Evaluating Headwater Stream Buffers: Lessons Learned from Watershed-scale Experiments in Southwest Washington

Peter A. Bisson, Shannon M. Claeson, Steven M. Wondzell, Alex D. Foster, and Ashley Steel

Abstract

We present preliminary results from an experiment in which alternative forest buffer treatments were applied to clusters of watersheds in southwest Washington using a Before-After-Control-Impact (BACI) design. The treatments occurred on small (~2- to 9-ha) headwater catchments, and compared continuous fixed-width buffered, discontinuous patch-buffered, and unbuffered streams to an adjacent unlogged reference catchment. Eight treatment clusters were monitored from 2001 to 2006; four were located in the Black Hills (Capitol State Forest) and four in the Willapa Hills of the Coast Range. Logging took place in 2004 or 2005, depending on the cluster. The study streams were too small to support fishes, but catchments did harbor amphibians, aquatic invertebrates, and riparian mollusks. In addition to biota, we examined water quality, discharge, and organic matter inputs. The intent was to monitor the sites two years pre-treatment and three years post-treatment, although unforeseen circumstances caused some exceptions. Overall, results suggested that relatively small but measurable changes in ecological condition occurred in most catchments where logging occurred. Changes were most apparent in streams having no buffers. In catchments with no buffers, summer water temperature increases were largest, organic matter inputs declined, and drifting invertebrates increased or decreased depending on their trophic guild. Changes in catchments with discontinuous patch buffers were often complex and generally less detectable, and streams with continuous fixed-width buffers tended to exhibit the fewest changes in invertebrate communities and organic matter inputs relative to reference sites.

Analyses of ecological response, both physical and biological, were fraught with difficulty. Difficulties in executing the study and analyzing results include operational planning and scheduling, spatial and temporal variability, and unplanned environmental disturbances. Based on our experience, we offer suggestions for future research on riparian management in small watersheds. First, think of watershed-scale studies as interventions that are carried out so as to maximize what we can learn from them, rather than as true experiments. Second, if the study covers several locations, thoroughly characterize differences in physical and biological features among the sites before treatments are applied to avoid misinterpreting results. Third, be prepared to accommodate uncontrolled environmental disturbances (e.g., droughts, floods, wildfires, etc.) that are inevitable in multi-year investigations. Fourth, once the basic experimental layout is

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established, resist the temptation to switch treatments midway through the study, which will only confound analyses. Finally, when surprises occur, be flexible enough to monitor their effects to take advantage of learning opportunities.

Keywords: riparian management, buffers, headwater streams, small watershed studies, experimental design, BACI.

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Introduction

When the Northwest Forest Plan (NWFP) was implemented in the late 1990s, one of the most significant information gaps was whether the new buffer guidelines in the NWFP conserved sufficient amounts of riparian forest to protect fish habitat in headwater streams. The default guideline in the NWFP called for riparian buffers as wide as or wider than a site-potential tree height on each side of stream channels. While there were theoretical reasons to believe that wide buffers protected headwater stream functions and processes (Sedell et al. 1994), field studies verifying the efficacy of wide buffers were lacking. Furthermore, the NWFP's riparian conservation areas were focused on habitat protection for salmonid fishes, but questions remained about whether the guidelines adequately protected the habitats of other aquatic and riparian-associated species or the processes that provide organic matter and sediment to downstream salmon-bearing rivers.

To address the scarcity of information about the effectiveness of buffers of different width and forest age, we initiated a study in 1996 (Riparian Ecosystem Management Study—REMS) that employed a synoptic survey approach in which a large number of streams in watersheds with different logging histories and different buffer patterns were compared. The research was located on Washington's Olympic Peninsula, and included both aquatic and riparian-associated vertebrates (Bisson et al. 2002; Raphael et al. 2002). As a part of REMS, faunas were compared for small, perennially-flowing streams that included sites where no buffers were left during timber harvest, sites where narrow (<20 m) buffers were left, sites where wide buffers that had been thinned were left, and sites where intact unmanaged buffers >100 m were left. Using correlation analysis, we found that fewer fish were associated with late-seral riparian buffers; however, fish densities were strongly influenced by local stream habitat conditions such as the abundance of pools. Further, fish abundance was positively correlated with riparian characteristics associated with increased primary production —light gaps, nutrients and deciduous organic matter inputs. Stream-dwelling amphibians were positively correlated with late-seral forest in riparian zones and the amount of late-seral forest in their watersheds. Their abundance was also negatively associated with roads.

While these observations were interesting and our results were consistent with the findings of other stream investigations in the Pacific Northwest, we felt that a broad-scale survey approach relying on correlation analysis did not

yield many new insights into how environmental factors related to riparian buffers directly or indirectly influenced the vertebrates of interest. To learn more about the effects of different-sized buffers on headwater streams, we undertook a second phase of REMS that applied alternative riparian buffer treatments to entire small watersheds using a Before-After-Control-Impact (BACI) experimental design. We anticipated monitoring the study catchments for two years prior to treatment and three years after treatment. We also decided to focus our research on the very smallest non-fish bearing headwater streams. The influence of riparian buffers on very small headwater streams had been little studied, and the current Forest Practices Act in Washington state does not require that buffers be left along non-fish bearing channels on state- or privately-managed lands.

This paper describes the planning, design, implementation, and results of the second phase of the REMS study. We examine the problems and unforeseen surprises encountered over the course of the study, and the lessons learned that might be applicable to future watershed-scale studies of a similar nature. The BACI approach is widespread in ecological science (Downes et al. 2002), but we experienced a number of design, implementation, and analytical hurdles in applying it to our situation. The concluding section of the paper suggests general guidelines for investigators contemplating similar studies. These lessons are largely independent of the specific results of REMS, but hopefully will be of some use in future research.

Site Selection

We experienced many of the difficulties in conducting landscape-scale experiments in AMAs (Adaptive Management Areas—see *http://www.reo.gov/ama/index.htm*) that were discussed by Stankey et al. (2003). Our original intention was to compare the NWFP default buffers for headwater streams with other buffer configurations on the Olympic Peninsula where the first REMS phase was completed. We hoped the study could take place in an AMA that was designated as a place where departures from the standards and guidelines in the NWFP could be tested experimentally. We found, however, that the Olympic National Forest was unable to implement a watershed-scale experiment that involved different stream buffers in any AMAs on the Olympic Peninsula. Two other national forests in western Washington also declined to participate in the study. Several reasons were given, including the scale at which we wanted to apply the treatments, concerns over potential environmental litigation, and conflicts between our proposed experiment and other forest management plans. Taken together, these restrictions made it impossible for the research to take place in an AMA.

We then approached the Washington Department of Natural Resources (DNR) to explore opportunities for implementing watershed-scale buffer trials in western Washington on state-managed lands. Under DNR's guidance several study areas were located in the Coast Range of southwestern Washington (fig. 1) where alternative buffer treatments could be applied to small non-fish bearing streams in headwater catchments of approximately 5 ha.

Objective and Treatments

The objective of REMS Phase 2 was to compare the ecological effects of 15- to 20-m continuous ("fixed-width") buffers on each side of the channel, discontinuous ("patch") buffers, and no buffers ("clearcuts"), with adjacent reference sites of mature second-growth forest. One to two site-potential tree height buffers in the NWFP were not included among the treatments, as these were not an option under DNR's Habitat Conservation Plan (Washington State Department of Natural Resources 2005).

Current Washington state law does not require buffers on small non-fish bearing headwater streams, i.e., those extending from the channel

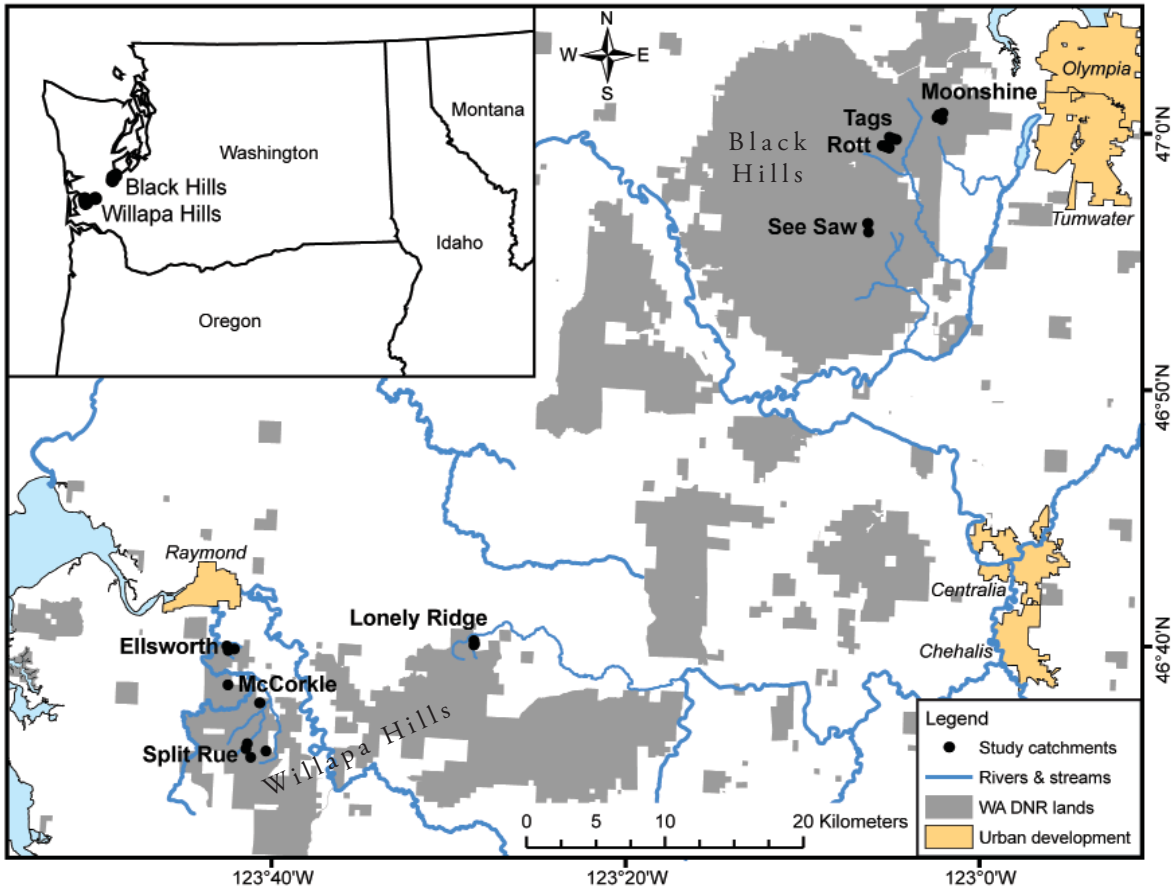

Figure 1—Location of eight clusters of non-fish-bearing headwater streams sampled in REMS Phase 2 (Riparian Ecosystem Management Study) within two regions of southwestern Washington, USA. Capitol Forest sites are located in the Black Hills area near Olympia, WA, and the Willapa Hills sites are located near Raymond, WA.

initiation point to the confluence with a larger stream. These streams, however, are protected from heavy equipment intrusion and in-channel yarding during logging operations. The fixed-width and patch-buffer treatments in our study represented alternative protection measures that would provide additional protection, beyond heavy equipment exclusion, for very small headwater streams, but these buffers would not include the relatively wide site-potential tree height buffers that were the default standard on federal lands. Thus, our study was most directly applicable to headwater streams in state and private industrial forests.

We selected clusters of small watersheds in two locations—Capitol State Forest southwest of Olympia, Washington, and the Willapa Hills, an area of mixed forest ownership which drains into Willapa Bay north of the Columbia River (fig. 1). Each location contained four clusters of headwater streams, and each cluster contained a reference catchment and a combination of treatment types (table 1). It was our goal at the outset to establish a balanced experimental design with each treatment represented in each cluster; however, this turned out to be impossible for operational reasons. Cluster size ranged from three to five catchments because timber sale plans and forest stand characteristics did not always accommodate harvesting small watersheds in the exact configuration we desired. Occasionally a cluster would possess two similar treatments and omit one treatment type. Furthermore, engineering considerations mandated that reference catchments be located at an end of a cluster in order to avoid roads and other harvest-

Table 1—Physical characteristics of eight clusters of 30 headwater catchments in the Pacific Coast Range, Washington, USA. Table modified from Janisch et al. (2012).

Cluster	Logging initiation	1st post-logging year	Catchment logging treatment	Area[a] (ha)	Channel length[b] (m)	Bankfull width[c] (m)	Channel gradient[d] (%)	Aspect	Elevation[e] (m)	Seasonally dry
Black Hills (Capitol Forest)										
Moonshine	July 2005	2006	Reference	8.5	173	1.2	36	W	393	yes
			Patch	8.5	270	0.6	32	W	287	no
			Fixed-width	2.7	176	0.4	35	W	318	yes
			Patch	4.8	176	1.8	37	W	390	no
Rott	April 2004	2004	Reference	6.0	391	0.9	29	S	246	no
			Fixed-width	7.3	403	0.8	37	S	288	no
			Clearcut	4.5	123	0.9	39	S	303	yes
			Patch	5.1	165	0.4	42	S	314	no
See Saw	September 2003	2004	Reference	6.5	173	2.3	27	N	336	no
			Clearcut	2.1	229	0.4	13	N	212	no
			Fixed-width	4.2	273	0.7	18	NW	212	no
Tags	January 2004	2004	Reference	5.5	206	—	46	NE	193	no
			Fixed-width	3.9	241	—	41	NE	203	no
			Patch	4.4	270	—	45	NE	218	no
			Patch	5.4	280	—	40	NE	230	no
			Clearcut	4.9	297	—	38	NE	234	no
Willapa Hills										
Ellsworth	February 2005	2005	Reference	1.9	111	—	17	NW	64	no
			Clearcut	3.5	255	0.7	18	SW	28	no
			Fixed-width	8.1	375	0.7	11	SW	12	no
Lonely Ridge	March 2004	2004	Reference	2.8	209	0.5	24	E	168	yes
			Clearcut	1.9	184	0.4	30	E	168	no
			Fixed-width	3.3	263	0.6	21	E	168	no
			Patch	3.1	282	1.4	25	E	169	no
McCorkle	November 2003	2004	Reference	2.7	311	1.8	24	NW	121	no
			Fixed-width	2.6	146	0.4	17	SE	110	no
			Fixed-width	3.5	155	0.5	18	S	110	no
Split Rue	May 2004	2005	Reference	6.2	229	—	26	N	225	no
			Clearcut	4.9	168	—	22	NE	205	no
			Fixed-width	8.1	480	—	21	N-NE	292	no
			Clearcut	3.4	203	—	27	SE	186	no

[a]Headwater catchment area derived from stereo pairs and ERDAS Stereo Analyst®.
[b]Channel length defined as the confluence to headwall or uppermost point of channel definition.
[c]Channel bankfull width calculated as the weighted mean of sub-segments in 2003.
[d]Channel gradient calculated as the weighted mean of sub-segments.
[e]Elevation of the channel confluence as determined from state 30-m DEM.

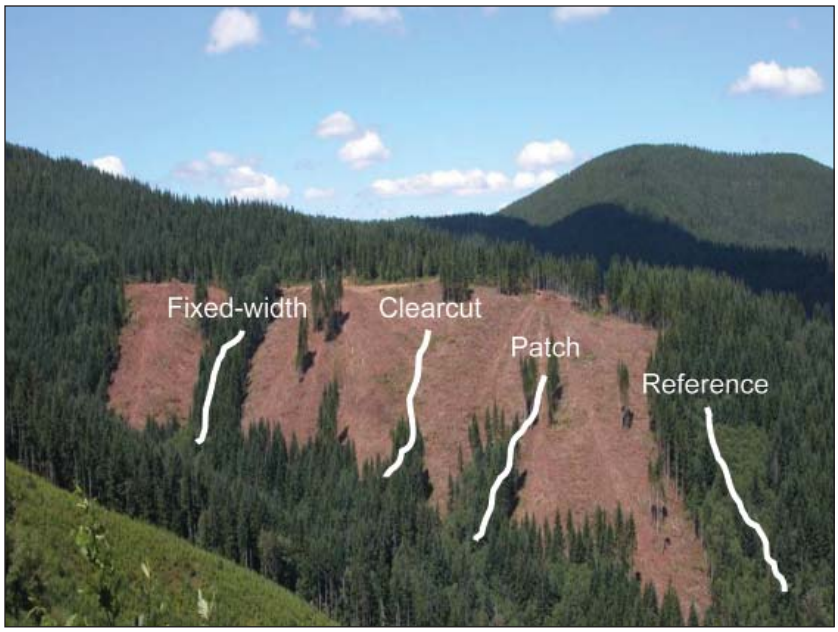

Figure 2—One of the experimental buffer treatment clusters in Capitol Forest showing fixed-width, clearcut, patch, and reference catchments, respectively, from left to right. Stream channels are shown in white. (Photo by Randall Wilk, USFS.)

related factors. For example, the reference catchments for the See Saw and Split Rue clusters were located 1.2 and 3.3 km, respectively, from the treatment catchments. Apart from these constraints, treatments were assigned randomly to individual catchments. A photograph of one of the clusters is shown in figure 2. The lower reaches of the study streams were always forested because they passed through the riparian management zone (RMZ) of the fish bearing stream into which they discharged. All measurements of treatment effects were made at, or above, the boundary of the RMZ into which the headwater stream flowed. Typical views of riparian areas after treatments were applied are shown in figure 3. The streams maintained surface flow continuously during the wet season, but the channels occasionally became intermittent during the dry season, and surface connections with the parent stream were often disrupted, although subsurface flow pathways likely remained active.

Implementation and Monitoring Surprises

We experienced several surprises during the pre-treatment, treatment, and post-treatment periods of the research. Some surprises were management-related; others resulted from natural events. In addition to being unable to establish a complete and balanced study design and to fully randomize treatment assignments, the desired one-year treatment interval was spread over two years when shifts in stumpage prices delayed several timber sales. Furthermore, the delay in harvesting some catchments took place when the regional weather pattern was changing from a multi-year warm, dry period to a cool, wet period caused by a transition from an El Niño to a La Niña weather cycle (Bumbaco and Mote 2010). The change in temperature and precipitation during the treatment period altered the hydrologic patterns of the study streams and, coupled with the harvesting delay in some clusters, added to the difficulty of sorting out treatment effects from weather-mediated changes in the parameters that were monitored.

In 2006, after the timber had been harvested from all study locations, some of the harvest units were treated with aerially applied herbicide as part of routine site preparation prior to reforestation. Although the helicopter operators exercised caution when applying the herbicide, it is likely that occasional overspray or wind drift occurred and some chemical entered the streams; however, precise wind strength and direction data

Figure 3—Post-logging appearance of typical reference and experimental treatments. (Photos by Peter Bisson.)

at the sites were not available. Because this was an unanticipated management action, we were not fully prepared to sample its effects on stream biota, and the fact that some harvest units, but not others, were sprayed made it impossible to determine if herbicide application significantly affected the results of the study.

Additional surprises occurred when severe windstorms in the winters of 2005, 2006, and 2007 caused extensive damage to some of the buffer treatments, especially at several of the Willapa Hills catchments (fig. 4). As a result of the windthrow, some fixed-width buffer treatments were altered to such an extent that they resembled patch treatments. Stream channels at sites with extensive disturbance often contained large amounts of fine sediment that entered streams when root systems were upturned. The windthrow also damaged a number of monitoring stations for forest litter and riparian-associated mollusks. Although windstorms were not factored into the original experimental design, we note that coastal watersheds in western Washington frequently experience high wind events in fall and winter, so occurrence of such storms during the REMS study was not out of the ordinary.

Finally, in November 2007 an intense storm passed through southwestern Washington and some watersheds received >30 cm of rain within 48 hours. Mass wasting was extensive throughout the Coast Range, and a large landslide and accompanying debris flow occurred in one of the patch-buffered study streams (Moonshine cluster) in Capitol Forest. The debris flow caused massive channel scouring and entrainment of riparian vegetation and destroyed existing monitoring equipment. Despite all these challenges, we feel the work yielded some valuable information with relevance to headwater stream management and it taught us several important lessons about

Figure 4—Top: extensive windthrow in a fixed-width buffer site after a windstorm in 2005. Bottom: view of the stream channel showing sediment input caused by upturned rootwads. (Photos by Peter Bisson.)

pitfalls in designing and executing watershed-scale manipulations.

Results

Results are presented for stream temperature, aquatic and riparian invertebrates and terrestrial litter inputs (fine particulate organic matter and coarse particulate organic matter). Due to time and budget constraints some samples were obtained only from a subset of study locations (table 2). Other aspects of the study were headed by other investigators and those aspects are not reported here. We briefly describe methods and statistical tests, but additional details of sampling methods and analytical techniques will appear in subsequent technical reports and publications. Below we present findings of the research thus far.

Stream Temperature

All streams were instrumented with recording temperature loggers and comparisons among treatments focused on the daily maximum temperatures during the July-August period of peak yearly temperatures (Janisch et al. 2012). Relationships between daily maximum stream temperatures in the treatment catchments and their associated reference sites during the pre-treatment phase of the study were used to predict daily maximum temperatures during the warmest part of the summer. After treatments were applied, departures from expected values were computed to determine if the treatment streams differed significantly from what was expected (H_o = no difference). Results for the first post-treatment year, which would be assumed to show the greatest treatment effects, are displayed in figure 5.

Stream temperatures in the unlogged reference catchments generally matched predictions, i.e., no significant changes occurred during the first post-treatment year. Average increases in daily maximum stream temperature were greatest in the clearcut sites, but the other two treatment types also exhibited significant thermal increases. Somewhat surprisingly, stream temperatures in the fixed-width buffer treatments exhibited a greater increase in the first post-treatment year than those in patch buffer treatments, although air temperature increases in the fixed-width buffer sites increased less than air temperatures in the patch buffers (Janisch et al. 2012). Maximum daily summer stream temperatures rose in each treatment category; however, overall temperature increases were small, the trends did not entirely match expected treatment effects (changes tended to be greater at sites with continuous fixed-width buffers than sites with patch buffers), and the responses were highly variable within treatment categories. Further analyses by Janisch et al. (2012) indicated that water temperature

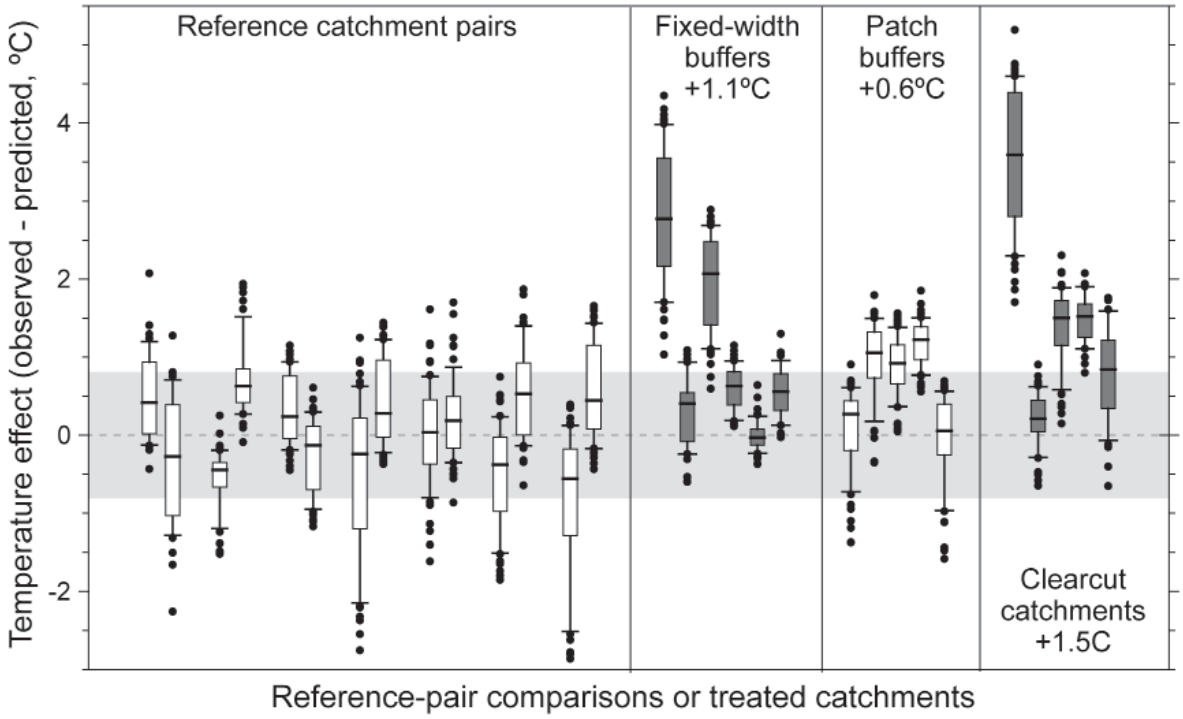

Figure 5—Changes in the maximum daily stream temperature (°C) in the first post-treatment year (or post-calibration year for reference catchments) during July and August at each site. The box and whisker plots denote the mean, quartiles, and 10- and 90-percentiles of water temperature deviations (observed minus predicted daily maximum temperature) in each catchment. Points represent more extreme daily departures from predictions in the first post-treatment year. The 95 percent prediction intervals for the daily random disturbance (in temperature variation; grey-shaded zone) was calculated as 0.00 ± 1.96 · SD of the single largest SD of all pairwise comparisons among reference catchments in the calibration year. The average change in maximum daily stream temperature for each treatment is indicated by a bold line in each box (after Janisch et al. 2012).

was strongly influenced by streambed texture, the presence of small wetlands in the catchment, and the length of channel exposed to sunlight. This suggests that, in addition to the presence or absence of forest canopy, the length of exposed stream channel and the amount of hyporheic water exchange are important factors regulating headwater stream temperatures.

Aquatic Invertebrates

The role of non-fish bearing streams in exporting aquatic invertebrates (potential food items) to fish-bearing streams is poorly understood; however, there is evidence that invertebrates exported from non-fish bearing streams provide important food resources for salmonids downstream (Wipfli and Gregovich 2002). The extremely heterogeneous substrates of the headwater channels, which consisted of a wide range of sizes of unsorted inorganic particles as well as tree roots and a variety of coarse and fine wood fragments, precluded benthic invertebrate sampling. Instead, invertebrates present in the surface water drift were sampled at gauging stations located at the downstream boundary of the headwater streams where the channel entered the riparian management zone of the larger stream into which it flowed.

Drifting invertebrates were collected from six of the eight clusters, three clusters in Capitol Forest and three in Willapa Hills (table 2). Weirs with pass-through PVC pipes were placed in each study stream so that drift nets attached to the pipes were able to capture the entire surface flow during spring, summer, and fall sampling. Drifting invertebrates were collected in 250-µm mesh tubular nets, fished for one full day

Table 2—Type of samples collected from eight clusters of headwater catchments in the Coast Range of Washington, USA. "yes" indicates that metric was sampled, "no" indicates that metric was not sampled at that particular catchment.

Region	Cluster	Treatment	Water temperature	Aquatic invertebrates	Riparian mollusks	Riparian literfall
Black Hills (Capitol Forest)	Moonshine	Reference	yes*	no	no	no
		Patch	no	no	no	no
		Fixed-width	no	no	no	no
		Patch	yes	no	no	no
Black Hills (Capitol Forest)	Rott	Reference	no	yes	no	yes
		Fixed-width	no	yes	no	yes
		Clearcut	no	no	no	yes
		Fixed-width	no	yes	no	yes
Black Hills (Capitol Forest)	See Saw	Reference	yes*	yes	no	yes
		Clearcut	yes	yes	no	yes
		Fixed-width	yes	yes	no	yes
Black Hills (Capitol Forest)	Tags	Reference	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
		Patch	yes	yes	yes	yes
		Patch	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
Willapa Hills	Ellsworth	Reference	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
Willapa Hills	Lonely Ridge	Reference	yes*	yes	no	yes
		Clearcut	no	yes	no	yes
		Fixed-width	yes	yes	no	yes
		Patch	yes	yes	no	yes
Willapa Hills	McCorkle	Reference	yes	no	no	no
		Fixed-width	yes	no	no	no
		Fixed-width	yes	no	no	no
Willapa Hills	Split Rue	Reference	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes

*Stream was dry during the pre-treatment (calibration) year, so an alternative reference catchment was used for analysis.

at monthly (2003) or bi-monthly (2004–2006) intervals. Water temperature and stream discharge were measured at the beginning and end of each 24-hour period. Drift density (individuals·m^{-3} water) was calculated by dividing invertebrate export by the mean discharge of the 24-h sampling period. Invertebrates were categorized as insect or non-insect, aquatic or terrestrial, and by their functional feeding group as defined by Merritt and Cummins (1996). Drift samples were obtained in spring, summer, and fall, but snow prevented access to some of the sites in winter and no winter data are reported here. We tracked all natural log-transformed, post-year minus pre-year differences of seasonally averaged drift densities. Simple linear regression models were used to detect significant effects ($p < 0.05$) among buffer treatments, post-harvest year, and clusters. Because only three of the six clusters received a patch treatment, the sample size was too small for statistical modeling; therefore, no data from the patch treatments are presented here.

The most common terrestrial invertebrates in the drift were mites (Acari) and springtails (Collembola). Drift density proportions of terrestrial invertebrates increased significantly in clearcuts during summer compared to changes at the fixed-width buffer and reference locations (fig. 6). The proportion of terrestrial invertebrates also increased at clearcut sites relative to reference sites in fall. In spring, no significant differences were observed among the treatment and reference streams.

Mayflies (Ephemeroptera) were abundant aquatic invertebrates in all headwater streams, where they were represented by taxa adapted to scraping periphyton from the substrate (scrapers) or taxa that consumed organic detritus (collector-gatherers). No significant differences in mayfly drift densities were observed among treatment or reference streams in the spring or summer, but mayfly densities in both clearcut and fixed-with buffer treatments declined relative to reference catchments in fall (fig. 7). This was surprising because small mayflies have been reported to prosper in streams where canopy openings have favored increased periphyton production (Hawkins et al. 1983). It is possible that logging and windthrow-related fine sediment inputs

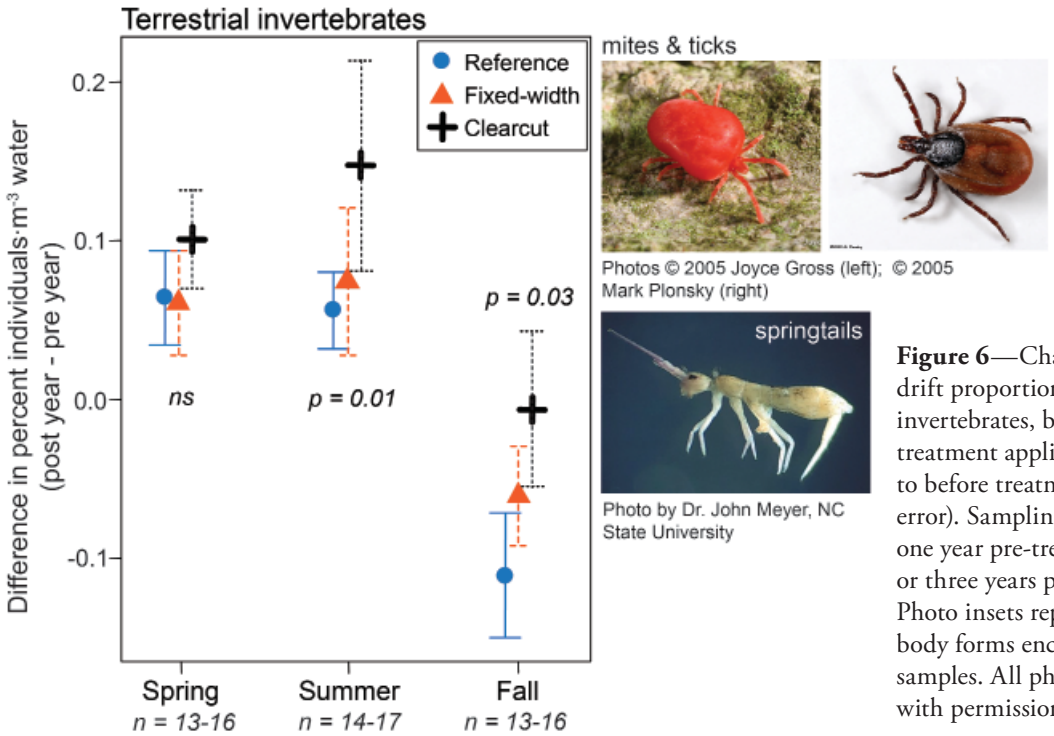

Figure 6—Change in the mean drift proportions of terrestrial invertebrates, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

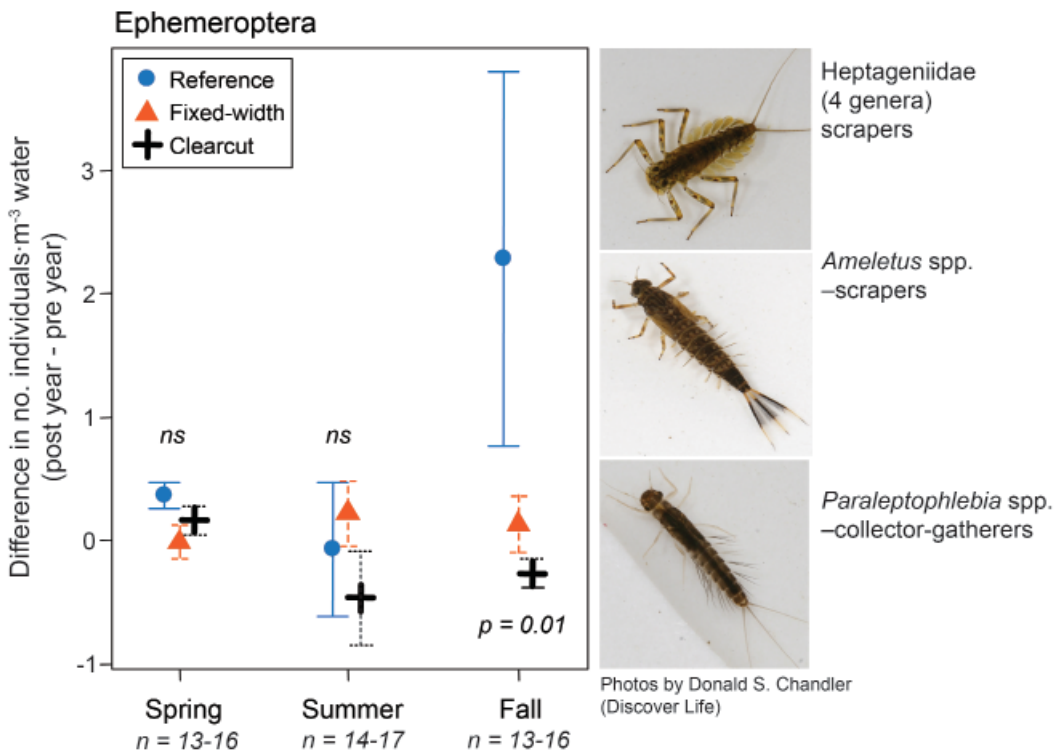

Figure 7—Changes in the mean drift densities of mayflies, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

contributed to mayfly reductions at the study locations, as taxa that occurred there are known to prefer coarse substrates.

Midge larvae (Chironomidae) are frequently the most ubiquitous invertebrates in small streams in the Pacific Northwest, and two midge groups (Orthocladiinae and Tanytarsini) were well represented in drift samples. Small-bodied midge larvae commonly enter the drift and often constitute a substantial portion of the diet of rearing salmonids (Chapman and Bjornn 1969). Midges decreased in clearcut catchments relative to the fixed-width buffer and reference catchments in the fall (fig. 8). In the spring and summer, no significant differences among treatment and reference streams were observed. Extremely high variability in spring prevented an apparent increase in midge drift densities in the treatment catchments from being statistically significant.

We classified aquatic invertebrates in drift samples into feeding guilds to determine if buffer characteristics affected functional feeding groups. Feeding guilds included scrapers (organisms that feed on periphyton), shredders (organisms that "shred" leaves and other riparian inputs), and collector-gatherers (organisms that feed on fine particulate organic material). No changes in shredder density proportions were observed in spring, summer, or fall (fig. 9). Significant declines in scrapers occurred in fall at clearcut catchments relative to fixed-width buffer and reference locations. Likewise, collector-gatherers decreased at clearcuts in fall, but not during other seasons. Seasonal density proportions of other feeding guilds, such as collector-filters or predators, did not differ among treatment or reference catchments.

Riparian Mollusks

We sampled the ground-dwelling riparian mollusk community at three of the clusters (table 2) to determine if buffer treatments affected the spring and fall densities of snails and slugs.

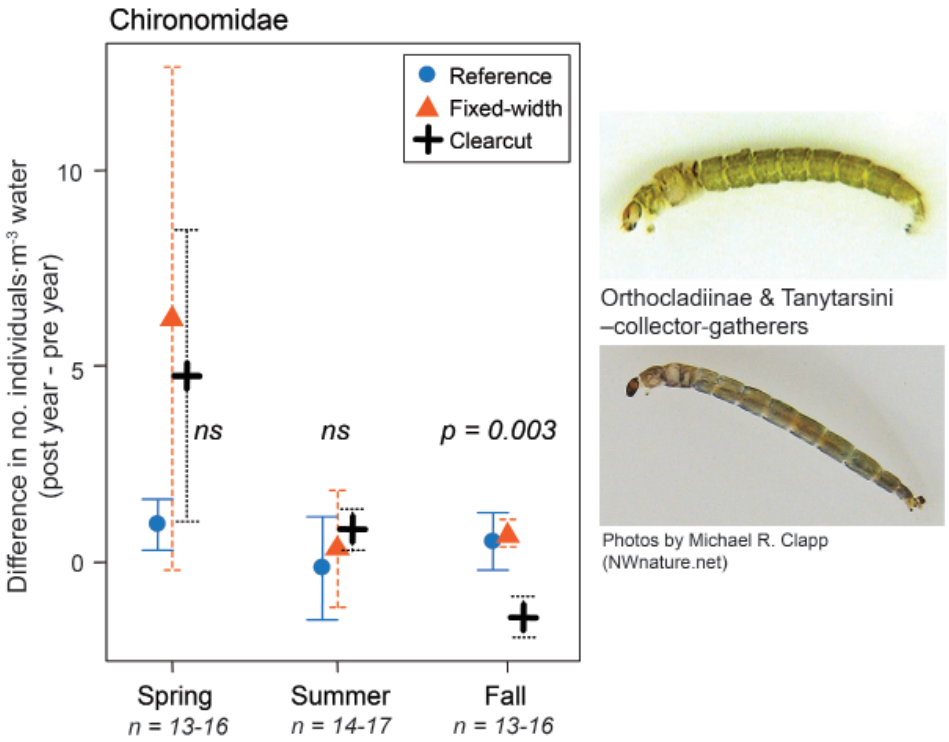

Figure 8—Changes in the mean drift densities of midge larvae, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

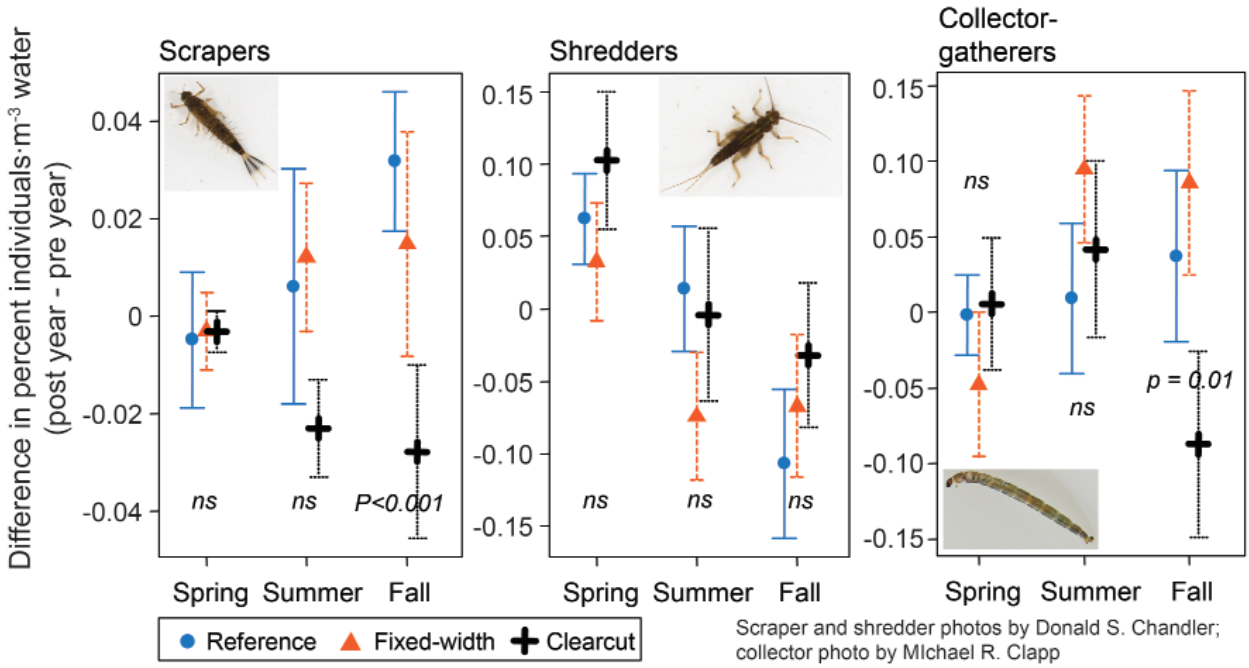

Figure 9—Changes in the mean drift proportions of scrapers, shredders, and collector-gatherers, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include at least one year pre-treatment and two years post-treatment. All photos reproduced with permission.

We focused on spring and fall seasons because mollusks burrow deep into the riparian forest litter in summer and some of the sites were inaccessible in winter. We used arrays of 30 x 30-cm artificial substrates made of laminated paperboard set out in evenly spaced patterns adjacent to the streams. After allowing several weeks for weathering and colonization, the laminations were peeled back and inspected under magnification for mollusks and other organisms. Total mollusk densities (all species) in three of the watershed clusters are presented in figure 10. Once again, because a limited number of catchments with patch buffers were sampled, only data for clearcut, fixed-width, and reference catchments are presented.

In general, mollusk densities between treatment and reference catchments did not change after logging (fig. 10). The reason mollusks were less abundant in catchments designated for no riparian buffers (i.e., clearcut) even before treatments were applied is not known, although we suspect that watershed features unrelated to logging (e.g., presence of seeps) played an important role in influencing them, and the

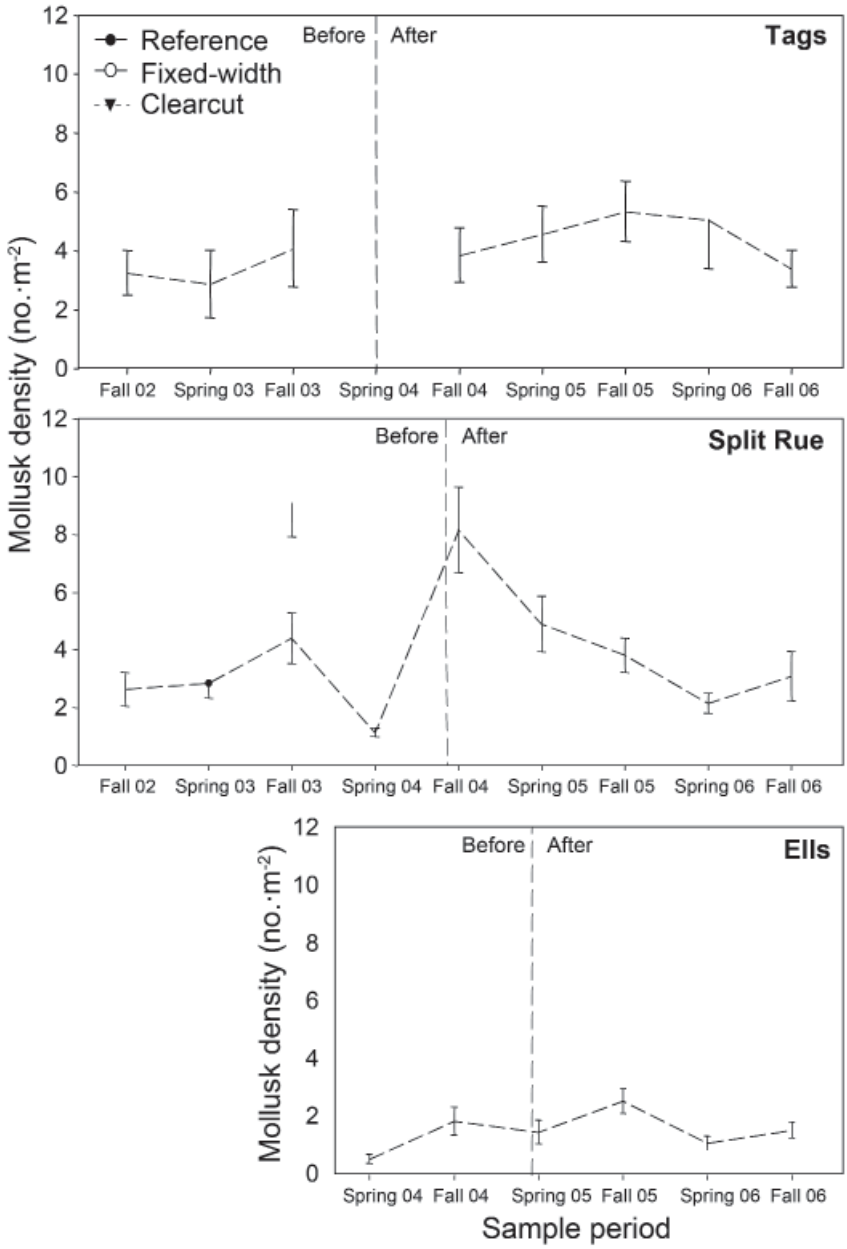

Figure 10—Mean total mollusk density (+ 1 standard error), by season, before and after treatment application (vertical dashed line) in three watershed clusters.

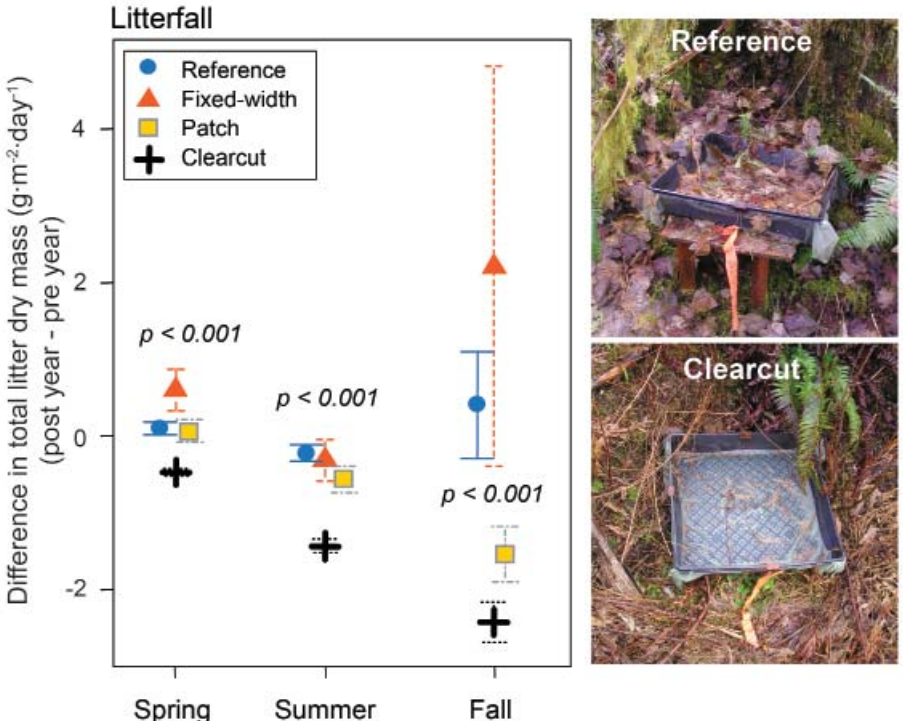

Figure 11—Changes in mean riparian litterfall, by season, after treatment application compared to input rates before treatment (± 1 standard error). (Photos by Shannon Claeson and Alex Foster.)

catchments selected for no buffer treatments happened to be less suitable for riparian mollusks by chance. As with aquatic invertebrates, the greatest divergence in mollusk abundance among treatment and reference locations occurred in fall.

Riparian Litter Inputs

Litterfall was sampled at six of the eight clusters, three in Capitol Forest and three in Willapa Hills, with an array of 40 x 40 x 10-cm traps spaced along the headwater channels. We placed litter traps 0 m, 10 m, and 20 m from the stream's edge on both sides of the channel. Monthly or bimonthly samples were collected from spring through fall (generally April through November). In the laboratory, samples were dried, sorted into various categories (coniferous litter, deciduous litter including seeds, wood, lichens, and other material), and weighed. All dry weights were divided by the sampling interval and converted to $g \cdot m^{-2} \cdot day^{-1}$. Deciduous litter inputs were greatest in fall (October–December), whereas coniferous litter inputs dominated in summer (July–September). The wood component of litterfall was variable throughout the year, and lichen and other components were generally a small fraction of total inputs.

Riparian litterfall in clearcut catchments significantly decreased to very low levels after treatment application (fig. 11), although inputs were beginning to recover in the second post-treatment year. Litterfall also declined significantly along patch-buffered catchments, but only in the fall (fig. 11). Fixed-width and patch-buffer litter inputs were highly variable post-logging, in part due to differences in windthrow among sites. No differences in litter amounts were observed at increasing lateral distances from the stream (0 m, 10 m, or 20 m).

Preliminary Ecological Conclusions

Preliminary findings suggest that biophysical changes in the headwater streams and riparian zones were detectable, but not large, after logging. Biological differences were most apparent in clearcut sites, especially in fall. An overall response to treatments was detectable,

although the signal was not strong. At no streams were dramatic changes observed in the biophysical response metrics we monitored after treatment. Temperature changes were relatively minor and, by themselves, did not likely have a major impact on the biota of the headwater stream communities. Changes in the composition of aquatic invertebrate communities were observed, but we did not see major reductions in macroinvertebrate diversity or feeding guild structure after logging. It was also apparent that changes were most discernible at catchments where no buffers were left along headwater streams.

Differences between continuous fixed-width buffers, patch buffers, and reference catchments were much more difficult to detect. Because of the variability in the data and the small number of replicates, it was unclear whether these buffers provided some form of protection or whether we simply did not have the statistical power to detect differences between buffered and reference streams. For aquatic invertebrates, riparian mollusks, and terrestrial litter, differences among treatments were most apparent in fall. Concern often focuses on conditions in mid-summer, when streams are warmest, but our research suggests that monitoring in fall may be a more sensitive time to look for management impacts.

Watershed-Scale Experiments: Lessons Learned

In addition to the scientific results, we learned much about conducting experiments involving operational forest harvest at the scale of whole catchments as well as designing and implementing large-scale management research. These lessons may be useful to others considering similar types of studies.

Watershed-scale manipulations are interventions, designed to maximize what we can learn from them. They are not classical experiments. Due to numerous implementation problems, we found it impossible to achieve a complete and balanced experimental design with all treatments applied to all watershed clusters. We believe that an ideal, balanced BACI statistical design with precisely replicated treatments and controls may be impractical to achieve in a broad landscape setting. This is not to imply that careful attention should not be given to setting up a study where management actions can be evaluated, but rather to accept that implementing such studies with normal operating constraints in disturbance-prone environments may well prove impossible.

One of the major differences between a classical experiment and watershed-scale ecological experiment is the interpretation of control sites. The classical definition of a control would be a replicate that is identical to the treated replicated except that it does not receive the treatment. In lab studies or greenhouse experiments, the ability to impose control is an essential part of statistical analysis. However, at watershed scales, neither identical replicates nor rigid environmental control are feasible. Researchers must adapt the strengths of designed experiments to the realities of reference reaches or reference streams.

Logistical constraints can prevent true random allocation of experimental treatments, and therefore "control" sites are different from treated sites before treatments are administered. Even though reference and treated units in the REMS study were in close proximity, there were differences between catchments that presented analytical challenges. Where control or reference streams do not follow a similar trend pre-treatment, the use of control sites in a BACI design may actually reduce statistical power (Roni et al. 2005). In the REMS study, natural differences between reference and treated sites made it difficult to detect results that could be directly attributed to management actions. Based on our experience, it seems prudent to select reference locations as close as possible and as similar as possible to experimental treatment areas, and even then to be prepared for unanticipated variation among locations.

Natural variability can make it difficult to detect responses to experimental treatments, especially with the small sample sizes that are common in large-scale interventions. We were surprised that background levels of variability were so high, even in the absence of management actions. Temporal variability makes detection of ecological change difficult across a wide range of ecological phenomena. Temporal variability decreases statistical power, making it difficult to detect significant treatment effects, and thereby necessitating either a larger sample size or a longer measurement window, or both. Roni et al. (2003), for example, re-analyzed data from four streams in coastal Oregon and concluded that it might take more than 70 years to detect a doubling of Coho Salmon (*Oncorhynchus kisutch*) smolt production in response to habitat restoration, even using a BACI design. Korman and Higgins (1997) found a similar result. Natural variability makes detecting effects of management very difficult. This poses a particular problem for watershed-scale studies that are limited with respect to sample size. Having a large sample size and randomizing treatments across study units can improve statistical power and reduce the potential for bias.

Had we performed statistical power analyses to determine the time or sample size needed to evaluate the null hypothesis that no differences were attributable to buffer treatments, we would likely have had to either greatly expand the number of study sites or extend both the pre- and post-treatment monitoring intervals to a decade or more, and the post-logging forest would be regenerating anyway. The importance of long-term monitoring to the evaluation of environmental change has been a cornerstone of the U.S. Long Term Ecological Research network (*http://www.lternet.edu/*). This program has undoubtedly contributed to our understanding of watershed management. However, in an environment where decision-makers do not have the luxury of multi-decade time horizons for evaluating policy choices, we were essentially constrained to carry out the study in less time than was necessary to properly implement a BACI study that could accommodate natural variation in headwater areas. For similar reasons, increasing the study's sample size was logistically and financially impossible. Natural variability is a general problem in designing ecological experiments. A common tendency among scientists is to more intensively monitor a small set of study sites for a longer and longer period of time in order to truly understand patterns at a particular site; this is exemplified by the Alsea Watershed Study in Oregon and the Carnation Creek Study in British Columbia. However, significant statistical gains can be made by simply increasing the sample size, allowing the larger number of observations to absorb the natural variability so that trends across space and time can be quantified (Liermann and Roni 2008).

We had difficulty determining whether observed changes in response variables were related to buffer treatments or to long-term changes that had nothing to do with the experimental design. The use of a BACI design with reference watersheds is intended to help control for temporal variation but, when all treatments are not administered in the same year, it is impossible to completely separate the year and treatment effect. Further, if the pre-treatment period does not include conditions experienced during the post-treatment phase, the assumption of stationarity is violated and the strength of the BACI design is weakened. This was the situation in the REMS study. Staircase designs (Walters et al. 1988), in which treatments are implemented over a series of years in stepwise fashion, are intended to separate the effects of environmental variation over time from treatment effects, and to control for time-treatment interactions. These designs are also more complicated and expensive to carry out at large spatial scales. The specific questions being asked (e.g., How do buffers influence the export of invertebrates to larger streams?) will determine the most appropriate study design, given natural variability.

Experiments over large spatial scales are often confounded by unanticipated events. During the study we encountered both natural and anthropogenic events that contributed to the difficulty of detecting treatment effects. The large wind storms that struck the Willapa Hills catchments in the first two post-treatment years caused so much windthrow in some of the fixed-width and patch-buffer sites (fig. 4) that these treatments departed significantly from the original experimental design. In addition, heavy sediment inputs that accompanied the storms influenced many response variables, adding to already high levels of background variation. The landslide and debris flow that occurred in one of the Capitol Forest catchments in 2007 essentially terminated the study at that site. Finally, we did not anticipate that some of the catchments would be treated with herbicide as part of the routine site-preparation process. None of these events were factored into the original study design; however, in retrospect we believe that such surprises are not uncommon in multi-year, watershed-scale studies and may need to be accepted as an inevitable part of the research.

Scientists and managers need to work together to design, conduct, and interpret, results of large-scale studies. Scientists and managers should work together to establish realistic expectations for what can be done to maximize learning opportunities where rigid control of factors other than the variables of interest cannot be achieved. For managers, this may mean forgoing some operational flexibility to maintain as much treatment consistency as possible across study locations. For scientists, this may mean having to make some concessions in the types of treatments and the location and timing at which they are applied. Both managers and scientists should also collaborate to establish realistic expectations of results relative to the questions being asked. For example, we were naïve to think that we could fully assess the issue of headwater buffer efficacy in a 5-year study. Post-treatment changes were still occurring when the study ended, and thus our work should be seen as an examination of the short-term responses of experimental catchments in southwestern Washington to different buffer configurations.

Implement the design and stick with it until you resolve the important question(s) or until the data show that actual uncontrolled variation is so different from what was assumed during the planning phase that the design is not adequate to resolve the question. It is essential not to alter treatments in the middle of a study, even if the temptation to do so arises. In our study there was interest in salvaging trees that were blown down in the post-treatment storms. We were able to convince the managers that the additional disturbances created by salvage logging operations would only make it more difficult to evaluate post-treatment monitoring results at the affected locations. It is also important to recognize that when experimental results are simply so variable that there is no hope of answering the original questions, there may be no compelling reason to continue the study. It may take several years (or one exceptionally large disturbance) to reach this conclusion, but there is little to be gained by continuing research that cannot lead to new insights, even if treatment consistency is maintained.

Do not be afraid to investigate novel response metrics. When surprises occur, be flexible enough to monitor their effects to maximize learning opportunities. It may become apparent after a study has been initiated that adding a new metric to the suite of response variables in the monitoring plan can yield important information. Even if the metric or method is relatively untested in the context of the study questions, the benefits of incorporating something novel with the potential to shed new light on ecological processes that control system response may outweigh the risks of ignoring it. The metric may not provide useful information, and after a reasonable trial period it can be dropped. The new metric might not help answer

the original questions, but instead may contribute information of different value. For example, we collected riparian millipedes along with mollusks because we found many millipedes in the artificial substrate samples. In the process, we captured several taxa that were previously unknown to science (Foster and Claeson 2011). Finally, surprise occurrences such as extreme weather events or unexpected management activities can add to the difficulty of evaluating experimental hypotheses, but they can also provide unique opportunities for learning in a setting where a monitoring program is already in place. For example, this study could have provided a useful design for addressing the question, "Are different buffer configurations prone to windthrow at different rates?" Taking advantage of these rare opportunities by investigating the effects of the event may add to the overall utility of the research.

Acknowledgments

Support for REMS was provided by the USDA Forest Service, Pacific Northwest Research Station. We express our sincere appreciation to the Washington Department of Natural Resources, especially to Richard Bigley and Jeffrey Ricklefs, for assistance in all aspects of the REMS study, and to William Ehinger, Jack Janisch and the Washington Department of Ecology for their contribution to the study of stream and air temperatures. We are also grateful to the University of Washington, Olympic National Forest, and the Washington Department of Fish and Wildlife for help with various aspects of the investigations. This paper benefited from helpful reviews provided by Sherri Johnson, John Colby, and an anonymous referee.

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