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Biological and water quality responses to hydrologic disturbances in third-order forested streams

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ABSTRACT

We compared stream ecosystem responses to two types of disturbances: flood and debris flows. A large storm in February 1996 disturbed four similarly sized sub-watersheds of the Calapooia River, in the Cascade Mountains of Oregon, USA. All sub-watersheds had flood and in two, 31 and 81% of the perennial channel, a debris flow affected the channel. For 8 years, we used a suite of approaches: stream temperature, nutrient regime, periphyton and macroinvertebrate assemblages, and resident trout abundance and habitat to evaluate the persistence of instream impacts. Differences in stream temperatures and nitrate-nitrogen concentrations were evident, probably, due to the removal of vegetation and modification of riparian soils at the debris flow sites. Instream biological responses varied. After the event, fish, including trout, were rare with no fish at the debris flow sites. Within 6 years, trout densities (*Oncorhynchus mykiss* and *Oncorhynchus clarki*) were similar and young-of-the-year trouts were common. In contrast, periphyton and macroinvertebrate assemblages differed. Periphyton biomass was lower and nitrogen-fixing periphyton was more abundant at the debris flow sites. Macroinvertebrate assemblage diversity was higher at the debris flow sites due to fewer dominant taxa. Macroinvertebrate functional feeding groups also differed with fewer gatherers and more scrapers at the debris flow sites. Debris flow impacts related to loss of riparian canopy will probably persist until mature red alder stands are re-established along stream-reaches affected by debris flows to provide nitrogen input and shade. Copyright © 2011 John Wiley & Sons, Ltd.

KEY WORDS alder; debris flow; macroinvertebrates; nitrate-nitrogen; periphyton; trout

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INTRODUCTION

Extreme hydrologic events are major factors in the structure and function of fluvial systems (Bayley, 1995; Tockner *et al.*, 2000). In steep mountainous terrain, flood disturbance differs from lowland flooding as water is more confined and flow peaks shorter (Swanson *et al.*, 1998). The high precipitation that causes the flood initiates landslides in the steep upper reaches of watersheds, which create debris flows (Wu and Chen, 2009). These debris flows are one of the dominant geomorphic processes in montane landscapes (Benda and Dunne, 1997). Debris flows initiate when a landslide enters a stream channel and temporarily blocks the flow. Breaching of the debris dam generates a torrent, which carries a large mass of mud, rock, organic matter, and water downstream (Slaymaker, 1988). Once initiated, debris flows typically do not stop until the channel gradient decreases to $<1\%$ (Swanson *et al.*, 1987). Factors such as topography, climate, soils, and vegetation play a key role in determining the susceptibility of a landscape to debris flows (Dunne, 1998; Dietrich *et al.*, 2001; Wu and Chen, 2009). In the montane forested watersheds of the Pacific Northwest, debris flows shape aquatic ecosystems in small and medium-sized streams (Bilby *et al.*, 2003; Bigelow *et al.*, 2007) and are a consistent disturbance mechanism.

Large precipitation events, severe enough to induce debris flows in some streams, likely produce flows sufficient to disturb instream habitat conditions in all nearby streams. In some situations, antecedent conditions such as snowpack can exacerbate flow volume (Jones and Perkins, 2010). These disturbance-inducing storms are typically large enough that watershed scale or regional impacts cause widespread impacts. Unlike floods in low gradient systems where overbank flow disperses peak flow (Bayley, 1995), in steeper, more confined terrain, the flood will translate into greater velocity, more stream power, and the hydraulic capacity to reorganize substrate conditions and reposition or flush out roughness features like large wood. Flood events are also consistent disturbance mechanisms in montane watersheds (Swanson *et al.*, 1998).

Pattern and rate of recovery by aquatic biota in stream ecosystems following floods has received considerable attention, particularly low gradient systems (Schlosser, 1985; Junk *et al.*, 1989; Danehy *et al.*, 1998). However, recovery from flood and debris flow disturbances in steeper mountain streams has received less effort

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(Richardson and Danehy, 2007). While both disturbance types alter the habitats available for aquatic communities, debris flows drastically reconfigure channels (Gomi *et al*., 2002; Hassan *et al*., 2005). The prevalence of exposed bedrock in debris flow channels provides poor habitat for many aquatic invertebrates, and pools that do exist often lack cover, making them less than ideal for fish (Martin *et al*., 1986). Nonetheless, fish do colonize soon after the disturbance in available habitats (Lamberti *et al*., 1991; Roghair *et al*., 2002; Carline and McCullough, 2003; Cover *et al*., 2010). The limited work on lower trophic levels has consistently observed rapid colonization (Anderson, 1992; Snyder and Johnson, 2006) although major shifts in functional groups are evident (Anderson, 1992; Kobayashi *et al*., 2010).

A fundamental difference between flood and debris flow is the obliteration of the riparian corridor by the debris flow as riparian vegetation is either incorporated into it or swept aside. Following the disturbance, it is rare for conifer species to occupy an area impacted by a debris flow without pioneer species, such as red alder (*Alnus rubra*) and willow (*Salix* spp.), becoming established first (Pabst and Spies, 2001). The riparian responses of low gradient streams and small rivers to periodic flooding disturbance and the fluvial biogeomorphologic succession in large alluvial rivers have been described and documented (Bendix and Hupp, 2000; Grunell *et al*., 2001; Steiger *et al*., 2005; Corenblit *et al*., 2007, 2010). While interactions and feedback between landforms and vegetation will occur in these debris flow channels as well, the debris flow disturbance is a severe resetting of riparian conditions. Ten years after a debris flow in the Oregon Coast Range, Pabst and Spies (2001) found red alder to be dominant and providing shade, but there were few conifers established. Typically, conifers replace these early successional species over a period of century. Bilby *et al*. (2003) suggests 150–250 years for not only conifer establishment, but also for complexity in stream habitat conditions to develop with large wood recruitment to the channel. Therefore, in mountain streams the succession dynamics being observed in low gradient streams or large alluvial rivers may differ and perhaps may involve longer maturation timelines.

Recurrence intervals of debris flows for stream channels in the Oregon Coast Range are estimated to range from 98 to 357 years (May and Gresswell, 2003). A substantial body of evidence for forest management, especially forest harvest and road construction, has increased the frequency of landslides (Swanson and Dryness, 1975; Schmidt *et al*., 2001; Imaizumi *et al*., 2008; Turner *et al*., 2010). Loss of tree canopy and subsequent decomposition of tree roots following timber harvest reduces the capacity for roots to stabilize steep terrain during major storms (Sidle and Swanston, 1982; Johnson and Jones, 2000). Road construction side-cast, which until the 1980s, was used to support roadbeds on steep mountain slopes, becomes prone to failure with the degradation of organic material buried in the side-cast. Current forest practices have reduced the occurrence of management-related debris flows (Pyles and Skaugset, 1998). However, debris flow risks are still elevated for many roads that were constructed before more stringent standards were established (Pyles and Skaugset, 1998).

We selected four similarly sized tributaries to the Calapooia River after a large storm to investigate biological and water quality responses to hydrologic disturbances in third-order forested streams. All these streams endured prolonged flood and in two streams the storm caused debris flows. Initial fish surveys found no fish in debris flow streams. Within the small area (<10 km apart), assumed to have similar rainfall, we tracked changes to fish populations and investigated lower trophic community responses. We compared stream conditions by disturbance type, as the storm affected all watersheds in western Oregon. Given the severe channel and riparian disturbance, we expected changes to be apparent and focussed on similarities and differences in interrelated responses.

METHODS

Site description

The Calapooia River watershed drains the west slope of the Cascade Mountains and enters the Willamette River near Corvallis, OR (Figure 1). Douglas fir (*Pseudotsuga menziesii*) is the dominant upland tree species as the watershed is managed for timber production (Table I). The primary forest has been harvested with stand ages ranging from <5 to >50 years. Current forest practice regulations require riparian buffers on all fish bearing streams, prior to 1980 riparian harvest was allowed, which facilitated colonization of riparian zones with pioneer species like red alder (*A. rubra*).

Heavy rains during 5–9 February 1996 initiated landslides that caused debris flows on multiple stream channels in the Calapooia River watershed. In addition to the 10–20 cm of rainfall, there was a significant snow pack at higher elevations that melted and contributed to runoff. There is no flow gage on the Calapooia River; however, flows at gages in the adjacent watersheds to the north (South Santiam River) and south (Mohawk River) experienced flows estimated to represent a 50-year flood event.

We compared physical, chemical, and biological conditions in two streams impacted with a debris flow during this storm (McKinley and Hands Creeks) with two streams affected only by flooding (Potts and Blue Creeks). Characteristics of the study sites were comparable with watershed areas ranging from 563 to 843 ha, reach gradients from 3·6 to 5·8%, and elevations from 302 to 439 m (Table I). The watersheds were <10 km apart (Figure 1). Prior to the storm, all the sites had intact riparian buffers in which aerial photographs (1995) indicated were dominaed by red alder (Table I).

Physical habitat

Within 2 months after the storm, we conducted extensive habitat surveys, measuring pool depths and wood loading throughout the watershed. In 2004, we conducted a

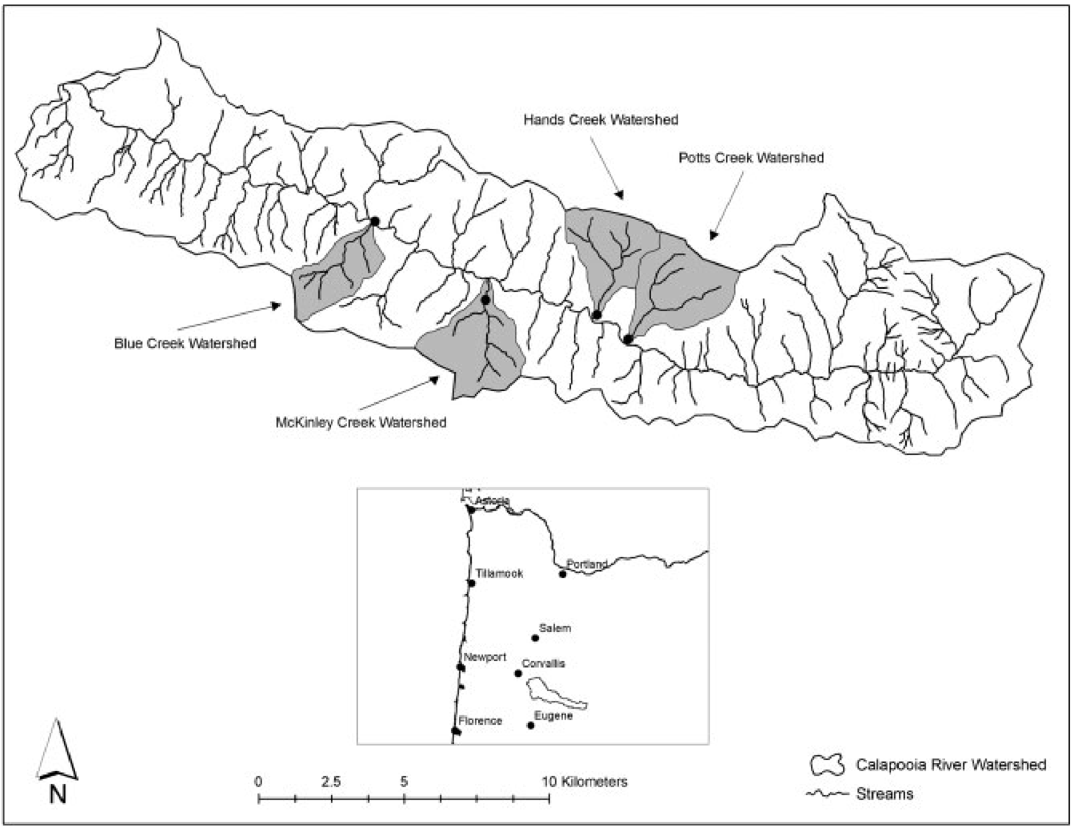

Figure 1. Four sub-watersheds upper Calapooia watershed. Hands and McKinley sub-watersheds had a debris flow in 1996 that affected 81 and 31% of perennial channel, respectively.

Table I. Characteristics of four sub-watersheds and study site reaches in the Calapooia River watershed (Ar–*Alnus rubra* and Pm–*Pseudotsuga menziesii*).

Disturbance type	Potts flood	Blue flood	McKinley debris flow	Hands debris flow
Area (ha)	848	563	769	698
Reach elevation (m)	421	302	439	426
Channel gradient (%)	3·6	5·1	5·8	5·1
% Perennial stream with debris flow	<1	<1	31	81
Mean debris flow width (m)	—	—	17	26
Dominant riparian tree species (pre-storm)	Ar	Ar	Ar	Ar
Dominant upland trees species	Pm	Pm	Pm	Pm
Wood no.·(100-m)	7	2	0	0
Baseflow discharge (m^3·s^{-1})	0·027	0·019	0·039	0·021

detailed assessment of physical habitat for 100-m reaches in the four streams. Substrate characteristics were quantified along the 100-m reach by the same sampler visually estimating the percent composition of bedrock, boulders, cobble/rubble, gravel, and sand (<2 mm) at successive 10-m sections within the bankfull channel. Bankfull widths were measured in 10-m increments. Functioning wood (>20 cm diameter) within the wetted channel was counted. Residual pool depth was measured at every pool. Canopy cover was measured from hemispherical photographs taken with a Nikon Coolpix 995 with a fisheye adapter at 25, 50, and 75 m. Direct below radiation (DBR) at the summer solstice was calculated from the photos using the software Hemiview©. Widths of debris flow tracks were measured at five transects. We determined the percent of perennial channel impacted by debris flows from aerial photographs (1996) to assess the extent of disturbance upstream from each study-reach.

Water quality

Thermographs (Onset tidbit™) measured water temperature in a shaded location near the lower end of each study-reach from May to October 2004. We checked instruments for accuracy with pre- and post-deployment quality control checks (±0·5 °C). Water samples were collected from the study streams, for eight times, between July 2004 and March 2005. These samples were analysed for total nitrogen, nitrate–nitrite nitrogen (nitrate-N), total phosphorus, ortho phosphorus, and total organic carbon (DOC). Before analysis, samples were vacuum-filtered (0·45 μm). We analysed total nitrogen and dissolved organic carbon using a Shimadzu TOV-Vcsh Total

Organic Carbon Analyser and nitrate-N, ammonium-nitrogen, total phosphorus, and ortho-phosphate using a Lachat Quick Chem 4200 analyser. Minimum detection limits for the Lachat were 0·04 mg l^{-1} for nitrogen and 0·01 mg l^{-1} for phosphorus. We observed nitrate-N and phosphate to be occasionally below detection limits and used the detection limit values in analysis for those measurements.

Biological sampling

During April and May 1996, we conducted a reconnaissance survey of fish distribution in tributaries of the Calapooia River. The spring sampling was before young-of-the-year (YOY) trout would typically be out of the redd and susceptible to electrofishing. The four study sites were included in this survey. We electrofished every pool in stream-reaches that were known to have supported fish in an upstream direction. If no fish were observed in debris flow streams in the first 100 m, then only the higher quality pool habitats (e.g. >0·3-m residual pool depth) were shocked. In 2002 and 2004, quantitative fish density surveys were conducted at the four study streams by blocking 100-m reaches with nets and using a two- or three-pass removal technique (second pass where $\geq$1+ trout catch on first pass was 70% of total). We targeted 1+ age trout (>75 mm) for comparisons due to the effort required to reduce YOY adequately for accurate density estimates. YOY trout were counted. A minimum of 1 h elapsed between passes. Fish were identified and fork lengths of each individual measured. Fish were kept in holding wells until the sampling was completed and distributed along the reach after all data were collected.

We collected periphyton, diatoms (Bacillariophyceae), green algae (Chlorophyta), and blue-green algae (Cyanophyta) by placing five sets of two unglazed tiles (150 cm^2) in mid-channel at each site in mid-July to the end of August in 2004. One tile was used for taxonomic identification (preserved in 5% formaldehyde), and the second for chlorophyll *a* determination. We mounted an aliquot of algal suspension from each sample in NAPHRAX to identify and enumerate periphyton. We counted a minimum of 600 diatom valves at 1000× magnification using a Nikon E600 Eclipse microscope with phase contrast. Diatoms were identified to species. References used for diatom taxonomy included Krammer and Lange-Bertalot (1986, 1988, 1991a,b) and Patrick and Reimer (1966, 1975). We counted, and identified to genus, a total of 300 non-diatom, algal units in a Palmer Counting Chamber at 400× magnification. Chlorophyll *a* was extracted using 90% acetone buffered with magnesium carbonate at 4 °C in the dark and measured fluorometrically before and after acidification to correct for pheophytin (American Public Health Association, 1992).

We sampled macroinvertebrates in June 2004 by taking three Surber samples (250-µm mesh) in erosional areas. For each sample, 500–600 individuals were randomly selected and identified. Macroinvertebrates were identified to genus for most taxa, including Chironomidae, with some of the aquatic insects identified to species. Non-insect groups, primarily ostracods, oligochaetes, and nematodes, were not identified beyond class (phylum for nematodes).

Data analysis

Site means of abiotic characteristics were compared between debris flow and flood sites with one-way analysis of variance (ANOVA) (Proc MIXED, SAS 9·1), except for nutrients. Multiple measures of insolation (3), stream morphology (10), and substrates (10) allowed estimate of site means. We calculated stream temperature metrics in 2004 by randomly selecting 5 days during a 34-day period of no rainfall (7/15–8/18) as replicates for comparisons. Nutrient levels were compared between disturbance types with repeated measures models (Proc MIXED, SAS 9·1). We selected the autoregressive covariance structure after comparing AICC (Akaiki Information Criterion Corrected) values from various alternative covariance structures. All tests evaluated differences at $\alpha \leq 0{\cdot}05$.

Differences in trout (only trout >75 mm), macroinvertebrate, and periphyton metrics were compared with one-way ANOVA ($\alpha \leq 0{\cdot}05$) (Proc MIXED, SAS 9·1). Trout (>75 mm) population sizes were estimated for 2002 and 2004 with a multi-pass removal method (Seber and Le Cren, 1967). Fish densities for 2002 and 2004 were calculated by dividing estimated population size of trout (>75 mm) by stream area at the time of sampling. We counted YOY trout (<75 mm) to assess habitat potential to support spawning and early rearing, but did not test for differences, as consistent reduction was difficult. Periphyton metrics included density, percent of Bacillariophyceae, Chlorophyta, and Cyanophyta, and percentage of nitrogen-fixing periphyton (Cyanophyta and *Epithemia turgida,* a diatom, DeYoe *et al*., 1992; Brock, 1973). Macroinvertebrate metrics: taxa richness, dominance (percent contribution of three most abundant taxa), diversity [Shannon–Weaver H′ (log e)] (Magurran, 1988), percentages of common groups (Ephemeropterans, Plecopterans, and Trichopterans combined (EPT) and Chironomidae), and functional feeding groups (FFG) were compared between disturbance types. We assigned categories based on the designations of Merritt *et al.* (2008).

RESULTS

The debris flows affected 81 and 31% of the length of perennial channel in Hands and McKinley Creeks, respectively (Figure 1). The study areas of the flood sites at Potts and Blue Creeks were not directly impacted by debris flows in both watersheds but some minor debris flow impact (<1% of the total length of perennial channel above the study site) did occur in areas upstream (Table I). Along the debris flow study-reaches, bank and overstory vegetation was obliterated for mean widths of 26 and 17 m at Hands and McKinley Creeks, respectively. After 8 years, the riparian corridor was largely

Table II. Comparisons of stream morphology, riparian influence (DBR), thermal pattern, and substrate composition of four third-order streams 8 years after flood and debris flow disturbances with one-way ANOVA.

Variable	Flood	Debris flow	*P*-value[a]
Stream/Riparian			
Bankfull width (m)	1·57	1·54	ns
Residual pool depth (cm)	28·5	21·5	ns
DBR at solstice ($MJ \cdot m^2 \cdot day$)	112	274	*
Thermal pattern			
July T°C maxima	15·4	18·2	*
July T°C minima	13·6	14·3	ns
July maximum diurnal range	1·8	3·8	*
Substrates			
Bedrock (%)	10·3	23·1	ns
Boulder (%)	26·5	24·1	ns
Rubble/Cobble (%)	28·1	30·3	ns
Gravel (%)	18·7	12·2	ns
Sand (<2 mm)(%)	16·8	10·3	ns

[a] not significant (ns) = >0·05;
* $P < 0{\cdot}05$.

Table III. Comparison of disturbance type, time period, and the treatment × time interaction based on the repeated measures analysis for nutrient concentrations collected from July 2004 to March 2005 (eight sample dates) at four third-order streams. All streams had flood and two had a debris flow in February 1996 in the Calapooia watershed.

	Flood mean	Debris flow mean	Disturbance type[a]	Time[a]	Interaction[a]
Total nitrogen (mg l^{-1})	0·320	0·169	ns	ns	ns
NO_3-N+NO_2-N (mg l^{-1})	0·189	0·044	*	**	*
Total phosphorus (mg l^{-1})	0·019	0·018	ns	*	ns
PO_4-P (mg l^{-1})	0·011	0·012	ns	***	ns
DOC (mg l^{-1})	5·21	4·94	ns	***	ns

[a] ns = $P > 0{\cdot}05$;
* $P < 0{\cdot}05$;
** $P < 0{\cdot}01$;
*** $P < 0{\cdot}001$.

young red alder and willow (*Salix* spp.) with little understory vegetation and a surface of bare ground and exposed rock. Overstory vegetation at the flood sites was mature red alder. The debris flow sites had higher solar insolation and maximum July water temperatures during summer than the flood sites (Table II). At all sites there were low amounts of functioning in-channel wood, with no large wood found at the debris flow sites (Table I). Stream width, pool depth, and substrate composition did not differ between disturbance types (Table II). The flood sites exhibited higher nitrate-N concentrations than the debris flow sites (Table III). The other nutrients did not differ (Table III). Nitrate-N was consistently low throughout the year at the debris flow sites, whereas at flood sites it exhibited an annual pattern of increasing concentration in fall through winter, then a decline in spring (Figure 2). There were seasonal differences for all nutrients except total nitrogen, although the pattern was similar between disturbance types (Table III).

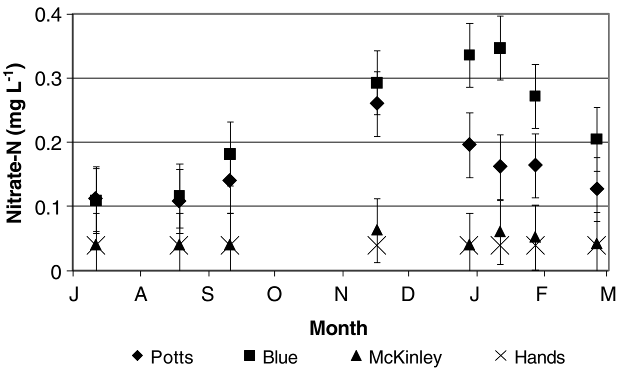

Figure 2. Interaction plot showing nitrate-N concentration (mg l^{-1}) in eight samples collected from 25 July 2004 to 5 March 2005 at two flood only sub-watersheds (Potts and Blue Creeks) and two debris flow sub-watersheds (McKinley and Hands Creeks). Error bars are $\alpha = 0{\cdot}05$.

Fish community

Post-storm fish surveys in early spring of 1996 were conducted throughout the watershed. Within the study-site reaches no fish were captured at the two debris flow sites (or any other debris flow reach surveyed) and only trout at the flood sites. We collected three cutthroat trout (*Oncorhynchus clarki*) at Blue Creek and at Potts Creek five cutthroat trout, and one rainbow trout (*Oncorhynchus mykiss*) (Table IV). Cutthroat trout ranged in size from 90 to 189 mm in Potts Creek and 82 and 131 mm in Blue Creek in 1996 (Table IV). The 2002 and 2004 trout density estimates in June for trout >75 mm (*O. clarki* and *O. mykiss* combined) varied little between the debris flow and flood sites (Table V). The numbers of trout captured in 2002 and 2004 were substantially greater than in 1996, with >30 age 1+ at each site. While it was too early in the year to find YOY trout in 1996, in 2002 and 2004 we found YOY trout outnumbering >1+ trout at all sites.

Only one or two fish species (both trout) were found at the sites in 1996. *Cottus* spp. were absent at all sites. By 2002, fish species richness had increased at three of the sites, with Hands Creek supporting the most species, five (Table IV). *O. clarki* and *O. mykiss* were found at Hands and Potts Creeks and at least two species of sculpin (*Cottus perplexus* and *Cottus rhotheus*) have repopulated three streams. In McKinley Creek, which is above a barrier, no fish were collected in 1996 but by 2002, cutthroat trout had repopulated the study reach (Table IV).

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Table IV. Number of fish species (Species) and cutthroat trout maximum length (mm) (Max CT L) in 1996, 2002, and 2004 fish surveys at study sites in the Calapooia watershed.

Stream	Disturbance type	1996		2002		2004	
		Species	Max CT L	Species	Max CT L	Species	Max CT L
Blue	Flood	1	131	3	160	3	166
Potts	Flood	2	189	4	188	4	175
Hands	Debris flow	0	—	5	170	5	168
McKinley	Debris flow	0	—	1	195	1	190

Table V. Comparison of disturbance type means with one-way ANOVA of fish, macroinvertebrate, and periphyton assemblage metrics in third-order streams that all had flood and two a debris flow in February 1996 in the Calapooia watershed.

Variable	Flood	Debris flow	P-Value[a]
Fish			
2002 Trout density (fish m^{-2})	0·099	0·104	ns
2004 Trout density (fish^{-2})	0·084	0·101	ns
Macroinvertebrates			
Diversity	2·79	3·29	*
Taxa richness	50·3	64·2	ns
Dominance	54·3	33·3	*
% EPT	25·3	36·2	ns
% Chironomidae	38·2	23·3	*
% Gatherers	75·1	63·3	*
% Predators	11·2	15·8	ns
% Shredders	6·5	8·8	ns
% Scrapers	4·5	9·2	*
% Filter feeders	2·4	2·2	ns
Periphyton			
Chlorophyll a (g m^{-2})	0·015	0·007	*
Periphyton density (cells cm^{-2})	$7·7 \times 10^{5}$	$1·2 \times 10^{5}$	*
% Diatom	98·8	87·7	ns
% Nitrogen fixers	0·5	9·1	*

[a] ns = $P > 0·05$; * $P < 0·05$.

Periphyton community

Periphyton summer density was sixfold higher at flood sites than debris flow sites (Table V) and chlorophyll *a* concentration was about double (Table V). Diatoms (Bacillariophyceae) dominated the periphyton community at all sites, comprising 98·8 and 87·7% of the abundance at the flood and debris flow sites, respectively (Table V). We detected green algae (Chlorophyta) only at Hands Creek. More nitrogen-fixing periphyton, both blue-green algae and a diatom, *E. turgida*, inhabited the debris flow sites (Table V). *E. turgida* was absent from the flood sites.

Macroinvertebrate community

Macroinvertebrate taxa diversity was higher and dominance (percentage of three most abundant taxa) lower at the debris flow sites (Table V). Two taxonomic metrics, EPT and Chironomidae, comprised 63·5 and 59·5% of the assemblages at the flood and debris flow sites, respectively (Table V). Chironomidae were more abundant at the flood sites than the debris flow sites. There also were differences in the FFG with more gatherers at the flood sites and more scrapers at the debris flow sites (Table V). Each site supported eight or nine scraper taxa but was dominated (89%) by four taxa. Two (*Cingmula* spp. and *Glossosoma* spp.) were well represented at all sites and two (*Optioservus* spp. and *Epeorus longimanus*) were more abundant at the debris flow sites.

DISCUSSION

We compared the responses of multiple trophic levels of aquatic organisms to disturbance caused by flood flows and debris flows. We classified sites by extent of debris flow occurrence and assumed given their proximity that they received similar amounts of rainfall on similar depths of snow. These post-storm data provided us with an opportunity to conduct a case-study evaluation on these systems to evaluate relative changes in conditions after 8 years. The extent and range of physical impacts from this storm have been described by Johnson *et al.* (2000) and Nakamura *et al.* (2000) in an adjacent watershed. Our observations are consistent with those studies that even sites with the 'less' severe impacts (flood sites), probably had flows with enough power to move and rearrange most of the streambed. There is limited research on biotic recovery of aquatic systems from debris flows and while we do not have pre-disturbance instream biological data, the study provides an 8-year viewpoint. Only the fortuitous circumstance of having a study ongoing at the time of a debris flow occurrence allowed Lamberti *et al.* (1991) and Roghair *et al.* (2002) to use the before and after approach. We do know that these similarly sized streams had similar riparian conditions based on aerial photography. Ongoing differences in light, thermal regime, and nitrate-N concentrations created distinctions in the periphyton and macroinvertebrate communities between disturbance types.

The 15–20 cm of rainfall from 5–9 February on snow covered hills in 1996 created strong hydrologic and hydraulic consequences for these ecosystems. The low fish populations and richness following the storm at flood sites indicate that the flood did affect fishes. The debris flows, however, represent a much more intense type of disturbance, as evidenced by the complete absence of fish at these sites in 1996. Nonetheless, within 6 years all study streams supported trout populations (*O. clarki* and *O. mykiss* combined) of comparable densities. These trout densities (0·08–0·1 fish m^{-2}) are within the range reported by Platts and McHenry (1988) as typical for western US rivers. Lamberti *et al.* (1991) found cutthroat trout densities to return to pre-disturbance levels within a year after a debris flow in Quartz Creek, a stream nearby our study sites and Cover *et al.* (2010) observed abundant rainbow trout in streams with recent debris flows in the Klamath Mountains of northern California. Roghair *et al.* (2002) and Carline and McCullough (2003) also found rapid recovery of brook trout (*Salvelinus fontinalis*) populations in streams impacted by debris flows in Virginia and West Virginia. These results indicate that trout populations often recover rapidly following severe fluvial disturbances.

In addition, the flood sites and the debris flow site without a fish barrier supported fish communities as diverse as is typically seen in small streams in the western Cascade Mountains in the Pacific Northwest by 2002, 6 years after disturbance (Fransen *et al.*, 2006). We did not have information on fish community composition prior to the storm, so it is uncertain whether the fish community has fully recovered to pre-1996 storm condition. However, in addition to the trout rebounding, we observed a return of two species of *Cottus* spp. at the three sites not above a barrier. We do not know if these species repopulated the reaches by upstream migration from the mainstem of the Calapooia River, or if some fish were able to persist somewhere in a tributary and subsequently repopulated the study reach. In addition, we observed a repopulation of resident cutthroat trout at the McKinley Creek study-reach despite the fact that there is a waterfall barrier between the study-reach and the mainstem Calapooia River. Therefore, even in a physically isolated stream-reach, fish surviving the debris flow in an upstream refuge can repopulate the reach in a few years.

The differences in light, thermal regime, and nitrate-N concentrations after 8 years, between the flood and debris flow sites, influenced the organization of lower trophic levels of the stream ecosystems. Impacts at the flood sites may also persist but comparisons with sites without flood impacts are not possible, since the storm affected the entire region. Lamberti *et al.* (1991) found a rapid increase in periphyton abundance after a debris flow on Quartz Creek. They attributed this increase to increased light and reduced macroinvertebrate grazing, hypothesizing that algal productivity provided the bioenergetic-basis for recovery of aquatic productivity. While increases in algal production could also have occurred shortly after the storm at our study streams, 8 years later, periphyton was less abundant at the debris flow sites.

Two lines of evidence suggest that nitrate-N limitation was contributing to the lower levels of periphyton production at debris flow sites. Light levels were higher at the debris flow sites, yet chlorophyll *a* was twice as high at the flood sites, indicating that a factor other than light was limiting. The role of red alder both in riparian and upland habitats in nutrient cycling has been well-studied (Binkley *et al.*, 1994), as has its contribution to instream nitrogen concentrations (Compton *et al.*, 2003; Evans *et al.*, 2011). Because of the debris flows, red alder has begun to reoccupy the riparian corridor along the debris flow streams. In contrast, the flood sites are bordered by a dense stand of mature red alder. The lower nitrate-N concentration at debris flow sites indicates periphyton production may continue to be depressed until red alder is well established along the stream. A second indicator of low nitrogen on periphyton production at the debris flow streams was the higher occurrence of species capable of fixing atmospheric nitrogen, Cyanophyta (blue-green) algae and the diatom *E. turgida*, at the debris flow sites.

The flood sites supported a less diverse macroinvertebrate assemblage, largely due to greater dominance by a few taxa. Two of the dominant taxa, the chironomids (*Tvetenia bavarica* gr. and *Micropsectra* spp.) comprised 15–30% of the assemblage at flood sites. These taxa feed by gathering fine particulate organic matter (FPOM) and a principal source of FPOM is terrestrial plant material. The developed and functioning riparian canopies of the flood sites produce more organic matter, both locally and from upstream, as the watersheds were largely without debris flow impacts. Diversity at the debris flow sites was also enhanced by higher abundances of scrapers. Snyder and Johnson (2006) in the Blue Ridge Mountains of Virginia found a similar shift towards scrapers after debris flows. Anderson (1992) observed higher macroinvertebrate diversity at Quartz Creek after a debris flow and hypothesized that this response was related to increased diversity of substrate types, which he termed 'habitat patches'. We were unable to discern differences in substrate composition. All sites had low instream roughness and substrate composition was highly variable and poorly sorted.

Initial patterns of recovery are strongly influenced by the rate at which riparian vegetation develops following disturbance. Therefore, while succession will occur from the bare soil to maturing riparian conditions, riparian development to sufficiently influence instream aquatic habitat conditions will likely require conifer recruitment. Biogeomorphologic succession in these mountain streams should be different from that seen in large alluvial rivers (Steiger *et al.*, 2005; Corenblit *et al.*, 2010). Nevertheless, maturation of the riparian corridor will provide positive influences to instream communities. Thermal conditions after debris flows will change as the riparian canopy becomes re-established over the channel (Johnson and Jones, 2000). Channel substrate conditions will likely not fully return to pre-disturbance conditions until wood pieces, large enough to maintain position in the channel, are delivered; although some recruitment of less durable wood such as alder can initiate for some sorting of substrates. Nitrogen availability will increase as red alder reoccupies the riparian zone along the debris flow streams. It is unknown how long it will take the riparian alder to raise nitrate-N concentrations to pre-disturbance levels, but our results found nitrate-N still low after 8 years. A reduction in nitrogen-fixing periphyton species at debris flow sites may provide a good indication of recovery in nitrogen availability and be a useful tool in assessing recovery.

Each of these fundamental drivers to stream ecosystems–thermal conditions, substrate, and nutrient regimes –are tied to riparian development. A conifer canopy would not be expected to replace the deciduous trees until a century or more (Beschta *et al.*, 1987). Pabst and Spies (2001) found that conifers were unable to establish for 10 years after a debris flow in the Oregon Coast Range and no conifer were established at our debris flow sites after 8 years, demonstrating the lag in the arrival of these species at highly disturbed sites. Therefore, input of conifer wood may take centuries before it attains rates comparable to those before disturbance (Bilby *et al.*, 2003). Given that the reoccurrence interval of debris flows for some channels (May and Gresswell, 2003) has been estimated to be less than the time required for the re-establishment of mature conifer trees. Mature conifer riparian condition may never be attained along some small streams in steep terrain in the Pacific Northwest.

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