

Ecological Impacts of Three Lower Kenai Peninsula, Alaska, ATV Stream Fords

by

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Contents

Tables and Figures *iv*

Introduction *1*

Methods *3*

 Study Sites *3*

 Study Approach *4*

 Sediment Deposition – Sediment Collectors *4*

 Suspended Sediment – Siphon Samplers *5*

 Substrate Composition – Substrate Photography

 Biological Response – Benthic Macroinvertebrates

Results *7*

 Sediment Deposition – Sediment Collectors *7*

 Suspended Sediment – Siphon Samplers *7*

 Substrate Composition – Substrate Photography *8*

 Biological Response – Benthic Macroinvertebrates *11*

Discussion *15*

Literature Cited *17*

Tables and Figures

Tables

1. Sediment collector installation dates, removal dates, and number installed 15
2. Total suspended solids and total suspended inorganic sediment from each siphon sampler
3. Physicochemical measurements taken simultaneously with macroinvertebrate sampling
4. Mean biomass of common macroinvertebrate taxa and results for ANOVA comparing biomass of common taxa among positions

Figures

1. Location of three ATV stream ford study sites within the Anchor River basin 2
2. Aerial views of Site 1 ATV ford 5
3. Site 1 ATV ford approaches facing south and north 8
4. Aerial views of Site 2 ATV ford 9
5. Site 2 ATV ford facing south and facing north 10
6. Aerial views of Site 3 ATV ford 10
7. Site 3 ATV ford 12
8. Sediment collector array and individual sediment collector 12
9. Schematic of siphon sampler 13
10. Mean mass of sediment deposited within sediment collectors at Site 1
11. Mean mass of sediment deposited within sediment collectors at Site 2
12. Mean mass of sediment deposited within sediment collectors at Site 3
13. Total suspended inorganic solids from each siphon sampler at Site 1
14. Total suspended inorganic solids from each siphon sampler at Site 2
15. Total suspended inorganic solids from each siphon sampler at Site 3
16. Percent coverage of fine sediment measured by streambed photography
17. Site 1 macroinvertebrate metric values from summer and fall samples
18. Site 2 macroinvertebrate metric values from summer and fall samples
19. Site 3 macroinvertebrate metric values from summer and fall samples
20. Macroinvertebrate populations showing significantly different biomass among positions.

Introduction

Inorganic sedimentation is a natural process in streams, and inorganic sediment (hereafter referred to as sediment) at naturally occurring levels is necessary for stream function. However, some land use practices can introduce large quantities of sediment to streams. The U.S. Environmental Protection Agency (2000) recently cited sediment as the primary source of water quality impairment in the United States. Major documented sediment and turbidity sources within Alaska include urban and suburban development, road construction, timber harvest with associated roads, and mining.

Biological impacts from sedimentation are pervasive and occur at every trophic level within the stream ecosystem. Increased turbidity limits light penetration, which can greatly decrease the primary productivity of benthic algae, the base of the stream food web (Lloyd, Koenings, and LaPerriere 1987). Sediment can further reduce algal stocks by scouring and smothering (Van Nieuwenhuysen and LaPerriere 1986). Sedimentation can limit macroinvertebrate abundance through a reduction in algal food resources, mechanical scouring, and smothering when fine particles fill interstitial spaces in the streambed (Rosenberg and Wiens 1978), potentially leading to food limitation among fish populations (Bruton 1985). Incubating salmonid eggs and alevin rely on gravel permeability for oxygen delivery and the flushing of metabolic waste. Gravel permeability and, in turn, egg-to-fry survival can be reduced markedly by modest increases in streambed sediment (Everest et al. 1987). Exposure to suspended sediment has been directly linked to increased mortality in many fishes, including Arctic grayling (*Thymallus arcticus*), rainbow trout *Oncorhynchus mykiss*, and coho (*O. kisutch*), chinook (*O. tshawytscha*), and chum (*O. keta*) salmon (Newcombe and Jensen 1996).

All-terrain vehicle (ATV) stream fords (i.e., shallow water crossings) are very common on Alaska's landscape, occurring haphazardly wherever backcountry routes meet streams. It is possible that fords cause significant sedimentation in some areas of Alaska, but little or no research has addressed this. The few studies published to date have found increased sedimentation in association with vehicle fords (e.g. Brown 1994; Hinckley, Iverson, and Hallett 1984), but these were conducted in arid environments with precipitation, geology, and vegetation so different from those found in Alaska that extrapolation is not meaningful.

Although anthropogenic sedimentation can be detrimental to stream ecosystems and fish production, direct measurement of the existence and severity of sediment inputs is problematic. Bedload sediment, suspended sediment, and turbidity can fluctuate dramatically both temporally and spatially, making detection by direct measurement difficult and inconsistent. Electronic automated sampling devices can collect water continuously at short intervals, but the expense and maintenance of such units is prohibitive to most budgets. Furthermore, direct sediment measurements give no information about the impact of sediment to the stream ecosystem.

Macroinvertebrate biomonitoring is an accepted tool for the monitoring and detection of disturbances in aquatic systems, and most states (including Alaska) use biomonitoring protocols in their water quality programs. Macroinvertebrate communities can sustain long-lasting alteration in response to pollutants, which makes possible the detection of ephemeral impacts often missed by direct measurement. Most biomonitoring protocols are designed to detect the cumulative effects of multiple impacts (e.g. the Alaska Stream Condition Index; Major et al. 2001). Other protocols are designed to detect specific impacts and may offer increased sensitivity in situations where a particular pollutant is suspected to be impairing stream quality. Recent biomonitoring advances include the development of sediment-specific biomonitoring protocols (Relyea, Minshall, and Danehy 2000; Zweig and Rabeni 2001), which may prove useful for the detection of anthropogenic sedimentation in Alaska streams. This could provide agencies, tribes, and nonprofit organizations throughout Alaska with a tool for documenting stream sediment impacts and guiding restoration and recovery efforts.

This study by the University of Alaska Anchorage's Environment and Natural Resources Institute (ENRI) was funded by the Alaska Department of Environmental Conservation. It was conducted during the summer and fall of 2002 within the Anchor River watershed of the lower Kenai Peninsula. The goals of the project included: (1) provide a first look at the extent to which ATV stream fords influence sedimentation in streams, (2) determine the distance of sediment transport downstream of ATV fords, (3) evaluate the efficacy of a biotic index designed specifically for the detection of sedimentation, (4) determine the impact of ATV stream fords on the abundance and biomass of macroinvertebrates (i.e., salmonid food sources), and (5) provide information regarding results to the professional community and the public.

Methods

Study Sites

The Anchor River has been listed as a high-risk water body with high data collection priority by the Alaska Department of Environmental Conservation (303d list). This system hosts runs of chinook, coho, and pink (*O. gorbuscha*) salmon, which drive locally important commercial and sport salmon fisheries. Resident and anadromous rainbow trout and Dolly Varden char (*Salvelinus malma*) also contribute to the sport fishery. Spruce bark beetles have caused extensive white spruce (*Picea glauca*) and Lutz spruce (*Picea X lutzii*) mortality in the area. A consensus reached at a January 2002 interagency meeting regarding lower Kenai Peninsula streams was that sedimentation is the primary factor limiting water quality and salmon production. Also noted was that new logging roads have created access to seismic trails and backcountry areas, ATV use has increased dramatically in recent years, and ATV stream fords were a likely source of sedimentation.

This study was conducted at three ATV stream fords on regularly traveled trails in separate basins of the upper Anchor River watershed (Figure 1). These fords represented the upstream-most human disturbance within each of these basins, which eliminated any potentially confounding influence of upstream sediment sources. Although traffic volume was not successfully measured, ATVs were regularly observed using all three fords and all fords appeared to be used with similar frequency. All three streams were of moderate gradient (~2%) with predominantly gravel/cobble substrate. These sites offered contrasting soils, vegetation, and topography in the vicinity of the ford.

Site 1 was on a small, 2nd order, unnamed headwater tributary to the Anchor River (59°48.925' N, 151°11.655' W; Figure 2). The stream valley was relatively narrow and the trail sloped relatively steeply (10% grade) into the stream for a distance of 30 m on the south and 50 m on the north (Figure 3). The soil in the vicinity of the ford and on both approaches consisted mainly of fine, unconsolidated sand and silt with some gravel and cobbles. Considerable downcutting was apparent on both approaches, indicating that sediment was actively being eroded into the stream channel. Channel morphology was altered considerably at the ford, the channel being more than three times wider and much shallower in the ford than in the surrounding stream reach. Stream substrate within the ford and for 5 m downstream consisted entirely of fine sediment, rather than the gravel/cobble substrate that dominated the surrounding streambed. Local woody vegetation consisted mainly of Barclay willow (*Salix barclayi*) and white spruce, many of which have succumbed to spruce bark beetle infestation.

Site 2 was on Beaver Creek, a 3rd order tributary to the Anchor River (59°45.086' N, 151°28.784' W; Figure 4). To the north of the ford the trail was gently sloped for the first 200 m, beyond which the grade increased to approximately 10%. To the south the trail approached the ford at an average of 7% for approximately 100 m, beyond which point the trail leveled

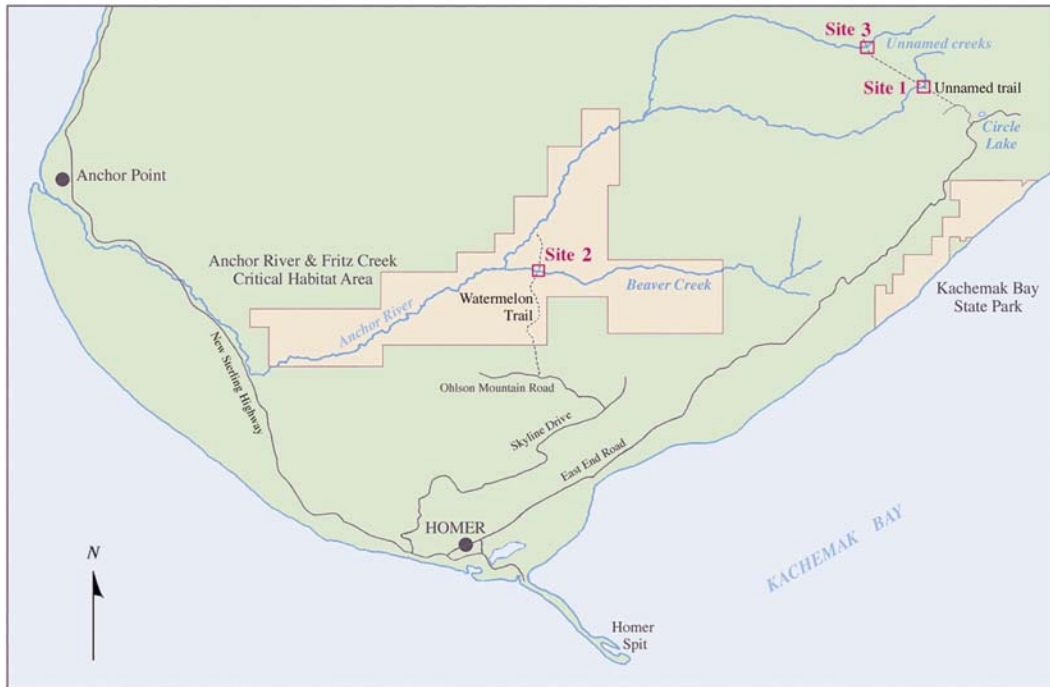


Figure 1. Location of three ATV stream ford study sites within the Anchor River basin.

(Figure 5). The trail surface consisted of sand and gravel, except within the immediate vicinity of the ford where it was predominantly gravel and cobble. Likewise, the stream substrate within the ford consisted entirely of gravel and cobble. Very little downcutting was evident on either trail approach, but sediment-laden surface water was observed flowing into Beaver Creek from the north during periods of rain (Figure 5). Within the ford, the stream channel was more than three times wider and slightly shallower than in the surrounding stream reach. Local woody vegetation consisted of Barclay willow and widely scattered white spruce.

Site 3 was on an unnamed 2nd order Anchor River tributary (59°49.932' N, 151°13.919' W; Figure 6). The stream valley was broad and gently sloped. Adjacent to the stream channel, vegetation consistent with discharge slope wetland was present (Gracz et al. 2003), dominated by Barclay willow. The trail approached the ford on a grade averaging 4% for 100 m to the north and 200 m to the south. Both approaches to the ford consisted of braided, deeply rutted trails with exposed, saturated organic soil firmly bound by roots (Figure 7). Relative to the surrounding stream channel, the channel was more than three times wider and much shallower within the ford; channel substrate within the ford consisted of cobble and gravel.



Figure 2. Aerial views of Site 1 ATV ford. In upper photo, stream flows from upper left to lower right. In lower photo, stream flows from top to bottom. (Photos courtesy of Michael Weidmer, Alaska Department of Fish & Game [ADF&G].)



Figure 3. Site 1 ATV ford approaches facing south (left) and north (right).

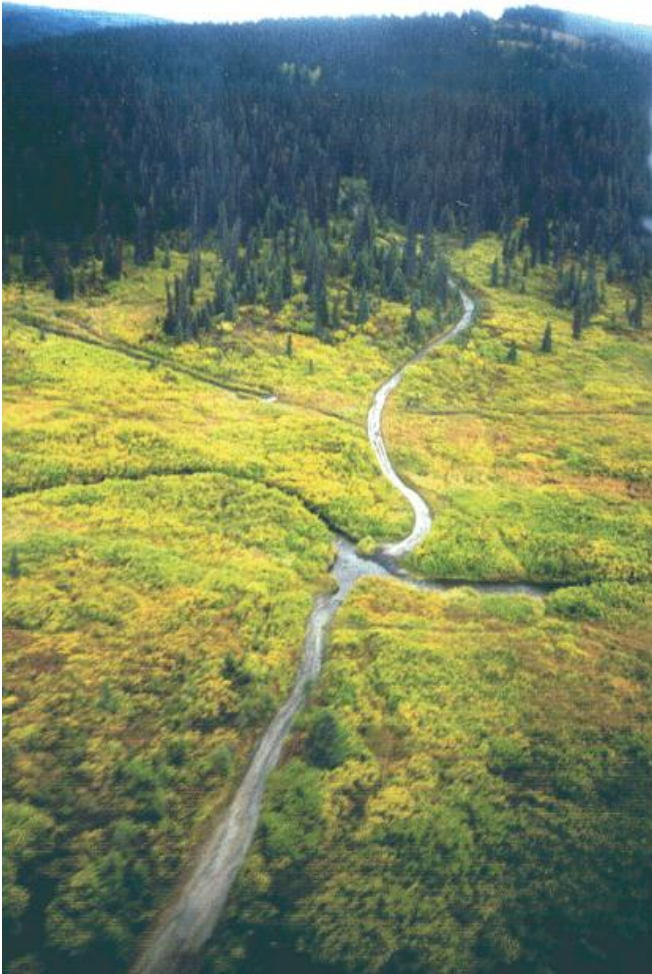


Figure 4. Aerial views of Site 2 ATV ford (Beaver Creek at Watermelon Trail). In left photo, stream flows from left to right. In right photo, stream flows from right to left. (Photos courtesy of Michael Weidmer, ADF&G.)



Figure 5. Site 2 ATV ford facing south (left photo) and facing north (right photo). Note sediment-laden surface flow on trail in right photo.



Figure 6. Aerial views of Site 3 ATV ford. Stream flows from right to left in both photos. (Photos courtesy of Michael Weidmer ADF&G.)



Figure 7. Site 3 ATV ford (facing north).

Study Approach

The primary goal of this study was to isolate the influence of ATV stream fords on stream sedimentation and biota. As such, all parameters were measured at various distances upstream and downstream of the ford. As originally conceived, this study was to use all sample dates at three sites as replicates, thus offering the ability to statistically test hypotheses (albeit with relatively low power). Upon working at these sites it became obvious that physical and vegetative differences had profound influences on the character and sedimentation potential of the fords and that, because of this, the three fords differed remarkably. Treating the three fords as replicates would have required the assumption that all fords functioned similarly and that these three fords represented the range of fords in this geographic area—an oversimplification that would require a much larger sample size to overcome. Therefore, we examined each ford independently, comparing upstream positions to downstream positions.

Sediment Deposition – Sediment Collectors

We used sediment collectors to measure day-to-day sediment deposition during the study period. Sediment collector construction and application followed the technique in Brown (1994). Each sediment collector consisted of a Petri dish base (~85 mm diameter) into which we affixed a coarse filter paper with silicone-based adhesive. We assigned each collector a unique identification number (written on the collector in permanent marker) to keep track of attributes of

the collector throughout the study. After allowing the adhesive and marker to dry for at least 48 hours, we weighed each sediment collector on a calibrated balance to the nearest 0.0001 g.

For instream deployment, we nested the base of each sediment collector into its lid, which formed a stable base, and fastened the entire apparatus to a flat anchoring rock with two crisscrossing rubber bands (Figure 8). The anchoring rocks were seated into the streambed so the bottom of each sediment collector was level with the adjacent streambed. We positioned arrays of sediment collectors at 50 m and 10 m upstream (denoted as +50 m and +10 m) and 10 m, 100 m, 250 m, and 500 m downstream (denoted as -10 m, -100 m, -250 m, and -500 m) of each ATV ford. For the first sampling period we used arrays of 15 collectors (5 rows of 3; Figure 8). Since both collector loss and variation in deposition among collectors in a given position were relatively low, we used arrays of 9 (3 rows of 3) in subsequent sampling periods. To minimize the influence of hydrologic variability on sediment deposition, we positioned sediment collectors in a standard depth of 0.3 m and a standard current velocity of 0.3 m/s (Marsh-McBirney model 2000).



Figure 8. Sediment collector array and (inset) individual sediment collector.

After a period of deployment (Table 1), sediment collectors were gathered (any water in the dish was carefully removed with a pipette so as to not lose sediment), dried at room temperature for 48 hours, oven dried at 45 °C until constant mass, and re-weighed. We calculated sediment deposition as the difference between the post-deployment mass and the pre-deployment mass. We used the Kruskal-Wallis nonparametric ANOVA to test for differences among sediment deposition at the various positions (SPSS Base 9.0). This test was chosen because the data did not satisfy the parametric ANOVA homogeneity of variance assumption (i.e., variance was higher at high sediment levels) and square root transformation did not bring the data into compliance. When the ANOVA indicated a significant difference ($\alpha = 0.05$), we used Dunnett's C post-hoc test for pairwise comparisons (SPSS Base 9.0).

Suspended Sediment – Siphon Samplers

Siphon (single-stage) samplers were used to draw water samples for suspended sediment analysis. Siphon samplers were constructed following specifications in Edwards and Glysson (1999). Siphon samplers are designed to draw and hold water during the rising limb of the hydrograph (i.e., as water level rises above the curve in the siphon tube in response to rainfall; Figure 9), a period when suspended sediment was expected to be highest and when any sediment originating from ATV stream fords was expected to be mobilized.

Siphon samplers were attached (using a large hose clamp) to steel fence posts driven into the streambed. Samplers were deployed in duplicate (to collect concurrent replicates and allow an estimate of sample collection precision [Mueller, Martin, and Lopes 1997]) at distances of 50 m and 10 m upstream of (denoted as +50 m and +10 m) and 10 m, 100 m, and 250 m downstream of (denoted as -10 m, -100 m, and -250 m) each ford. To maximize the potential that all siphon

Table 1. Sediment collector installation dates, removal dates, and number installed.

Site	Date Installed 2002	Date Retrieved 2002	Sediment Collectors per Array
1	21 July	8 Aug	15
	11 Aug	10 Sep	9
	10 Sep	30 Sep	9
2	23 July	10 Aug	15
	10 Aug	9 Sep	9
	9 Sep*	29 Sep	9
3	9 Aug	8 Sep	9
	8 Sep*	1 Oct	9

* unsuccessful sampling period due to sediment collector loss

samplers collected water during the same rising-limb event, siphon samplers were deployed in runs of similar hydrologic dimensions (e.g., channel width, water depth, current velocity). The top of the siphon tube, which determines the rise in stage height necessary to trigger the siphon sampler, was placed at a similar distance above the water surface in all samplers.

Siphon samplers were collected and replaced after rainfall of sufficient magnitude to activate the samplers. Laboratory suspended sediment analysis of siphon samples followed standard methods (American Public Health Association 1998). Each sample was homogenized and a measured volume (graduated cylinder) was filtered through a pre-ashed, pre-weighed (to 0.0001 g using a calibrated balance) glass fiber filter. After filtration, filters were dried to constant mass at 104 °C and re-weighed (calibrated balance) to yield measurement of total suspended solids. The filters were then ashed at 550 °C and re-weighed to yield a measure of suspended inorganic solids. Split replication was conducted on 10% of samples to allow estimation of laboratory precision (Mueller, Martin, and Lopes 1997).

Substrate Composition – Substrate Photography

We followed the U.S. Geological Survey photographic technique for measuring coverage of fine sediment methods (Whitman, Moran, and Ourso 2003). Streambed photography employed a 35 mm SLR camera (28–200 mm lens) mounted on a Plexiglas-bottomed view bucket (Whitman et al., in press). Upstream and downstream of each ford, we established 11 transects at 13 m intervals. At each transect we shot three close-up slide photographs of the streambed: one in the thalweg and one equidistant between the thalweg and each of the streambanks. At those transects where the thalweg was adjacent to a streambank, we shot one photograph in the thalweg and two photographs equally spaced between the thalweg and the opposite bank. Streambed photographs were taken between 29 September and 1 October 2002.

We used a high-resolution scanner (4000 dots per inch) to digitize each slide and imported the resulting images into GIS software (ArcView 3.2) to digitize the coverage of fine sediment and quantify its percent coverage. For 10% of all slides, different staff members quantified fine sediment coverage in duplicate to estimate laboratory precision.

Biological Response – Benthic Macroinvertebrates

Benthic macroinvertebrate sampling and processing generally followed the techniques in Major and Barbour (2001). We sampled during summer base-flow conditions (8–10 August) and during fall high-flow conditions (29 September–1 October) to allow for comparison of consistency across seasons and flow conditions. At each site we established three 100 m sample reaches: immediately upstream of the ford (referred to as UP1 in this report), immediately downstream of the ford (referred to as DS1 in this report, and 500 m downstream of the ford (referred to as DS2 in this report). Within each sample reach, 20 kicks, each covering approximately 0.14m², were collected (using a 350 µm mesh D-frame kick net) from riffle areas

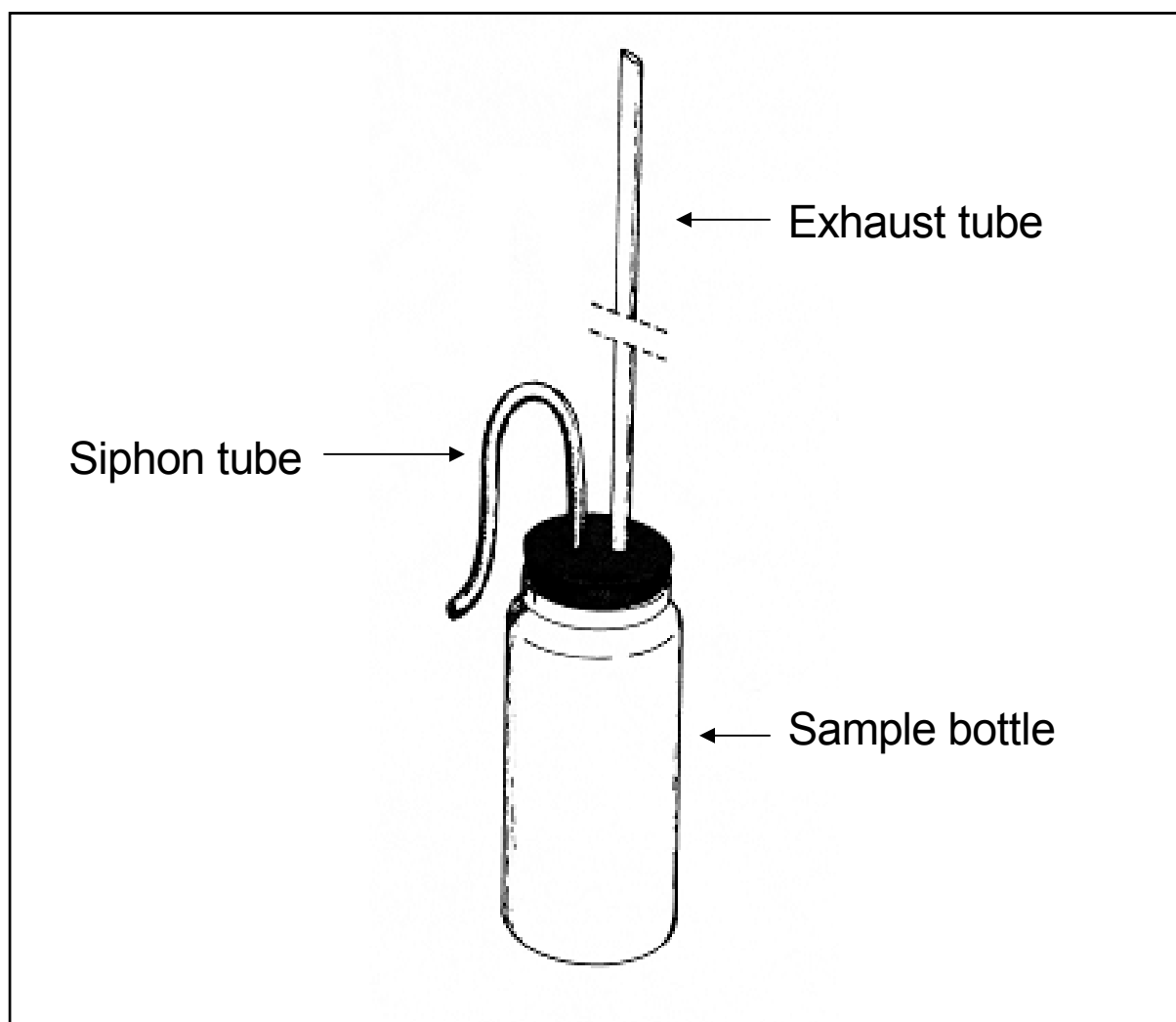


Figure 9. Schematic of siphon sampler.

and composited into a single sample. We conducted each kick by placing the kick net firmly on the stream substrate and manually disturbing the substrate and rubbing rocks upstream of the net; dislodged macroinvertebrates were carried into the net by flowing water. Samples were preserved in the field with 70% ethanol and returned to the lab for processing. Physicochemical measurements (discharge, temperature, pH, dissolved solids, conductivity, and dissolved oxygen) were taken at each stream simultaneously with macroinvertebrate collections.

From each sample we randomly subsampled 300 ($\pm 20\%$) macroinvertebrates and conducted a timed pick (5 minutes) to ensure that all large and/or rare taxa were represented in each sample (see Major and Barbour 2001 for details). Subsampling was used to standardize effort across all samples and to reduce (to a feasible level) the number of organisms requiring identification. All insects in the subsample were identified to genus except the Chironomidae, which were

identified to the subfamily or tribe level (Merritt and Cummins 1996). All non-insects were identified to higher taxonomic levels, typically class or order.

Various attributes of the macroinvertebrate community were quantified for comparison among positions within each stream. Sites with consistent, detectable sedimentation originating from the ford will yield the most information on the influence of fine sediment on macroinvertebrates. We calculated macroinvertebrate taxa richness, density, total biomass, and Fine Sediment Biotic Index (FSBI; Relyea, Minshall, and Danehy 2000) for each sample. Biomass was calculated for each individual in the sample (using the length-dry mass regressions in Benke et al. 2000) and summed to yield the total biomass. FSBI, designed to detect stream impacts due to fine sediment in the northwestern contiguous United States, was tested as a cost-effective, sedimentation-specific tool for documenting biological impairment. FSBI assigns a sediment tolerance value to insect taxa (1 = sediment tolerant, 8 = sediment intolerant) and ranks streams by summing tolerance values for all taxa present. We selected FSBI for use in this study because it showed remarkable taxonomic overlap with Alaska stream fauna and because it has shown sensitivity to suspended sediment levels during preliminary work conducted by ENRI (Rinella 2003).

We also compared biomass of each common taxon (i.e., those species or genera with ≥ 5 individuals in each sample for a given site and date) among the three positions. Each individual served as a replicate so statistical comparison was possible. Due to homogeneity of variance violations, we compared positions with Kruskal-Wallis nonparametric ANOVA and Dunnet's C nonparametric post-hoc test (SPSS Base 9.0). Because multiple taxa were compared, we used Bonferroni corrections to maintain an "experimentwise" alpha of 0.05 for each site and date (adjusted $\alpha = 0.05/\text{number of comparisons}$).

Results

Sediment Deposition – Sediment Collectors

Three sediment collector-sampling sessions were successfully conducted at Site 1 (see Table 1); for clarity we will refer to them by the date of installation. The 21 July and 11 August sampling sessions showed very similar patterns of sediment deposition, with the -10 m collectors accruing approximately 5 g of sediment per sampler, while all other positions accrued approximately 1 g per sampler (Figure 10). Despite this similarity in pattern, only the 21 July samples showed significant differences among positions ($p < 0.001$), with the -10 m position showing greater sediment accrual and the -250 m and -500 m positions showing lower sediment accrual than the other positions (Figure 10). The lack of statistical significance for the 11 August samples ($p = 0.055$) may have been related to the lower number of sediment collectors used in this sampling effort (i.e., 9 collectors per array vs. 15; Table 1). The sampling effort initiated on 10 September showed considerably less sediment deposition (likely related to increased discharge) than the earlier two sampling dates, and no significant difference among positions was detected ($p = 0.370$; Figure 10).

Two sediment collector-sampling sessions were successfully conducted at Site 2 (see Table 1). The session initiated on 23 July 2002, with a high of 7.4 g sediment at -10 m and a low of 5.2 g sediment at -250 m, showed no difference among positions ($p = 0.48$; Figure 11). The 10 August sampling session showed significant differences among positions ($p < 0.001$) but the deposition pattern did not appear to be related to the ford: sediment deposition was highest at the +10 m position (15.4 g sediment) and tapered off both upstream and downstream (Figure 11).

One sediment collector sampling session, initiated on 9 August 2002, was successful at Site 3. Sediment deposition was low at all positions (< 0.4 g sediment; Figure 12) and no significant difference among the positions was detected ($p = 0.58$).

For all three sites, sediment deposition was most variable at the position immediately downstream of the ATV ford (-10 m). Sediment collector data collection was cut short by extensive flooding during October and November of 2002.

Suspended Sediment – Siphon Samplers

Patterns of suspended sediment concentration were similar for total sediment and inorganic sediment, so only values for inorganic sediment are displayed graphically. Table 2 gives values for total and inorganic suspended sediment. Duplicate samplers indicated that average field sampling error was relatively high for both total solids (29%) and inorganic solids (30%), with individual sampler pairs showing discrepancies as high as 170%. Laboratory

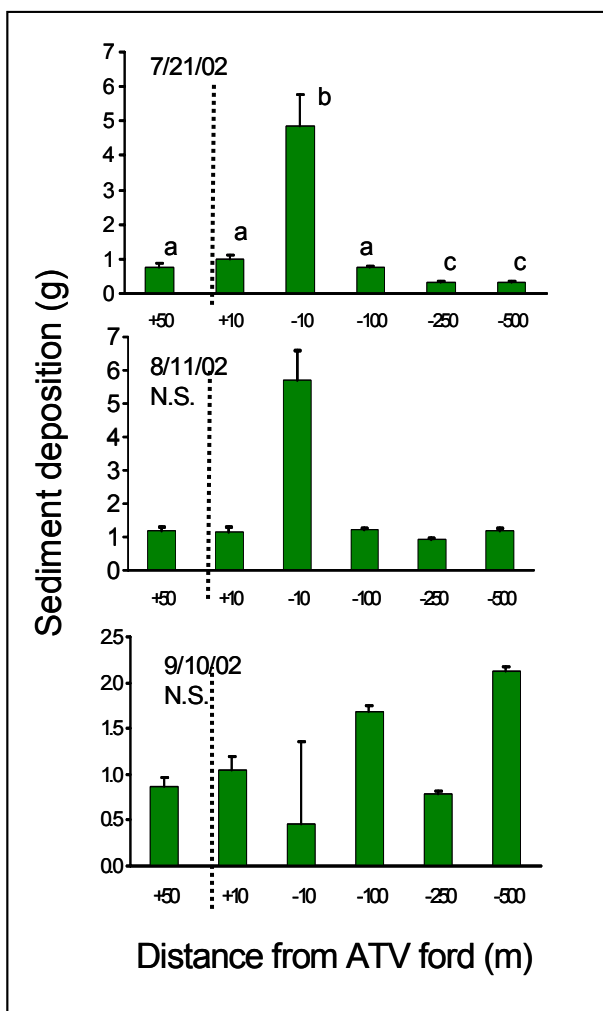


Figure 10. Mean (+ SE) mass of sediment deposited within sediment collectors at Site 1. The dashed line represents the ATV ford; X-axis values represent distance upstream (+) and downstream (-) from the ford. N.S. = no significant difference among positions; bars that do not share a lower case letter are significantly different.

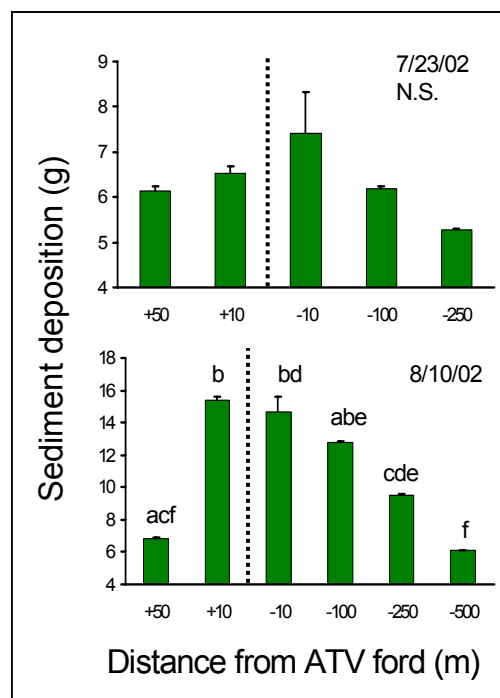


Figure 11. Mean (+ SE) mass of sediment deposited within sediment collectors at Site 2. The dashed line represents the ATV ford; X-axis values represent distance upstream (+) and downstream (-) from the ford. N.S. = no significant difference among positions; bars that do not share a lower case letter are significantly different.

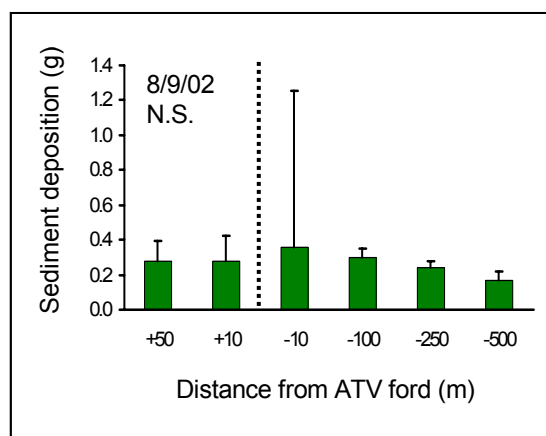


Figure 12. Mean (+ SE) mass of sediment deposited within sediment collectors at Site 3. The dashed line represents the ATV ford; X-axis values represent distance upstream (+) and downstream (-) from the ford. No significant difference was detected among

Table 2. Total suspended solids and total suspended inorganic sediment from each siphon sampler. Position values represent distance (in m) upstream (+) and downstream (-) from the ford.

Site: Date Retrieved 2002	Position	Total Suspended Solids	Total Inorganic Solids
Site 1: 30 Sep	+50	32	30
	+50	20	26
	+10	39	24
	+10	36	12
	-10	166	149
	-10	61	49
	-100	56	44
	-100	56	44
	-250	63	45
	-250	63	51
Site 2: 29 Feb	+50	62	50
	+10	63	52
	-10	350	323
	-100	789	678
	-100	64	53
	-250	17	14
Site 3: 8 Sep	+50	182	154
	+50	241	161
	+10	202	131
	+10	213	178
	-10	229	171
	-10	237	180
	-100	213	160
	-100	232	174
	-250	234	177
	-250	266	205
Site 3: 1 Oct	+50	48	49
	+50	317	88
	+10	114	33
	-10	54	36
	-10	96	73
	-100	404	317
	-250	391	298

replication indicated relatively low sampling error: 10% for total solids and 11% for inorganic solids. Siphon-sampling effort was cut short by extensive flooding during October and November of 2002.

One siphon-sampling session was successfully completed at Site 1; these samplers were retrieved on 30 September 2002. The longitudinal pattern of suspended sediment concentrations suggests that sediment was being introduced at the ATV ford, as sediment concentrations were consistently higher at downstream positions (44–149 mg/L) than at upstream positions (12–30 mg/L; Figure 13). However, there was a considerable difference between the -10 m samples.

One siphon-sampling session was completed at Site 2 (Figure 14). Sediment concentrations were increased at the 10 m and 100 m downstream positions, relative to upstream positions, suggesting that sediment may have washed into the stream at the ford. However, only a fraction of the samplers were activated and there was considerable variation both among and within positions.

Two siphon-sampling sessions were completed at Site 3; data from neither session suggested sedimentation from the ATV ford (Figure 15). The samplers collected on 8 September 2002 showed little variation among or between positions. The samplers collected on 1 October 2002 showed high suspended sediment concentrations at the -100 m and -500 m positions with lower values (and little variation) at the +50 m, +10 m, and -10 m positions.

Substrate Composition – Substrate Photography

Laboratory duplication of sediment coverage showed sampling error to be relatively low (6%). Site 1 substrate photographs showed fine sediment coverage approaching 100% at the upstream three transects (Figure 16). These transects were located within a beaver pond (see Figure 2), where sediment deposition and coverage were expected to be high regardless of sedimentation rates. Below the beaver pond, upstream transects showed 0.8%–21% fine sediment coverage. The first two transects below the ATV ford showed increased sedimentation relative to upstream positions (66% and 97% fine sediment). Below the third downstream transect, located within a steep riffle, fine sediment coverage generally decreased with distance downstream, returning to levels similar to upstream by 75 m downstream.

Site 2 substrate photographs showed relatively low, fine sediment coverage, with all transects $\leq 15\%$ coverage (Figure 16). There was no longitudinal pattern to suggest that fine sediment coverage was increased downstream of the ATV ford. Likewise, Site 3 showed low fine sediment coverage ($\leq 7\%$ coverage) with no pattern suggestive of increased fine sediment coverage downstream of the ford.

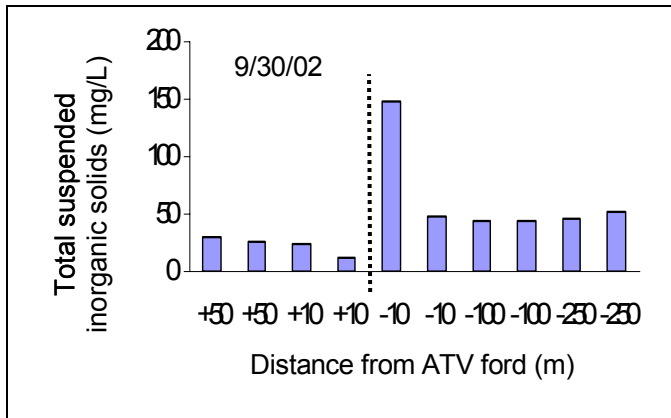


Figure 13. Total suspended inorganic solids from each siphon sampler at Site 1. Installation date is given within the figure. The dashed line represents the ATV ford; X-axis values represent distance (in m) upstream (+) and downstream (-) from the ford.

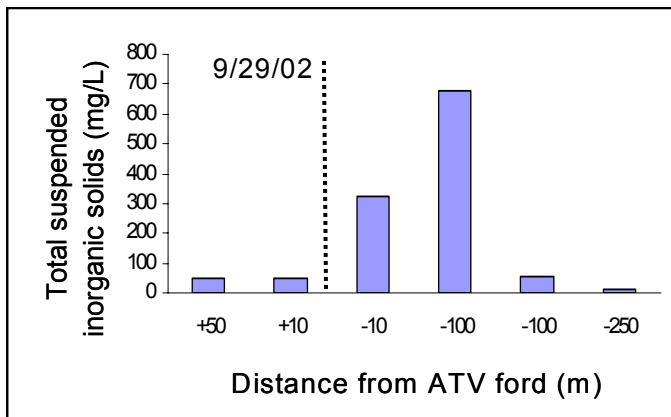


Figure 14. Total suspended inorganic solids from each siphon sampler at Site 2. Installation date is given within the figure. The dashed line represents the ATV ford; X-axis values represent distance (in m) upstream (+) and downstream (-) from the ford.

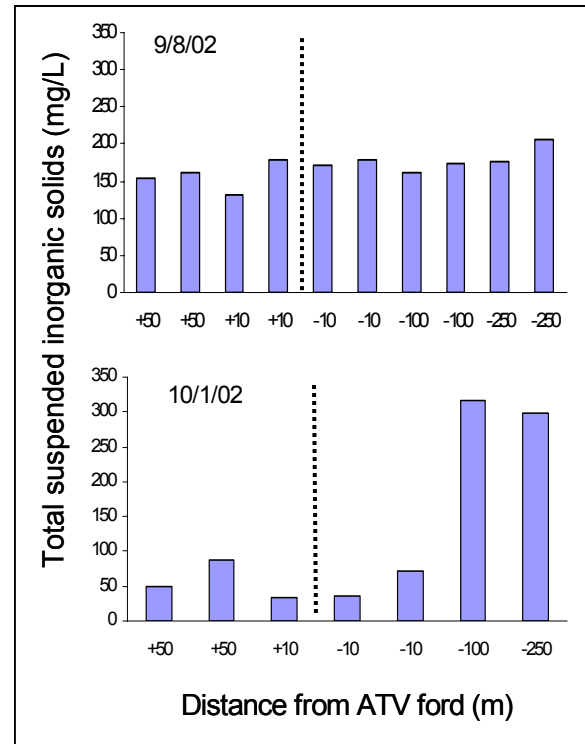


Figure 15. Total suspended inorganic solids from each siphon sampler at Site 3. Installation dates are given within each panel. The dashed line represents the ATV ford; X-axis values represent distance (in m) upstream (+) and downstream (-) from the ford.

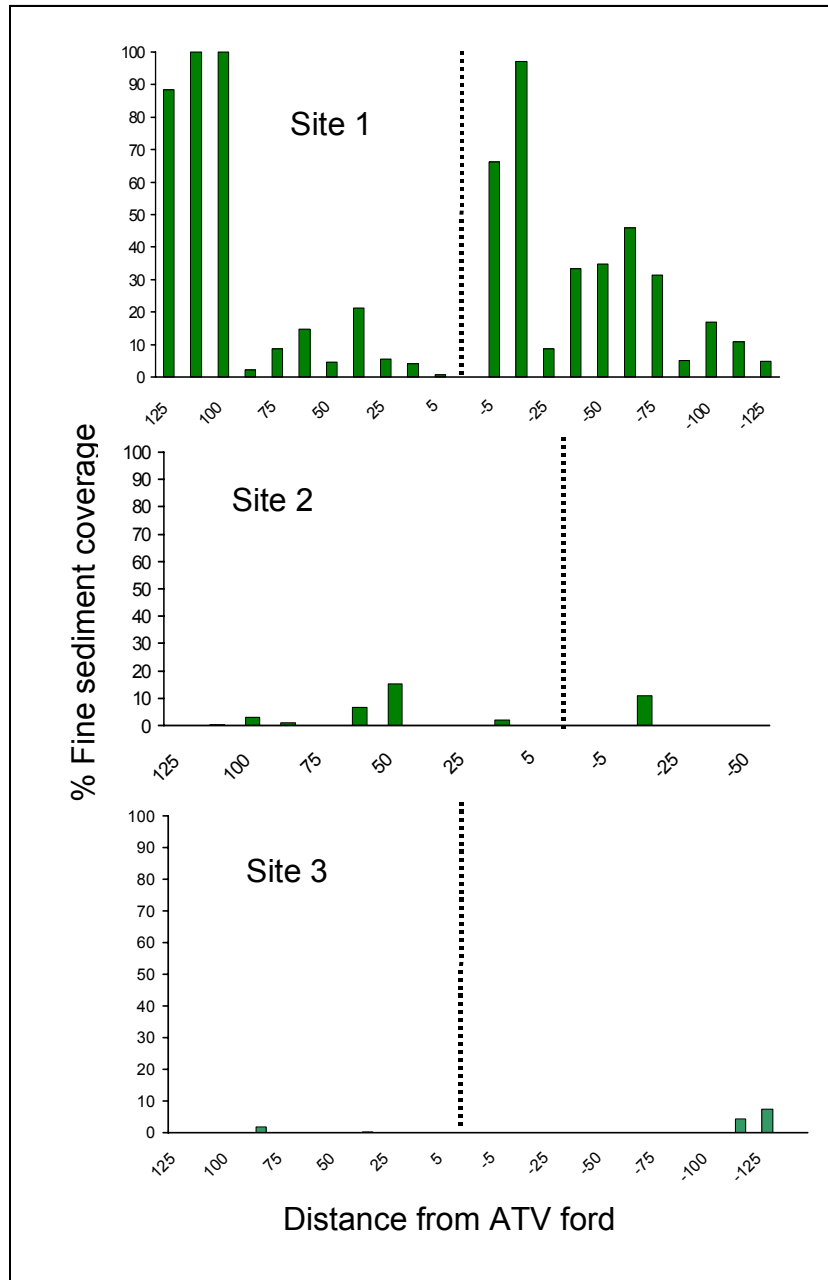


Figure 16. Percent coverage of fine sediment measured by streambed photography (29 September–1 October 2002). The dashed line represents the ATV ford; X-axis values represent distance (in m) upstream (positive values) and downstream (negative values) from the ford.

Biological Response – Benthic Macroinvertebrates

Physicochemical measurements taken simultaneously with macroinvertebrate sampling are presented in Table 3. For both sampling dates, Site 1 had the lowest discharge and Site 2 had the highest. Within each site, temperature was lower and discharge was at least two times higher during the later (i.e., fall) sampling period. Total dissolved solids and conductivity were lower during the fall, likely the result of dilution. Dissolved oxygen was near saturation and pH was near neutral at all three sites.

Of the three sites examined in this study, only Site 1 appeared to be receiving significant and consistent sediment inputs from the ATV stream ford (Figures 10, 13, 16). Therefore, this site is of particular interest as it allows comparison of macroinvertebrate assemblages receiving varying degrees of sedimentation. In the summer sample, taxa richness was slightly reduced downstream and showed some recovery after 500 m (Figure 17). This pattern was not observed in the fall sample, as the DS1 position was similar to the US1 control. During summer FSBI was reduced at the DS1 position and intermediate at the DS2 position; FSBI values were relatively similar during the fall, with the DS2 position having the lowest value. During summer, density was highest at the US1 position and lowest at the DS1 position; during fall density was greatest at the DS2 position. Biomass was highly variable among positions and seasons, showing a sevenfold difference between low and high values. In the summer samples, biomass was slightly reduced at the DS1 position and considerably higher at the DS2 position relative to the US1 control. In the fall samples, biomass was markedly lower at the DS1 position and showed some recovery after 500 m (DS2).

At Site 2, FSBI scores were highest at the DS1 position for both summer and fall samples (Figure 18). For all other metrics, downstream positions showed reduced values (relative to the US1 control) for both seasons. The DS2 position showed considerable variation in relation to the other positions.

Table 3. Physicochemical measurements taken simultaneously with macroinvertebrate sampling.

Site	Date 2002	Discharge (cfs)	Temperature (C)	pH	Total Dissolved Solids (mg/L)	Conductivity (µs/cm)	Dissolved Oxygen (% Saturation)
1	9 Aug	0.6	13.08	7.43	0.0255	39.7	91.3
	30 Sep	1.2	5.32	6.94	0.0127	20.0	93.3
2	10 Aug	14.0	8.27	7.60	0.0444	69.2	95.7
	29 Sep	36.0	5.05	6.95	0.0255	40.0	95.1
3	8 Aug	1.1	9.34	7.42	0.0282	43.9	88.4
	1 Oct	7.3	6.12	6.80	0.0091	14.1	92.1

At Site 3, all metric scores were higher at the DS1 position than at the US1 control position (Figure 19). With the exception of fall biomass, metric scores at the DS2 position were remarkably similar to the US1 control position.

Twenty-one macroinvertebrate populations were abundant enough to allow for biomass comparison among the three positions within a site (Table 4). Of these, three populations showed significant differences among positions (Figure 20; Table 4). At Site 2 during summer, the mayfly *Drunella* were larger at the US1 position (0.046 g per individual) than at the DS1 position (0.029 g) and the DS2 position (0.028 g). At Site 3 during summer, the blackfly *Simulium* were larger at the US1 position (0.17 g per individual) than at the DS1 position (0.10 g). At Site 3 in the summer, the mayfly *Baetis bicaudatus* were larger at the US1 position (0.062 g per individual) than at the DS1 position (0.038 g) and the DS2 position (0.032 g).

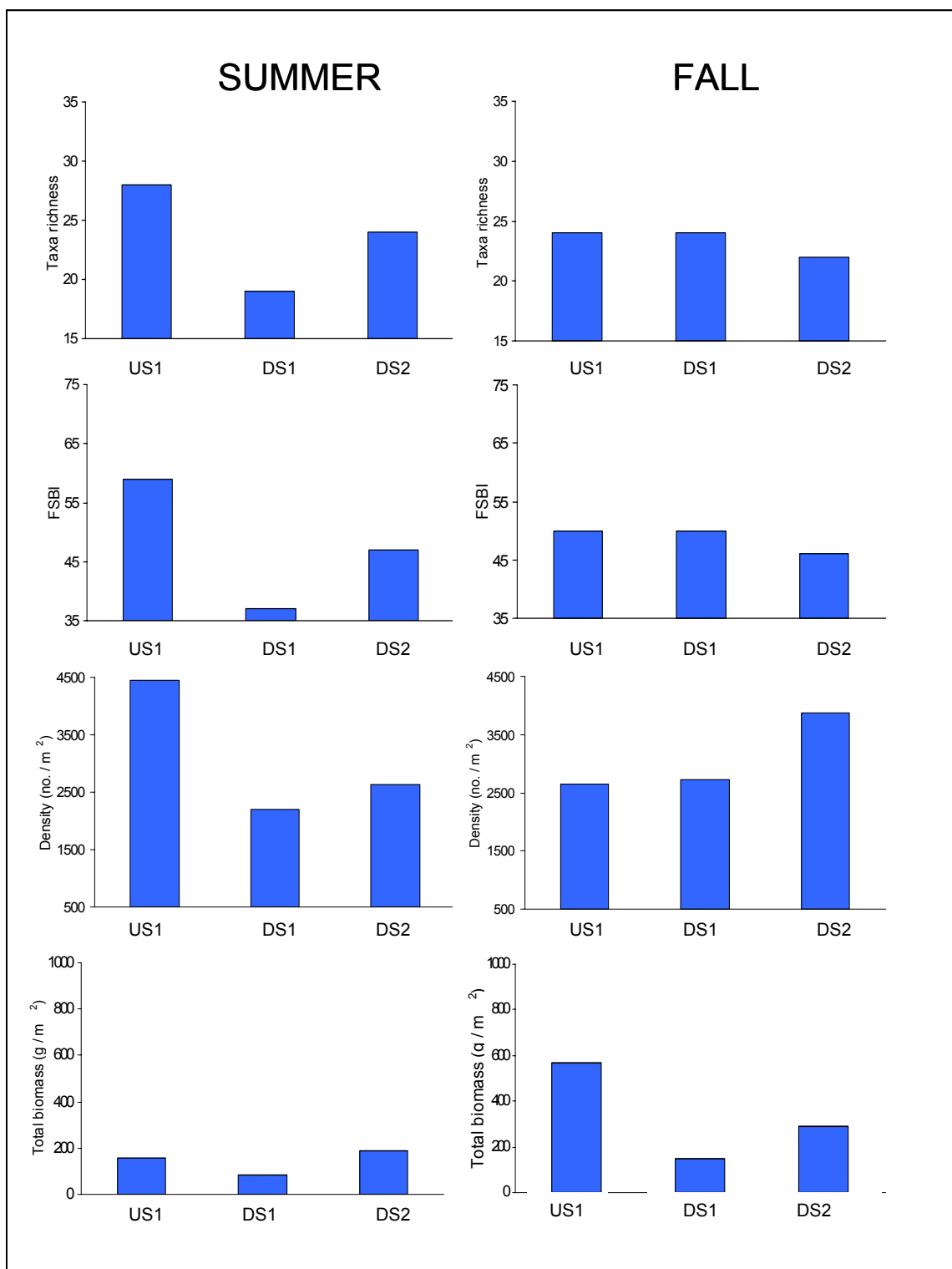


Figure 17. Site 1 macroinvertebrate metric values from summer and fall samples.

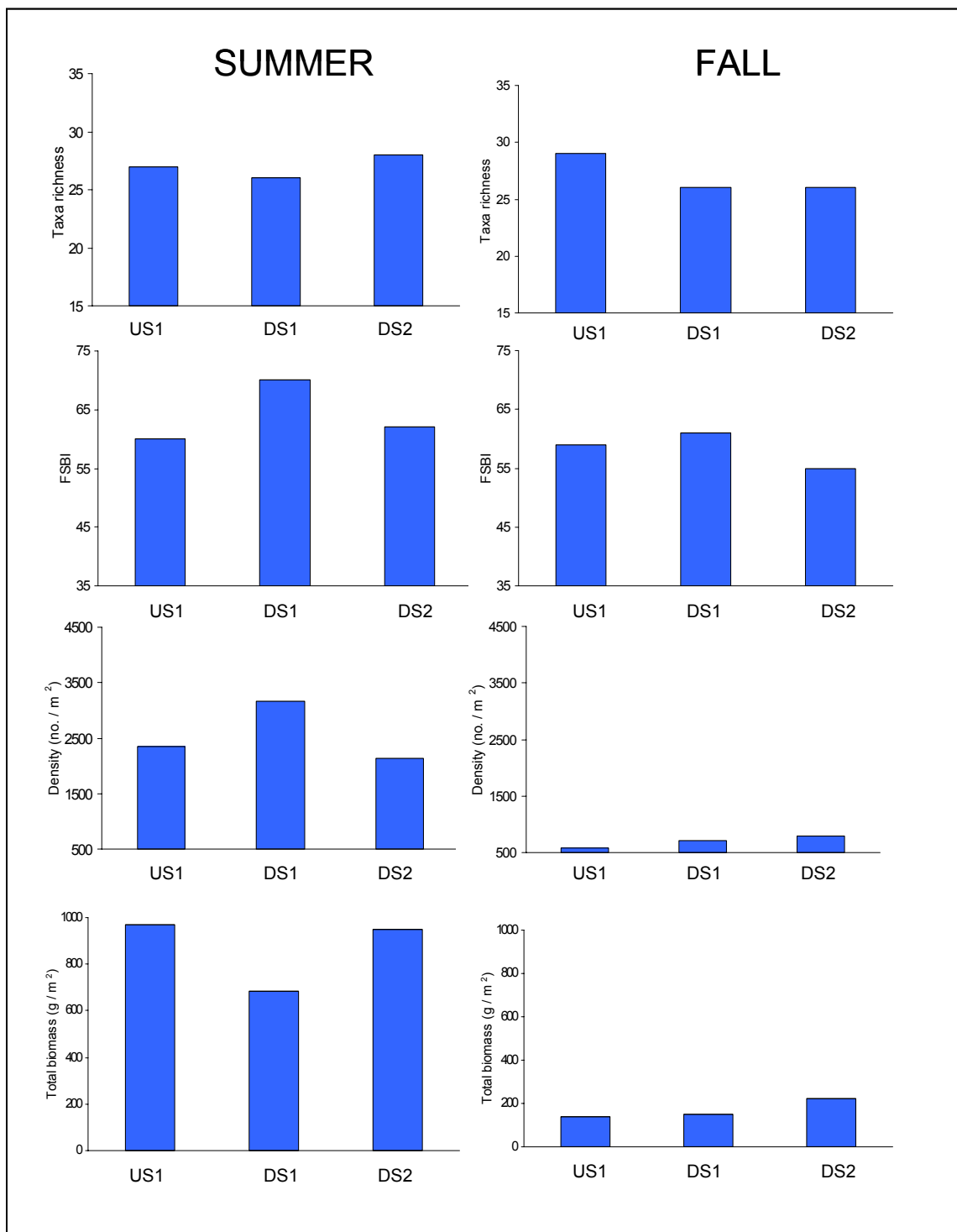


Figure 18. Site 2 macroinvertebrate metric values from summer and fall samples.

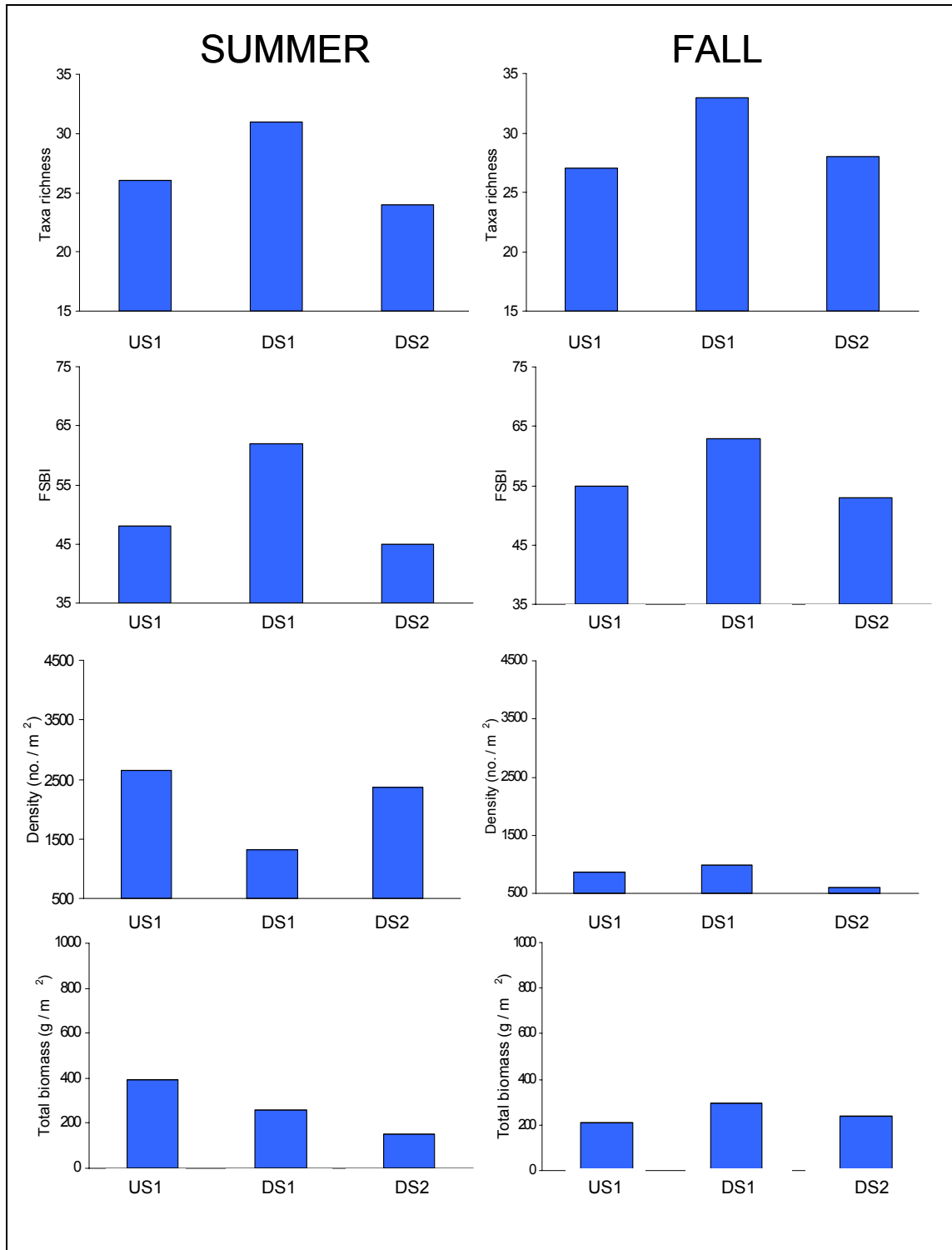


Figure 19. Site 3 macroinvertebrate metric values from summer and fall samples.

Table 4. Mean biomass (by sample) of common macroinvertebrate taxa and results of ANOVA comparing biomass of common taxa among positions.

Site: Season	Taxon	Mean Individual Biomass in g Dry Mass and [Sample Size]			<i>p</i>	Adjusted α
		Upstream	Downstream	500 m Downstream		
1: Summer	<i>Baetis bicaudatus</i>	0.0265 [6]	0.0800 [10]	0.092 [20]	0.495	0.017
	<i>Prosimulium</i>	0.0124 [81]	0.00958 [35]	0.00886 [28]	0.664	0.017
	<i>Zapada</i>	0.0194 [9]	0.00736 [5]	0.137 [15]	0.031	0.017
1: Fall	<i>Baetis bicaudatus</i>	0.0593 [49]	0.0627 [30]	0.0531 [63]	0.498	0.013
	<i>Brachycentrus</i>	0.0769 [17]	0.0318 [16]	0.0333 [21]	0.320	0.013
	<i>Simulium</i>	0.109 [68]	0.0988 [19]	0.122 [19]	0.197	0.013
	<i>Zapada</i>	0.0172 [12]	0.0172 [17]	0.0836 [42]	0.179	0.013
2: Summer	<i>Brachycentrus</i>	0.633 [28]	0.469 [16]	0.864 [11]	0.149	0.013
	<i>Drunella</i>	0.0461 [29]	0.0294 [26]	0.0281 [24]	0.002*	0.013
	<i>Glossosoma</i>	1.03 [27]	0.436 [22]	1.06 [15]	0.020	0.013
	<i>Zapada</i>	0.0883 [23]	0.0663 [21]	0.0657 [11]	0.784	0.013
2: Fall	<i>Baetis bicaudatus</i>	0.0442 [60]	0.0477 [50]	0.0294 [23]	0.038	0.013
	<i>Brachycentrus</i>	0.0654 [23]	0.157 [19]	0.322 [12]	0.162	0.013
	<i>Drunella</i>	0.305 [29]	0.323 [20]	0.371 [21]	0.466	0.013
	<i>Glossosoma</i>	1.63 [16]	1.80 [13]	2.38 [21]	0.036	0.013
3: Summer	<i>Probezzia</i>	0.258 [5]	0.445 [5]	0.402 [11]	0.229	0.025
	<i>Simulium</i>	0.172 [15]	0.104 [17]	0.120 [5]	0.002*	0.025
3: Fall	<i>Baetis bicaudatus</i>	0.0617 [56]	0.0378 [93]	0.0320 [19]	<0.001*	0.013
	<i>Brachycentrus</i>	0.0937 [11]	0.387 [19]	0.628 [11]	0.030	0.013
	<i>Drunella</i>	0.554 [15]	0.609 [6]	0.461 [9]	0.263	0.013
	<i>Zapada</i>	0.109 [46]	0.207 [57]	0.079 [42]	0.376	0.013

* denotes statistically significant difference among positions

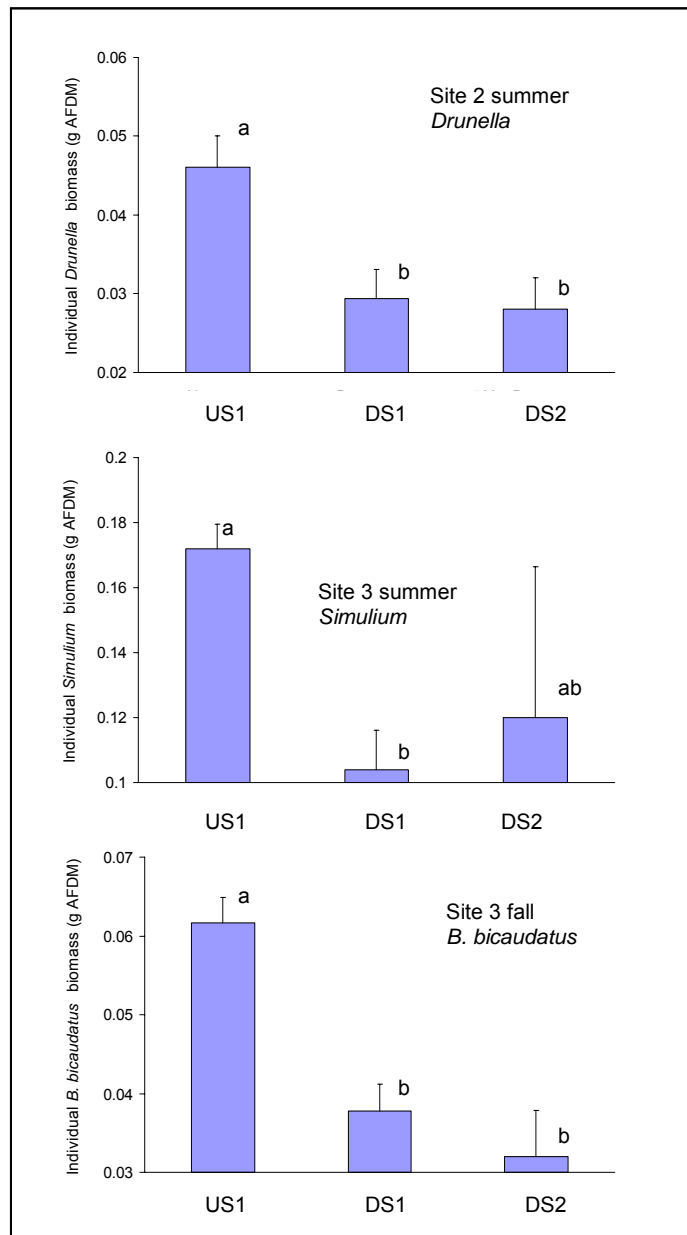


Figure 20. Macroinvertebrate populations showing significantly different biomass among positions. (Bars that do not share a lower case letter are significantly different.)

Discussion

The potentially high error between duplicate siphon samplers (up to 170%) and the fact that some samplers drew water while the duplicate sampler remained unfilled indicates that samplers placed side-by-side were not necessarily drawing water simultaneously (i.e., at the same stage of the rising hydrograph). This casts doubt that samplers at different positions within the same stream were drawing samples simultaneously, despite our efforts to deploy samplers in stream reaches of similar channel dimensions. For future siphon sampler use, we strongly suggest the development and use of an electronic timer that can indicate the time at which samplers were filled. Furthermore, if the opening of the exhaust tube becomes submerged, the sample bottle may receive a continuous flow of water that allows potentially large amounts of sediment to settle inside the sample bottle, potentially leading to overestimation of the suspended sediment concentration by multiple orders of magnitude (Edwards and Glysson 1999). To avoid this, we recommend that crest gages be installed adjacent to all sets of siphon samplers. When the crest gages indicate that water had reached sufficient depth to submerge the exhaust tube, samples should be discarded.

Brown (1994), in the paper that introduced the design of the sediment collectors used in this study, expressed sediment deposition as a rate (i.e., g/cm²/d), which makes the assumption that the efficiency of sediment collectors will be constant throughout the sampling period. Our field experience suggested that sediment collectors trapped sediment more efficiently when first deployed, then less efficiently as a layer of sediment began to fill the dish. Therefore, the expression of sediment deposition as a rate would tend to underestimate the true value, so we expressed sediment deposition as an absolute value (g/collector).

Our data suggest that the degree of sedimentation impact at an ATV ford can be influenced by local vegetation, soil, and topography. Of the three ATV stream ford sites used in this study, only Site 1 appeared to receive consistently measurable sedimentation from the ford. This is likely because both ford approaches at Site 1 had ample unconsolidated fine sediment available for transport into the stream. A heavy sediment plume was visible for 10 minutes or more upon ATV passage. Furthermore, the approach topography, which sloped directly into the stream from both sides, appeared to act as a sediment conduit during periods of rain. Stream volume may also mediate stream sedimentation—Site 1 was the smallest of the three streams in this study, and larger streams offer greater dilution potential.

Sediment data from Site 2 indicated no increased sediment deposition or alteration of the substrate composition downstream of the ford. The trail in the vicinity of the ford was naturally hardened with gravel and cobble, which appeared to prevent sediment from entering the stream during day-to-day ATV use (no sediment plumes were observed during vehicle passage). The approach topography was relatively flat, which likely prevented surface flow from entering the stream during all but heavy rains. We personally observed sediment-laden surface flows entering the stream during two heavy rains, and siphon sampler data suggest that sediment

concentrations downstream of the ford may be markedly increased during rain episodes. Site 3 sediment data suggested no increased sedimentation downstream of the ford. The approaches consisted of saturated organic soil that was firmly bound by roots. Although both ford approaches were marked by considerable wetland damage for some distance, little sediment was available for transport into the stream. No sediment plumes were visible during ATV passage at Site 3.

Because Site 1 showed the most pervasive sedimentation impacts, this site is of particular interest for examining the influence of ATV ford sedimentation on stream biota. For all metrics (taxa richness, FSBI, density, and biomass), the summer macroinvertebrate sample showed reduced values at the DS1 position and partial or full recovery at the DS2 position. Although not statistically testable, the consistency of this pattern across all metrics is compelling evidence of a biotic response to sedimentation. This pattern did not hold during fall sampling, where only biomass was reduced at the DS1 position. This may be related to the dramatically higher discharge (2x) observed during fall sampling that can cause invertebrates to drift from areas of upstream refuge and redistribute in downstream habitats (Bird and Hynes 1981).

Due to documented sedimentation, Site 1 provided a good test of FSBI. Relative to the US1 position, FSBI was markedly reduced at the DS1 position in the base-flow summer sample. Sediment photography data, collected concurrently with fall macroinvertebrate sampling, showed increased coverage of fine sediment downstream of the ford. However, FSBI showed essentially no difference between upstream and downstream positions, suggesting that FSBI did not respond to increased streambed fine sediment composition during fall. Again, this may be related to increased macroinvertebrate drift during high stream flow (Bird and Hynes 1981). Sites 2 and 3 consistently showed somewhat increased FSBI scores at the DS1 position, a pattern for which we have no immediate explanation. Taken together, our data suggest that FSBI may be useful for detecting sedimentation impacts during summer base-flow conditions.

Further studies of ATV stream ford impacts on lower Kenai streams are warranted and should be guided by the knowledge gained in this study. A replicated study covering a large number of fords and measuring a range of landscape variables that likely mediate sedimentation (i.e., soil type, topography, degree of trail downcutting, riparian vegetation, traffic volume, etc.) will identify landscapes that are prone to sediment impacts and, more importantly, landscapes that are impervious to sediment impact. With a large number of sites, rapidly measured response variables will be necessary and, based on the experience gained in this study, we recommend the USGS photographic method (Whitman, Moran, and Ourso 2003) in conjunction with measures of tractive force to establish the expected streambed particle size. Information from simulated ATV convoy experiments and/or simulated rain experiments should supplement the ambient data. From these data it would be possible to develop a model for predicting sedimentation at ATV fords based on landscape attributes that could be used for designing new trails and rerouting existing trails to stable landscapes, prioritizing existing ATV fords for mitigating improvements, and guiding the development of BMPs for ATV stream crossings.

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