development, mineralization and nutrient input (Facelli and

Pickett, 1991). However, neighboring forest vegetation close enough to provide a seed source may not consist of species tolerant of soil saturation, drought, low nutrients, or pH extremes. This illuminates the potential benefits of introducing a variety of native riparian plants to determine which species possess tolerance of such extreme soil conditions (Ash et al., 1994).

While the colonization of Lake Mills post-dam-removal sediment may prove challenging for many native plants, a multitude of neighboring invasive grasses, forbs and shrubs have the capability to thrive in such substrates (Orr and Stanley, 2006). In 2008, seed bank analysis of sediment cores collected from Lake Mills revealed a higher presence of invasive plant species than native species as potential colonizers of exposed sediment beyond the range of forest edge seed rain (Brown and Chenoweth, 2008). Invasive weeds such as Himalayan blackberry (*Rubus armeniacus*) favor substrates dominated by gravel and other coarse materials (Caplan and Yeakley, 2006), like those found on the Lake Mills delta. Hence, weed control is vital to the establishment of a riparian environment conducive to stream shading, nutrient input, and contribution of woody debris to support a riverine ecosystem. Invasive plants can modify succession of riparian plant communities through alteration of microbial communities, competition for natural resources, allelopathy, and habitat alteration (Haubensak and Parker, 2004; Rudgers and Orr, 2009; Peltzer et al, 2009).

Examples of habitat alteration from non-native plant invasion are plentiful. Local invasive grasses such as tall fescue (*Lolium arundinaceum*) host the fungal endophyte *Neotyphodium coenophialum*, known to slow the progression of disturbed grasslands to native forests over time through the growth inhibition of non-host plants (Rudgers and Orr, 2009). In southern Wisconsin, following the removal of the Oak Street Dam in Baraboo and the Rockdale Dam in Rockdale, invasive reed canary grass colonized exposed post-dam-removal sediment, even in the absence of an on-site seed source (Orr and Stanley, 2006; Orr and Koenig, 2006). Attempts to establish primarily graminoid native vegetation through outplanting showed little success three years following dam removal (Stanley and Doyle, 2003; Orr and Koenig, 2006). Establishment of native woody plants may aid more effectively in the establishment of riparian forests, providing soil formation, terrestrial wildlife forage, and bank stabilization in addition to weed control (Castro et al., 2004; Chenoweth et al., 2011).

Previous studies have revealed the potential for greater homogeneity in post-dam-removal sediment than in underlying topsoils. Such environments may favor colonization by invasive plants such as reed canary grass, which can thrive in wet substrates with homogeneous chemical, textural and nutrient properties. This advantage over native plants, many of which fare better in heterogeneous substrates, can allow this grass to alter the successional pathways of native vegetation (Wells et al., 2008). Conversely, the establishment of native plant communities of sufficient biodiversity and species richness may create a competitive environment more likely to resist invasion by exotic plant species (Kennedy et al., 2002).

Post-dam-removal sediment is not a comparable substrate to developed soil which supports most plant communities. Therefore, no existing plant communities or associations can serve as models on which to base the species composition of the ONP Lake Mills revegetation efforts (Dave Allen, 2012, personal communication). For this reason, testing a variety of native vegetation types may be equally important to invasive weed control in the dewatered Lake Mills reservoir.

### 3.2 Research Question

The post-dam-removal sediment remaining in the dewatered Lake Mills reservoir affords an environment which is potentially inhospitable to native plant colonization. Establishment of native riparian vegetation is an important component of the restoration of a riverine ecosystem (Chenoweth et al., 2011). The research question guiding this study centers around which woody plants, native to the lower Elwha River, show the greatest survivability in the post-dam-removal sediments of Lake Mills. Does species selection influence woody plant survival in specific sediment textures? An additional research question focuses on the revegetation treatments employed by ONP. Do planting adjacent to woody debris or adjacent to other vegetation affect plant performance? Answering these questions may aid in the selection of suitable species for future planting efforts at the former Lake Mills reservoir site. Additionally, these findings may help set the precedent for future post-dam-removal plant restoration efforts.

## IV. Methods

### 4.1 Study Sites

The purpose of this study was to determine whether certain woody plant species perform better than others in Lake Mills post-dam-removal sediments, and additionally whether proximity to WD and other vegetation enhances plant performance. To meet the objectives of this study, the survival and growth performance of six native woody plant species were compared in three different Lake Mills sediment textures: 1) fine sediment: composed of alluvial silt and clay; 2) coarse sediment: composed of cobbles, gravel and sand; and 3) a mixed sediment composed of silt, clay and sand. The restoration plots assessed in this study lie on the southern, southwestern, and southeastern shores of the former Lake Mills reservoir, numbered Site 1, Site 2, and Site 3, respectively (**Map 1**). Site 1, the southernmost study site containing coarse sediment, lies on a bench at the Boulder Creek delta of the former Lake Mills. Here, several planted sites totaling 0.30 ha were included in the study. Sites 2 and 3 lie on the southwestern and southeastern shores, totaling 0.61 ha and 0.20 ha, respectively. Depending upon aspect, each site is subject to varying degrees of wind exposure, with sites 1 and 3 experiencing especially windy conditions. The interactive effects of heavy wind, low soil moisture, and high temperature can negatively influence the performance of woody plants (Heiligmann and Schneider, 1974).

Three planting prescriptions, with different planting densities and lifeforms, were assessed in this study (**Table 1**). Site 1 plots in this study include three, 0.10-ha plantings. Two of these plots were planted at a density of 8,000 plants per acre of shrubs only, totaling 12,000 plants (prescription 1). The third plot, east of and separated from the other two plots by a small gully, contains trees only at 3,000 plants per acre (prescription 3).


**Map1.** Lake Mills woody plant restoration 2012 study sites

The majority of plants monitored lie in Site 2, south of Stukey Creek and 0.53 km north of Boulder Creek. In this 0.65-hectare site consisting of fine sediment, six, 0.10-hectare plots were planted with trees, shrubs and forbs, at a density of 9,000 plants per acre (prescription 2); the other three plots were planted according to prescription 3. Site 3, directly across the drained reservoir from Site 2, is 0.20 ha in size, with a sediment texture of mixed fine and sandy sediment. At this site, two 0.10-ha plots were planted according to prescriptions 2 and 3. The plot planted according to prescription 2 lies downslope from and west/northwest of the plot planted according to prescription 3, in a terrace formation.

**4.2 Selection of Woody Plants and Prescriptions**

We selected woody plant species to include in this study by reviewing primary literature, consulting plant ecologists, and basing our decisions on planting trials conducted by ONP

| Site | Location | Site Size | Planting Density/ lifeforms | Expos. | Sediment Texture | Prescriptions Applied |
|---|---|---|---|---|---|---|
| **1** | **South** (Delta) | 0.60 ha (3, 0.20-ha plots) | **8,000/acre** Shrubs only | N | **Coarse** (Gravel, cobbles, sand) | **1, 3** |
| **2** | **Southwest** | 0.60 ha (6, 0.10-ha plots) | **9,000/acre** Trees, shrubs, herbs | NE | **Fine** (Silt, clay) | **2, 3** |
| **3** | **Southeast** | 0.20 ha (2, 0.10-ha plots) | **3,000/acre** Trees only | NW | **Mixed** | **2, 3** |

**Table 1.** Study sites and planting prescriptions, Lake Mills 2012 survivability study

during the fall of 2010. The following plants were selected for this study (**Table 1**): oceanspray (*Holodiscus discolor*), Nootka rose (*Rosa nutkana*) (ONP Planting trials, 2011), thimbleberry (Michel et al., 2011), western redcedar (D'Amore et al., 2009), black cottonwood (*Populus balsamifera* ssp. *trichocarpa*) (Naiman and DeCamps, 1997, 2005; Naiman et al., 2010), and Douglas-fir (*Pseudotsuga menziesii*) (Joshua Chenoweth and Dave Allen, personal communications).

Ocean spray and Nootka rose were selected due to their performance in experimental sediment planter boxes installed at ONP's Matt Albright Native Plant Center near Agnew, WA. In this experiment, three substrates were placed in multi-celled wooden planter boxes with the following substrates: 1) coarse sediment and 2) fine sediment, both collected from the northern Lake Mills shoreline neighboring the Glines Canyon dam; and 3) a potting soil control. Oceanspray and Nootka rose plugs were planted in the cells, tagged and observed over time for survival, mortality and growth performance. The majority of these shrubs survived, with only two oceanspray mortalities which likely resulted from sudden freezing temperatures at the ONP nursery site (Joshua Chenoweth, personal communication). Thimbleberry was selected for this study due to its tolerance of a variety of substrates and moisture regimes, general hardiness in riparian environments, dense growth pattern, ability to spread via rhizome and seed, as well as its successful germination and growth in fine sediment in a recent seed rain study (Michel et al., 2011).

Trees selected in this study included both late and early-successional riparian species. Western redcedar was selected due to its abundance in the lower Elwha River and tolerance of relatively inundated growing conditions (D'Amore et al., 2009; Naiman et al., 2010). Additionally, cedar trees can grow in nitrogen-poor substrates, and contribute calcium-rich litterfall which may increase N turnover over time by increasing pH (D'Amore et al., 2009). The major challenge for this tree may be its survival during drier periods, particularly in fine sediment. Douglas-fir seedlings were abundant in plantings due to highly successful germination, and this tree is prolific in late-seral forests along the lower Elwha River (Wendel and Zabowski, 2010). While Douglas-fir is a later-successional tree in riparian environments, the dynamic and transitional nature of the restoration sites warrants its inclusion in order to test a wide range of species.

Black cottonwood was selected due to its survivability in a variety of disturbed environments and substrates. Cottonwoods and willows tend to establish in flood-prone exposed sites, but impoundment of the Elwha River has reduced the extent and frequency of flood disturbance conducive to creating such appropriate environments (Shafroth et al., 2002). Cottonwood trees can perform well in poorly drained hypoxic soils, largely due to the development of adventitious roots (Van DerKamp and Gokhale, 1979; Hanson, 1997; Naiman and Decamps, 1997; Haycock et al., 2003; Naiman et al. 2010). Black cottonwood trees and willows of the family Salicaceae are phreatophytes, possessing extensive, deep root systems capable of accessing riparian water tables during dry periods (Naiman, Decamps and McClain, 2005). The roots of these trees grow especially fast as soil moisture diminishes, in some cases over 38 cm within the first two months (Naiman, Decamps and McClain, 2005). Establishment of *Populus* varieties can be enhanced in environments where the spring season brings high levels of moisture or inundation followed by dry summer seasons (Naiman, et al., 2005).

In addition to their survivability in extreme conditions, black cottonwoods can provide many ecological benefits in stream and riparian restoration. Cottonwood trees are known for their ability to colonize unfruitful, inundated or semi-arid riparian habitats that may hinder initial establishment of other trees, creating a facilitative canopy cover, and also possess resprouting capabilities when water movement shears saplings and transports them downstream (Rood et al., 2003; Gurnell et al., 2005). Nitrogen-fixing bacteria housed in the stems of cottonwoods allow these trees to produce nitrogen even in extremely nitrogen-poor soils (Doty et al., 2009; Gang Xin et al., 2009). Additionally, cottonwood may play an essential role in post-dam-removal floodplain and perched terrace stabilization, and contribute LWD to the Elwha River. The short, 80-150 year lifespan and rapid growth of this tree species enables a more rapid production of LWD available to streams than that of longer-lived native conifers, and larger WD than that of other short-lived early successional broadleaf trees such as red alder (Collins and Montgomery, 2002; Chenoweth et al., 2011).

Red alder, while included in the 2012 ONP Lake Mills plantings (**Table A1, Appendix A**) and currently more common than black cottonwood in floodplain environments of the lower Elwha River, may not be as well-suited as other early-successional riparian trees to post-dam-removal fluvial geomorphology (Shafroth et al., 2002; Chenoweth et al., 2011). Therefore, this tree was not included in the 2012 woody revegetation study. Large quantities of sediment, carried and deposited by the river, may create hypoxic burial and inundation states which can be inhospitable to red alder survival, as these trees have shown poor performance in fine, hypoxic sediments (Shafroth et al., 2002). Additionally, red alders lack the deep rooting systems of cottonwoods and willow crucial to accessing increasingly lowered water tables. In 2003 seeding trials conducted by ONP, red alder did not survive beyond one growing season (Chenoweth et al., 2011).

### 4.3 Experimental Design

Plantings assessed in this study were installed by the ONP Elwha Revegetation Crew and Washington Conservation Crew (WCC) according to a technique which facilitates shading, moisture retention, nutrients and protection from herbivory, known as a facilitation patch strategy (del Moral and Wood, 1993; Chenoweth et al., 2011). Dense islands of trees and/or shrubs were installed, followed in certain treatments by the interplanting of less dense corridors of woody plants, forbs and graminoids. Dense placement of woody plants affords a nurturing microsite which may aid in the survival and establishment of other plantings and volunteer seeds, and eventually may serve as a seed source in exposed sediments >50 m from forest edge (Chenoweth et al., 2011). The shed leaves and litter of the planted shrubs in turn contribute organic matter to the litterdepauperate sediment, providing a more suitable environment for seed establishment. Eventually, these islands of woody plants may reach a state of plant species competition leading to natural selection of the most site-suitable species (Chenoweth et al., 2011).

Additional techniques of ONP planting strategies entail plant placement. Plants were placed among onsite woody debris when possible, and “armored” thorny shrubs, less palatable to herbivores, were strategically placed around plants more desirable to herbivores (Padilla and Pugnaire, 2006). Western redcedar and berry-producing shrubs in particular require these or similar protection measures. Over time, seeds of established berry-producing plants may be distributed throughout the exposed reservoir through zoochory (Chenoweth et al., 2011). The three planting prescriptions tracked in this study are different variations of facilitation patches in three southern Lake Mills locations. Plantings were positioned at least 50 m from the edge of the intact forest. Within this distance to forest edge, seed rain from established trees and vegetation aide in natural plant regeneration, negating a need for artificial regeneration efforts (Chenoweth et al., 2011; Michel et al., 2011). Predictive models in previous vegetation studies depicted a drastic decline in seed dispersal beyond 170 m from forest edge (Greene and Johnson, 1996).

### *Field Study Design*

For each of the 6 woody species tracked in this study, 30 replicates per 0.10-ha restoration plot were tagged, totaling N=860 individual trees across all sites. The exception was western redcedar, for which only 20 plants were installed in each plot (**Table 2**). The field component of this study required two major stages: 1) selection and marking of replicates, followed by 2) monitoring throughout the growing season. Plants

| Woody Plants Selected for 2012 Lake Mills Survivability Study | | | | |
|---|---|---|---|---|
| **Species** | **Totals** | **Totals Site 1** | **Totals Site 2** | **Totals Site 3** |
| Oceanspray (*Holodiscus discolor*) | 180 | 60 | 90 | 30 |
| Nootka rose (*Rosa nutkana*) | 120 | 0 | 90 | 30 |
| Thimbleberry (*Rubus parviflorus*) | 180 | 60 | 90 | 30 |
| Western redcedar (*Thuja plicata*) | 80 | 0 | 60 | 20 |
| Douglas fir (*Pseudotsuga menziessii)* | 150 | 30 | 90 | 30 |
| Black cottonwood (*Populus balsamifera*) | 150 | 30 | 90 | 30 |
| ***Grand Total =*** | **860** | **180** | **510** | **170** |

**Table 2.** Native plant species assessed in Lake Mills study, and totals number of individuals for each species per site

assessed in this study were selected using random number generators for azimuth and distance, and marked with numbered aluminum tags. Plant performance assessments included survival, qualitative vigor (leaf coloration, size and shape), and minimal growth performance measurements. The temporal scale of this study (i.e., one growing season) is likely not sufficient for significant measurable plant growth. Growth can be underestimated because bare root and potted plants, when newly installed into developed soil, invest much of their initial energy in root growth rather than foliar growth during the first 1 to 2 growing seasons, and may exhibit lower root growth rates than those of cuttings (Alpert et al., 1999). The Lake Mills restoration sites lack surficial developed soil and pose yet a greater challenge for plant growth beyond root establishment. It may require at least two to three growing seasons for substantial between-site growth differences to be significant (Woodruff et al., 2002). Nevertheless, the height and diameter or lateral growth of 155 tagged plants were randomly measured using a meter stick and dial caliper (**Table 17, Appendix B**). To establish baseline data, measurements were initially conducted in April 2012, early in the growing season, and repeated in late

September 2012. Diameter measurements were taken from 5 cm aboveground.

In the process of plant selection and tagging, vegetation and woody debris (WD) within $1m^2$ of replicates were documented (Eränen and Kozlov, 2007). Vegetation in close proximity to plantings may result in either beneficial or harmful interactions (Callaway, 1995). Competition for moisture and space can occur between adjacent plants both above and belowground, particularly during dry periods or other environmental conditions that intensify plant stress (DeSteven, 1991; Roberts et al., 2005; Smit et al., 2006; Sthultz et al., 2007). Conversely, the presence of the neighboring plants may serve as "nurse" vegetation, aiding in shading, moisture retention or provide a nurturing microclimate for certain species (Coffman, 1975; del Moral and Wood, 1993; Callaway, 1998; Padilla and Pugnaire, 2006). For newly introduced seedlings which are not adapted to an open, disturbed environment, shelter of adjacent plants may be especially beneficial (DeSteven, 1991). WD provides moisture retention, a shaded microclimate and nutrients to adjacent vegetation (Naiman et al., 2010). The moisture retention neighboring plants receive from WD is of potentially greater benefit than that provided by adjacent vegetation, as WD creates no interspecific competition (Coffman, 1975). LWD in particular can serve as nurse logs, providing the aforementioned benefits as well as elevating plants which are less tolerant to inundated conditions (Fetherson et al., 1995). Finer woody debris, particularly litter, can potentially produce a positive or negative effect on seed and seedling establishment, contributing to competition for light and moisture or moisture retention and nutrient competition (Goodson et al., 2003). Woody debris encountered was classified as: coarse (C) = diameter of 10 cm or greater; fine (F) = diameter of <10 cm; and litter (L) = diameter of 1 cm or less.

### 4.4 Onsite Sediment Analysis

Post-dam-removal sediment may create an inhospitable environment for plant growth in comparison to developed soil. As a means of measuring environmental stress and testing the hypothesis that substrate influences plant performance, three abiotic factors were measured for sediment at each site: 1) moisture using two different methods; 2) particle size; and 3) nutrient analysis. Soil moisture, coupled with texture, particle size and nutrient content, can largely impact plant survival and growth (Sthultz et al., 2007), particularly potted and bare root specimens. The Lake Mills post-dam-removal sediments are subject to greater seasonal moisture extremes than are developed soils.

Sediment moisture was assessed every 3 to 4 weeks using a Quick Draw 2900F1 portable tensiometer (Soil Moisture Corp, Santa Barbara, CA). Tensiometers measure the capillary force or suction which roots must exert in order to draw water from soil particles at different moisture contents and in a wide range of sediment textures. In other words, these instruments measure the amount of moisture available to be utilized for plant growth in units of positive or negative pressure (Soil Moisture Equipment Corp, 2012). With measurements in centibars (1 centibar = 1/100$^{th}$ of a bar, 1 kilopascal or 0.14 pounds per square inch (PSI)), they reflect plant root stress within 1.5% accuracy throughout the growing season while affording complete onsite assessment Stoeckler and Aamodt, 1940; Robertson et al., 1999). Adhesion of soil particles to water varies with soil pore size and texture. Soil particles possess an attraction to water contained the soil; the finer the soil particles, the greater their binding to water molecules, and the lower the availability of water to plant roots (Soil Moisture Corp, May 2012). Finer sediments, while inundated during wetter periods of a growing season, provide less available water for plants during drier seasonal periods, since the water contained therein is “bound” (Stoeckler and Aamodt, 1940; Robertson et al., 1999). To gather heuristic baseline sediment moisture data, gravimetric water content (GWC) was also assessed with collection of sediment cores from each moisture sample site, at 20 cm depths, in May and late September (Black, 1965).

Sediment moisture sampling sites were divided among planted sites, with 10 to 11 sample sites in each 0.10-ha restoration plot ($n$ = 111) (Gotelli and Ellison, 2004). The tensiometer probe was placed at depths of 20 cm to measure available moisture (Roberts et al., 2005). High variability between and within plots and planting sites required a large sample size for moisture measurement, and this number proved to be the maximum number feasible in relation to time constraints. Moisture sampling sites were randomly selected using the same technique as for replicate tagging (**Appendix A**). In addition to moisture assessment, pH was recorded at each sample site through the use of a portable Kelway® soil pH and moisture meter (Kel Instruments, Inc., Wyckoff, NJ).

### 4.5 Vegetation Monitoring

Monitoring, conducted from May through September of the 2012 growing season, was a vital component of assessing plant survival, performance and changes over time. The primary aspect of plant performance monitored in this study was the survival or mortality of tagged plants in fine, coarse and mixed fine-coarse sediment textures. Visual inspection of live biomass on each plant was the primary method for determination of survival or mortality (Schaff and Pezeshki, 2003; Greer et al., 2006). Mortality was indicated by a lack of foliage, all or mostly dead branches and twigs, and scratching trunks or stems to inspect for green tissue if necessary. It was assumed in this study that plants which did not produce foliage by late summer or fall, beyond the peak of the growing season, did not survive; however, future monitoring will determine whether this assumption was correct.

Plant vigor was assessed qualitatively through leaf color, leaf shape (i.e., curled under, curled up or other deformation) and rough size (“full” or “diminished”). Leaf color can be linked to nutrient availability in a given substrate, or environmental conditions. In particular, nutrient limitation may have a greater influence and be more visible in younger, early successional plants (Chapin et al., 1986). Nitrogen is especially critical to riparian plant and ecosystem development (Bradshaw, 1982). Yellow leaves may indicate chlorosis from a lack of nitrogen, while red leaf color may indicate sun stress (Stewart, 1999). Leaf pigmentation results from anthocynanin production, which is associated with the protection of photosynthetic structures in leaf tissue during periods of excessive light exposure. Anthocyanin production has also been associated with environmental stresses such as drought or nutrient limitation (Hoch et al., 2000; Hoch et al., 2003), two key factors that may affect plant growth in the dewatered Lake Mills restoration sites. The water-soluble pigments created in anthocyanin production responsible for red leaf coloration may aid leaves in water retention during periods of drought (Chalker-Scott, L., WSU). “Scorched” leaf edges or spots can be indicative of low soil potassium levels (Stewart, 1999). Additionally, leaf scorch can result from drought, drying winds or poor root development (Purdue University Plant and Pest Diagnostic Laboratory, 2002). Herbivory, an additional factor affecting plant performance, was also documented during monitoring. Signs of plant predation such as chewed or broken off stems or leaves were recorded, including signs of insect herbivory on leaves (i.e., holes or skeletonization).

### 4.6 Laboratory Analysis

#### ***Sediment Moisture and Nutrient Analysis***

Sediment cores were collected at all moisture sample sites in May and late September at depths of 20 cm (Henderson et al., 1989; Roberts et al., 2005). To assess gravimetric water content (GWC), sediment samples were sieved and dried at 70°C for 72 h; samples were weighed after 48 h of drying, then at 72 h to ensure samples had completely dried. Ideally, soil samples are dried at the standard 105°C temperature (Black, 1965). However, two soil ovens were required to dry a large number of samples with limited time constraints, and one of these ovens reaches a maximum temperature of only 70°C. In order to be consistent with methodology, this temperature had to be applied to all sediment samples.

Using an AQ1 discrete autoanalyzer (Seal Analytical Inc., Mequon, Wisconsin) a subset (N=34) of the fall 2012 sediment cores collected from moisture sample sites were tested for nitrate-N and phosphate-P concentrations (mg N/L and mg P/L). Ten g of wet sediment were extracted into 100 ml of potassium chloride (KCl). Bottles were shaken for 1 h and allowed to settle overnight. The supernatant was then filtered through KCl-infused Grade 42 Whatman filters to create an extract, and frozen until analysis (Mulvaney, R.L., 1996). Extracts were tested for nitrate-N and phosphate-P using the AGR 31-B Nitrate-N+Nitrite-N and AGR-204-B Orthophosphate-P in 2 M KCl Extracts of Soil methods, respectively. Reagents employed included 2% copper sulfate ($C_uSO_4$), a high standard reagent, KCl, and a nitrate buffer.

Sediment particle size can shape organic matter-derived nutrient retention capabilities during early soil development (Naiman et al., 2010). In addition to moisture and nutrient analysis, sediment particle size was characterized from sixteen randomly selected subsamples of the Lake Mills sediment. Nine of these sediment samples were assessed using the hydrometer method (Bouyoucos, 1962; Gee and Bauder, 1986). A Bouyoucos ASTM No. 152H hydrometer with Bouyoucos scale in g/L was employed to measure clay, silt and fine sand (H-B Instrument Company, Trappe, PA). Due to a very low presence of organic matter in all study sites, sediment samples were not pretreated with a hydrogen peroxide ($H_2O_2$) digestion. Seven coarse samples from Site 1 were assessed through dry sieving to separate gravel (particles >2mm), sand (0.05 - 2 mm), silt (0.002-0.05 mm) and clay (<0.002 mm) (Caplan et al, 2006).

### 4.7 Statistical Analysis

The categorical and continuous nature of the data collected in this study and the variables hypothesized to influence plant performance required the use of both parametric and non-parametric analytical methods. R and JMP software were utilized to conduct statistical analysis. Categorical variables included survival or mortality, plant species, planting in woody debris, presence of other vegetation within $1m,^2$ leaf color, and sediment type. Continuous data consisted of plant growth, sediment moisture, percent mortality and pH. Therefore, Pearson’s Chi-square tests for independence were appropriate to test for interactions between plant survival and treatments. Levene’s tests for equality of variance and Shapiro-Wilk’s tests of normality for plant growth data revealed non-normal datasets, inappropriate for analysis of variance (ANOVA) tests (Alpert et al., 1999). Plant height growth by sediment type, moisture, and pH were analyzed by performing Kruskal-Wallis rank sum tests. Percent leaf color and percent plant mortality were transformed using an acrsine-square root transformation to perform one-way ANOVAs. Total leaf color by species and sediment type were also examined using one-way ANOVAs, to detect relationships between leaf color, plant species and sediment type. Tensiometer and GWC data were analyzed with repeated-measures ANOVAs, testing the influence of sediment texture and month on moisture availability and GWC. Percent moisture data from GWC assessments were transformed through arcsine square root transformations prior to these tests.

## V. Results

### 5.1 Site Conditions

Overall, weather during the Lake Mills 2012 growing season remained cool and wet until the months of late July through September (**Table 3**). By October, rainfall resumed and temperatures dropped by nearly 9° C (http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?wa2548). Windy conditions were frequent at all sites. The Boulder Creek delta (Site 1) received especially high gusts during 2012 surveys. Archival 2009 and 2010 wind data collected from an anemometer formerly installed on the Glines Canyon Dam, reported average wind speeds of 10 kph, with wind speeds periodically reaching speeds of greater than 29 kph (USGS data, courtesy of ONP GIS office, 2012).

### 5.2 Plant performance

Overall, mortalities of tagged plants were low at all sites during the 2012 growing Season (**Table 4**); however, variation was detected in plant performance among species and sites. During the month of May, a total of 30 plants had died (mortality = 23

| Lake Mills Weather Data Spring through Fall 2012 | | | | | |
|---|---|---|---|---|---|
| **Month** | **Average Max Air Temp (C)** | **Average Min Air Temp (C)** | **Average Mean Air Temp (C)** | **2012 Rainfall (cm)** | **Average Rainfall (cm)** |
| Apr | 13.4 | 3.3 | 8.4 | 9.2 | 8.5 |
| May | 16.4 | 4.8 | 10.6 | 3.3 | 4.5 |
| Jun | 17.1 | 6.9 | 12.0 | 6.3 | 3.1 |
| Jul | 22.2 | 9.8 | 16.0 | 2.0 | 1.9 |
| Aug | 23.9 | 10.7 | 17.4 | 0.2 | 2.8 |
| Sep | 22.2 | 8.5 | 15.3 | 0.3 | 4.4 |
| Oct | 13.5 | 5.1 | 9.3 | 20.1 | 13.3 |

**Table 3**. Lake Mills 2012 weather data, taken from the Elwha Ranger Station Weather Station

| | Site 1 – Coarse Sediment | | | Site 2 – Fine Sediment | | | Site 3 - Fine/Sandy Sediment | |
|---|---|---|---|---|---|---|---|---|
| **Species** | **Survival** | **Mortality** | **Species** | **Survival** | **Mortality** | **Species** | **Survival** | **Mortality** |
| HOLDIS | 57 | 3 | HOLDIS | 89 | 1 | HOLDIS | 30 | 0 |
| RUBPAR | 59 | 1 | RUBPAR | 89 | 1 | RUBPAR | 30 | 0 |
| *ROSNUT | _ | _ | ROSNUT | 87 | 3 | ROSNUT | 30 | 0 |
| *THUPLI | _ | _ | THUPLI | 56 | 2 | THUPLI | 17 | 0 |
| PSEMEN | 12 | 18 | PSEMEN | 57 | 33 | PSEMEN | 27 | 3 |
| POPBAL | 30 | 0 | POPBAL | 90 | 0 | POPBAL | 29 | 1 |
| *Total* | **158** | **22** | *Total* | **470** | **40** | *Total* | **166** | **4** |

**Table 4.** Survival and mortality by species and site. *Nootka rose and western redcedar were not planted in Site 1. HOLDIS=oceanspray, RUBPAR=thimbleberry, ROSNUT= Nootka rose, THUPLI=western redcedar, PSEMEN=Douglas-fir, POPBAL=black cottonwood

Douglas-fir, 4 oceanspray, 3 Nootka rose). During the month of July, mortality had increased to 43 (additional mortality = 10 Douglas-fir, 1 thimbleberry, 2 western redcedar). By August, tagged plant mortality reached 59 (additional mortality = 14 Douglas-fir, 1 cottonwood, 1 thimbleberry). In the final September 2012 monitoring, plant mortality totaled 66 for all sites, with the remaining 7 mortalities occurring among Douglas-fir seedlings. Of the 6 species selected for this study, Douglas-fir had the highest mortality rate at 36% of tagged Douglas-fir seedlings and 82% of total tagged plant mortalities. Proportionally, the highest percentage of plant mortalities occurred in Site 1 (coarse sediment), at 12% mortality of onsite tagged plants, compared to 8% in Site 2 (fine sediment) and 2% in Site 3 (fine/sandy sediment). By individual species, Douglas-fir, oceanspray and thimbleberry experienced the highest mortality, relative to individual site sample size, in the coarse sediment (**Table 5**). Nootka rose and western redcedar mortalities occurred only in the fine sediment (these species were not planted in the coarse prescriptions tracked in this study), while the only black cottonwood mortality occurred in the fine/sandy sediment, possibly as a result of browsing.

| Lake Mills 2012: Percent Mortality for Plant Species by Individual Site | | | | | | |
|---|---|---|---|---|---|---|
| | **HOLDIS** | **ROSNUT** | **RUBPAR** | **POPBAL** | **THUPLI** | **PSEMEN** |
| **Coarse** | 5% | N/A | 2% | 0% | N/A | 60% |
| **Fine** | 1% | 3% | 1% | 0% | 3% | 36% |
| **Fine/Sandy** | 0% | 0% | 0% | 3% | 0% | 10% |

**Table 5.** Plant species percent mortalities, proportional to sample size, by individual sites

According to Pearson's Chi-Square tests for independence for individual study sites, sediment type and plant species significantly influenced plant survival for sites 1 and 2, but not for Site 3 in the mixed sediment (**Table 6)**. Adjacent woody debris showed significant influences only in the fine sediment (Site 2), while woody debris size was not significant to survival in any of the study sites. Adjacent vegetation significantly influenced survival in the fine and fine/sandy mixed sites, while browsing Significantly influenced plant survival in the coarse and fine sediment types. Pearson's Chi-Square tests for independence for combined study sites by month revealed significant

| Pearson's Chi-Square Test for Independence, Individual Sites: Plant Survival by Treatments, Browsing | | | | | |
|---|---|---|---|---|---|
| **Sediment** | **Species** | **WD** | **WD Size** | **Other Veg** | **Browsed** |
| **Coarse** | $X^2$ = 77.16<br>df = 3<br>**p-value = <0.0001** | $X^2$ = 1.89<br>df = 2<br>p-value = 0.39 | $X^2$ = 5.84<br>df = 5<br>p-value = 0.32 | $X^2$ = 0.87<br>df = 1<br>p-value = 0.35 | $X^2$ = 11.61<br>df = 2<br>p-value = **0 .003** |
| **Fine** | $X^2$ = 135.86<br>df = 10<br>**p-value = <0.0001** | $X^2$ = 233.86<br>df = 6<br>p-value = **<0.0001** | $X^2$ = 14.67<br>df = 12<br>p-value = 0.26 | $X^2$ = 341.07<br>df = 4<br>p-value =**<0.0001** | $X^2$ = 19.88<br>df = 2<br>p-value = **<0.0001** |
| **Fine/sandy** | $X^2$ = 13.88<br>df = 10<br>p-value = 0.18 | $X^2$ = 4.53<br>df = 4<br>p-value = 0.34 | $X^2$ = 1.77<br>df = 10<br>p-value = 1.00 | $X^2$ = 9.11<br>df = 2<br>**p-value = 0.01** | $X^2$ = 1.06<br>df = 2<br>p-value = 0.59 |

**Table 6.** Chi-Square table: plant survival compared to treatments and browsing by sediment

| Pearson's Chi-Square Test for Independence, Combined Sites: Plant Survival by Treatments, Browsing | | | | | |
|---|---|---|---|---|---|
| **Month** | **Species** | **WD** | **Vegetation 1 m$^2$** | **Sediment type** | **Browsed** |
| **May** | $X^2$ = 12.31<br>df = 10<br>p-value = 0.27 | $X^2$ = 6.66<br>df = 4<br>p-value = 0.18 | $X^2$ = 28.43<br>df = 4<br>p-value = **<0.0001** | $X^2$ = 13.00<br>df = 4<br>**p-value** = **0.01** | $X^2$ = 5.47<br>df = 2<br>p-value = 0.06 |
| **July** | $X^2$ = 38.30<br>df = 15<br>**p-value = 0.001** | $X^2$ = 4.68<br>df = 6<br>p-value = 0.67 | $X^2$ = 31.01<br>df = 6<br>**p-value = <0.0001** | $X^2$ = 23.50<br>df = 6<br>p-value = **0.0006** | $X^2$ = 7.96<br>df = 3<br>p-value = **0.05** |
| **August** | $X^2$ = 41.41<br>df = 20<br>**p-value = 0.003** | $X^2$ = 3.45<br>df = 8<br>p-value = 0.90 | $X^2$ = 862.06<br>df = 8<br>**p-value = <0.0001** | $X^2$ = 26.02<br>df = 8<br>**p-value = 0.001** | $X^2$ = 10.00<br>df = 4<br>p-value = **0.04** |
| **September** | $X^2$ = 19.96<br>df = 20<br>**p-value= 0.03** | $X^2$ = 2.79<br>df = 4<br>p-value = 0.59 | $X^2$ = 344.59<br>df = 4<br>**p-value = <0.0001** | $X^2$ = 23.31<br>df = 4<br>**p-value = 0.0001** | $X^2$ = 10.22<br>df = 2<br>p-value = **0.006** |

**Table 7.** Chi-Square table, plant survival compared to treatments for all sites

differences in plant survival by species and adjacent vegetation for the months of July through September (**Table 7**), while woody debris did not significantly influence survival for any month of the growing season. The influences of sediment type and browsing on plant survival were significant for all or most months of the 2012 growing season.

Leaf color varied among the three Lake Mills study sites, but yellow and red leaf coloration increased throughout the growing season in all sites. Leaves were primarily green during the first assessment in May, with red and yellow leaf coloration making up only 3 and 7% of all plants examined, respectively (**Table 8**). By August, red leaf coloration reached its peak at 54% of the total studied plant population and highest in the coarse sediment, declining slightly by the late September assessment. Overall yellow leaf coloration rose to 45% by late September, with the highest percent in the fine/sandy sediment. Scorched leaf edges, leaf spotting, leaf curling and diminished leaf size also increased throughout the growing season for most plant species in all sites (**Table 9).**

| Month | % Yellow Leaf Color By Sediment Texture | | | % Red Leaf Color By Sediment Texture | | |
|---|---|---|---|---|---|---|
| | **Coarse** | **Fine** | **Fine/Sandy** | **Coarse** | **Fine** | **Fine/Sandy** |
| **May** | 0% | 3% | 2% | 0% | 7% | 0% |
| **July** | 18% | 33% | 46% | 58% | 44% | 24% |
| **August** | 32% | 34% | 50% | 61% | 45% | 55% |
| **September** | 41% | 38% | 55% | 56% | 26% | 22% |

**Table 8.** Yellow and red leaf coloration (% of total tagged plants) by sediment

Percent leaf color by species and month varied considerably. By the end of the growing season, Nootka rose and western redcedar displayed the highest percentage of yellow leaf coloration at 70% and 60% (**Table 9).** Highest red leaf coloration was observed in Douglas-fir and thimbleberry, both at 59% by fall 2012. Thimbleberry displayed the greatest percent scorched leaf edges (71%) and leaf deformation (69%); oceanspray the

**2012 Lake Mills Qualitative Leaf Data by Species**

| Species | Month | % Yellow Lvs. | % Red Lvs. | % Scorched Edges | % Spotting | % Lvs. Deformed | % Lvs. Diminished |
|---|---|---|---|---|---|---|---|
| **Oceanspray** | May | 0 | 7 | 0 | 0 | 0 | 0 |
| | Jul | 22 | 63 | 3 | 32 | 6 | 69 |
| | Aug | 30 | 52 | 25 | 54 | 11 | 69 |
| | Sept | 33 | 52 | 30 | **66** | 27 | 30 |
| **Black cottonwood** | May | 2 | 1 | 0 | 0 | 0 | 0 |
| | Jul | 19 | 22 | 0 | 9 | 15 | 96 |
| | Aug | 21 | 15 | 2 | 24 | 23 | 94 |
| | Sept | 39 | 10 | 1 | 22 | 29 | 5 |
| **Douglas-fir** | May | 5 | 8 | 0 | 0 | 0 | 0 |
| | Jul | 50 | 45 | 0 | 1 | 23 | 41 |
| | Aug | 33 | 63 | 0 | 3 | 27 | 45 |
| | Sept | 39 | **59** | 1 | 3 | 32 | **42** |
| **Nootka rose** | May | 3 | 0 | 0 | 0 | 0 | 0 |
| | Jul | 55 | 18 | 1 | 3 | 8 | 85 |
| | Aug | 63 | 14 | 13 | 35 | 8 | 83 |
| | **Sept** | **70** | 18 | 23 | 39 | 15 | 14 |
| **Thimbleberry** | May | 0 | 3 | 0 | 0 | 0 | 0 |
| | Jul | 24 | 57 | 8 | 23 | 45 | 51 |
| | Aug | 34 | 60 | 57 | 31 | 62 | 57 |
| | Sept | 35 | 59 | **71** | 43 | **69** | 41 |
| **Western redcedar** | May | 0 | 10 | 0 | 0 | 0 | 0 |
| | Jul | 40 | 44 | 1 | 0 | 8 | 79 |
| | Aug | 44 | 39 | 3 | 3 | 9 | 79 |
| | Sept | 60 | 29 | 4 | 24 | 11 | 16 |

**Table 9.** Qualitative Leaf data by plant species and month

| One-Way Analysis of Variance: Leaf Color and Mortality by Month, Sediment | | | |
|---|---|---|---|
| | **Red Leaves** | **Yellow Leaves** | **Mortality** |
| **Month** | F = 7.99<br>DF = 3<br>**P = 0.009** | F = 19.30<br>DF = 3<br>**P = 0.0005** | F = 0.31<br>DF = 3<br>P = 0.82 |
| **Sediment type** | F = 0.13<br>DF = 2<br>P = 0.88 | F = 0.44<br>DF = 2<br>P = 0.65 | F = 32.23<br>DF = 2<br>P = **<0.0001** |
| **Species** | F = 1.10<br>DF = 5<br>P = **0.006** | F = 0.43<br>DF = 5<br>P = 0.74 | F = 3.08<br>DF = 5<br>**P = 0.01** |

**Table 10.** One-way ANOVA, leaf color by month, sediment and species

greatest leaf spotting (66%); and Douglas-fir the highest percentage of diminished needles (41%). During July and August up to 96% of black cottonwood specimens had diminished leaves, 73% of which was due to caterpillar defoliation, but late summer foliar growth reduced this percentage to 5% by late September. Results of One-way ANOVAs indicated that month during the growing season significantly influenced percent leaf color, while sediment type and plant species significantly influenced plant mortality (**Table 10**).

Plant growth increases generally proved minimal, but varied by site (**Table 11; Figure 1**). Measured oceanspray and thimbleberry were browsed in all three sites, resulting in negative or minimal growth trends in the coarse and mixed sediments. By individual site, the fine/sandy sediment site experienced the highest plant growth, with western redcedar, Nootka rose and cottonwood total height growths reaching 285, 127 and 165 mm, respectively (**Table 11)**. Following a Kruskal-Wallis rank sum test for all sites (**Table 12**), species, pH and GWC significantly influenced plant growth while sediment type did not. Testing the relationship between growth of individual plant species, sediment type and pH, sediment significantly influenced oceanspray lateral

**Plant Growth by Species**

| Height (mm) | Lateral or Diameter (mm) |
|---|---|

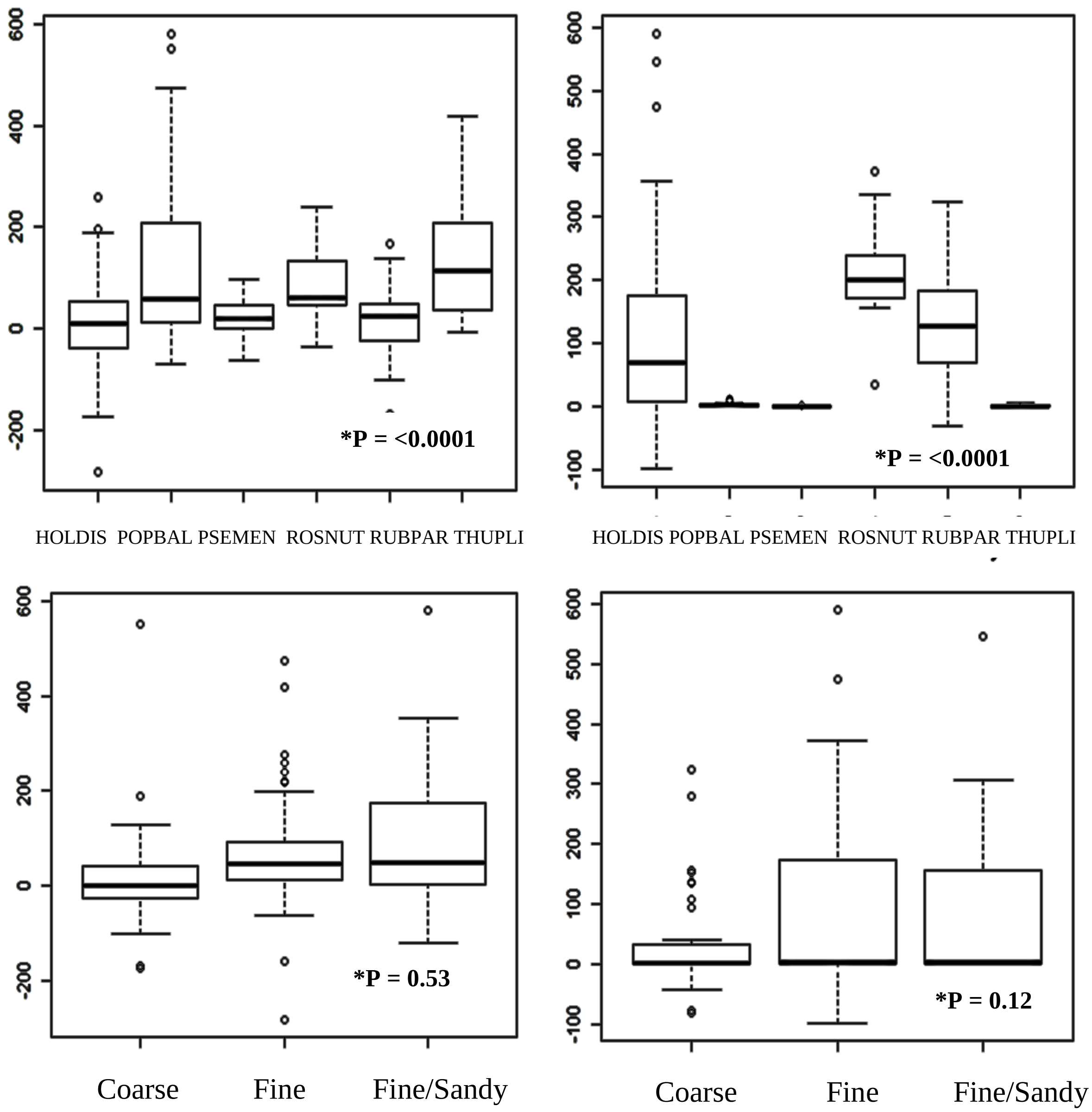


**Figure 1.** Median plant growth by species and sediment type. *Kruskal-Wallis rank sum tests for plant growth by species and sediment texture

| Species | Height (mm) | | Lat. Grwth. (mm) | | Diam. (mm) | | Comments |
|---|---|---|---|---|---|---|---|
| | Mean | Stdev | Mean | Stdev | Mean | Stdev | |
| **Site 1 - Coarse** | | | | | | | |
| *Holodiscus discolor* | -8.00 | 94.72 | -4.25 | 50.01 | ---- | ---- | Browsed |
| *Rubus parviflorus* | -27.92 | 65.49 | 99.41 | 113.95 | ---- | ---- | Browsed |
| *Pseudotsuga menziessii* | 8.00 | 20.30 | ---- | ---- | 0.11 | 0.28 | |
| *Populus balsamifera* | 123.29 | 192.66 | ---- | ---- | 1.04 | 0.61 | |
| **Site 2 - Fine** | | | | | | | |
| *Holodiscus discolor* | 26.00 | 124.28 | 184.31 | 175.09 | ---- | ---- | |
| *Rosa nutkana* | 76.35 | 78.12 | 204.07 | 80.17 | ---- | ---- | |
| *Rubus parviflorus* | 47.93 | 56.97 | 138.07 | 57.21 | ---- | ---- | |
| *Thuja plicata* | 111.84 | 119.16 | ---- | ---- | 0.79 | 1.57 | |
| *Pseudotsuga menziessii* | 19.57 | 32.1 | ---- | ---- | 0.40 | 0.76 | |
| *Populus balsamifera* | 101.38 | 135.98 | ---- | ---- | 1.76 | 1.20 | |
| **Site 3 - Mixed** | | | | | | | |
| *Holodiscus discolor* | -66.33 | 51.47 | 257.67 | 48.21 | ---- | ---- | Browsed |
| *Rosa nutkana* | 127.00 | 79.68 | 252.67 | 79.68 | ---- | ---- | |
| *Rubus parviflorus* | 1.33 | 88.10 | 170.67 | 21.01 | ---- | ---- | |
| *Thuja plicata* | 285.33 | 21.01 | ---- | ---- | 3.22 | 88.10 | |
| *Pseudotsuga menziessi* | 39.75 | 17.07 | ---- | ---- | 0.58 | 17.07 | |
| *Populus balsamifera* | 165.11 | 200.16 | ---- | ---- | 5.52 | 200.164 | |

**Table 11.** Height and lateral (shrub) or diameter (tree) growth of selected studied plants

growth, thimbleberry height, and western redcedar lateral growth. No apparent significant relationship was found between pH and plant growth by individual species. However, GWC appeared to significantly influence plant height growth (**Table 12**).

| Plant Growth | Wilcoxon/Kruskal-Wallis Rank SumTests: Plant Growth by Site Conditions<br>Sediment | Species | pH | Moisture* |
|---|---|---|---|---|
| **Height** | $X^2$ = 4.96<br>Df = 2<br>p-value = 0.53 | $X^2$ = 30.32<br>Df = 5<br>**p-value = <0.0001** | $X^2$ = 30.29<br>df = 14<br>**p-value = 0.007** | $X^2$ = 38.88<br>Df = 26<br>**p-value = 0.05** |
| **Lateral or Diameter** | $X^2$ = 14.28<br>df = 2<br>p-value = 0.12 | $X^2$ = 78.99<br>df = 5<br>**p-value = <0.0001** | $X^2$ = 25.53<br>df = 14<br>**p-value = 0.05** | $X^2$ = 34.19<br>Df = 26<br>p-value = 0.13 |

**Table 12.** Kruskal-Wallis rank sum test data plant growth. *GWC difference

**Lake Mills 2012 Sediment Moisture Averages by Site**

| | March | April | May | June/July | August | September | October | Increase |
|---|---|---|---|---|---|---|---|---|
| | | | **Tensiometer (centibars)** | | | | | |
| **Site 1** | 6.3 | 7.6 | 13.1 | 19.7 | 41.8 | 46.1 | ----- | 35 |
| **Site 2** | 4.7 | 5.7 | 26.6 | 49.2 | 68.6 | 74.7 | ----- | 68 |
| **Site 3** | 9.4 | 10.6 | 18.8 | 20.3 | 43.1 | 50.6 | ----- | 41 |
| | | | **Gravimetric Water Content (%)** | | | | | **Decrease** |
| **Site 1** | ---- | ---- | 0.13 | ----- | ----- | ----- | 0.07 | 6% |
| **Site 2** | ---- | ---- | 0.39 | ----- | ----- | ----- | 0.19 | 20% |
| **Site 3** | ---- | ---- | 0.32 | ----- | ----- | ----- | 0.23 | 9% |

**Table 13.** Mean centibars suction and GWC for the 2012 Lake Mills growing season

### 5.3 Sediment Field and Laboratory Analysis

Tensiometer suction in centibars (cb) increased at all sites throughout the growing season, indicating increasingly lower water availability in post-dam-removal sediments throughout the summer months. GWC decreased during the growing season; periods of hotter, drier conditions corresponded with the timeframe in which tensiometer suction

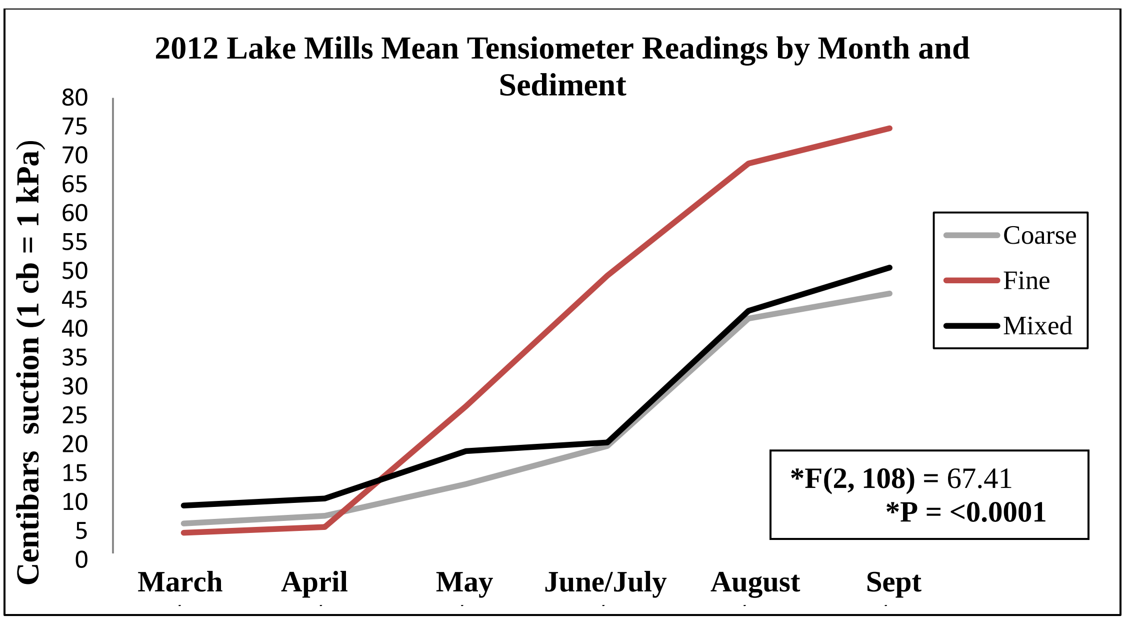


**Figure 2.** Mean tensiometer readings (available water) by site and month. *Repeated Measures ANOVA, monthly readings by sediment texture

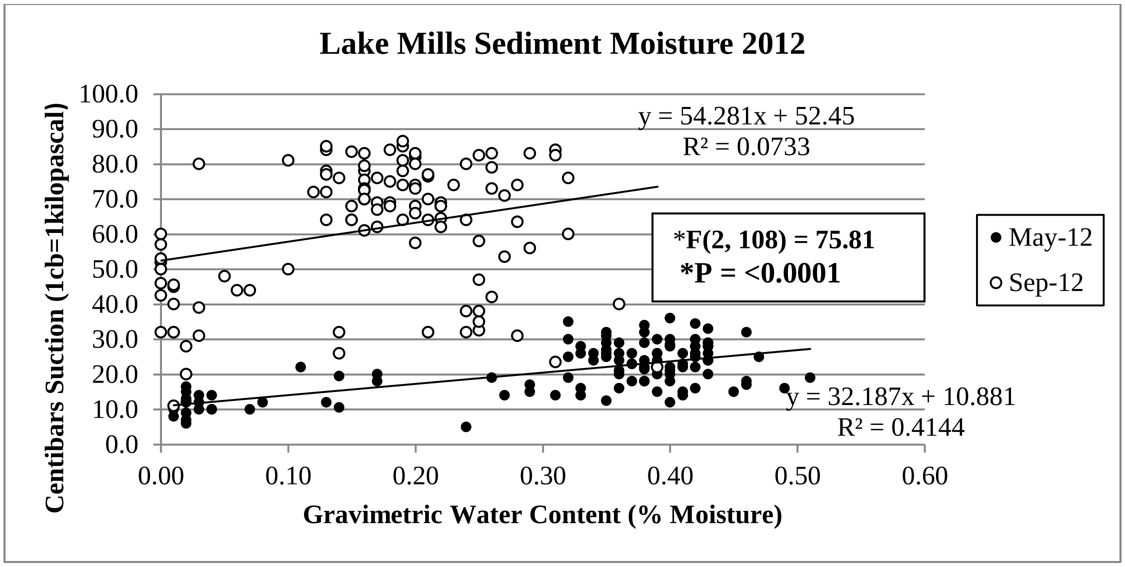


**Figure 3.** Available moisture versus percent moisture content of sediment in all study sites. *Repeated-measures ANOVA, May and October GWC by sediment

was high and GWC low (**Table 13; Figures 2, 3)**. It is noteworthy that these figures are averages, and readings were variable within each plot and site. By the end of the 2012 growing season, individual tensiometer readings reached 85 cb in all sites, particularly in northern Site 2. Mean seasonal pH for coarse sediments was 6.7, fine sediments 6.0, and mixed sediments 6.3 (**Table 15,** p. 55). Repeated-measures ANOVAs for tensiometer readings and GWC by month showed significant effects of sediment type **(Figures 2, 3)**. As a general trend, water availability and GWC showed the greatest decrease over time in fine sediment, while overall GWC was consistently the lowest in coarse sediment.

Following particle size analysis, characterization of sediment cores by site was as follows (**Table 14**): 1) predominantly sand, followed by gravel/cobbles and silt/clay in the coarse sediment site; 2) predominantly silt, followed by sand and clay in the fine and fine/sandy sediments (in the fine sediment, percent sand content was only marginally higher than percent clay). Additionally, hydrometer analysis showed highly variable

| 2012 Lake Mills Sediment Particle Size Characterization<br>Mean Particle Size by Sediment Type | | | | | | |
|---|---|---|---|---|---|---|
| **Particle Size Class (mm)** | **Coarse** | | **Fine** | | **Fine/sandy** | |
| | **Sieve** | **Hydrometer** | **Sieve** | **Hydrometer** | **Sieve** | **Hydrometer** |
| **% Gravel/Cobbles (>2.0)** | 27% | ------ | ------ | ----- | ----- | ----- |
| **% Sand (0.05-2.0)** | 61% | 38% | ------ | 17% | 15% | 36% |
| **% Silt (0.002-0.05)** | 11% | 38% | ------ | 69% | 85% | 57% |
| **% Clay (<0.002)** | ---- | 24% | ------ | 15% | ---- | 7% |

**Table 14.** Sediment particle size ranges by sediment type

particle size ranges in all sites. Fine sediments ranged from 1% to 32% sand, 18% to 34%clay, and 58% to76% silt, while the mixed sediment samples ranged from 15% to 40% sand, 7% to 8% clay, and 54% to 85% silt. Two samples included in hydrometer assessments from the predominately coarse sediment site (Site 1) were actually exposed fine sediment, with particle sizes ranging from 31% to 44% sand, 23% to 26% clay, and 33% to 43% silt. Sieve analysis of coarse sediments revealed gravel/cobble content ranging from 14% to 31%, sand content from 42% to 79%, and silt content from 3% to 37%.

***Sediment Nutrient Analysis***

Discrete analyzer nutrient tests revealed low available nitrate and phosphate concentrations in the 34 sediment cores assessed (**Table 15**). Nitrate-N/L content was highest in the fine sediment, and ranged from 1) below levels of detection (-0.006) to 0.118 mg Nitrate-N/L in coarse sediment; 2) 0.005 to 0.197 mg N/L in the fine sediment; and 3) below levels of detection (-0.001) to 0.551 mg N/L in the fine/sandy mixed sediment. Orthophosphate content was highest in the coarse sediment, and ranged from 1) 0.01 to 0.156 mg phosphate-P/L in coarse sediment; 2) below levels of detection (-0.004)

| 2012 Lake Mills Mean $NO_3$-N , $PO_4$-P and pH by Sediment Type | | | | | |
|---|---|---|---|---|---|
| **Sediment** | **$NO_3$- mg N/L** | **SE** | **$PO_4$- mg P/L** | **SE** | **Mean pH** |
| **Coarse** | 0.05 | ± 0.01 | 0.08 | ± 0.01 | 6.7 |
| **Fine** | 0.10 | ± 0.02 | 0.03 | ± 0.007 | 6.0 |
| **Fine/sandy** | 0.10 | ± 0.03 | 0.05 | ± 0.006 | 6.3 |

**Table 15.** Mean Nitrate-N and Phosphate-P concentrations and pH by Sediment

to 0.111 mg phosphate-P/L in the fine sediment; and 3) 0.004 to 0.13 mg P/L in the fine/sandy mixed sediment. In Site 3, an anomalous spike in nitrate-N was detected in one sample, with an outlier of 0.551 mg Nitrate-N/L. While the majority of samples at this site read lower than 0.10 mg Nitrate-N/L, this outlierresulted in mean Nitrate-N concentrations of 0.099 mg/L, just below the mean concentrations in the fine sediment. One-way Analysis of Variance tests were performed to determine if effects of sediment type significantly influenced nitrate and phosphate concentrations. While phosphate content was significantly influenced by sediment type, nitrate content was not (**Figure 4**).

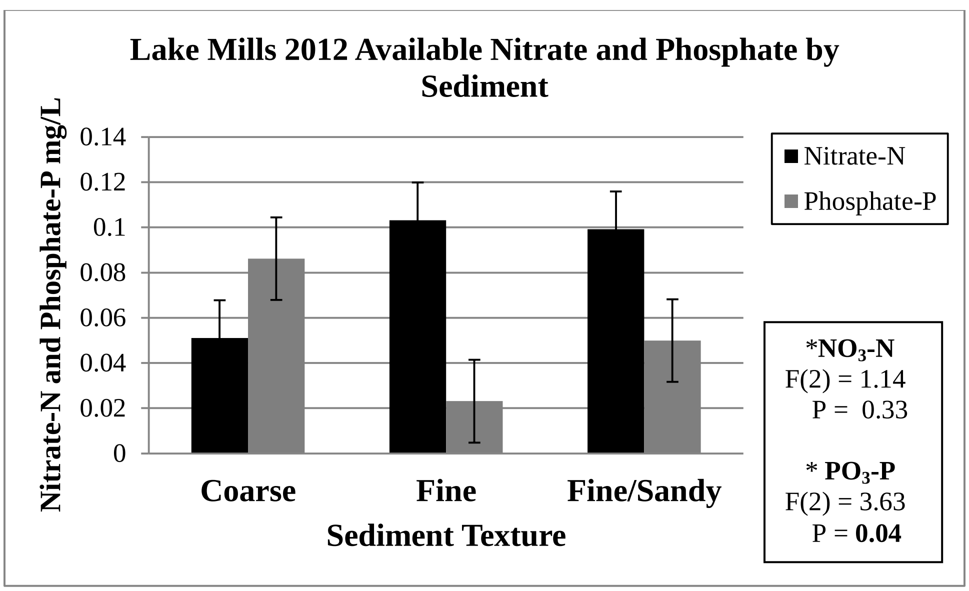


**Figure 4.** Nitrate-N and phosphate-P (mg/L) by sediment type. *One-way ANOVAs for nutrient concentrations by sediment

### 5.4 Natural Plant Regeneration

At all restoration sites, native graminoids and horsetails rapidly recolonized moist areas. Horsetail (*Equisetum arvense*) showed the highest frequency (41%, ONP 2012 vegetation surveys). Baltic rush (*J. balticus*), Common rush (*Juncus effusus*), dagger-leaved rush (*J. ensifolius*), Bolander's rush (*J. bolanderi*), toad rush (*J. bufonius*), spreading rush (*J. supiniformis*) and tapered rush (*J. acuminatus*) were common species among native, naturally regenerating rushes (Hitchcock and Cronquist, 1973; Pojar-Mackinnon, 1994; ONP 2012 vegetation surveys). Common regenerating native grasses included slender hairgrass (*Deschampsia elongata*), spike bentgrass (*Agrostis exarata*), blue wildrye (*Elymus glaucous*), and bromes (*Bromus* spp.). Native sedges included thick-headed sedge (*Carex pachystachya*), saw-bead sedge (*C. stipata*), and dewey sedge (*C. deweyana)*, and others not observed flowering (Hitchcock and Cronquist, 1973; Pojar- Mackinnon, 1994; ONP 2012 vegetation surveys). Over time, as the exposed sediment changes from mesic to xeric, the rushes and other naturally regenerating plants prone to moist conditions may not persist (Shafroth et al., 2006).

Naturally regenerating forbs included several varieties of forget-me-not (*Myosotis* sp.), fringed willowherb (*Epilobium ciliatum*), chaparral willowherb (*E. minutum*), pearly everlasting (*Anaphalis margaratacea*), sow thistle (*Sonchus spp.*), western dock (*Rumex occidentalis*), American speedwell (*Veronica americana*) and goats beard (*Aruncus dioceae)* (Hitchcock and Cronquist, 1973; Pojar- Mackinnon, 1994; ONP 2012 vegetation surveys). Germinants of Douglas-fir, grand fir (*Abies grandis*), western hemlock (*Tsuga heterophylla*), bigleaf maple (*Acer macrophyllum*) western redcedar, and oceanspray were also commonly encountered at all sites. An additional naturally regenerating native woody plant, observed in greater abundance than expected throughout the dewatered Lake Mills reservoir, is resprouting Sitka willow (*Salix sitchensis*) (Hitchcock and Cronquist, 1973; Pojar-MacKinnon, 1994; ONP 2012 surveys).


**Map 2**. Creeks and Whiskey Bend Road, bordering Lake Mills

In addition to naturally regenerating native plants, a variety of non-native forbs and graminoids also emerged in the exposed Lake Mills sediments. A trail on the west lakeshore and an unpaved road above the east shore coupled with several creeks flowing from these human developments to the dewatered reservoir likely serve as weed vectors (**Map 2**). Three common invasive grasses and several common forbs included: common velvet grass (*Holcus lanatus*), creeping bentgrass (*Agrostis stolonifera*), orchard grass (*Dactylus glomerata*), mainly at northern boundaries of the dewatered reservoir, hairy cat's ear (*Hypochaeris radicata*), bull thistle (*Cirsium vulgare*), and creeping buttercup (*Ranunculus repens*) (Hitchcock and Cronquist, 1973; Pojar- Mackinnon, 1994; ONP 2012 vegetation surveys). Common invasive forbs observed to less frequently were: Canada thistle (*Cirsium arvense*) and Robert's geranium (*Geranium robertianum*) (Hitchcock and Cronquist, 1973; Pojar-Mackinnon, 1994). Robert's geranium was generally encountered within forested edges of the northeastern and northwestern shores of the reservoir, where Whiskey Bend Road and the West Lake Mills Trail serve as potential weed seed sources from which adjacent creeks carry the seed to the edges of the exposed sediment (**Map 2**). The Sedge Creek vicinity harbors a large infestation of Robert's geranium originating from Whiskey Bend Road which requires periodic treatment and monitoring.

### 5.5 Herbivory Observations

During the July 2012 monitoring, small green caterpillars appeared on the leaves of black cottonwood at all study sites. The caterpillars skeletonized affected leaves and led to a decline in qualitative vigor data for many tagged trees. In all, 73% of the cottonwoods in this study showed varying degrees of caterpillar herbivory. For the purpose of moth identification, several caterpillar specimens were collected from Sites 2 and 3 and reared in a jar containing green leaves (Carri LeRoy, 2012, personal communication). One moth emerged and was submitted to moth specialist Dr. Lars Crabo, co-founder of Western Washington University's "Pacific Northwest Moths" website, for identification. The moth was identified as the grey midget (*Nycteola cinereana*), Neumögen & Dyar, 1893, in the family Nolidae (Lars Crabo 2012, personal communication).

Grey midget caterpillars are semi-translucent, green and relatively hairless, bearing long setae. Caterpillars feed on members of the tree family Salicaceae (*Populus* and *Salix* sp.). Larvae can forage as colonial tent-makers where populations are high, or as "single defoliators," as with the Lake Mills population. Moths are light grey-brown-and-white mottled, with rectangular wings reaching 2.5 to 3.1 cm in wingspan (www.entomology.museums.ualberta.ca_Searching_species_details.php?s=2776). Life history traits include a broader "single peak" of emergence in early summer than other *Nycteola* species, which are generally bimodal in spring and late summer. Grey midget moths can overwinter in either larval or caterpillar life stages. A common physical characteristic of the moth is a dark "mustache" from the basal area to posterior margin. This moth is generally found from Newfoundland to southern B.C., although sightings in western Washington State and Oregon are not uncommon. This moth is not known to be a significant pest in Pacific Northwest forests or croplands (http://pnwmoths.biol.wwu.edu).

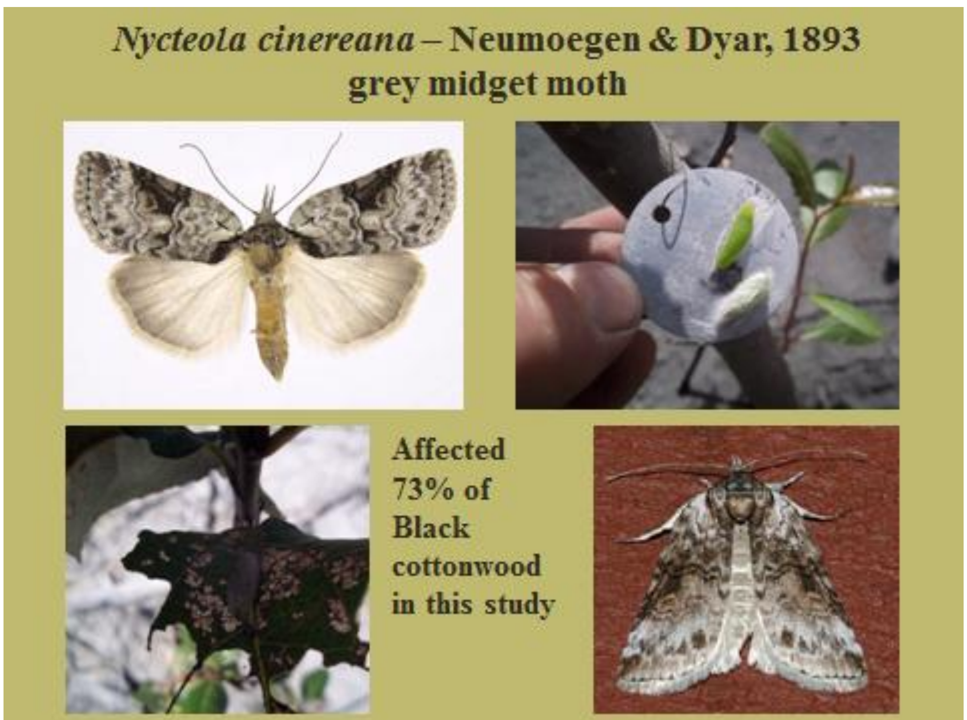


**Figure 5.** Photos of the grey midget moth as mature moth (upper left, lower right) and in caterpillar and larval stages (lower left, upper right)

Ungulate herbivory was apparent in all sites, particularly with thimbleberry, oceanspray and western redcedar. In all sites, potted plants were pulled up by Roosevelt elk (*Cervus canadensis roosevelti*) or blacktail deer (*Odocoileus hemionus columbianus*). Elk and deer tree herbivory and damage had been observed during the 2012 growing season, and became increasingly apparent by the winter and spring of 2013 following this study. In addition to grazing on a variety of planted forbs, shrubs and young trees, a resident elk bull utilized these saplings to rub the velvet from its antlers during molting, resulting in significant bark shredding. In some cases this led to displacement of black cottonwood tags from the 2012 survivability study. This bull, initially sited during this study at on the West Lake Mills Trail during the winter of 2012, was apparently displaced from a neighboring herd upon reaching maturity and is currently being tracked via radio collar by ONP wildlife biologists.

## VI. Discussion

### 6.1 Plant Performance and Site Conditions

2012 Lake Mills revegetation plant mortality, while lower than expected, showed a distinct trend in relation to species. It was not entirely unexpected that Douglas-fir showed the highest morality rate as this tree is generally not an early successional riparian species and often requires a greater degree of organic matter and nutrients, lacking in the Lake Mill substrates, in order to establish. The most likely cause of the poor Douglas-fir seedling performance is believed to be a lack of mycorrhizal fungi (Dave Allen, Joshua Chenoweth, personal communications). Douglas-fir trees are dependent on ectomycorrhizal fungi in order to establish and survive. Mycorrhizae allow host plants to maximize available nutrients in nutrient-poor, low-fertility soils typical in severely disturbed lands such as the dewatered Lake Mills reservoir. Additional benefits trees receive from mychorrhizae are disease resistance (Haselwandter and Bowen, 1996), greater absorption of available water, and increasing tolerance to drought stress and high temperatures (Simmard, 2009). With the exception of the periodic, naturally re-establishing plants in moister areas, the exposed post-dam-removal sediments were devoid of live trees, the roots of which serve as carbon sources for mychorrhizal fungi, prior to replanting efforts (Jones et al., 2003). Nursery inoculation of Douglas-fir seedlings with locally-derived ectomychorrizal fungi may enhance their survivability and performance in future ONP restoration efforts (Chenoweth et al., 2011).

Environmental factors coupled with low organic matter and poor soil development likely contributed to high Douglas-fir mortality during the 2012 Lake Mills growing season. Windy site conditions may have been detrimental to Douglas-fir performance and that of other installed plants. High winds can increase the transpiration rates of seedlings, resulting in stomatal closure and lowered water potential (Heiligmann and Schneider, 1974). Additional research (Philipson, 1988) indicates that Douglas-fir seedlings are dependent on current photosynthate for new root growth. In other words, Douglas-fir trees must obtain carbohydrates for new root development from foliage, rather than from buds, cambium or starch reserves as with other conifer seedlings such as Sitka spruce (*Picea sitchensis*) (Philipson, 1988). Additionally, root damage incurred from rough handling during transport and planting can exacerbate drought stress in Douglas-fir seedlings, reducing the potential absorption of water lost during transpiration in periods of drought due to inherently low root growth potential (RGP) (Philipson, 1988).

The high survivability of black cottonwood in this study despite nutrient-poor, undeveloped substrates supports further examination of its performance in Lake Mills post-dam-removal sediments. In addition to their deep rooting capabilities as an adaptation to drying soil conditions (Naiman and Decamps, 2005), recent research indicates that cottonwoods and other trees of the family Salicaceae share symbiotic microbial relationships which further enhance their adaptability to nutrient-poor substrates (Doty et al., 2009). Extraction of cells from the surface-sterilized stems of cottonwood (*Populus trichocarpa*) trees revealed the presence of endophytic bacteria (*Burkholderia vietnamiensis*, wild poplar strain B) responsible for nitrogen fixation. In a nursery experiment, inoculation of Kentucky bluegrass (*Poa pratensis*) with this endophytic bacterium enhanced its growth in a nitrogen-free substrate (Doty et al., 2009;

Xin et al., 2009). Black cottonwood was included in the 2013 ONP plantings of the dewatered Lake Mills and Lake Aldwell reservoirs, and will likely be included throughout the remaining planting trials through 2017. Planting black cottonwood, willow and other native woody and herbaceous species adapted to nutrient-poor soils may be a key strategy in the survival of the revegetation in Lake Mills, and the eventual development of late-seral forests critical to river restoration.While black cottonwood showed the greatest survivability of woody species included in this study, their long-term survival remains to be seen as water tables drop over time (Scott et al., 1999).

In addition to the highest and poorest-performing woody plant species from this study, the other species were assessed for their utility in future restoration efforts. The remaining four woody plants tested during the 2012 ONP planting trials, oceanspray, Nootka rose, thimbleberry, and western redcedar, all experienced low mortality rates and were included in the 2013 ONP Elwha restoration plantings. Additionally, woody plants such as western bittercherry (*Prunus emarginata*), red alder, and all of the *Rubus* and *Ribes* species in the 2012 Lake Mills planting trials were unofficially observed to perform well in all sediment types where planted. Low germination success rates and challenges with rodent seed predation prevented the inclusion of high numbers of western bitter-cherry in the 2013 plantings, but this species will be propagated over the coming 2013 ONP seed collection season for future plantings.

Pearly everlasting (*Anaphalis margaritacea*) and fireweed (*Chamerion angustifolium*), included in the 2012 Lake Mills plantings and also observed as natural regeneration in all planted sites, were known to colonize low-nitrogen volcanic depositions following the 1980 Mt. St. Helens pyroclastic volcanic eruption in southern Washington (del Moral and Wood, 1993). Pearly everlasting was observed to adapt to the severe substrate in these post-eruption environments with the development of branched root systems (Chapin, 1995). It remains to be seen how these and other pioneer plants may or may not persist in post-dam-removal sediments, or whether they will aid in the native vegetation colonization, but these plants may show greater persistence than the current, predominantly mesic rush and sedge regeneration as exposed sediments dry out (Shaffroth et al., 2006). During the 2012 Lake Mills woody plant performance studies, all plant species in all restoration sites appeared to have completed their flowering and fruiting cycles, and a resulting seed legacy and new germinants may be detected over the coming growing 2013 season.

Low plant growth measurements were not unexpected during the relatively short timeframe of this study. Follow-up monitoring over the next several years may reveal increasing growth rates of surviving vegetation once plant roots are established and energy is transferred to stem growth (Woodruff et al., 2002). Growth measurements will continue along with monitoring survivability and vigor of the replicates included in this study during the next several growing seasons.

While the fine sediments in Lake Mills were originally considered to be of greater concern than coarse sediments for native plant establishment, the 2012 planting trials have changed this perspective. Although this sediment affords poor drainage and hypoxic growth conditions, its cohesive texture protects plant roots more effectively than the highly erosive coarse sediment, particularly in the windy site conditions experienced at

Lake Mills. These very properties also inhibit herbivores from uprooting plantings, in comparison to high occurrences of uprooting observed in the planted coarse substrates. Limited vegetation survival in coarse sediments can potentially delay the development of this substrate into soil, lacking the structure provided by organic matter, root contact points and capillary draw (Angers and Caron, 1998).

### 6.2 Sediment Field and Laboratory Analysis

Nutrient limitations, particularly N, may delay the establishment of native vegetation in the dewatered Lake Mills reservoir, as was the case with mudflow deposits out of the range of legacy seed  and organic matter following the eruption of Mount St. Helens (del Moral and Clampitt, 1985). Past analysis of Lake Mills fine sediments showed total N contents of 9 to 11%, compared to 15 to 22% in climax western hemlock forest values, and P contents of 1.0 to 2.0 ppm, compared to 7.4 to 9.2 ppm in climax western hemlock forests (Henderson et al., 1989; Chenoweth et al., 2011). Additionally, pre-dam-removal assessments of Lake Mills sediments conducted by ONP revealed low concentrations of potassium, calcium and magnesium content and the micronutrient boron (Chenoweth et al., 2011). The fall 2013 nitrate and phosphate testing of Lake Mills sediment, with concentrations ranging from below level of detection to 0.10 mg/L further supported low nutrient content in the post-dam-removal sediments. Phosphate concentrations detected during this study were higher in coarse sediment than in fine, contrary to earlier nutrient assessments of Lake Mills sediments (Cavaliere and Homann, 2012). However, the sediment nutrient assessment conducted in this study reflects only a single snapshot in time during the fall of 2012. Ideally, numerous nutrient assessments would have taken place over time from these sample sites to detect changes during the

2012 growing season, since detectable levels of available N and P can vary during different seasonal periods (Carri LeRoy, personal communication).

During early stages of forest development, nutrient-poor sediments are not uncommon in Pacific Northwest river floodplains (Naiman et al., 2010). Over time (>50-100 years of floodplain soil development), sediment erosion and N-fixing plants such as red alder, cottonwood, willow or lupines (*Lupinus* sp.) (del Moral and Wood, 1993) provide rapid nutrient input. Fine silt and clay sediment particles can be efficient in the adsorption of C and N obtained through organic matter during the initial 100 years of soil development (Naiman et al., 2010), while coarse sediments may not readily retain nutrients due to leaching, typical of an “open” system. A “closed” system is attained with the formation and accumulation of soil and organic matter and the development of plant-soil nutrient recycling more typical in later-seral forests (Naiman et al., 2010). N retention and availability are especially dependent upon the presence of organic matter (Chappell et al., 1991). Therefore, nutrient assessments of sediments in the dewatered Lake Mills reservoir should continue over time as more vegetation becomes established.

The low available $NO_3$ concentrations detected in Lake Mills sediment cores potentially explains the high yellow leaf coloration of 38 to 55% of tested woody plants in the 2012 Lake Mills plantings by September (Stewart, 1999). However, further research and consideration must be given pertaining to the different responses (i.e., vigor) of individual plant species to environmental stress. Hot, dry weather potentially contributed to the rising percentages of plants displaying red leaf coloration by late summer 2012 (Hoch et al., 2001, 2003). The highest red leaf coloration among all plants occurred in the coarse sediment, in which windy site conditions coupled with erosive, low-moisture substrate may have exacerbated drought and root stress. The introduction of potted or bare root plants into a new site can result in “transplant shock,” where trees and shrubs display leaf scorching during the first two to three years of development while undergoing root establishment (Purdue University Plant and Pest Diagnostic Laboratory, 2002). Increasing leaf scorching as high as 27% by the end of the 2012 Lake Mills growing season followed a pattern indicative of drought stress and/or root damage, where dry winds during the late summer drought were likely the greatest contributors.

Mean pH ranges by sediment type measured during this study were more basic than those of climax western hemlock forest ranges (Henderson et al., 1989), but consistent with those found in a preliminary ONP Lake Mills sediment assessment (6.0-6.5) (Chenoweth et al., 2011). Some fine sediment sample sites, however, yielded pH readings in the range of 5.4 – 5.9, more consistent with baseline climax western hemlock forest ranges (Henderson et al., 1989).

Overall, the negative GWC trend from the early to late growing season was consistent with what may be expected in increasingly dry site conditions between May and late September 2012, coupled with lower water availability (**Figure 1**). In coarse sediment, due to large gravel and sand pore sizes providing greater moisture availability, lower suction in centibars, compared to fines occurred in conjunction with low GWC. However, average tensiometer readings in coarse sediment were 46 cb by late summer and fall. In agricultural operations, soils high in coarse sand would receive irrigation upon reaching 20 to 40 cb (Soil Moisture Corp, 2012). In fine sediment, low porosity and fine sediment particles yielded higher suction in centibars, coupled with GWC significantly higher than that of the coarse substrate. Tensiometer readings had climbed as high as 85 cbs suction in the fine sediment by August and September. Soils high in silt and clay, in agricultural production, would require watering upon reaching 50 to 70 cb, depending on clay content (Soil Moisture Corp, 2012). The permanent wilting point for most soils is 150 cb (Tolk, 2003). The fine/sandy sediment moisture availability and GWC proved more variable within-site, but showed a greater trend of decreasing water availability during tensiometer measurements than in the coarse sediment, displaying some of the moisture characteristics. The Heterogeneity encountered in both GWC and available water assessments in this study within individual sites is not uncharacteristic of substrates lacking developed vegetation, since lower density of living plants means the presence of fewer plant roots to uniformly remove soil moisture at any given time (Bethlahmy, 1962; Adams et al., 1991). Results from 2012 Lake Mills sediment moisture assessments illuminate the likelihood of increasingly low moisture storage in all post-dam-removal sediment types as they dry with continued exposure.

Use of a tensiometer in coarse sediments proved challenging during this study, particularly due to the fact that this substrate is far coarser than in developed soils given this characterization. Abrasive, loose rock often served as a barrier to insertion of the tensiometer core and probe, shortened the life of the ceramic probe tip due to gouging and surface abrasion, and high porosity coupled with low cohesion provided less surface area contact for the probe tip than in the fine and fine/sandy sediments. While the use of a tensiometer suited the needs for onsite moisture assessment in this study due to extensive time required in the field, this method may not be well-suited for future high-volume use in similar coarse sediments.

Tensiometer readings will be conducted by ONP for the remainder of the Elwha revegetation project, subsampling 20 of the 111 2012 Lake Mills sediment moisture sites (Sites 1 and 2), in addition to 30 new sample sites within the coarse and fine 2013 planted sites on west Lake Mills. Reassessment of the 2012 sediment moisture sites may indicate the degree to which sediment water availability declines post-dam-removal with continued exposure, potentially shifting the plant species these sites can support as conditions change from mesic to xeric (Shafroth et al., 2002).

Particle size characterization in this study appeared fairly consistent with that of previous efforts. In two previous Lake Mills studies, fine sediments were found to contain 82% to 85% silt, 18% to 6% clay, and up to 8% sand while coarse sediment contained <1% to 4% clay, 2% to 11% silt and 85% to 98% sand (Mussman et al., 2008; Cavaliere and Homann, 2012). Sediment particle size in this study proved variable due to within-site heterogeneity, to a lesser degree in the fine sediment (**see Section 5.3**), and was consistent with the variability detected in sediment moisture between sites.

### 6.3 Natural Plant Regeneration and Invasive Plants

Several invasive plants encountered in all study sites will be monitored and treated for the remainder of the Elwha River restoration revegetation efforts. Robert's geranium is of particular concern due to its presence in the Lake Mills vicinity and ability to establish beneath closed forest canopies and well-established dense native shrub communities (Chenoweth et al., 2011; Washington State Noxious Weed Control Board, http://www.nwcb.wa.gov/detail. asp?weed=55). This herb has been known to colonize waste sites, spoil heaps, limestone quarries and a multitude of disturbed areas worldwide, and can eject seeds as far as 4.5 to 6.0 m. Robert's geranium prefers moist sites, which are currently abundant on the dewatered Lake Mills reservoir (Tofts, 2004). Common velvet grass was common in all sites, and is currently treated by park staff through mechanical removal when detected. This grass was linked to a slowing of litter decomposition in coastal California prairies, by deterring key detritivore macroinvertebrates responsible for grass litter loss (Barstow, 2008). While the lower Elwha River ecosystem may not be impacted by common velvet grass in the same manner as the California prairie environments, it is worth future monitoring due to its potential effects on long-term soil formation.

### 6.4 Future Research

For at least the next 3 to 4 growing seasons, the survivability and vigor of the Lake Mills 2012 tagged woody plants will be monitored. This long-term monitoring will capture more fully the plant survival and establishment rates which may not have been reflected during the first year following their installment. It not uncommon in similar plant restoration efforts for the majority of potted and bareroot seedling mortalities to occur within the first growing season; however, gradual mortality over the following several growing seasons may also be expected (DeSteven, 1991). Additionally, treatments such as planting adjacent to woody debris, woody debris size class, planting density, proximity of natural regeneration or dense neighbor species may show greater influence as time passes.

During the winter of 2013, a student of Peninsula College, Port Angeles, WA is conducting a nursery ectomycorrhizal inoculation study with the seeds and seedlings of

Douglas-fir and grand fir at the ONP nursery. During late February and early March 2013, inoculated soil (256 ounces or one, 2-gallon container) was collected from the base of Douglas-fir stands or trees at several locations between Glines Canyon Dam and the ONP park boundary along Olympic Hot Springs road. For both Douglas-fir and grand fir, 400 seeds were inoculated in a 3-layered substrate mesocosm of collected soil and nursery seeding soil (Sunshine peat mix with 40% perlite), with an additional 400 seeds for each species sown into seeding soil only as a control. In all, 8 seeding flats were prepared: six flats of Douglas-fir, (three flats inoculated and three control, 200 seeds per flat); and two flats of grand fir (one inoculated, one control, 200 seeds per flat). Additionally, 300 1-year-old Douglas-fir and grand-fir seedlings were inoculated in pots, accompanied by potted control seedlings, 300 for each species. Inoculated and control seeding flats are currently stored in the ONP greenhouse, while inoculated and control potted seedlings reside onsite outdoors. The performance of seedlings will be tested in the 2014 ONP post-dam-removal restoration efforts, to determine whether ectomycorrhizal conifer inoculation improves their survivability and performance in different substrates. These findings may build on and at least partially explain the high mortality rate of Douglas-fir in the 2012 Lake Mills woody plant trials.

A University of Washington graduate student from the Master of Environmental Horticulture program will conduct thesis research similar to the 2012 Lake Mills woody plant revegetation study. This research will test the performance of 2-year-old Douglas-fir seedlings, to determine whether performance may be more favorable with older specimens (2012 plantings were 1-year-old seedlings). Additionally, grand fir (*Abies grandis*), western white pine (*Pinus monticola*), bigleaf maple (*Acer macrophyllum*), and

Scouler's willow (*Salix scouleriana*) will be included to broaden knowledge of tree and tall shrub survivability in Lake Mills fine and coarse post-dam-removal sediments. The plant survivability studies of 2012, 2013 and those of future ONP plant restoration efforts will capture an increasingly larger picture of native woody plant species performance in post-dam-removal sediments.

# VII. Significance

## 7.1 Broader Impacts of study

Findings from the Lake Mills native plant restoration and similar efforts may set the precedent for future plant restoration following dam removals, particularly in the Pacific Northwest. Riparian plant establishment is key to maintaining healthy river ecosystems which support fish and other associated aquatic and terrestrial organisms. While abundant research has been conducted on effective riparian plant restoration strategies, little to no research has focused on selection of plant species suited for specific types of residual reservoir sediment. Findings from this study will aid in the development of planting strategies for the remaining five of the seven-year plant restoration efforts of Olympic National Park. The ONP riparian plant restoration efforts will be carried out in stages, determining which species to propagate and plant based on survival rates in previous planting efforts. Determining which woody species to plant in specific post-dam-removal sediment textures will aid in a more efficient use of time and resources for the duration of the project.

The removal of Glines Canyon Dam is nearly complete, with 20 m or 30% of the dam remaining. In March 2013, 4.9 million $m^3$ of sediment had passed from the dewatered Lake Mills reservoir to downstream reaches of the Elwha River. This is estimated to be 25% of the sediment load accumulated during the presence of the Glines Canyon Dam (USGS Sediment Team, 2013). An additional 1.1 million $m^3$ of sediment have passed downstream from Lake Aldwell (USGS Sediment Team, 2013). This exceeds estimates from previous Lake Mills sediment core studies (Shaffer et al., 2008;

Winter and Crain, 2008). Additionally, due to historic topographic mapping errors, sediment depth was found to be 6.1 m deeper than previously estimated (USGS Sediment team). Thus, Lake Mills is believed to have contained 21.7 million $m^3$ of accumulated sediment (USGS Sediment Team, 2012). This quantity of sediment is approximately 2.3 billion $m^3$ less than the total sediment released into the north and south forks of the Toutle River during the eruption of Mt. St. Helens (Bednarek., 2001). Establishment of native riparian vegetation is therefore key to the eventual development of organic matter and eventual topsoil with large quantities of sediment remaining in the former lake.

Due to high levels of suspended sediment in the lower Elwha River, further dam deconstruction has been delayed until September of 2013 at the earliest, and before the expiration of the dam removal contract in September 2014 at the latest (Mapes, 2013). The Elwha Water Treatment Plant (EWTP), constructed in 2010 in order to mitigate sedimentation-related harm to water consumers following dam removal, requires improvements to process the recently high levels of coarse debris flowing out of the dewatered Lake Mills reservoir. The EWTP treats water utilized by the City of Port Angeles, Nippon Paper Company (a mill located on Ediz Hook), the Washington Department of Fish and Wildlife's hatchery rearing channel, and the LEKT fish hatchery (ONP Press release, February 1, 2013). The need for EWTP amendments became apparent during the fall of 2012 when fish screens and pumps were overwhelmed with organic matter and sediment, lessening the feasible amount of water the plant could process and increasing the required equipment maintenance time. The EWTP was built to process up to 53 million gallons of water per day, with sediment loads as high as 40,000 parts per million (ppm) Total suspended solids. Sediment loads thus far have only reached 10,000 ppm TSS (Maynes, ONP Press release, February 1, 2013). The inadequacies of the EWTP have become so chronic that personnel must be continually onsite just to keep it running, when the treatment facility was originally designed to be operated remotely (Mapes, 2013). The Nippon Paper plant is currently the only recipient receiving treated water from the EWTP, and the City of Port Angeles has had to rely on its well for drinking water (Mapes, 2013).

### 7.2 Cultural Impacts of Dam Removal on Lower Elwha Klallam Tribe

The Lower Elwha Klallam Tribe (LEKT), inhabiting the Lower Elwha River and neighboring bluffs since as early as 750 B.C., are acknowledged in the U.S. 1855 Treaty of Point No Point as having resided in this location since "time immemorial." Current Elwha tribal lands make up close to 405 ha of the Elwha River watershed. Ancestral land of the LEKT included villages on the north and south sides of the Strait of Juan de Fuca. Areas of residence south of the Strait included the regions of Hoko, Clallam Bay, Pysht, Deep Creek, Freshwater Bay, the Elwha valley and river mouth, the Port Angeles vicinity and areas east of modern-day Port Angeles (Valadez, 2002). The current LEKT reservation was not officially designated a reservation until 1968, when the U.S. Federal government assessed stipulations the Treaty of Point No Point and determined that it entitled the tribe to their own historic land (http://www.elwha.org/elwhariver restoration.html; Valadez, 2002).

When the Treaty of Point No Point was signed in 1855, the LEKT were expected to relocate to the Skokomish reservation on the Hood Canal of the southeast Olympic Peninsula. The LEKT, however, resisted this push to be relocated, preferring instead to reside in their homeland where their ancestors were buried (Valadez, 2002). European settlers began occupation of LEKT land in the 1860s, displacing tribal members from their traditional villages and hunting and fishing grounds. Because the LEKT were not U.S. citizens by technical definition, they were unable to purchase their own land until the signing of the Indian Homestead Act in 1884 (Valadez, 2002). While this act allowed as many as thirteen Klallam tribal families to own land in their historic residences, the price of such ownership was the severance of LEKT homesteaders from their tribal relations. While some individuals were able to adapt to life as independent homesteaders, many did not wish to break away from their tribal heritage and as a result lost their land. The 1934 Wheeler-Howard Act passed under the Franklin D. Roosevelt administration, also named the Indian Reorganization Act, eventually provided federal funding for the LEKT to acquire private land in their usual and accustomed dwellings until the official designation of a reservation in 1968 (Valadez, 2002).

The construction of the Elwha River dams has adversely affected the culture and economy of the LEKT for nearly a century through a drastic decline in salmon runs (http://discoveringourstory.wisdomoftheelders.org/history-of-the-lower-elwha-klallam-people). Historically, the culture and economy of the Elwha tribe centered around salmon fishing of all five Pacific salmon species. The tribe has actively proposed the removal of the dams since the 1980s, and has been increasing the reestablishment of traditional tribal knowledge since this proposal proved closer to reality (Valadez, 2002). In addition to serving as barriers to anadromous fish, the dams led to loss of LWD delivery into the lower Elwha River. Woody debris formerly carried downstream from upstream reaches of the river was retained by the Glines Canyon Dam and to a lesser degree the Elwha

Dam for nearly a century following their construction. This loss of woody debris delivery to the lower reaches of the river eliminated logjams which had been key structural elements in formation of pools, side channels and other features needed for salmon resting and spawning grounds. To address this problem, the Elwha tribe has been installing engineered log jams (ELJ) since 1999 (Chenoweth et al., 2011). ELJs have been proven potentially effective for restoration of juvenile Elwha River salmon (Pess et al., 2012), but natural WD delivery and placement from an undammed river may further enhance salmon habitat.

In order to enhance fisheries recovery in the lower Elwha River, a 5-year fishing moratorium on the Elwha River has been proposed by the LEKT, State of Washington Department of Fish and Wildlife (WDFW), and ONP (Warren, 2010). In accordance to this moratorium, fishing on the Elwha River ceased in the spring of 2012 and will not resume until the spring of 2017. It is hoped that this temporary fishing cessation will allow the reestablishment of native anadromous fish stock in the middle and upper reaches of the Elwha River (Warren, 2010).

The LEKT manages numerous fish hatchery programs based on the Hatchery and Genetic Management Plan (HGMP). The HGMP was created cooperatively by the LEKT and Hatchery Scientific Review Group (HSRG), in addition to a number of federal and state government agencies such as National Marine Fisheries, U.S. Fish and Wildlife, and National Park Service. This plan was intended to reduce endangerment risks to threatened and endangered salmonid species (Case 3:12-cv-05109-BHS, Document 126, 2013).

A controversial fisheries recovery measure proposed by the LEKT entailed the stocking and upstream release of non-native, Chambers Creek steelhead into the Elwha River. The last release of Chambers Creek steelhead into the lower Elwha River took place in 2011, before major dam deconstruction had taken place. Tribal members in favor of the non-native steelhead stocking believed the LEKT had waited long enough to reap the benefits of their historic fishing rights. Opponents emphasized the risk of nonnative steelhead competing with native fish for food and resources, and the potential loss of genetic integrity resulting from nonnative steelhead breeding with Elwha River native fish.

Upon announcement of the plan to release non-native steelhead into the Elwha River, immediate legal actions were taken against the LEKT and National Park Service. Several nonprofit interest groups, including the Wild Fish Conservancy, Wild Steelhead Coalition, Federation of Fly Fishers Steelhead Committee, the Wild Salmon Rivers and the Conservation Anglers, filed lawsuits suing the LEKT and other agencies involved with the steelhead restocking, under the claim that this action violates the Endangered Species Act (Case 3:12-cv-05109-BHS, Document 126, 2013). In February 2013, U.S. District Court Judge Benjamin Settle threw out the lawsuit, due to the fact that the tribe had obtained all necessary federal fisheries permits to carry out their hatchery operations since the initial complaint. The LEKT, in the meantime, agreed to cease the release of any additional Chamber Creek steelhead, opting only to catch the remainder of hatchery stock released in 2011. As an alternative option the tribe will attempt to obtain permits to fish wild native Elwha River steelhead following the moratorium, even if these fish are still listed as endangered, so long as this action can be deemed as non-harmful to the fish populations (Case 3:12-cv-05109-BHS, Document 126, 2013).

Just one year following initial dam deconstruction, changes in the lower Elwha River seemed hopeful for the LEKT and the prospect of restoring historic fish runs. By the summer and fall of 2012, steelhead and Chinook, coho and pink salmon were observed upstream of the former Elwha dam (http://www.usgs.gov/ blogs/features/ usgs_top_story/elwha-one-year-later/). During the spring of 2013, however, hatchery Chinook released into the river experienced high mortality due to high river sedimentation. Acclimation of hatchery fish to conditions which better represent the current state of the lower Elwha River (i.e., cloudier water) and adjustments to the timing of release in relation to periods of high suspended sediment may prevent such fish kills in the future. Over time, the restoration of sediment delivery to the mouth of the Elwha River, particularly fine sediment, may restore shellfish beds, allowing an additional historic form of sustenance to be available once more to the LEKT (Shaffer, 2004).

Elwha tribal members are currently working with ONP, WDFW and the Bureau of Reclamation to restore the lower Elwha River. In addition to hatchery operations, fisheries monitoring and research, LEKT crews are actively revegetating and seeding the dewatered Lake Aldwell reservoir. Additionally, the LEKT crew treats invasive plants detected in the vicinity of the dewatered Lake Mills reservoir. Nearshore restoration efforts of the LEKT include the organization of cleanup at the abandoned, contaminated Rayonier mill site in Port Angeles, and the Ennis Creek Conceptual Restoration Plan, which entails removal of a pier, jetty and concrete to restore natural stream meanders and native riparian vegetation (The Watershed Company, 2011). It is hoped that this cooperative effort will restore not only the livelihood and traditionally sustaining salmon food source to the Elwha tribe, but also provide cultural renewal to a community that has triumphed in maintaining its homeland despite a legacy of strife (Valadez, 2002).

### 7.3 Elwha River Nearshore Habitat

As early as one year following the 2011 Elwha River dam deconstructions, changes at and beyond the mouth of the Elwha River were taking place as freed sediment was transported downriver. However, the complete restoration of Elwha River nearshore habitat requires interdisciplinary efforts among government and private entities. Testament to the large-scale sediment release following the full and partial removal of the Elwha River impoundments, sandbars were forming at the mouth of the Elwha River by fall 2012 (http://www.oregonlive.com/environment/ index.ssf/ 2012/12/ sandbars_forming_at_ mouth_of_r.html). While natural recovery processes such as downstream sediment delivery may improve the health of the lower Elwha River ecosystem over time, human development may impede other recovery processes. At least 66% of tidelands along the Strait of Juan de Fuca are privately owned, with <12% of shoreline bluffs or uplands adjacent to public beaches operating under public ownership (Shaffer, 2006).

Nearshore waters along the Strait are more tidally influenced than open marine waters, and therefore more subject to stratification and hypoxia, particularly in heavily populated areas. Deeper marine waters undergo mixing from high winds, currents and freshwater input from a multitude of rivers (Shaffer, 2006). Loss of sediment delivery formerly contained by the Elwha River dams had been a major culprit of beach loss in nearshore habitat, but armoring of privately owned bluffs is an additional contributor.

Pollutant runoff from industrial and everyday human activity in neighboring developments introduces toxins and risk of eutrophication to the Strait. Where government land regulation is replaced by a mosaic of private and city land ownership, collaborative changes in land use are critical to restore the beaches surrounding the mouth of the Elwha River and beyond.

Beyond the agency and tribal post-dam-removal restoration efforts of the lower Elwha River, nearshore recovery is being monitored by private entities. With downstream sediment delivery processes resumed, the possibility of rebuilding beaches and nearshore habitat arises for the first time in nearly a century (Shaffer et al., 2008). The Elwha Nearshore Consortium (ENC), coordinated by Anne Shaffer of Washington State Department of Fish and Wildlife, brings together scientsists, land managers, city planners and other stakeholders to address post-Elwha-River-dam-removal nearshore habitat recovery. This organization conducts research regarding public and private land management issues in shorelands between the mouth of the river and the City of Port Angeles (Shaffer, 2009). The ENC developed the DRFT Shoreline Management Plan in 2011. This plan operates under the priniciples of the Shoreline Management Act of 1971, which encompasses specific water of Washington State in addition to their shorelands. Focus in this Act is placed on shorelands associated with streams possessing a mean annual flow of at least 20 cfs, as well as lakes greater than 8.1 ha (20 acres). The Shoreline Management Act of 2011 defines shorelands as “lands projecting at least 200 feet in all directions on a horizontal plane from the high water mark,” and includes all wetlands and floodplains associated with streams, lakes, and tidal waters (Shaffer, 2006).

The ENC conducts a variety of studies related to nearshore, post-Elwha River-dam-removal recovery. Among these studies are the monitoring and mapping of Lower Elwha River river sedimentation and fish passage, particularly forage fish and their nursery beds, monitoring eelgrass populations, and monitoring for euchalon (also known as candlefish) in the Elwha estuaries. Historically, these estuaries served as critical feeding grounds for Chinook, coho, pink, and cutthroat salmon as well as anchovies, euchalon and surf perch (Shaffer, 2006, 2009). Of particular interest is whether the bull kelp beds of Freshwater Bay and throughout the nearshore will be replaced once again with eel grass as riverine-derived cobble material is replaced with finer sediment (Shaffer, 2006).

Several key components of nearshore restoration addressed by the ENC include beach loss, beach modification, and the pollution of marine waters. The ENC samples the beach profile from Crescent Bay to Discovery Bay to detect changes over time following dam removal (Shaffer, 2006). Bulkheads and dikes installed to prevent erosion of private coastal bluffs which actually serve as "feeder bluffs" for beach-building will continue to deprive beaches of material despite the post-Elwha-River-Dam-removal sediment delivery. Of equal concern is point and non-point-source pollution from residential and industrial sources in and around the City of Port Angeles, particularly that of a leaching decommissioned landfill constructed in the 1940s (Shaffer, 2006).
Other ENC restoration goals include the inventory of critical nearshore habitat, development of an Ediz Hook master plan, protection and restoration of native vegetation, and reconnection of feeder bluffs to the Strait of Juan de Fuca's marine system (Shaffer, 2006). The ENC is carrying out a 2-year study tracking the movement of

fine-grained sediment transported down the Lower Elwha River to the river mouth. Between November and June of 2012, an offshore sediment surface plume appeared to have increased in suspended sediment concentration by 60%, although this was later transported away from the delta by tidal action and a “net northeast travel direction” (www.coastalwatershed institute.org).

### 7.4 Interdisciplinary Context

Completion of this study required an expansion of my skill sets beyond that of my previous scholarly or work experience. Developing a better understanding of soil science was a necessary part of this study, and truly an asset. Learning the use and concepts of a tensiometer, planning and implementing the sampling of sediment water availability, and the collection and processing of 111 sediment cores for GWC and nutrient analysis, much of this with full-time college course and workloads, were genuine tests of endurance. Exposure to the use of an AQ1 discrete analyzer and the necessary soil preparations was also a new experience I am pleased to have gained. Particularly, the knowledge acquired in research design, data analysis, and professional writing skills will influence my future work.

In addition to expanding my scientific research skills, a variety of I have gained a greater understanding of the processes entailed in dam removal and post-dam-removal restoration. Through literature review, I learned of the legalities and functions which dictate the operation and licensing of hydroelectric dams. A greater understanding of the cultural history of the Lower Elwha Klallam Tribe, and the degree to which the LEKT had had to struggle to remain in their historical land, was gained through this project. I had taken for granted that the LEKT would have, at the very least, been designated reservation land to call their own long before 1968. Reading about the policies and legalities of post-dam-removal fisheries and learning more about the work of the ENC were additional benefits of this study. With much time, commitment and effort behind me, I leave the MES Program with a greater understanding of where my career path in the environmental sciences may lead.

**7.5 Conclusion**

In conclusion, the 2012 woody plant survival study is merely one piece of a much larger puzzle to the long term post-dam-removal recovery of the lower Elwha River. Species selection appeared to influence woody plant survival in Lake Mills post-dam-removal sediments, while planting treatments showed varying levels of influence by sediment type. Native riparian plant restoration of Lake Mills during the 2012 growing season saw high survival rates overall. Plant restoration efforts of ONP and the LEKT may aid the eventual provision of stream shading, channel complexity and other ecological contributions of a riparian forest which are key to maintaining a healthy river ecosystem. Interdisciplinary government and private entities will continue to work toward the full ecosystem of the lower Elwha River basin and nearshore habitat through active and passive restoration work and monitoring. Only time will tell to what degree this modified ecosystem returns to its pre-dam state following the largest U.S. dam removal operation to-date.

**Appendix A.**

**Table A1: Full Species Composition of ONP Planting Prescriptions**

| 2012 ONP Planting Prescriptions, Lake Mills: Species Composition | | |
|---|---|---|
| **Prescription 1** | **Prescription 2** | **Prescription 3** |
| **Shrubs Only - Site 1** | **Trees, Shrubs and Forbs - Sites 2, 3** | **Trees Only – Sites 1, 2, 3** |
| *Acer circinatum* | *Acer macrophyllum* | *Acer macrophyllum* |
| *Alnus viridis* | *Alnus rubra* | *Malus fusca* |
| *Cornus sericea* ssp. *Occidentalis* | *Crataegus douglasii* | *Populus balsamifera*ssp. *Trichocarpa* |
| *Gaultheria shallon* | *Holodiscus discolor* | *Prunus emarginata* var. *mollis* |
| *Holodiscus discolor* | *Oemleria cerasiformis* | *Pseudotsuga menziesii* |
| *Lonicera involucrata* | *Pseudotsuga menziesii* | *Salix scouleriana* |
| *Mahonia aquifolium* | *Rhamnus purshiana* | |
| *Mahonia nervosa* | *Rosa nutkana* | |
| *Oemleria cerasiformis* | *Rubus parviflorus* | |
| *Philadelphus lewisii* | *Sambucus cerulea* | |
| *Physocarpus capitatus* | *Symphoricarpos albus* | |
| *Ribes divaricatum* | *Thuja plicata* | |
| *Ribes lacustre* | | |
| *Ribes sanguineum* | **Herbaceous Plants** | |
| *Rubus leucodermis* | *Anaphalis margaritacea* | |
| *Rubus parviflorus* | *Aquilegia formosa* | |
| *Rubus spectabilis* | *Artemisia suksdorfii* | |
| *Rubus ursinus* | *Aruncus dioicus* | |
| *Salix sitchensis* | *Chamerion angustifolium* | |
| *Sambucus cerulea* var. *cerulea* | *Erigeron philadelphicus* | |
| *Sambucus racemosa* | *Fragaria vesca* | |
| *Spirea douglasii* | *Petasites frigidus* | |
| *Symphoricarpos albus* | *Solidago Canadensis* | |

**Table A1.** Plants included in ONP prescriptions: Full species list of prescriptions followed in this study. Note: Nootka rose and western redcedar not included in Site 1

## Appendix B: Methods of Sediment Moisture Assessment, and Limiting Factors in Study Sites

A portable tensiometer was utilized in this study to allow onsite assessment of sediment moisture. Many leave-in instruments, which are vulnerable to damage from freezing temperatures, are also very costly. Tensiometers, however, may not be advantageous in fine sediments during extremely dry periods, when the sediments desiccate and crack, and tensions of greater than 80 to 100 centibars are required for plant roots to extract water. During such periods, tensiometers may measure only 74-85% of the suction required of plant roots in order to draw water from binding soil particles (Stoeckeler and Aamodt, 1940).

Due to the wide range of sediment texture, particle size, and accompanying range of moisture retention potential between and within plots, no perfectly-suited instrument was available to accurately capture moisture levels in the highly variable sediments of the Lake Mills study sites (Robertson et al., 1999). Fine soils are commonly measured by the use of gypsum electrical resistance blocks and meters, while tensiometers are more commonly applied to measure available moisture in sandier soils (Robertson et al., 1999). post-dam-removal sediments found on the Lake Mills delta, however, do not resemble the substrates generally assessed for moisture in agricultural or other planting operations. Instruments such as neutron probes or capacitators were beyond the economic range of this study. The gravimetric method, in which core sediment samples are collected, pre-weighed, oven-dried and reweighted to measure gravimetric water content (GWC), is an empirical, accurate soil moisture assessment technique. However, the large number of samples required in this study (N=111), the time involved in sieving and processing samples, as well as 24-72 h for oven-drying, and long distances from study sites to the campus soil laboratory made this method a time-consuming and less realistic alternative. The onsite sediment moisture assessments afforded by a portable tensiometer proved a more feasible option, as this process could occur concurrently with vegetation monitoring. Additionally, tensiometers simulate the action of a plant root, measuring suction in units of pressure, providing a favorable representation of the stress undergone by plant roots during periods of drought.

### *Plant Growth Measurements*

The number of measured plants (**Table 17**) and their random selection for measurement was based upon their reasonable proximity to sediment moisture sample sites (i.e., the resulting feasible number of plant measurements which could be carried out while waiting for tensiometer readings).

| **Species** | **Number Measured** |
|---|---|
| Oceanspray | 31 |
| Nootka rose | 17 |
| Thimbleberry | 30 |
| Black cottonwood | 32 |
| Douglas-fir | 29 |
| Western redcedar | 16 |
| ***Total*** **=** | 155 |

**Table 17.** Total number of growth measurements by plant species

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