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[Note]

Effects of Road Decommissioning on Stream Habitat Characteristics in the South Fork Flathead River, Montana

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Abstract.—Previous studies have demonstrated the negative effects of roads on stream characteristics important for fish survival; however, few have examined whether decommissioning reverses these adverse impacts. We examined the relationships between the percentage of fine sediment in stream substrate and roads and looked at whether decommissioning had measurable effects on stream habitat in the Flathead National Forest, Montana. We conducted habitat surveys and substrate coring in 12 streams encompassing three watershed treatment types: (1) roadless areas, (2) areas with roads in use, and (3) areas with decommissioned roads. Significant positive correlations were found between the percentage of fine sediment in substrate and various measures of road impact (road density, roads in use, and number of stream crossings). Watersheds with roads in use had higher percentages of fine sediment than those without roads and those with decommissioned roads. Watersheds with high levels of vegetative regrowth on decommissioned roadbeds had a lower percentage of fines in stream sediment. Decommissioning efforts that enhance regrowth may improve stream habitat, although significant effects of these manipulations are difficult to detect through spatial comparisons. Future studies using either before–after or before–after–control designs to evaluate the effects of decommissioning practices on fish and wildlife habitat and populations are needed.

Roads primarily influence salmonid stream habitat by obstructing fish passage and degrading spawning, incubation, and juvenile rearing habitat (Furniss et al. 1991). Improperly designed culverts can impede or preclude fish passage and subsequently fragment aquatic habitat (Wofford et al. 2005). An excess of fine sediments resulting from soil erosion can degrade or completely destroy spawning habitat (e.g., Furniss et al. 1991). The successful incubation of salmonid embryos in stream gravels depends on intragravel water flow to provide oxygen and remove waste products (e.g., Bams 1969). Enhanced levels of fine sediment can reduce intragravel flow, impeding egg development as well as trapping and entombing emerging fry in the gravel (e.g., Phillips et al. 1975). Macroinvertebrate communities also respond negatively to fine sediments, thus influencing food availability for juvenile fish. In addition, excessive sediment delivery can decrease depth and number of pools thereby reducing the physical space available in the streams for rearing and overwintering of juvenile fish. If the riparian zone is compromised, then temperature, shade, and large wood would be altered, further affecting juvenile rearing habitat (Furniss et al. 1991). Although the effects of roads on fish habitat and production in any particular watershed are complex and a function of many interacting factors (Everest et al. 1987), their potential adverse effect on stream fish populations has prompted extensive restoration efforts.

In an attempt to mitigate the negative effects of forest roads the U.S. Forest Service is decommissioning about 3,200 km of roads each year and working to upgrade culverts and passage structures to facilitate fish migration (USFS 2002). Decommissioning roads can include a number of restoration strategies, ranging from blocking access to roads (with a berm or by bridge removal) to a complete removal of the roadbed and recontouring of the road prism to the original natural slope. However, in contrast with the wealth of information on the effects of existing roads (Forman and Deblinger 2000; Haskell 2000; Jones et al. 2000; Trombulak and Frissell 2000), almost no information exists on the effectiveness of road removal. Relatively few studies have documented that road decommissioning reduces road-related erosion (Kolka and Smidt 2001; Luce 1997; Madej 2001) and studies that have examined the effects of road decommissioning on wildlife are rarer still (Switalski et al. 2004). These studies have primarily examined decommissioning

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actions involving complete road removal and recontouring in highly erodible landscapes. Few published studies have focused on the effects of road decommissioning on fish habitat and species in Montana (but see Wegner 1999). Given the variation in geomorphology and related underlying erodibility of the landforms and soil across the country, as well as the variety of types of road decommissioning activities, the evaluation of various road decommissioning actions across multiple geographic areas is necessary for us to determine the effects of restoration efforts on fish and wildlife.

In Montana, bull trout *Salvelinus confluentus* and westslope cutthroat trout *Oncorhynchus clarkii lewisii* are two native coldwater salmonid species of conservation concern. Connectivity between high-quality stream spawning and foraging habitats is necessary for migratory life history forms of both bull trout and westslope cutthroat trout to complete their life cycles. Although many factors have contributed to the decline of these species, habitat degradation and fragmentation are two primary concerns in their conservation (Fraley and Shepard 1989; Lee et al. 1997; Liknes and Graham 1988; Rieman et al. 1997). Roads have been implicated in the degradation of bull trout habitat; forest roads are negatively correlated with bull trout stream use, abundance, and spawning activity (Baxter et al. 1999; Dunham and Rieman 1999). In addition, there is evidence of a significant inverse relationship between the percentage of fine sediment in the substrate and survival to the emergence of westslope cutthroat and bull trout embryos in incubation tests (Weaver and White 1985; Weaver and Fraley 1991).

On the Flathead National Forest, Montana, road decommissioning typically refers to blocking road entrances with earthen berms, which allows for natural revegetation and soil stabilization. In fish-bearing streams culverts are typically removed and stream banks recontoured. In some cases road entrances are gated. In addition to increased connectivity associated with culvert removal, road decommissioning is expected to reduce the delivery of sediment to streams, thus increasing the quality of spawning and rearing habitat for trout.

We evaluated streams within the Flathead National Forest that comprised three treatment types: (1) roadless watersheds; (2) watersheds with main roads still in public use but often with spurs that are decommissioned, gated, or both; and (3) watersheds with decommissioned roads (bermed and culverts removed). We did not consider roads that were only seasonally gated as decommissioned. We addressed two questions. First, is there a relationship between substrate composition (percentage of fine sediment) and road density? Second, if so, does road decommissioning have measurable effects on the percentage of fine sediment and other stream habitat characteristics that are important to fish?

Study Area

Research was conducted in the Flathead National Forest in northwestern Montana. As part of the Northern Continental Divide Ecosystem, the Flathead National Forest's 2.3 million acres is considered to be 47% wilderness and is one of the few forests left in the contiguous U.S. where a full complement of native trout species remains. Streams were sampled in the southern half of the Forest in the South Fork Flathead River Basin (Figure 1). This basin is bounded to the east by the Bob Marshall Wilderness Complex and to the west by the Swan Mountains and primarily comprises sedimentary rock with dense stands of coniferous forest that exhibit historical clear-cut logging and associated roads.

Of the 6,100 km of roads on the Flathead National Forest, 544 km have been decommissioned and an additional 612 km are slated for decommissioning (U.S. Forest Service, Flathead National Forest, unpublished database for road decommissioning projects). The effects that these roads have on aquatic habitat vary and are based on (1) watershed characteristics and geomorphology (e.g., slope and soil type, land use, road density and use); (2) proximity of the road to the stream (riparian buffers, number of road crossings); and (3) stream characteristics (the power of the stream to move or flush sediment from system [e.g. Duncan and Ward 1985; Luce et al. 2001]). Similarly, influences of road decommissioning will vary depending on the location and quantity of the decommissioned roads in the watershed as well as how and when they were decommissioned.

We chose 12 study streams with fairly similar watershed and stream size and gradient characteristics, which controlled for differences in stream power and watershed size while exhibiting differences in watershed road treatments (Table 1). Twin Creek had a significantly larger watershed area, but had characteristics similar to our other streams. Study watersheds generally had roads that paralleled the stream with a riparian buffer greater than 20 m. To minimize confounding effects, watersheds of study streams did not have recent (within 5 years) wildfires or timber sales within the watershed. Three watersheds had roads-in-use (Wheeler, Emery, and Quintonkon creeks), while three watersheds had entirely roadless watersheds (Riverside, Tin, and Twin creeks) to provide reference stream conditions. Of the six watersheds containing decommissioned roads, two streams had all roads within their watershed decommissioned (Slide Creek and Connor Creek) and the

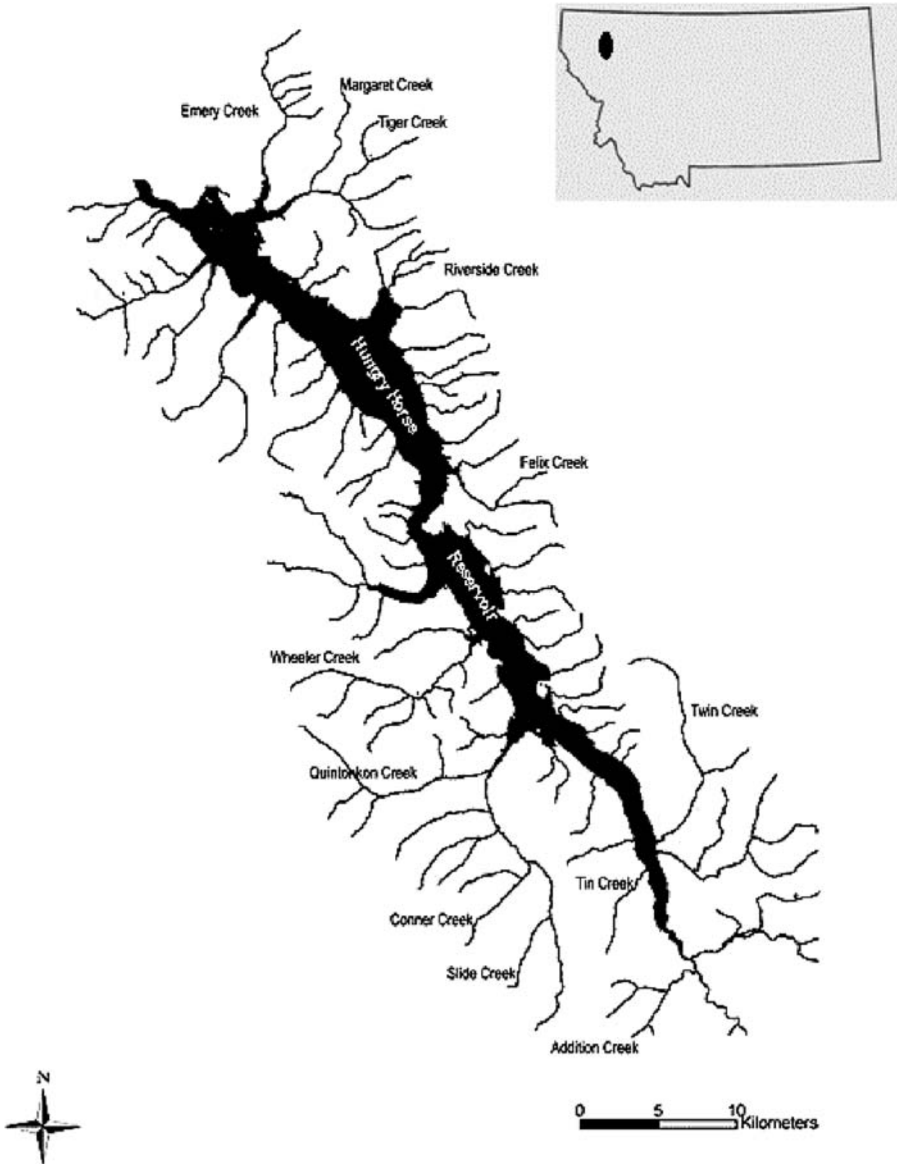

FIGURE 1.—Study area located in the Flathead National Forest (Hungry Horse and Spotted Bear Ranger Districts) in northwestern Montana. All of the study streams were tributaries to the South Fork of the Flathead River at or near Hungry Horse Reservoir.

remaining four watersheds contained a mix of road treatments (bermed with regrowth, as well as gated roads in administrative use; Table 1).

Methods

Basic road surveys were conducted to verify our categorization of roads within the different treatments. Watershed treatments were characterized as roadless, roads in use, or decommissioned. If they were decommissioned we qualitatively assessed the level of vegetative regrowth (sparse grass, dense grass, shrubs, or trees), and noted any signs of road activity (e.g., motorized vehicle tracks) for the primary road adjacent to the stream. We separated our decommissioned road treatments based on the level of revegetation of the primary road in the watershed. Conner, Slide, and Tiger creeks all had grass, bushes, and trees on the primary road and were classified as high

TABLE 1.—Stream and watershed habitat characteristics for 12 watersheds studied in the South Fork Flathead River basin.

Stream	Level of regrowth on decommissioned roads	Elevation (m)	Incline (%)	Total road density (km/km^2)	Road-in-use density (km/km^2)	Number of road crossings	Distance from Hungry Horse, Montana (km)	Watershed area (km^2)
Wheeler	Open road	1,364	3	0.61	0.24[a]	16	63	46.26
Emery	Open road	1,202	2	1.58	0.28	30	14	38.54
Quintonkon	Open road	1,223	2	0.69	0.25	15	70	55.45
Addition	Low	1,323	2	0.22	0.01[b]	4	93	38.87
Felix	High	1,139	2	1.51	0.00	4	46	19.91
Tiger	Low	1,234	2	0.13	0.00	0	21	17.46
Margaret	Low	1,295	5	0.67	0.00	3	19	10.42
Slide	High	1,347	2	0.33	0.00	0	84	12.90
Conner	High	1,299	2	0.53	0.07[c]	3	80	17.60
Riverside	Roadless	1,163	3	0.00	0.00	0	31	14.79
Tin	Roadless	1,184	2	0.02[d]	0.00	0	84	16.88
Twin	Roadless	1,113	2	0.00	0.00	0	79	116.92

[a] Seasonal closures.
[b] Gated during this study (summer 2004); decommissioned at the end of summer 2004.
[c] Bridge out at Sullivan Creek so no road in use in Conner watershed.
[d] Main road at base of watershed downstream from sampling.

regrowth watersheds. Other decommissioned watersheds (Addition, Margaret, and Felix) exhibiting only sparse grass were grouped together into a low regrowth category.

To quantify the influence that roads have on a watershed, we calculated a series of road-related variables for each of our watersheds. Both total road density (for all classes of roads) and the density of roads currently in use within each study watershed were calculated from U.S. Forest Service geographical information systems (GIS) road layers using the distance tool in ArcMap at a resolution of 1:24,000. Similarly, distance along roads between the stream access point to the closest town (Hungry Horse, Montana) was established as a surrogate measure of accessibility and hence traffic volume (Table 1). In addition, we noted any road crossings (e.g., bridge, culvert) in the field and used a Flathead National Forest map (scale: 1:126,720) to estimate the number of roads crossing streams within each watershed (Table 1).

Approximately 300 m of each stream was sampled both in the summer (June–July) and fall (September–November) of 2004, except at Wheeler Creek, which was only sampled in the fall. The lower half of each stream was divided into contiguous 100-m sections and three of these sections were randomly selected for sampling.

During summer sampling we performed habitat surveys (Overton et al. 1997). Proceeding upstream, we described each channel habitat unit (riffle, run, or pool) and measured its length (m), mean wetted width (m), middle depths of riffles (cm) and maximum depth of pools (cm). In each section we measured average incline and elevation, visually estimated bank stability, quantified large wood, and estimated canopy cover. Channel incline was assessed with a compass, elevation was obtained using a Garmin global positioning system (GPS), and canopy cover was estimated with a Moosehorn densitometer (80 readings along eight different cross-sectional transects per section). The stability of stream banks was rated on a scale from 1 to 4 as follows: 1 = banks were stable, less than 5% of the bank having signs of erosion or bank failure absent or minimal; 2 = banks were moderately stable, with infrequent, small areas (5–30% of bank in reach) indicative of erosion; 3 = banks were moderately unstable, 30–60% of the bank showing signs of erosion resulting in high erosion potential during floods; and 4 = banks were unstable, eroded areas being seen frequently along straight sections and bends (60–100% having erosional banks). We deployed a temperature logger (ibuttons, Maxim Dallas semiconductor) in each section to collect water temperature data every 90 min.

During fall sampling we measured pool habitat characteristics and performed substrate coring in study sections. Pool frequency and depth were analyzed for differences among the three treatment types using a Kruskal–Wallis test (Zar 1999). Ten substrate cores (McNeil core samples) per stream were collected in the lower two sections (five per section) to measure substrate composition and fine sediments. All cores were collected in flowing water, over cobble substrate, and at a pool–riffle break (as suggested by OPSW 1999). Cores were taken to a depth of 15 cm into the substrate. Oven-dried core samples were weighed after being shaken through sieves with mesh sizes of 75, 50, 25, 19, 12.5, 9.5, 6.3, 4.75, and 2.0 mm and 850, 425, and 63 μm. An average of 3.8 kg ± 0.058 (mean ± SE) of substrate was collected per core. Since fine

TABLE 2.—Characteristics of streams surveyed in the South Fork Flathead River basin (LWD = large woody debris, ND = no data).

Stream	Temperature (°C)	LWD/100 m Singles	LWD/100 m Aggregates	Bank stability[a]	Riffle area (%)	Average width (m)	Midstream depth (cm)	Pool frequency (per 100 m)	Maximum pool depth (cm)	Sediment <6.3 mm (%)
Wheeler	ND	6.8	1.9	1.0	85.9	8.9	35.3	1.2	89.5	23.88
Emery	7	23.1	8.0	1.0	89.1	4.4	29.7	4.7	67.3	34.22
Quintonkon	7	5.7	1.0	1.1	97.6	7.7	32.7	1.2	89.5	20.42
Addition	ND	16.8	4.2	1.1	83.0	6.7	52.7	6.1	79.7	22.02
Felix	9	11.4	3.0	1.6	97.9	5.7	19.3	3.0	71.7	20.64
Tiger	9	6.6	2.0	1.0	97.7	4.6	23.0	5.5	58.3	18.49
Margaret	8	15.7	4.0	1.0	97.9	3.8	22.3	6.1	59.3	20.78
Slide	8	12.1	3.9	1.3	94.9	5.0	27.7	3.0	77.7	14.81
Conner	10	3.0	1.6	1.1	96.2	5.4	29.0	3.7	54.3	14.45
Riverside	ND	8.9	5.0	1.0	58.1	4.9	38.8	7.4	66.3	21.36
Tin	8	7.8	1.5	1.1	97.3	4.8	33.7	4.0	61.7	13.36
Twin	10	0.0	0.0	1.1	83.2	9.3	31.7	3.0	77.7	18.53

[a] Scale = 1 to 4; see text for details.

material disturbed by coring typically remains in suspension within the corer and is often not included in the captured substrate, we improved our estimate of the fine particle component by agitating the remaining sediment within the core and extracting three 150-mL subsamples of water to measure suspended sediment. The height of the water within the corer was measured and converted to a volume based on a depth-to-volume curve produced in the laboratory specifically for this corer. These subsamples were returned to the laboratory where the sediment was settled and measured in Imhoff cones. The volume of fine sediment was then multiplied by the volume of water in the corer to determine the total fines. These wet volumes of fine sediment were then converted to a dry weight using a conversion factor developed by Shepard and Graham (1982) allowing us to add these measures of fine sediment to the fraction of our substrate composition data that was less than 63 μm in size.

Substrate composition was expressed as the percentage of substrate particles (SP) smaller than 6.3 mm. This is the size fractionation used to assess spawning habitat for both bull trout and westslope cutthroat trout (e.g., Shepard and Graham 1982; Weaver and Fraley 1991; 1993). We compared percent of substrate particles less than 6.3 mm among treatment types with a Kruskal–Wallis nonparametric analysis of variance (ANOVA) (Zar 1999). We examined correlations in our dataset associated with roads and watersheds. After eliminating highly correlated variables a step-wise multiple linear regression was used to evaluate which watershed characteristics (e.g., road crossings, watershed area) best predicted the percent of fine substrate. Finally, watersheds with decommissioned roads were analyzed using a stepwise multiple regression analysis to examine whether level of regrowth, number of road crossings, or watershed area explained a significant amount of variation in the percentage of substrate composed of fine sediment (Zar 1999).

Results

Stream Habitat Characteristics

Overall, stream sections were of similar size and gradient, and the habitat was dominated by riffles. Bank stability was high in all streams (Table 2).

Substrate Composition and Road Impacts

To investigate how roads affect the substrate composition of our study streams, we examined whether there was a correlation between the percentage of fine sediment (%SP $<$ 6.3 mm) in the substrate and total road density, density of roads in use, distance from the closest town (Hungry Horse), and the number of road–stream crossings within the watershed. There was a significant positive relationship between percentage of fine sediment and road density (Pearson correlation = 0.36; $P = 0.038$, $n = 12$), roads in use (Pearson correlation = 0.43; $P = 0.021$, $n = 12$), and the number of road–stream crossings (Pearson correlation = 0.84; $P = 0.001$, $n = 12$, Figure 2). Furthermore, there was a negative trend but nonsignificant correlation (Pearson correlation = −0.573; $P = 0.051$, $n = 12$) between percentage of fine sediment in the substrate and potential road use (i.e., distance from Hungry Horse). These measures of watershed road characteristics were all significantly correlated with each other making it impossible to separate their potential effects. Given the extent of the riparian buffers (typically $>$20 m) in these watersheds, personal observations of erosion at road crossings during the road surveys and the high correlation of road–stream crossings with percentage of fine sediment, we used number of road–stream crossings for our analyses to detect potential effects of decommissioning roads.

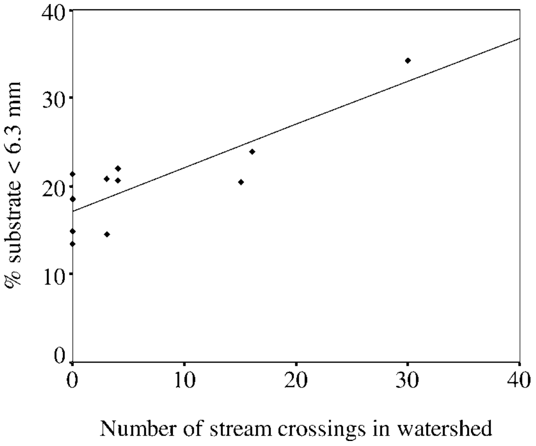

FIGURE 2.—Relationship between the percentage of substrate particles less than 6.3 mm in size and the number of stream crossings in the watershed ($r^2 = 0.70$, $P = 0.001$). Strong positive correlations were also found among total road density, the density of roads in use, and stream crossings.

Watershed Treatment Comparisons

There were no statistically significant differences in the number of pools per 100 m ($P = 0.981$) or maximum pool depth among our three treatment groups ($P = 0.207$; Table 2).

Watersheds with roads in use had the highest median percentage of substrate particles of less than 6.3 mm in stream cores and decommissioned and roadless watersheds exhibited similar percentages of fine sediment in stream cores, although the differences were not statistically significant (Kruskal–Wallis test: $\chi^2 = 3.15$, df $= 2$, $P = 0.20$; Figure 3a). Watersheds with decommissioned roads were tested for differences in percentage of fine sediment versus number of road crossings, watershed size, and amount of regrowth on roads. Percentage of fine sediment in stream substrate within the decommissioned watersheds was not significantly related to number of road crossings or watershed size. However, there was a significant effect of the level of regrowth with fine sediment (regrowth = 0.019, number of road crossings $P = 0.385$, area $P = 0.852$, final regression $F = 14.67$, $P = 0.02$, df $= 5$), whereby decommissioned roads with high levels of regrowth appeared to have a lower percentage of fine sediment in the stream substrate (Mann–Whitney test: $Z = -1.96$, $P = 0.05$; Figure 3b).

Discussion

Road building leads to increased sedimentation and a reduction in fish habitat quality (Gucinski et al. 2001; for reviews, see Meehan 1991; Trombulak and Frissell 2000) and areas without roads are often strongholds for native fish communities (Lee et al. 1997; Baxter et al. 1999). The percentage of substrate particles less than a given size for a specific species or guild is often considered the best indicator of fish habitat degradation from roads (Young et al. 1991). In this study, watersheds that had higher total road density, roads in use, and road–stream crossings exhibited higher percentages of fine sediment compared with those watersheds that had lower levels of road influence (Figures 2, 3a). These general trends tentatively support expectations about the relationship among roads, substrate composition, and potential for spawning success.

Other studies have found that as traffic increases, there are concomitant increases in sediment yields from roads (e.g., Reid and Dunne 1984). Using the distance along roads of each creek from the town of Hungry Horse as a surrogate measure of road use by vehicles, we found no significant relationship between increased percentage of fine sediment and increased potential traffic. Upon examination of our watersheds, we found that Emery Creek, which probably has the heaviest traffic volume given its relative accessibility (Table 1), exhibits the highest percent composition of fine sediment.

Having established a relationship between road density and crossings and fine sediment composition in streams (Figure 2), we examined whether road decommissioning is correlated with a lower percentage of fine sediment in stream substrate. We did not see significant differences among our three treatment groups. Watersheds with roads in use had higher levels of fine sediment in the substrate than either those without roads or those with decommissioned roads, but the high amount of fine sediment in Emery Creek had a large influence on these trends (Table 2).

Our lack of statistically significant results among treatments may stem from the combination of confounding factors and low power. For instance, in May 2004 (several weeks before sampling), our study area experienced a 14-year peak flood event with discharges approximately 50% higher than mean annual peak flow levels. This may have influenced our streams by flushing fine sediment from our study sites, thereby affecting our ability to detect differences in chronic sediment loading in these watersheds (U.S. Geological Survey, gauging station 12359800, South Fork Flathead River at Twin Creek near Hungry Horse).

There was a significant difference in the percentage of fine sediment in the substrate of streams with different levels of regrowth on decommissioned roads. Streams associated with watersheds containing a high amount of regrowth, whereby a mixture of trees, shrubs, and grasses had established themselves on the

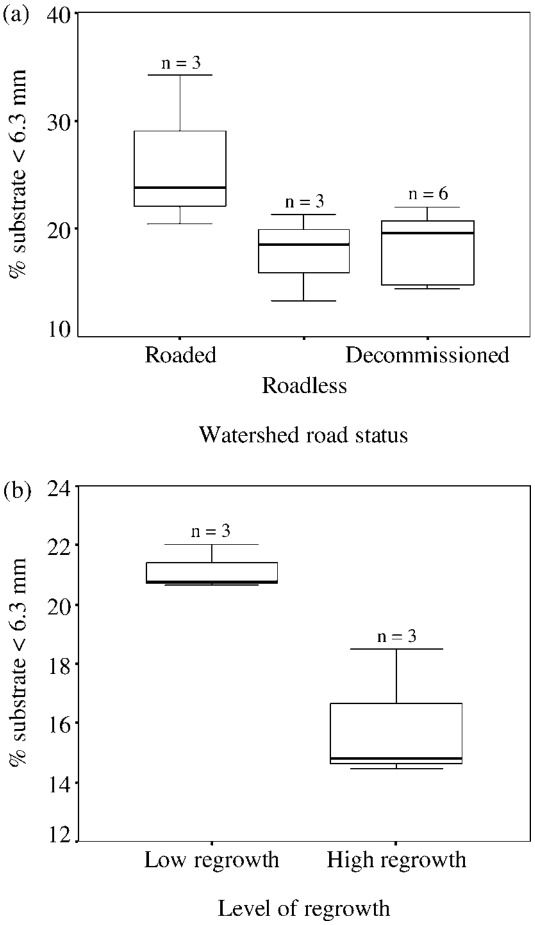

FIGURE 3.—Box plots showing **(a)** the percentage of fine substrate for each road treatment type ($\chi^2 = 3.15$, df = 2, $P = 0.20$) and **(b)** the percent of fine substrate with respect to the level of roadbed regrowth within decommissioned road treatments ($Z = -1.96$, $P = 0.05$). High regrowth refers to old roadbeds with a mixture of trees, shrubs, and grasses, while low regrowth refers to sparse grass. The heavy lines within the boxes represent the median values, the lower and upper boundaries of the boxes the 25th and 75th percentiles, and the whiskers the 10th and 90th percentiles.

old road, had lower percentages of fine sediment in the substrate than did those watersheds with only sparse grass (Figure 3b). Thus, as decommissioned roads become increasingly revegetated over time, the amount of fine sediment loading is reduced to the levels that existed before the roads were built.

The few studies in which road decommissioning has been shown to have large beneficial effects for fish habitat were conducted in areas with more erosive soils or higher susceptibility to mass wasting. The soils in the Flathead National Forest are not as erosive as some granitic soils where many of the most obvious road sedimentation problems exist (e.g., Clearwater National Forest, Megahan and Kidd 1972). However, large flood events and culvert blow-outs are not uncommon in this forest.

Our results suggest that road decommissioning that results in vegetative regrowth reduces fine sediment in streams, thereby conferring positive effects on stream habitat for bull and cutthroat trout in the Flathead National Forest. Other studies have demonstrated how upgrading passage barriers (e.g., perched culverts) can result in recolonization by juvenile bull trout (USFWS 2002). Bull trout populations also increased following full recontouring of the streams and culvert removals on the nearby Kootenai National Forest (Wegner 1999); a 48% decline in fine sediment and a 16% increase in bull trout redds was observed in the 5 years following decommissioning. Our study has attempted to elucidate differences in substrate composition associated with different road treatment types and levels of regrowth associated with decommissioning actions. Based on our results, we suggest that decommissioning roads that lead to high levels of revegetation probably reduces the amount of fine sediment in streams.

While road decommissioning appears to be an effective tool with which to mitigate many of the negative effects of roads on fish habitat, care must be taken when designing studies to demonstrate its effects. Given our estimated variance, detecting statistical significance would require large-effect sizes (~30–40% change in percentage of fine substrate), as well as large sample sizes ($n \geq 25$ streams). With the large amount of spatial variation in sedimentation that we observed among watersheds, even after controlling for watershed and stream characteristics, we recommend study designs in which streams serve as their own controls, that is, either replicated before–after or replicated before–after–control impact designs, to evaluate the effects of road decommissioning (Roni et al. 2005).

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