Biomonitoring for the Rocky Mount Power Dam Removal Project:

Establishing Baseline Conditions

Final report

Photo: N.P. Hitt

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Nathaniel P. Hitt[1,2]
Jeff Jones[1]
and
Ken Convery[1]

[1]Conservation Management Institute
Virginia Tech
1900 Kraft Drive, Suite 250
Blacksburg, Virginia 24061

[2]Current affiliation:
US Geological Survey
Leetown Science Center
11649 Leetown Road
Kearneysville, West Virginia 25443
nhitt@usgs.gov
(304) 724-4463

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Executive summary

The Rocky Mount Power Dam (RMPD) is located on the Pigg River in Franklin County, Virginia. The Virginia Rural Area Conservation and Economic Restoration (VA RACER) initiative has developed partnerships to evaluate the potential economic and ecological benefits of dam removal, as well as the potential risks to downstream populations of Roanoke logperch (*Percina rex*) and other species of concern. Here, we characterize the biological and physical condition of the Pigg River upstream and downstream from the RMPD. The intent of this report is to provide baseline data that will be useful for subsequent evaluations during and after dam-removal.

We documented significant variation in physical habitat, water chemistry, and biotic communities in the study area. The greatest differences were between sites located upstream and downstream from the RMPD: the upstream site was dominated by sandy substrate and less mesohabitat diversity than the downstream sites. Likely effects of the RMPD reservoir were observed in conductivity and turbidity lag-times between upstream and downstream locations. However, we note that assessing the effects of the dam directly would require comparisons to water quality data collected from sites separated by a similar distance but not separated by a reservoir (i.e., BACI design).

Fish and benthic macroinvertebrate communities showed increased richness downstream from the RMPD. We documented *P. rex* individuals in each study site, both upstream and downstream from the RMPD, demonstrating that dam removal will not create "new" populations of *P. rex* upstream, but may increase the abundance of *P. rex* in upstream areas. Mark-recapture methods for *P. rex* did not reveal movement, but permitted a more accurate estimate of total population size. Comparisons with a fish

community survey from 1978 revealed relatively stable fish communities through time. The greatest temporal changes were characterized by increases of *Moxostoma* and *Scartomyzon* suckers and Roanoke logperch in the study area.

Introduction

Franklin County, Virginia, in partnership with the Virginia Rural Area Conservation and Economic Restoration (VA RACER) initiative, the USFWS, and VDGIF, is working to remove the Rocky Mount Power Dam (RMPD) (Figure 1). The dam, measuring 240 feet wide and 25 feet high, was constructed in the early 1900's on the Pigg River to provide electrical power. The dam is now non-functional and provides few tangible benefits to Franklin County. Rather, concerns over the structural integrity of the dam and its negative ecological impacts have made its continued existence a liability.

Dam removal will provide several important economic and ecological benefits to Franklin County and the Pigg River, respectively. First, dam removal may serve to increase canoe and kayak recreational opportunities and tourism in Franklin County. Second, dam removal will alleviate residents' concerns about the dams' safety. Finally, the Pigg River contains important habitat for Roanoke logperch (*Percina rex*), a federally endangered species, and other fish species of conservation concern, including bigeye jumprock (*Moxostoma ariommum*) and orangefin madtom (*Noturus gilberti*). Long-term persistence of these populations will depend on habitat quality and connectivity among populations, and removal of the RMPD provides an important opportunity to improve habitat quality and connectivity for these species.

While dam removal may be an important ecological and economic restoration vehicle, it also presents potential short-term risks to downstream aquatic populations and habitat. Short-term ecological impacts of dam removal may include increased sediment load that may cause suffocation and physical damage to biota and habitats, and increased mobilization of contaminated material (if present) (Bednarek 2001). Ecological monitoring, therefore, is necessary to quantify the effects of dam removal on aquatic habitat and populations. For instance, the potential effects of sedimentation on downstream populations of Roanoke logperch (*P. rex*) must be considered. This information will contribute to understanding the ecological effects of dam removal, will assist civic leaders and regulators to make more informed decisions, and will help the affected human community to better understand the benefits of the project.

Here, we report on bioassessment surveys designed to establish baseline conditions in the study area prior to dam removal. Our assessments used established methods from the Environmental Protection Agency (EPA), including the Environmental Monitoring and Assessment Program (EMAP; Lazorchak et al. 1998) and the Rapid Bioassessment Protocols (RPB; Barbour et al. 1999). We also utilized methods from the Virginia Department of Environmental Quality (VDEQ), including their Stream Condition Index for Virginia (Burton and Gerritsen 2003).

Methods

Physical habitat

We established 3 sample sites to quantify physical habitat conditions upstream and downstream of the RMPD. One site was located upstream of the dam and two sites were located downstream from the dam (Figure 2). The upstream site (Site A) was located approximately 3.5 km upstream from the dam. This site location was selected because it includes the first riffle habitat in an upstream direction from the dam (i.e., much of the river between Site A and the dam was impounded and therefore not suitable for sampling). Two sites were located downstream from the dam (Sites B and C; Figure 2).

In each site, we used a stratified-random sampling protocol to quantify physical habitat within sites, including measures of substrate size and embeddedness, large woody debris, pool/riffle/run structure, thalweg depths (i.e., deepest point in a stream cross-section), and stream widths (Lazorchak et al. 1998). Reach lengths were delineated at 40x mean stream width. The total reach length was then divided in 10 equidistant "segments," and transects were delineated to bracket the upstream and downstream extent of each segment (total of 11 transects). Transects were approximately perpendicular to the direction of stream flow (Figure 3). Five sample "stations" where then located along each transect, including stations at the left-bank, left-center, center, right-center, and right-bank positions (Figure 3).

Substrate size and embeddedness were characterized at each sample station. To classify substrate size, a wading rod was dropped onto the substrate directly at each station. The size of the substrate directly underneath the wading rod was then classified as one of 6 categories: bedrock, boulder, cobble, course gravel, fine gravel, sand, or silt (following size definitions in Wolman [1954], Bain et al. [1985] and Platts et al. [1983]). In addition, substrate embeddedness was visually estimated within a region

approximately 0.25m^2, with the center at the sample station. Embeddedness ranged from 0-100%, where 0% embeddedness indicates loose substrates that are not "packed" with small particles and 100% embeddedness indicated a completely packed and consolidated substrate.

Woody debris, mesohabitat (pool/riffle/run structure), and thalweg depths were measured between transects (i.e., within segments). Pieces of large woody debris were counted within segments (i.e., between transects), including pieces of wood at least 10 cm in diameter at the small end and at least 1.5 m long (Lazorchak et al. 1998). Mesohabitat type and thalweg depths were measured at 10 evenly-spaced intervals between transects. Mesohabitats were classified as pools, riffles, and runs. Thalweg depths were measured with a wading rod. These methods were developed by the US Environmental Protection Agency (i.e., Environmental Monitoring and Assessment Program) for assessment of wadeable streams (Lazorchack et al. 1998) and have been applied elsewhere in the mid-Atlantic highlands region (McCormick et al. 2001). In addition, we conducted a 1.2-km thalweg survey to geomorphological structure throughout the study area downstream from the RMPD. Data for the thalweg survey were collected at 1-m sampling intervals.

Water quality

We quantified water temperature, dissolved oxygen, specific conductivity, pH, turbidity, and relative depth upstream and downstream of the RMPD. We deployed YSI 6920 V2 multi-parameter sondes in Site A (upstream) and Site C (downstream). Each sonde contained multiple sensors, including the YSI 6560 conductivity/temperature

sensor, the YSI 6561 pH sensor, the YSI 6136 optical turbidity sensor, and the YSI 6150 optical dissolved oxygen sensor.

The sondes were housed vertically in 4-inch PVC pipes and attached to submerged tree roots. The bottom 18 inches of both PVC pipes, where the sondes rested, had 1-inch holes drilled throughout and each hole was no more than 1 inch apart from another. This was done to ensure that adequate water flow would pass through the pipe and subsequently the probe sensors. Depth was measured at each site prior to deployment and it was ensured that the probes would rest in the middle of the water column throughout the assessment period (approximately 1-2 feet above the riverbed. Prior to initial deployment, the sondes were put through extensive testing, calibration, and QA/QC procedures following manufacture recommendations.

Calibration standards used in each calibration were YSI 1000 µs/cm solution for specific conductance, YSI 4.0 and 7.0 buffer for pH (2-point), 0.00 Nephelometric Turbidity Units (NTU) distilled water and YSI 123.0 NTU solution for turbidity (2 point). Dissolved oxygen was calibrated using the manufacturer recommended one-point "saturated air" method. Temperature was calibrated against a National Institute of Standards and Technology (NIST) certified liquid-in-glass thermometer.

Data were recorded at 15-minute intervals from May 2008 to April 2009. Approximately every 30 days the stations were visited to download recorded data, calibrate all sensors, and perform manufacturer-recommended QA/QC protocols. Nearly 33,000 water quality records were collected over the study period (Table 1), consisting of nearly 400,000 individual data points. Our sampling period captured a representative range of rainfall events, given that half of the sample months were above the 50% rank

for the 60-year period of record (1948-2008) at the NOAA weather gauge in Roanoke, Virginia (Appendix A). Sonde locations are shown in Appendices B and C.

Fish community

We sampled stream fish communities using electrofishing techniques in Sites A, B, and C. We collected fishes using a 1-pass sampling with 1 backpack electrofishing unit (Smith-Root LR-24). In riffle habitats, we shocked downstream into a 5-mm mesh seine, sampling all habitats within approximately 10 m of the seine (Lahey and Angermeier 2006). In run and pool habitats (i.e., where use of seines was impractical), we used dip nets to capture stunned fishes. All fishes were identified to species, counted, and released. Fish data were recorded within each sample segment (i.e., 10 even increments within each 40x-stream width site). Sampling was conducted in all sites during 2 periods: July 29-30 and November 7-8, 2008.

To estimate Roanoke logperch (*P. rex*) population size and to assess movement rates, we used visible implant elastomers (VIE) to mark *P. rex* individuals during the first sample period (Appendix D). VIE tags are small, biologically inert marks that are implanted subcutaneously and enable researchers to individually mark organisms and assess movement rates based on recapture locations. VIE tags have been demonstrated to not cause mortality in *P. caprodes* (J. Roberts, Virginia Tech, personal communication) and have been used successfully for Roanoke logperch in previous studies. Prior to VIE tag injections, fish were anesthetized in a dilute clove oil solution (Cuhna and Rosa 2006). Following injections, fish were placed in a freshwater tank and allowed to regain equilibrium before being released at the capture location. We recorded total length for each *P. rex* captured during both time periods.

Benthic macroinvertebrate community

We sampled benthic macroinvertebrates communities at Sites A, B, and C (Figure 2) using a stratified-random methodology. In each site, 500 micron D-frame net samples were taken from approximately 0.5 m^2 regions located at 1 m downstream from a randomly-selected point on each of the 11 transects. Cobbles or boulders lying 50 percent or more within the sampling area were held in front of the net, and scrubbed by hand to dislodge attached macroinvertebrates. The sampling area was then disturbed by kick-sampling for 30 seconds. Organic materials subsequently caught in the net were picked by-hand to remove invertebrates. If a particular transect was too deep to sample with a kick net, then a sample would be taken as close as possible to that transect. Transect samples where then pooled into a composite sample for each site. Composite samples were preserved in 95% ethanol for taxonomic identification. In the laboratory, organisms were identified to the family-level and counted.

Bioassessment metrics were then calculated from the Virginia Stream Condition Index score for each site (VSCI; Burton and Gerritsen 2003). VSCI is a multi-metric biological index calibrated from Virginia data for use in assessment of Virginia streams robust enough to be applied over a broad geographic region to allow a standardized comparison between sites both on and off the study area. Benthic macroinvertebrate community samples were collected during September 15, 2008 and April 4, 2009.

Results

Physical habitat

Sample sites exhibited important differences in physical habitat, with the greatest differences between the site upstream from the dam (Site A) and sites downstream from the dam (Sites B and C). Site A generally exhibited greater mean thalweg depths and narrower stream widths than Sites B and C (Table 2). Site A also exhibited more variability in thalweg depths than Sites B and C (Figure 4). Stream widths generally increased from upstream to downstream sites, but total volume (mean stream width x mean stream depth) did not exhibit a longitudinal trend (Table 2). A continuous survey of thalweg depths 1.2 km downstream from the dam revealed multiple riffle-run sequences and two large pools (i.e., depths > 1 m) (Figure 5).

Substrate sizes varied among sites. Site A was dominated by sand (>75% of area) with no particles larger than fine gravel observed (Figure 6). In contrast, sites B and C were characterized by the heterogeneity of substrate sizes, including silt, sand, fine gravels, coarse gravels, cobbles, boulders, and bedrock (Figure 6). Sites B and C were generally similar in substrate composition, but Site B contained a greater proportion of bedrock and boulders than Site C, and Site C contained a greater proportion of fine gravels and silt than Site B (Figure 6). Embeddedness (the proportion of large substrate particles that are "embedded" by fine substrate particles such as silt) was not estimated in Site A due to the nearly uniform sandy substrate. Embeddedness increased from Site B to Site C (Mann-Whitney U = 2013, $z = -2.99$, $p < 0.005$) (Figure 7).

Mesohabitats were dominated by runs throughout the study area (>75% of each sample site; Figure 8). Upstream from the dam, Site A contained no appreciable riffle

habitats and a relatively small proportion of pool habitat (Figure 8). Downstream from the dam, riffle habitats comprised approximately 10% of Sites B and C. Pool habitats occurred in both downstream sites, but occupied a relatively small proportion of the sample reaches (Figure 8). Concomitant with the decrease of pool habitat, large woody debris abundance also decreased from upstream to downstream sites (Figure 9). However, Sites B and C were similar in this regard, with an average of 1 piece of large woody debris per sample transect/site (Figure 9).

Water quality

Water quality parameters exhibited important temporal trends and differences between sites located upstream and downstream from the RMPD (Site A and C, respectively). Monthly-mean conductivity was generally higher in the summer and fall than in the spring or winter (Figure 10) and ranged from 96.5 to 132.3 µS/cm (Table 2). Site A exhibited higher monthly-mean peaks than Site C (Figure 10). However, Site C exhibited greater variability in 15-minute sample data than Site A in summer months (Figure 11). Conversely, conductivity measured in Site A was more variable than in Site C during October and November (Figure 11). At a 15-minute time-step, the downstream site tracked the upstream site more closely in winter (January) than in spring (April) or fall (October), where a 1-day travel time-lag was observed (Figure 12).

Turbidity exhibited significant seasonal and daily variation, as well as differences between the upstream and downstream sites. In both sites, monthly-mean turbidity exhibited lowest values in February and increased in a similar pattern to July (Figure 10). Site A showed the minimum value of 2.60 NTUs (February) and the maximum value of 29.08 NTUs (October) (Table 2). From August to November, monthly-mean turbidity

values increased substantially in the upstream site (Site A) but not in the downstream site (Site C). Moreover, variability among 15-minute samples was uniformly greater in Site A than Site C (Figure 12). In spring, summer, and fall, the upstream site exhibited higher daily maximum turbidity values than the downstream site (Figure 13). During the spring (April), peak turbidity levels in the upstream site exhibited approximately a 0.5-day travel lag-time before reaching the downstream site (Figure 13). However, in the winter (January), some peak-turbidity levels were higher in the downstream site than the upstream site (Figure 13).

Water temperatures exhibited temporal variation that was highly-consistent between upstream and downstream locations. In both sites, July exhibited the warmest stream temperatures (monthly-mean averages ~ 23°C) and January exhibited the coldest temperatures (monthly-mean averages ~ 3°C) (Table 2, Figure 10). However, mean-monthly downstream temperatures were marginally warmer than upstream temperatures (Table 2). Month-level variation in water temperature was highly consistent among downstream and upstream locations (Figure 11) but some differences were observed at a daily-level (Figure 14). During the winter (January), the downstream site generally exhibited higher daily-peak water temperatures than the upstream site, whereas spatial differences in the spring, summer, and fall were not as apparent (April, July, and October, respectively; Figure 14).

Relative depths showed an increasing trend from January to December in both sample sites (Figure 10). The maximum monthly-mean value was 0.70 m (Site A) and the minimum value was 0.07 m (Site C) (Table 2). The upstream and downstream site exhibited similar patterns of variation (i.e., standard deviation, Figure 11). Daily patterns of relative depth exhibited that Site A and C maintained similar patterns as observed at

across months (Figure 15). Time lags-between the upstream and downstream sites appeared to be smaller than for other water quality parameters. However, the magnitude of daily variation in relative depth was somewhat greater in Site A than Site C (Figure 15).

Dissolved oxygen levels exhibited seasonal and diurnal patterns as well as differences between the upstream and downstream sites. Monthly-mean dissolved oxygen levels were generally lowest in the summer (~ 7 mg/L) and highest in the winter (~ 12 mg/L) (Table 2, Figure 10). Variation among 15-minute records was highly-consistent among months for upstream and downstream sites (Figure 11). However, diurnal fluxes in dissolved oxygen levels exhibited important seasonal differences between upstream and downstream locations. During the spring (April), daily peak-levels of dissolved oxygen were greater in Site A whereas daily minima were comparable among sites (Figure 16). However, during the summer and fall (July and October), daily peak-levels were similar among sites, but Site A typically exhibited lower daily minima than Site C (Figure 16). In contrast, dissolved oxygen levels were generally higher during the winter (January) and did not exhibit such diurnal variation (Figure 16).

Neither the upstream or downstream site exhibited significant seasonal variation in monthly-mean pH (Figure 10), with both sites ranging from approximately 7.3 to 7.5 (Table 2). Although the upstream site was marginally lower pH (less alkalinity) than the upstream site, variation in sampled pH showed significant overlap between sites (i.e., ±1 standard deviation from the mean; Figure 12). However, we detected important seasonal and diurnal patterns at 15-minute time-steps. Spring samples (April) exhibited the greatest diurnal fluctuations (Figure 17) (i.e., from 7.2 to 7.8) and this pattern was observed in both Site A and Site C. During the summer and fall (July and October),

diurnal fluxes were reduced from spring levels and the downstream site exhibited consistently greater pH (more alkalinity) than the upstream site (Figure 12). In the winter (January), diurnal fluxes in pH exhibited the least variation of any season, and the greatest differences among sites: all samples from Site C were greater than Site A (Figure 17).

Fish community

Stream fish assemblages exhibited seasonal variation and differences among sites. A total of 3244 fish were captured, representing 5 families and 37 species (Table 3). Within sites, total abundance ranged from 1019 individuals (Site A, time 1) to 378 individuals (Site A, time 2). Cyprinid and percid fishes dominated the total abundance of the observed fish community, comprising 39% and 38% of the total catch, respectively (Table 3). Catostomids, centrarchids, and ictalurids comprised 14%, 7% and 2% of the total catch, respectively (Table 3). Observed species richness varied from 32 species (Site B, sample 1) to 23 species (Site A, sample 2) (Table 3; Figure 18). Summer samples (July, sample 1) showed higher species richness than winter samples (November, sample 2) and this effect was constant among sites (Figure 18).

In Site A (upstream from the RMPD), total abundances were dominated by glassy darter (*Etheostoma vitreum*) during the July sample and bluehead chub (*Nocomis leptocephalus*) during the November sample (i.e., sample 1 and 2, respectively; Figure 19). This effect was due to the drop in *E. vitreum* abundances between the first and second sample (Table 3). Relative abundances of rosefin shiner (*Lythrurus ardens*), Roanoke hogsucker (*Hypentelium roanoakense*), and white shiner (*Luxilus albeolus*)

Johnny darter (*Etheostoma nigrum*) also decreased in total abundance from the first to second sample, but the rate of decrease was smaller than observed for most other taxa (Table 3). We observed three species of catosomids (bigeye jumprock [*Moxostoma ariommum*], blacktip jumprock [*Moxostoma cervinum*], and golden redhorse [*Moxostoma erythrurum*]) during the summer sample but not during the winter sample (Table 3). We also observed 5 subadult Roanoke logperch (*Percina rex*) at this site during the summer but *P. rex* was not detected in the winter sample (see additional *P. rex* results below) (Table 3).

The sample site immediately downstream from the RMPD (Site B) showed the highest species richness in the study area (Figure 18). Total abundances in this site were dominated by 5 species which were shared among summer and winter sampling periods: bigeye jumprock (*M. ariommum*), bluehead chub (*N. leptocephalus*), blacktip jumprock (*M. cervinum*), fantail darter (*Etheostoma flabellare*), and Roanoke darter (*Percina roanoka*) (Figure 19). Margined madtom (*Noturus insignis*) and rock bass (*Ambloplites rupestris*) contributed to greater than 10% of the total catch during the summer sample but not the winter sample (Figure 19). (Most *Ambloplites* individuals exhibited black anal fin margins, and therefore are categorized as *A. rupestris*, see Discussion.) Conversely, stoneroller (*Campostoma anomalum*), satinfin shiner (*Cyprinella analostana*), and riverweed darter (*Etheostoma podostemone*) were observed in higher abundances during winter sample than the summer sample (Table 3, Figure 19).

In Site C, we detected 30 species during the summer sample and 25 species during the winter sample (Table 3, Figure 18). Total abundances were dominated by 4 species that occurred both sampling periods: bluehead chub (*N. leptocephalus*), fantail darter (*E. flabellare*), Roanoke darter (*P. roanoka*), and satinfin shiner (*C. analostana*).

We observed lower abundances of margined madtom (*N. insignis*), riverweed darter (*E. podostemone*), and white shiner (*L. albeolus*) in the winter sample than in the summer sample (Table 3, Figure 19). Conversely, stoneroller (*C. anomalum*) was detected at higher abundance in the winter than in the summer sample in this site (Table 3). Overall, fish communities in downstream sites (Sites B and C) were more similar to one another than to the site located upstream from the RMPD (Site A).

We observed 34 individual Roanoke logperch (*P. rex*) during this course of this study (Table 3). Abundances were lowest in Site A, where 5 subadult individuals were captured in a single seine haul during the summer sample. No adult *P. rex* were observed in Site A during either the summer or winter sample (Table 3). In Site B, 8 individuals were observed during the summer sample and 9 individuals were observed in the winter sample (1 of which was a recaptured individual, see below). In Site C, 7 individuals were observed during the summer sample and 6 individuals were observed during the winter sample (none of which were recaptured individuals, see below). *P. rex* were significantly larger in sites B and C than in the upstream sites (Figure 20). The smallest *P. rex* individual observed within the downstream sites measured 88 mm TL (Site C, sample 2).

Our application of visible implant elastomer (VIE) marks in *P. rex* during the first sample period (July) allows us to better estimate total population size for this species. Chapman's (1951) modification to the Lincoln-Peterson mark-recapture estimator accounts for some bias due to low sample sizes:

$$N = \frac{(M+1)(C+1)}{(R+1)} - 1$$

where N = estimated population size; M = number of fish marked and released during the first sample; C = number of fish captured and examined for marks during the second sample; R = the number of recaptures (marked fish observed during the second sample). In our surveys, we marked 20 *P. rex* individuals during the first sample (M). We captured and inspected 15 *P. rex* individuals during the second sample (C). Of these fish, only 1 was a recaptured individual (R). We therefore estimate that the total population size for *P. rex* in the study area is closer to 167 than to the count of individuals observed directly during a single sampling period (i.e., 20 individuals in time 1 or 15 individuals in time 2). The recaptured individual was observed in the sample transect within which it was marked and therefore no movement was detected.

Benthic macroinvertebrate community

Benthic macroinvertebrate communities varied among sites and among seasons (Fall 2008 and Spring 2009). Total observed richness ranged from 21 families (Site B, spring sample) to 12 families (Site A, spring sample) (Table 5; Figure 21). Seasonality did not exert a uniform effect on total richness, given that more taxa were detected in spring samples within Site B but more taxa were detected in fall samples in Site C (Table 5). Family richness within the orders Ephemeroptera, Plecoptera, or Tricoptera (i.e., metric "EPT", see Table 4) ranged from 6 taxa (Site A) to 8 taxa (Site B, spring sample) (Table 5). The percent of individuals as Ephemeroptera was greatest in Site A (64.8%, Table 5) and least in site C (30.0%). The percent of individuals as "scrapers" (i.e., feeding by scraping periphyton from substrate) ranged between 25 and 28% for all samples except Site B during the spring (18%). The percent of individuals as chironomid

larvae (i.e., non-biting midges in the order Diptera) ranged from 9.1% in Site B in the fall sample to 22.8% in the spring sample.

Taxonomic evenness within benthic macroinvertebrate assemblages varied among sample sites and seasons: Site A exhibited the highest proportion of the 2 most-dominant taxa (i.e., metric "%2Dom", see Table 4) at 50% (Table 5). Site B exhibited the greatest taxonomic evenness (i.e., %2Dom = 26.5%, Table 5). Site B also showed the greatest change in evenness among seasons, ranging from 26.5% in the fall sample to 40.1% in the spring sample (Table 5). Hilsenhoff Biotic Index scores (HBI, see Table 4) were similar among sites and seasons, ranging from 4.2 to 4.8 (Site A and Site C, respectively; Table 5). Following Burton and Gerritsen (2003), we calculated the Virginia Stream Condition Index (VSCI), a multi-metric index of stream quality, from the metrics presented in Table 4 and data presented in Table 5. Site A showed the lowest VSCI score (59.2) and Site C showed the highest VSCI score (70.6, fall sample). However, during the spring sample, Site B showed a marginally greater VSCI score than Site C (i.e., 67.3 versus 64.7, respectively; Table 5).

Discussion and conclusions

The goal of this study was to characterize the biological and physical condition of the Pigg River upstream and downstream from the RMPD. Our surveys revealed spatial and temporal trends in physical habitat, water quality, stream fish community structure, and benthic macroinvertebrate community composition. These data provide new information about the current condition of the Pigg River in the study area as well as the

range of natural variation within the study site. Moreover, our results provide some inferences about the likely effects of the RMPD based on comparisons of water quality lag-times from upstream and downstream locations. However, the most revealing applications of this dataset will probably be for evaluating biotic and abiotic changes during and after dam removal.

Our surveys demonstrated important differences in physical habitat between sites upstream and downstream from the RMPD. The upstream site (Site A) was dominated by sand substrate whereas downstream sites (B and C) contained a greater diversity of substrate sizes (Figure 6). Substrate conditions observed downstream from the dam are similar to substrate size composition observed in the mainstem of the Roanoke River (Vadas 1994; Stancil 2000; Rosenberger and Angermeier 2003). Rosenberger and Angermeier (2003) also reported an embeddeness profile for the mainstem of the Roanoke River that is similar to observed embeddedness in Sites B and C, despite the fact that the channel was wider for their study than for our Pigg River sites.

Physical habitat conditions in Site A are clearly influenced by extensive bank erosion observed at this site (Appendix E), but it is possible that the RMPD impoundment may also affect conditions in Site A. However, we cannot predict the effects of dam removal on conditions in Site A because the spatial extent of upstream impoundment from the RMPD has not been established. We recommend additional research to investigate the possible extent of geomorphological influences of dam removal on the Pigg River upstream at least to the Route 220 bridge (approximate upstream extent of Site A).

Our water quality data revealed important differences between upstream and downstream sites, and suggests some influences of the RMPD reservoir. Observed

conductivity was somewhat higher upstream of the dam than downstream (Table 2; Figure 10) but monthly-mean values did not suggest that conductivity was a likely stressor for biota in either site (i.e., "low conductivity" defined as < 500 µS/cm; Pond et al. 2008). However, monthly spikes in conductivity were observed in the upstream site but not the downstream site, suggesting that the reservoir may buffer the downstream delivery of ions and that this effect may be most important in summer and fall months. Observations at 15-minute intervals in October revealed a time-lag in downstream conductivity data and lowered peak levels (Figure 12). However, to quantify the effect of the RMPD directly, we would need to compare our observations against data from sites separated by a similar distance but without a reservoir in between them (i.e., BACI design).

Observed turbidity values exhibited a seasonal pattern observed in other Appalachian streams (Minckley 1962, as cited in Hynes 1972), reflecting variation in rainfall-runoff patterns. Similar patterns of downstream-buffering were observed for measures of conductivity: upstream peaks in monthly-mean turbidity levels were not observed in the downstream sonde (Site C) (Figure 10). This buffering effect was also observed for specific rainfall events in spring, summer, and fall months (Figure 13). The effect of the RMPD reservoir is further illustrated by the fact that relative depths in the downstream site may decrease after a rainfall event, but turbidity may continue to increase over that time period (e.g., April 22-22, 2008; Figures 13 and 15). Overall, observed turbidity values were consistently higher than would be expected for a mountain stream (approximately 1 NTU) and higher than drinking water standards (i.e., 0.5 NTUs) but not higher than would be expected for runoff events in rivers of this region.

Water temperatures followed expected seasonal patterns, showing maximum mean-monthly temperatures near 22°C and minimum temperatures near 3°C (Figure 10). In contrast to observations for conductivity and turbidity, time-lags between upstream and downstream locations were not observed, suggesting that localized factors may have a greater effect on water temperature than the RMPD reservoir. For example, local riparian cover and groundwater upwelling flows have important effects on stream water temperatures (Webb et al. 2008). However, some evidence of warming from the reservoir was observed in the downstream site during January (Figure 14), and the magnitude of this effect was approximately 1°C. We therefore do not expect the removal of the RMPD to dramatically affect water temperatures in the study area.

Dissolved oxygen is unlikely to present a stressor for biota in the study area, as observed levels were consistently above the Virginia's water quality criterion (4 mg/L). However, diurnal variation in dissolved oxygen was not trivial, and accounted for swings of more than 2 mg/L in some cases (Figure 16). Differences in water temperatures and microbial respiration rates between upstream and downstream locations may account for some of the observed differences in dissolved oxygen levels between these sites. Moreover, diurnal swings in dissolved oxygen content cannot be attributed simply to extensive algal growth or submerged aquatic vegetation in either the upstream or downstream sonde location. The diminished daily-low values for dissolved oxygen in the upstream site suggest that it may be influenced by greater rates of microbial respiration than the downstream site, and these differences cannot be attributed to water temperature effects alone. Diurnal variation in pH (Figure 17) also suggests localized differences in stream metabolism within the study area.

Our fish surveys provide some of the first quantitative community data within the study area. We observed between 23 and 32 species per site (Table 3), similar to species richness reported from other similarly-sized areas within the Pigg River basin (Lahey and Angermeier 2006) and the Roanoke River basin as a whole (Stancil 2000). In Site B, our observed species richness (32 in sample #1 and 28 in sample #2) was higher than that reported by James (1979) in his survey of the same location (17 species reported). However, this may reflect differences in sampling efficiency between studies (i.e., backpack electrofishing in 2008 versus seining-only in 1978) or in the length of stream sampled (i.e., 40x mean stream width in 2008 versus an unspecified sample length in 1978). It is also possible that the fish fauna observed in 1978 was recovering from an aqueous copper spill upstream from the study area in 1975 (James 1979). Notwithstanding these differences, 15 of the 17 species observed in 1978 were detected in our surveys of this site (i.e., except *A. cavifrons*, see below; and brown bullhead [*Ictalurus nebulosus*], although *I. nebulosus* was detected in our study in Site C). In contrast, the most significant changes over time were the addition of *Moxostoma* and *Scartomyzon* suckers and Roanoke logperch (*P. rex*) to the fish community in this site (see below).

Several of the species we observed are recognized as species of concern by the Virginia Department of Game and Inland Fisheries (VDGIF). Bigeye jumprock (*M. ariommum*) is classified as a Tier III conservation species, meaning that "populations of this species are in decline, have declined to low levels, or are restricted in range" and that "management action is needed to stabilize or increase these populations" (VDGIF 2005). We observed relatively high numbers of bigeye jumprock (*M. ariommum*) downstream from the RMPD (i.e., second most-abundant species in Site B, sample 1; Table 3). We

also observed this species upstream from the dam, albeit at relatively low numbers (Table 3). In contrast, Stancil (2000) reported relatively low numbers of *M. ariommum* and regional rarity in his survey of tributaries in the Roanoke River basin. James (1979) and Lahey and Angermeier (2006) also reported relatively low numbers of this species in other sections of the Pigg River basin. It is therefore possible that the Pigg River study area currently provides important habitat for this species of conservation concern.

Riverweed darter (*E. podostemone*) is recognized as a Tier IV species by VDGIF, meaning that this species "may be rare in parts of its range" and that "long term planning is necessary to stabilize or increase [these] populations" (VDGIF 2005). We observed *E. podostemone* in all sites, in all sample periods (Table 3), suggesting that this species can tolerate a wide variety of habitat conditions (Jenkins and Burkhead 1994). Moreover, the observed occurrences of this species may indicate that it is thriving in the study area. Although this species is endemic to the Roanoke River basin (and hence regionally rare), its status in the Pigg River study area is probably secure.

We did not detect two species that are recognized by VDGIF as conservation concerns. Orangefin madtom (*Noturus gilberti*), a state-threatened species, was not observed in any samples. Further, we did not identify Roanoke bass (*A. cavifrons*) in the study area because most individual *Ambloplites* exhibited a distinct black margin on the anal fin, a trait strongly suggesting rock bass (*A. rupestris*) (Jenkins and Burkhead 1994). Moreover, when black anal fin margins were weakly developed, cheek scale patterns suggested *A. rupestris*. However, we recognize that additional quantitative meristic and morphometric comparisons may be necessary to identify remnant *A. cavifrons* individuals in the study area. Moreover, we recognize that *cavifrons* x *rupestris* hybridization is

possible. In this case, genetic analyses would probably be necessary to assign *Ambloplites* individuals to *cavifrons* with certainty.

Roanoke logperch (*P. rex*) is a federally endangered fish species, and is recognized as a Tier I species by VDGIF (2005). In our surveys, we observed 34 individual Roanoke logperch during this course of this study (Table 3). Mark-recapture methods did not reveal movement (1 recapture at the marking-location transect) but do allow us to more precisely estimate the population size in habiting the study area. We estimate that the population size inhabiting the study area is closer to 167 (modified Lincoln-Peterson method) than to the sum of observed individuals. However, the low recapture rate could also be explained by inefficient sampling methods. Prior studies have used backpack shocking into seines to capture *P. rex* (e.g., Lahey and Angermeier 2006) but we used backpack electrofishing with a combination of dip nets and seines, depending on mesohabitat conditions. As expected, we found that seines alone were not able to capture fishes in deeper, boulder-type substrates and therefore we used dip nets in those conditions. We note that several *P. rex* were captured in dip net samples and thus would have been missed if we had not employed multiple methods. However, we recognize that additional sampling effort (i.e., increased number of backpack electrofishing units) may have improved overall sampling efficiency. Laboratory studies have shown very low mortality rates from VIE tags in darters (Coombs and Wilson 2008), and therefore we do not believe that this is a likely explanation for our low recapture rates. We emphasize that the greatest utility of our *P. rex* assessment will be to assess changes over time (i.e., post-dam removal).

Our surveys documented *P. rex* in the stream above the RMPD. This was an unexpected find, given the habitat preferences for this species (Rosenberger and

Angermeier 2003) and the 1975 industrial spill that affected this site (from Furnace Creek, see James 1979). Because upstream emigration is blocked by the RMPD, our results suggest that either (1) *P. rex* were not fully extirpated by the 1975 spill and/or (2) *P. rex* have recolonized the study area from locations upstream of the Furnace Creek confluence. In support of the recolonization hypothesis, James (1979) documented 1 *P. rex* individual near the Furnace Creek confluence. Although we cannot fully evaluate the relative support for these temporal change scenarios, we note that the practical implications are the same: dam removal will not establish "new" populations of *P. rex* but may increase population sizes upstream from the dam. It is also possible that dam removal could provide a "rescue effect" (Brown and Kodric-Brown 1977) for upstream *P. rex* populations that may otherwise have a high probability of extirpation.

We recommend population genetic analyses to evaluate the relation between populations upstream from the dam and those downstream. If populations downstream from the RMPD recolonized the study area from Big Chestnut Creek (i.e., known locations of *P. rex* in areas not affected by the 1975 spill), then the upstream (i.e., remnant) *P. rex* population may exhibit a distinct genetic profile. On the other hand, if the upstream populations contributed individuals to the downstream sites (i.e., survived the fall over the dam), then the population genetic profiles of upstream and downstream samples would be relatively similar.

The observed *P. rex* abundances in Sites B and C correspond with published literature on *P. rex* habitat use. Rosenberger and Angermeier (2003) concluded that Roanoke logperch are "substrate specialists and mesohabitat generalists," with preferred substrate sizes as gravels but occurring over a wide range of flow velocities. However, we note that the highest abundances of *P. rex* in our study were observed in Site B, but

that Site C contained more gravel substrate. Rosenberger and Angermeier (2003) demonstrated that *P. rex* use habitat differently in the Roanoke River and the Nottoway River, so it is also possible that differences in habitat availability in the Pigg River would also yield distinct habitat use patterns.

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Table 1. A comparison of sample site sizes. Site locations area mapped in Figure 2.

Site	Variable	Mean	Standard deviation
A	Depth (cm)	59.7	20.3
B		41.2	13.0
C		41.6	17.2
A	Width (m)	11.9	2.0
B		15.0	3.7
C		17.5	2.4

Table 2. Water quality summary data by month. Standard deviations from 15-minute interval data are presented in parentheses. ND = no data.

		Conductivity (µS/cm)		Dissolved oxygen (mg/L)		pH		Relative depth (m)		Temperature (C)		Turbidity (NTUs)	
	n	Up	Down	Up	Down	Up	Down	Up	Down	Up	Down	Up	Down
May 2008	2669	ND	ND	8.66 (0.504)	8.75 (0.413)	7.43 (0.077)	7.42 (0.076)	0.20 (0.086)	0.07 (0.075)	16.96 (1.604)	17.12 (1.557)	14.33 (18.277)	17.43 (11.849)
June 2008	2857	125.30 (10.543)	119.08 (13.277)	7.16 (0.504)	7.24 (0.404)	7.25 (0.142)	7.46 (0.072)	0.27 (0.058)	0.17 (0.046)	22.45 (1.911)	22.71 (1.874)	16.41 (24.954)	14.12 (7.950)
July 2008	1976	138.8 (22.272)	ND	7.17 (0.747)	7.32 (0.469)	7.37 (0.103)	7.50 (0.077)	0.30 (0.071)	0.22 (0.056)	23.04 (1.775)	23.23 (1.572)	12.65 (12.44)	12.69 (6.235)
August 2008	2961	183.00 (40.670)	132.27 (81.587)	6.80 (0.842)	7.29 (0.655)	7.28 (0.155)	7.47 (0.133)	0.28 (0.263)	0.21 (0.172)	21.86 (1.949)	21.98 (1.812)	32.21 (86.248)	15.34 (36.485)
September 2008	2869	123.49 (9.781)	124.34 (10.301)	7.88 (0.590)	8.05 (0.474)	7.40 (0.102)	7.44 (0.066)	0.40 (0.081)	0.26 (0.063)	19.68 (2.271)	19.98 (2.166)	17.61 (25.705)	12.70 (8.206)
October 2008	2965	132.58 (25.794)	122.16 (8.321)	8.93 (0.694)	9.52 (0.726)	7.32 (0.157)	7.49 (0.063)	0.39 (0.072)	0.20 (0.062)	12.92 (3.115)	13.24 (2.944)	29.08 (38.036)	7.14 (3.857)
November 2008	2675	187.41 (96.574)	127.67 (14.056)	11.42 (1.112)	10.14 (0.705)	7.13 (0.258)	7.17 (0.328)	0.30 (0.125)	0.20 (0.090)	7.40 (3.678)	7.96 (2.974)	13.66 (25.519)	5.95 (9.370)
December 2008	1278	111.65 (5.062)	112.04 (5.061)	11.68 (0.889)	11.77 (0.790)	7.33 (0.036)	7.49 (0.050)	0.47 (0.097)	0.33 (0.094)	5.91 (2.612)	5.88 (2.403)	7.40 (8.113)	8.12 (5.945)
January 2009	2961	106.77 (9.734)	108.22 (11.217)	12.55 (0.621)	12.63 (0.595)	7.24 (0.107)	7.41 (0.120)	0.58 (0.305)	0.40 (0.209)	2.71 (2.052)	2.69 (1.860)	20.38 (93.208)	18.82 (78.921)
February 2009	2673	103.36 (6.685)	103.08 (5.757)	11.84 (0.935)	12.06 (0.872)	7.41 (0.103)	7.53 (0.118)	0.57 (0.062)	0.43 (0.063)	4.68 (2.785)	4.70 (2.523)	2.60 (2.867)	3.35 (1.628)
March 2009	2841	101.28 (8.028)	101.81 (6.875)	11.15 (0.871)	11.02 (0.900)	7.43 (0.175)	7.54 (0.173)	0.70 (0.122)	0.52 (0.088)	8.45 (3.219)	8.55 (3.139)	10.42 (11.377)	9.90 (8.338)
April 2009	3174	96.50 (4.720)	98.47 (4.471)	9.86 (0.919)	9.51 (0.894)	7.35 (0.164)	7.46 (0.170)	0.63 (0.122)	0.51 (0.089)	14.24 (3.193)	14.34 (3.159)	13.18 (36.809)	12.52 (18.303)

Table 3. Relative abundances of fish species observed by site and sample period.

Family	Species	Code	Site A Time 1	Site A Time 2	Site B Time 1	Site B Time 2	Site C Time 1	Site C Time 2	Total
Catostomidae	Bigeye jumprock	BEJ	2	-	45	42	11	6	**106**
	Blacktip jumprock	BTJ	1	-	35	47	6	20	**109**
	Golden redhorse	GDR	2	-	4	2	3	5	**16**
	Northern hogsucker	NHS	1	8	-	1	4	-	**14**
	Roanoke hogsucker	ROH	76	44	8	18	-	18	**164**
	Shorthead redhorse	SHR	-	-	2	-	-	-	**2**
	White sucker	WIS	17	13	8	13	1	6	**58**
Centrarchidae	Bluegill	BLG	-	-	-	-	1	-	**1**
	Green sunfish	GSF	-	2	3	-	-	-	**5**
	Largemouth bass	LMB	3	1	1	-	-	-	**5**
	Pumpkinseed	PKS	6	4	5	4	5	4	**28**
	Redbreast sunfish	RBS	32	2	6	15	6	8	**69**
	Rock bass	RKB	8	4	33	7	18	10	**80**
	Smallmouth bass	SMB	-	-	2	4	1	1	**8**
Cyprinidae	Blacknose dace	BND	1	-	6	3	3	-	**13**
	Bluehead chub	BHC	210	115	28	48	58	55	**514**
	Bluntnose minnow	BNM	-	1	-	-	-	-	**1**
	Bull chub	BUC	-	-	7	10	1	-	**18**
	Cresent shiner	CRS	24	4	6	2	2	-	**38**
	Cutlips minnow	CLM	-	-	1	1	-	-	**2**
	Mountain redbelly dace	MRD	2	-	1	-	4	1	**8**
	Rosefin shiner	RFS	96	32	7	22	12	16	**185**
	Rosyside dace	RSD	4	3	7	5	5	7	**31**
	Satinfin shiner	SFS	37	6	21	26	22	26	**138**
	Spottail shiner	SPS	-	-	1	-	1	-	**2**
	Stoneroller	STR	2	3	18	26	5	34	**88**
	Swallowtail shiner	STS	1	-	1	5	2	1	**10**
	White shiner	WHS	109	42	17	4	34	24	**230**
Ictaluridae	Brown bullhead	BBH	-	-	-	-	2	-	**2**
	Margined madtom	MMT	-	1	33	7	28	2	**71**
Percidae	Fantail darter	FTD	6	4	35	49	63	112	**269**
	Glassy darter	GSD	287	48	-	-	-	1	**336**
	Johnny darter	JND	35	19	8	15	1	23	**101**
	Riverweed	RWD	9	8	19	25	24	7	**92**

	darter							
	Roanoke darter	ROD	16	5	57	73	99	78	**328**
	Roanoke logperch	RLP	5	-	8	9	7	6	**35**
	Shield darter	SHD	27	9	2	5	10	14	**67**
	Total		**1019**	**378**	**435**	**488**	**439**	**485**	**3244**

Table 4. Benthic macroinvertebrate metrics and expected responses to degradation. These metrics were used to calculate the Virginia Stream Condition Index (VSCI, Burton and Gerritsen 2003) in Table 4.

Code	**Metric**	**Expected response to degradation**
Richness	Total number of taxa identified at the family-level	Decrease
EPT taxa	Number of taxa in the orders Ephemeroptera, Plecoptera, or Trichoptera	Decrease
% Ephem.	Percent of individuals as Ephemeroptera	Decrease
% PT. – hydro.	Percent of individuals as Plecoptera plus Trichoptera, minus hydropsychids (net-spinning caddisflies)	Decrease
% Scrapers	Percent of individuals as scrapers	Decrease
% Chiro.	Percent of individuals as chironomids	Increase
% 2Dom	Percent of individuals in the top 2 dominant taxa	Increase
HBI	Hilsenhoff Biotic Index (family-level)	Increase

Table 5. Benthic macroinvertebrate data and Virginia Stream Condition Index (VSCI) scores. Fall samples were taken on 9/15/08; Spring samples were taken on 4/4/09. Metric codes are given in Table 3.

	Fall 2008		**Spring 2009**		
Metrics	**Site B**	**Site C**	**Site A**	**Site B**	**Site C**
Richness	17	18	12	21	17
EPT taxa	7	7	6	8	6
% Ephem.	38.0	45.5	64.8	37.7	30.0
% PT – hydro.	10.7	9.8	9.3	10.1	11.1
% Scrapers	25.6	25.8	25.9	18.0	28.1
% Chiro.	9.1	11.4	9.3	22.8	19.0
% 2Dom	26.5	32.6	50.0	40.1	35.3
HBI	4.6	4.4	4.2	4.5	4.8
VSCI Score	**69.2**	**70.6**	**59.2**	**67.3**	**64.7**

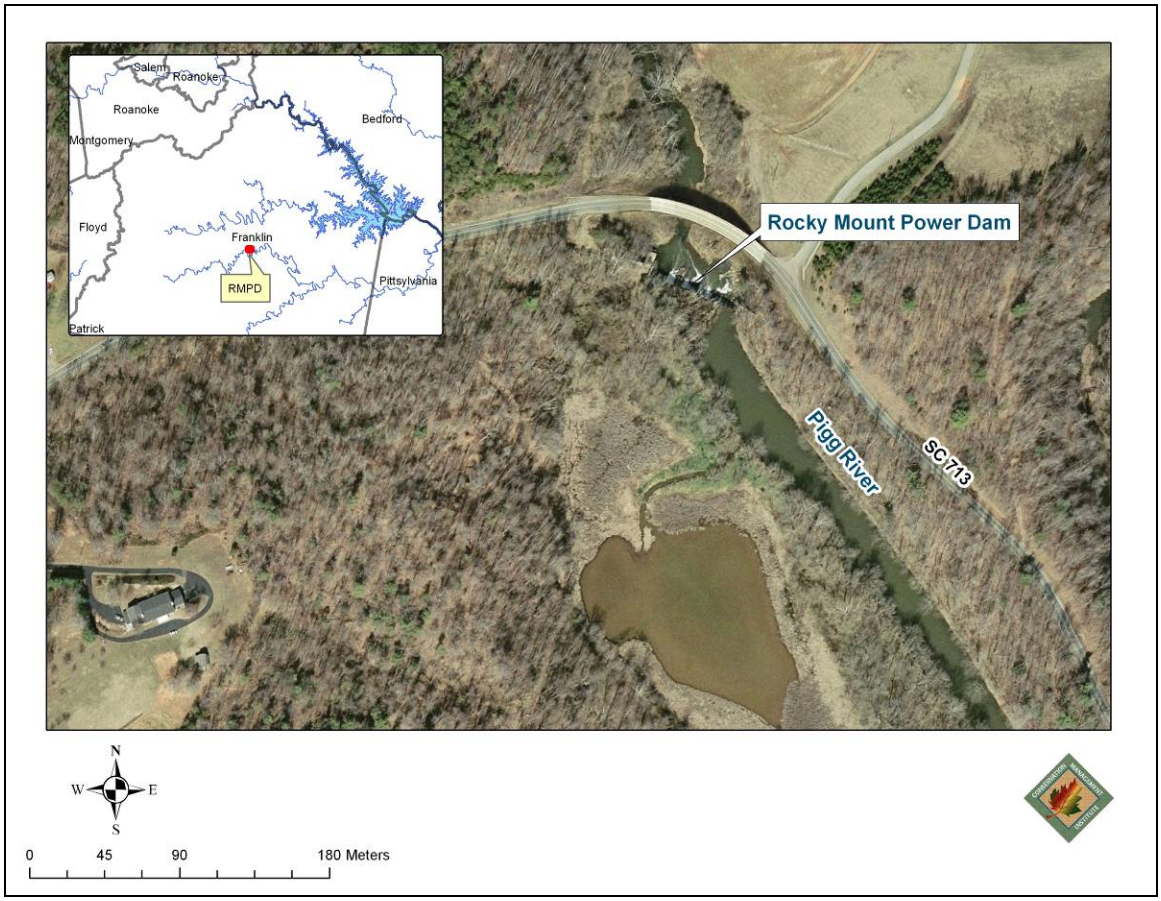

Figure 1. Map of the Pigg River study area, Frankin County, Virginia.

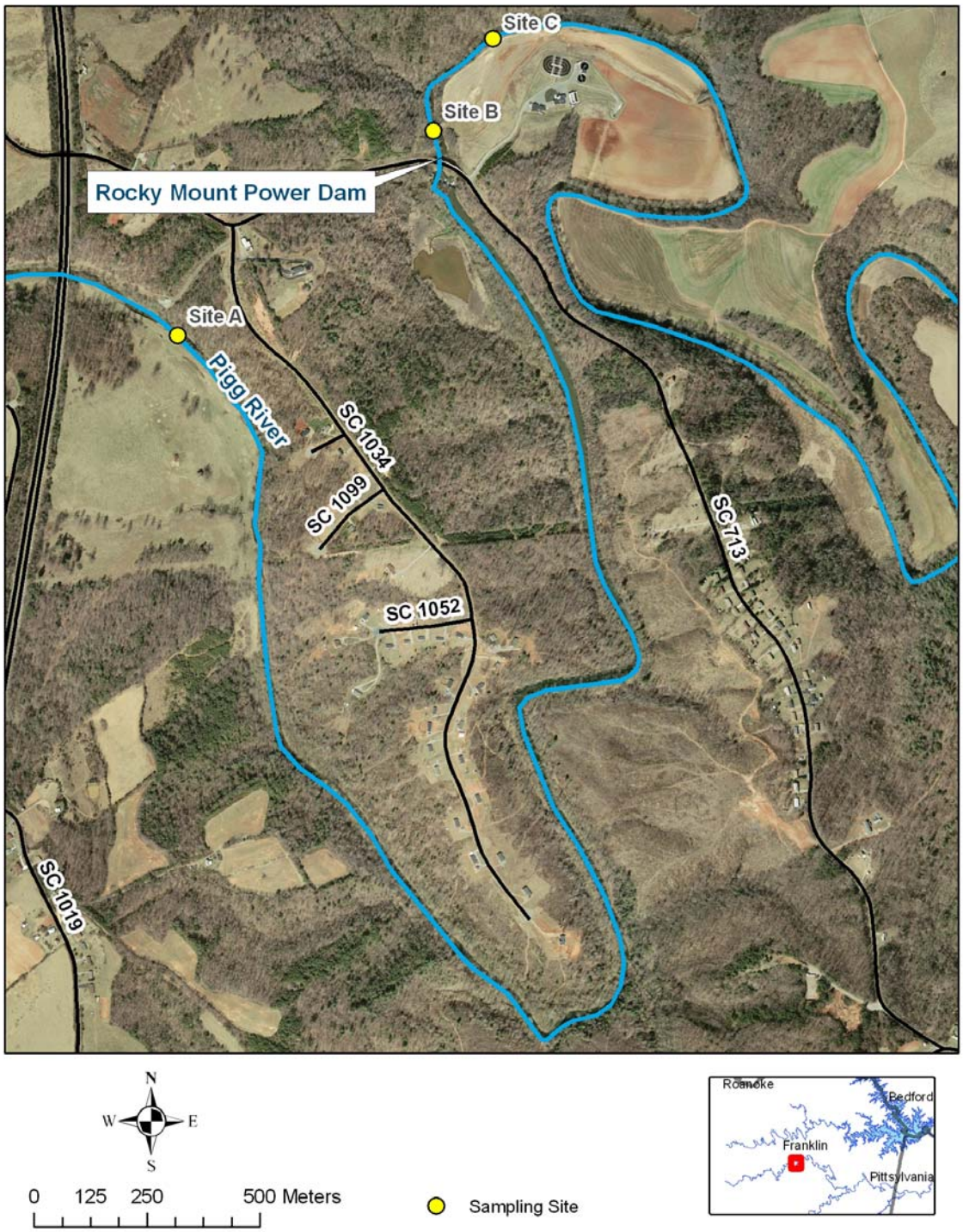

Figure 2. Map of the Pigg River study sites. Site A was upstream of the dam; Sites B and C were downstream from the dam.

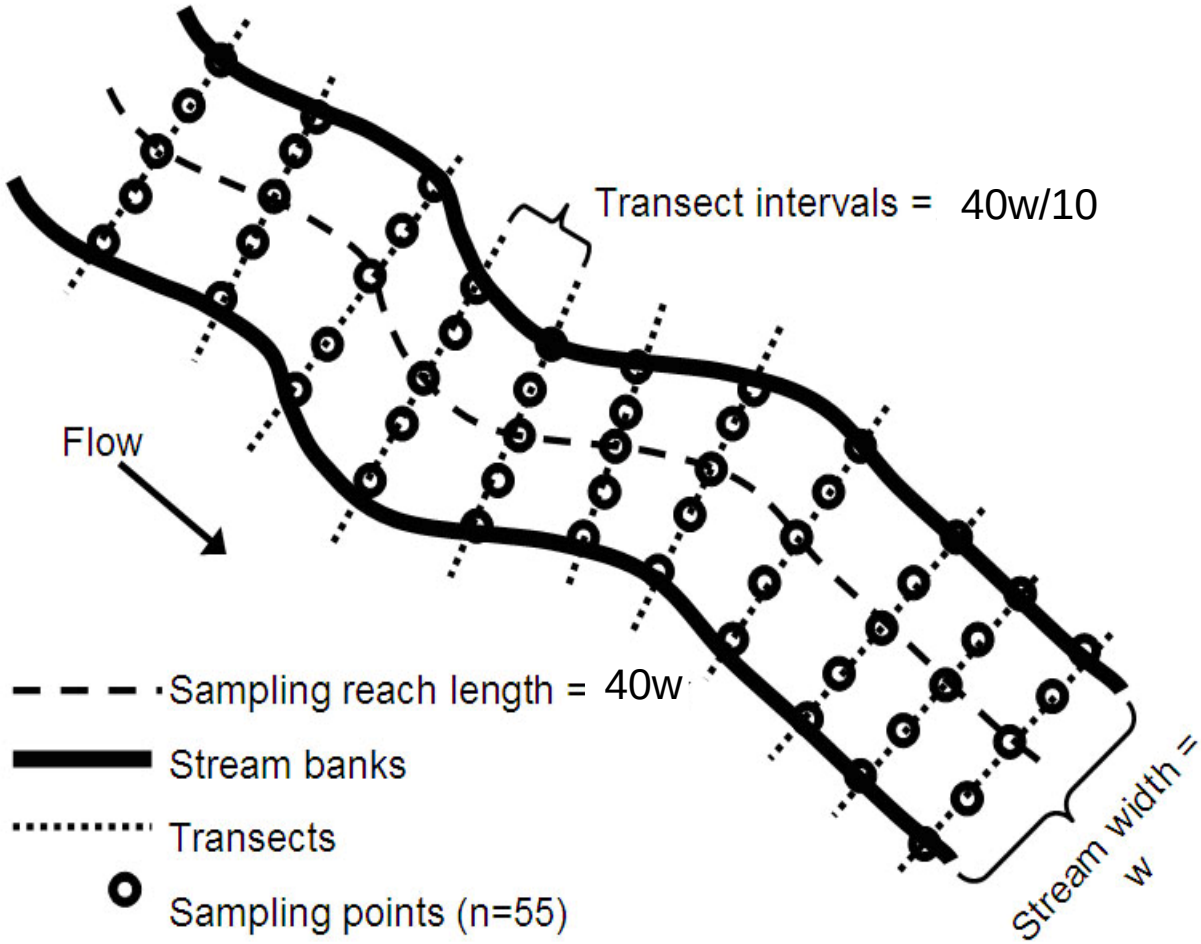

Figure 3. Physical habitat assessment protocol from USEPA Environmental Monitoring and Assessment Program (EMAP) methods (Lazorchak et al. 1998).

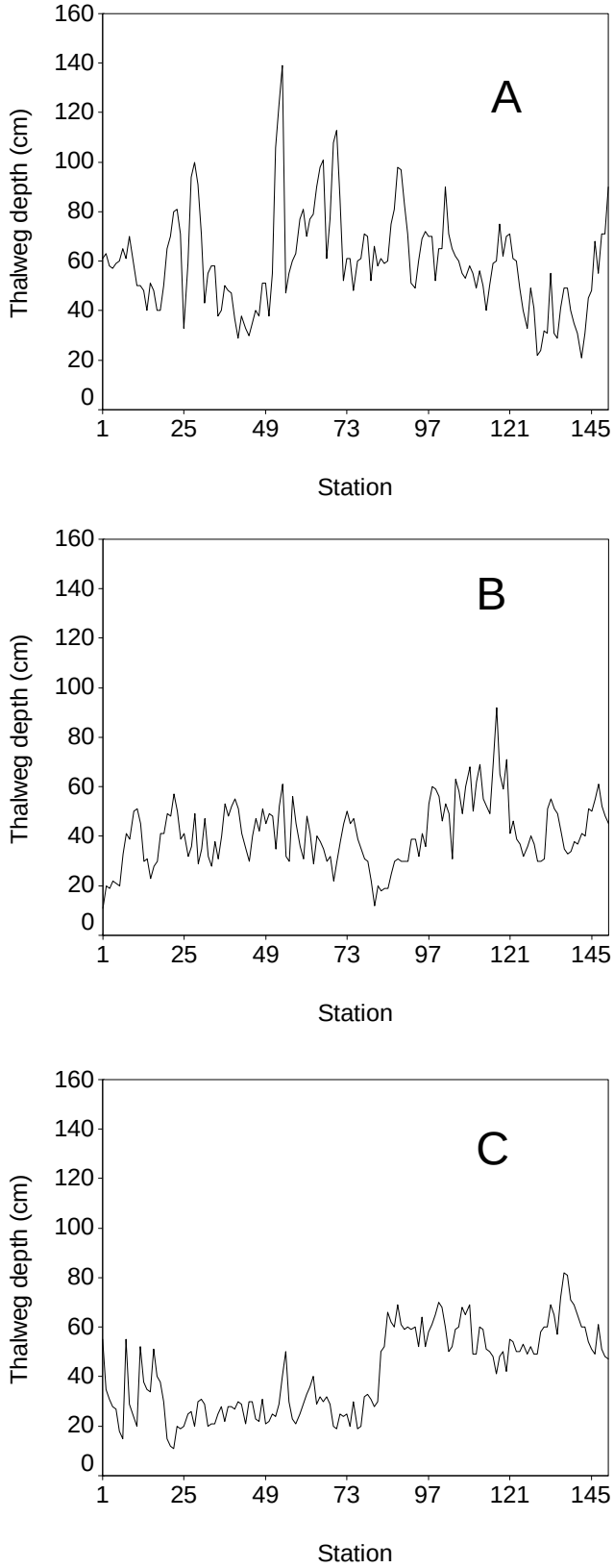

Figure 4. Longitudinal variation in thalweg depths for Sites A, B, and C. Data were collected at 1-m increments.

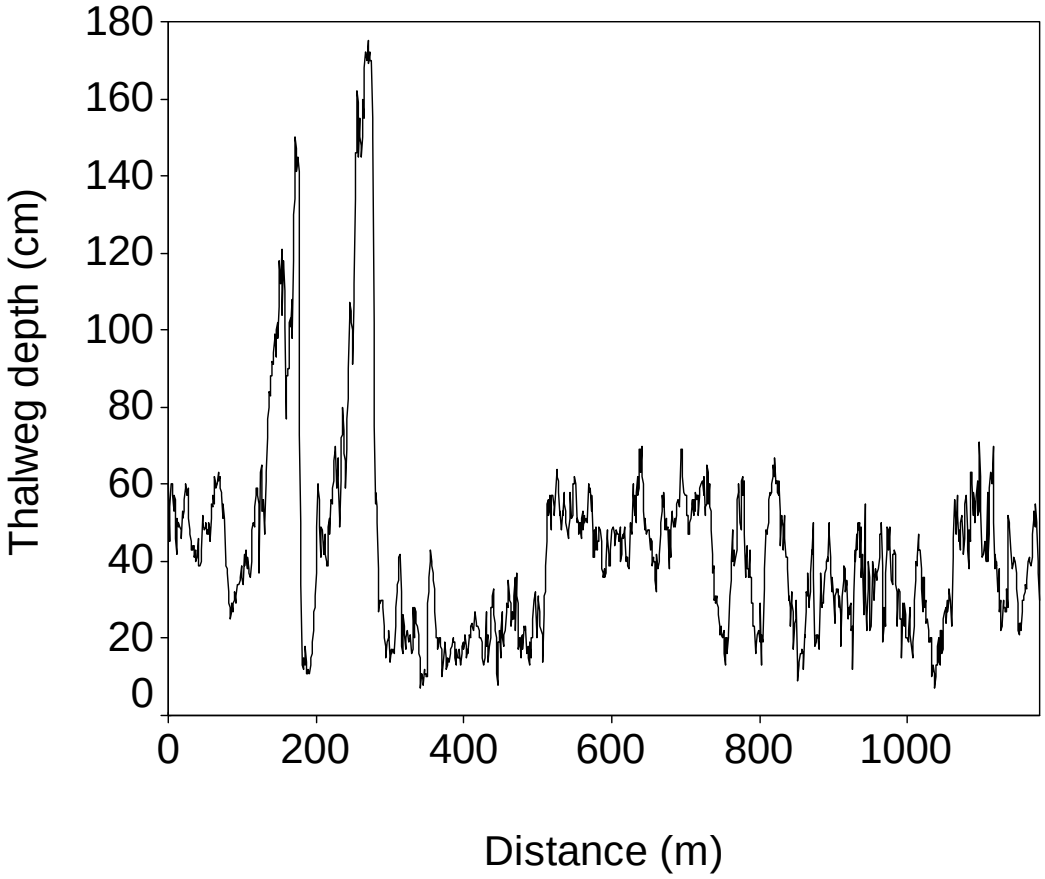

Figure 5. Longitudinal thalweg survey to 1.2 km downstream from the RMPD. The 0-distance is furthest downstream the RMPD and the upper distance stops at the last riffle below the dam. Data were collected at 1-m increments.

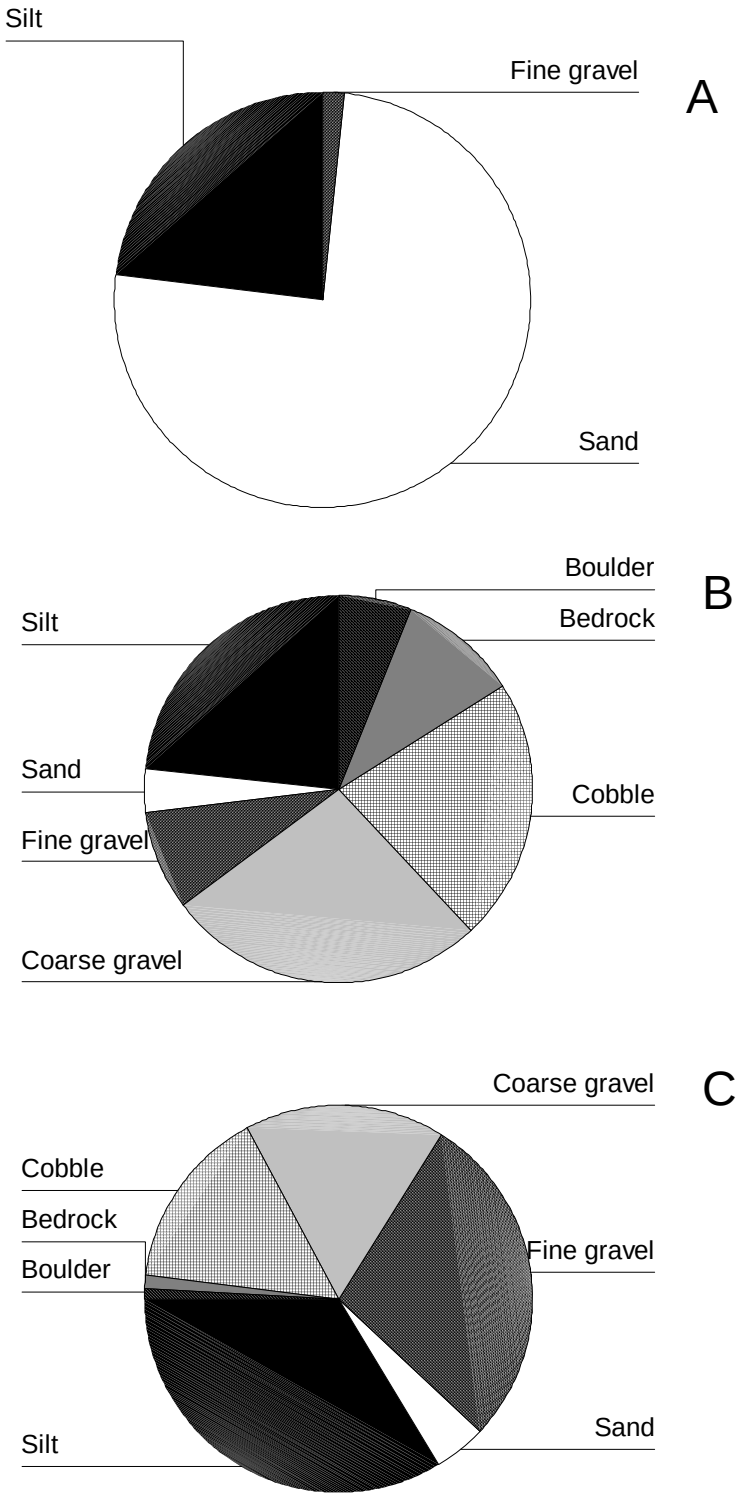

Figure 6. Substrate size distributions for Sites A, B, and C. Sizes followed definitions in Wolman (1954).

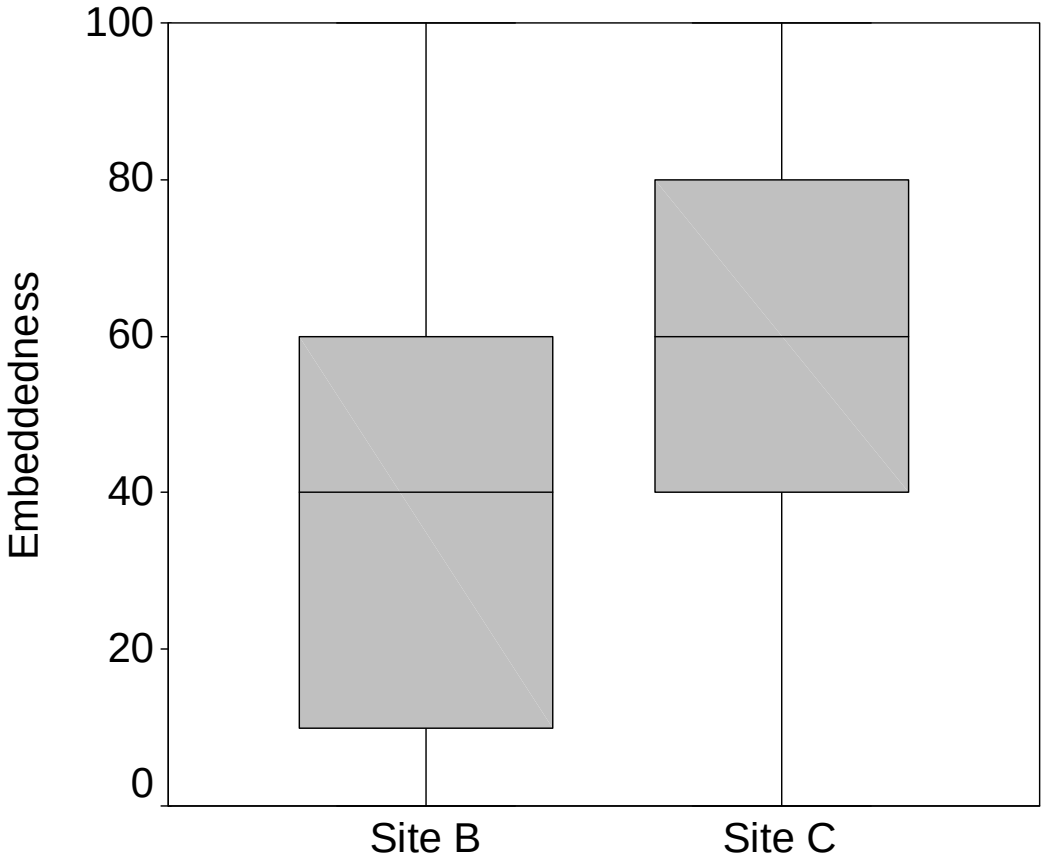

Figure 7. Boxplots for substrate embeddedness in sites B and C. Embeddedness was not estimated for Site A due to the nearly-homogenous sand substrate.

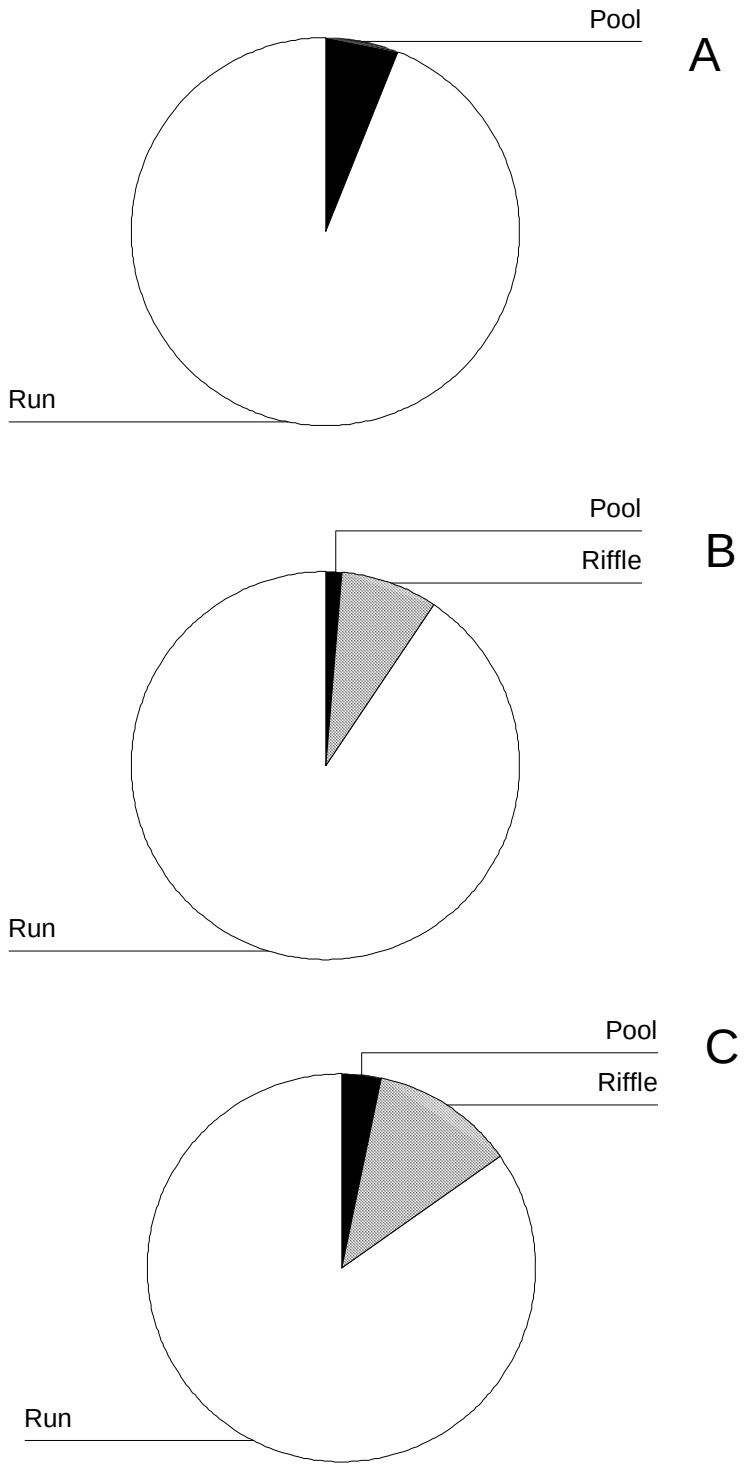

Figure 8. Mesohabitat variation within Sites A, B, and C.

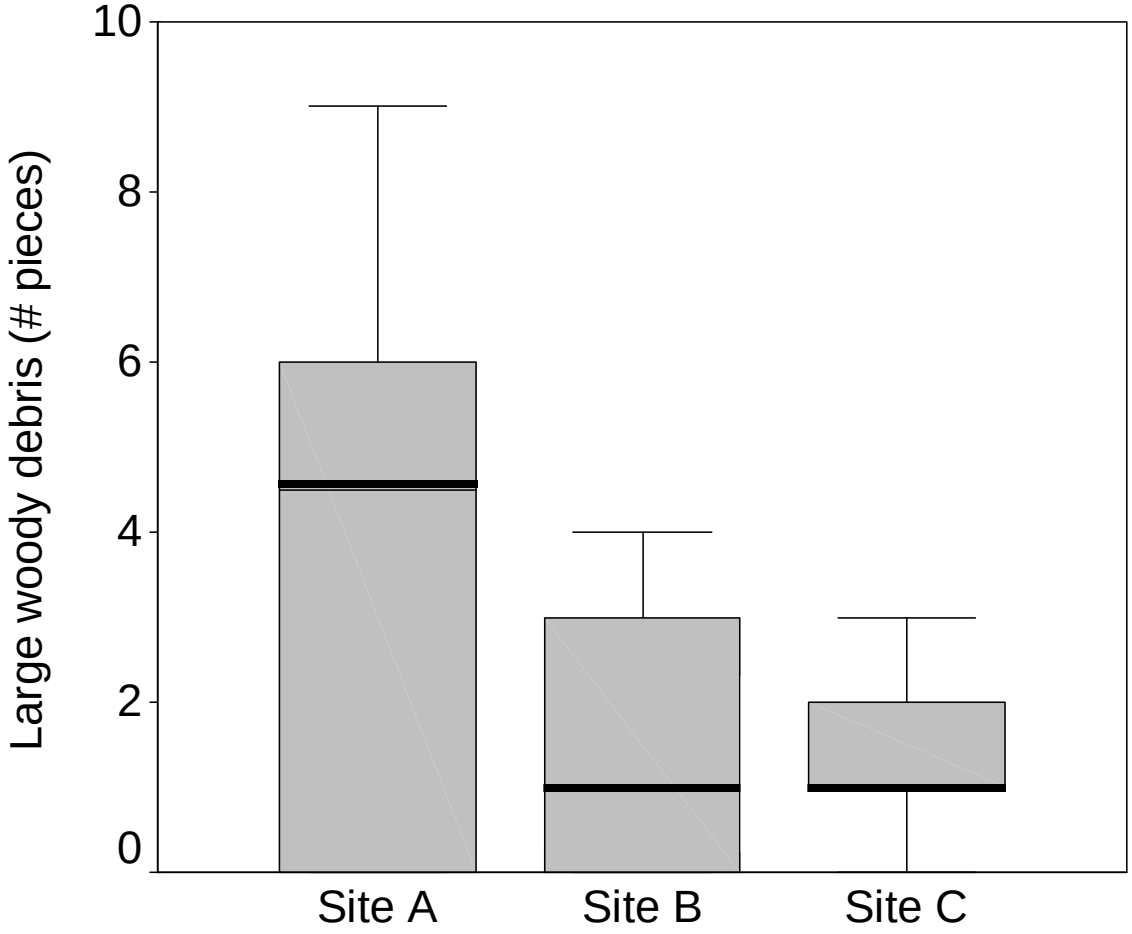

Figure 9. Box plots for large woody debris abundance in Sites A, B, and C.

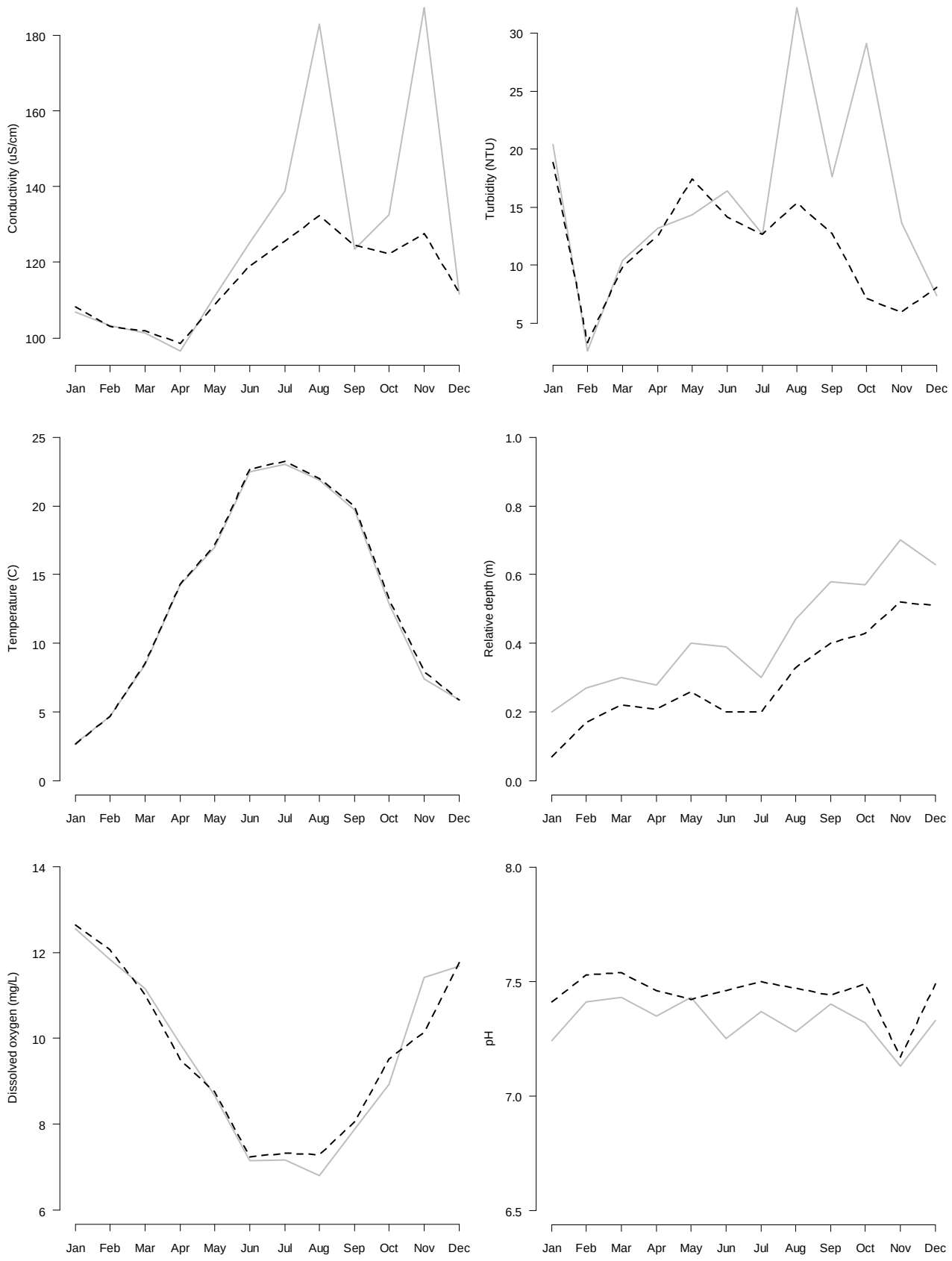

Figure 10. Mean-monthly water chemistry data. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

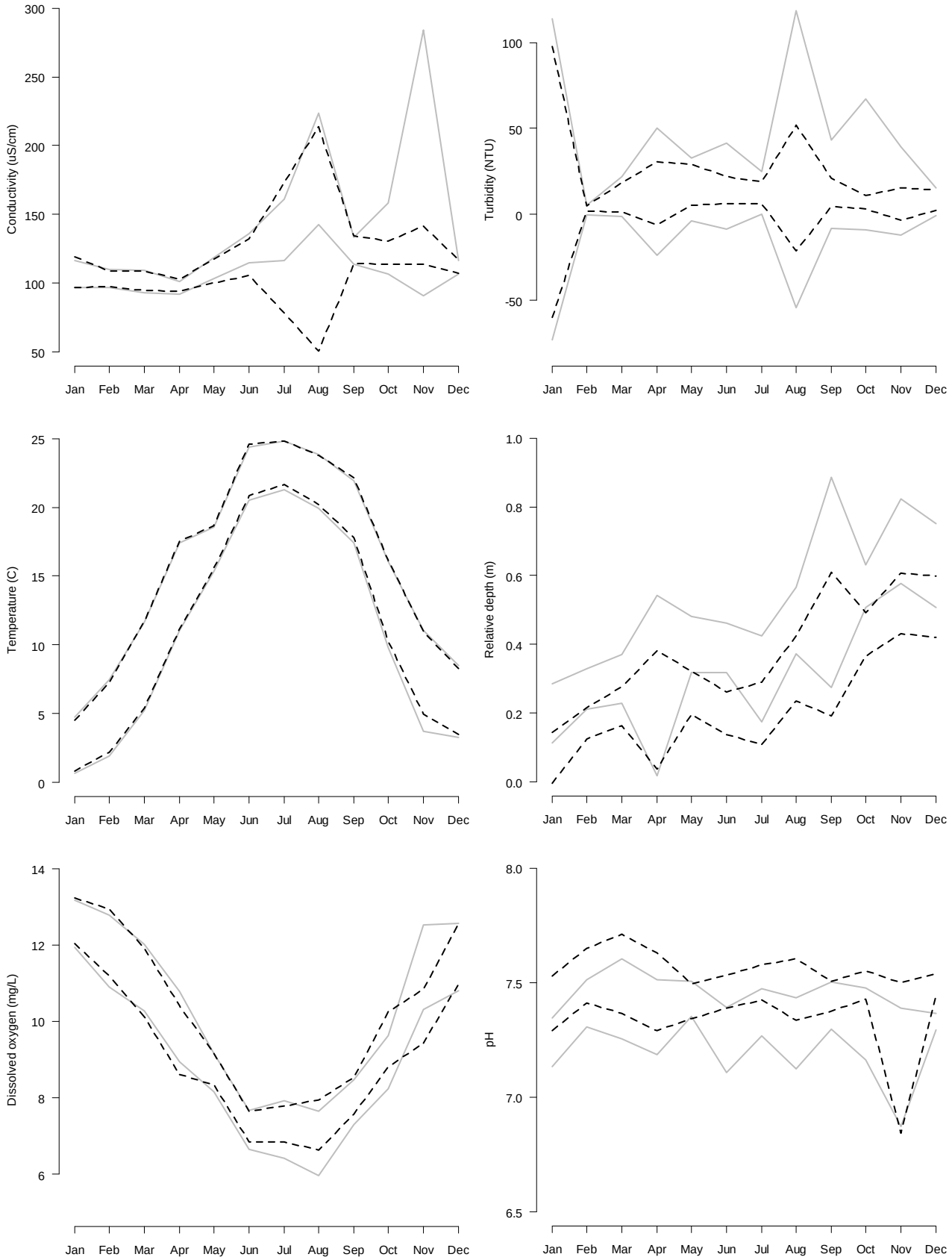

Figure 11. Mean-monthly water chemistry data plus and minus 1 standard deviation for the mean. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

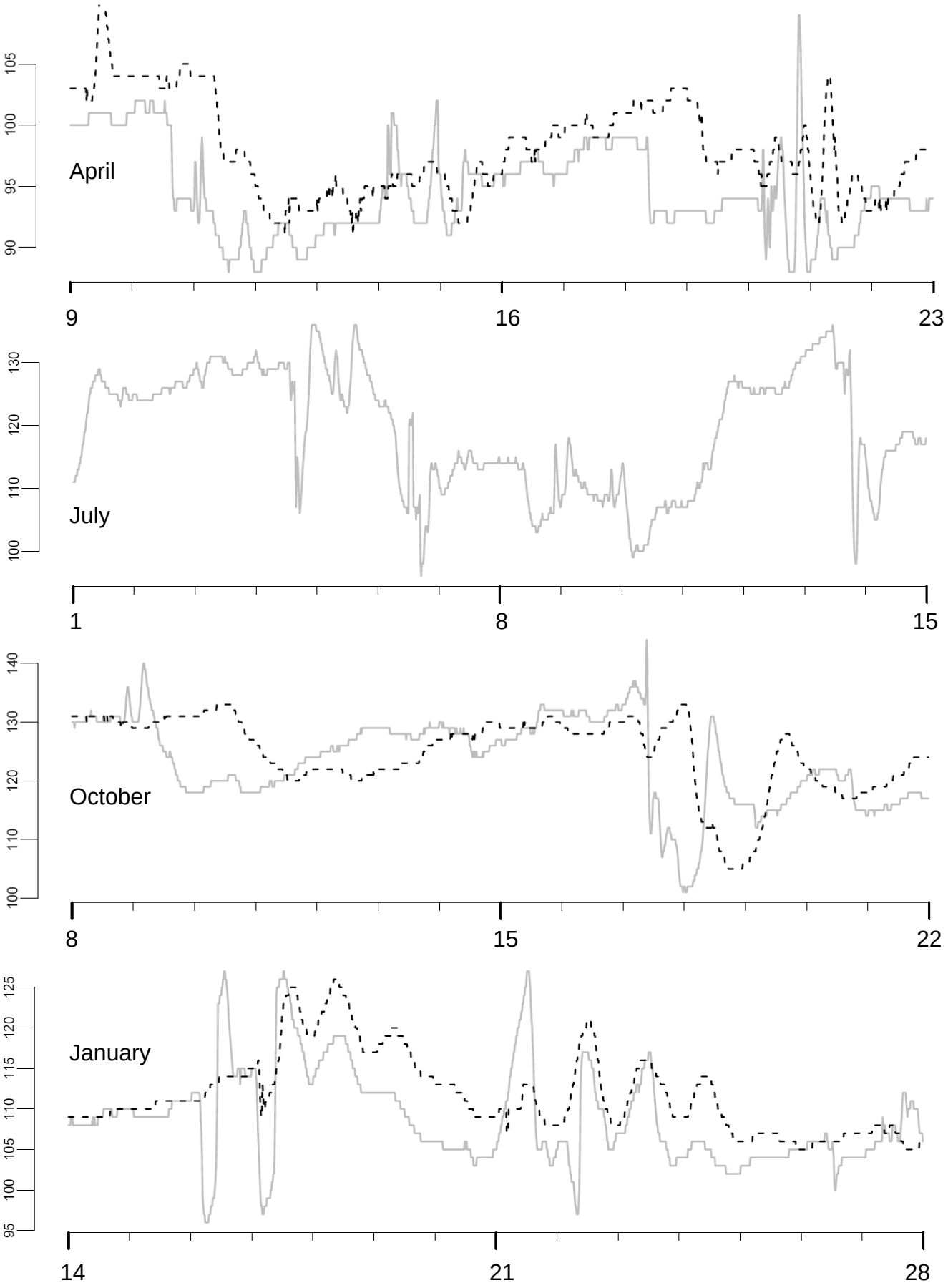

Figure 12. Conductivity data (µS/cm) for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C). No data were collected at Site C during July.

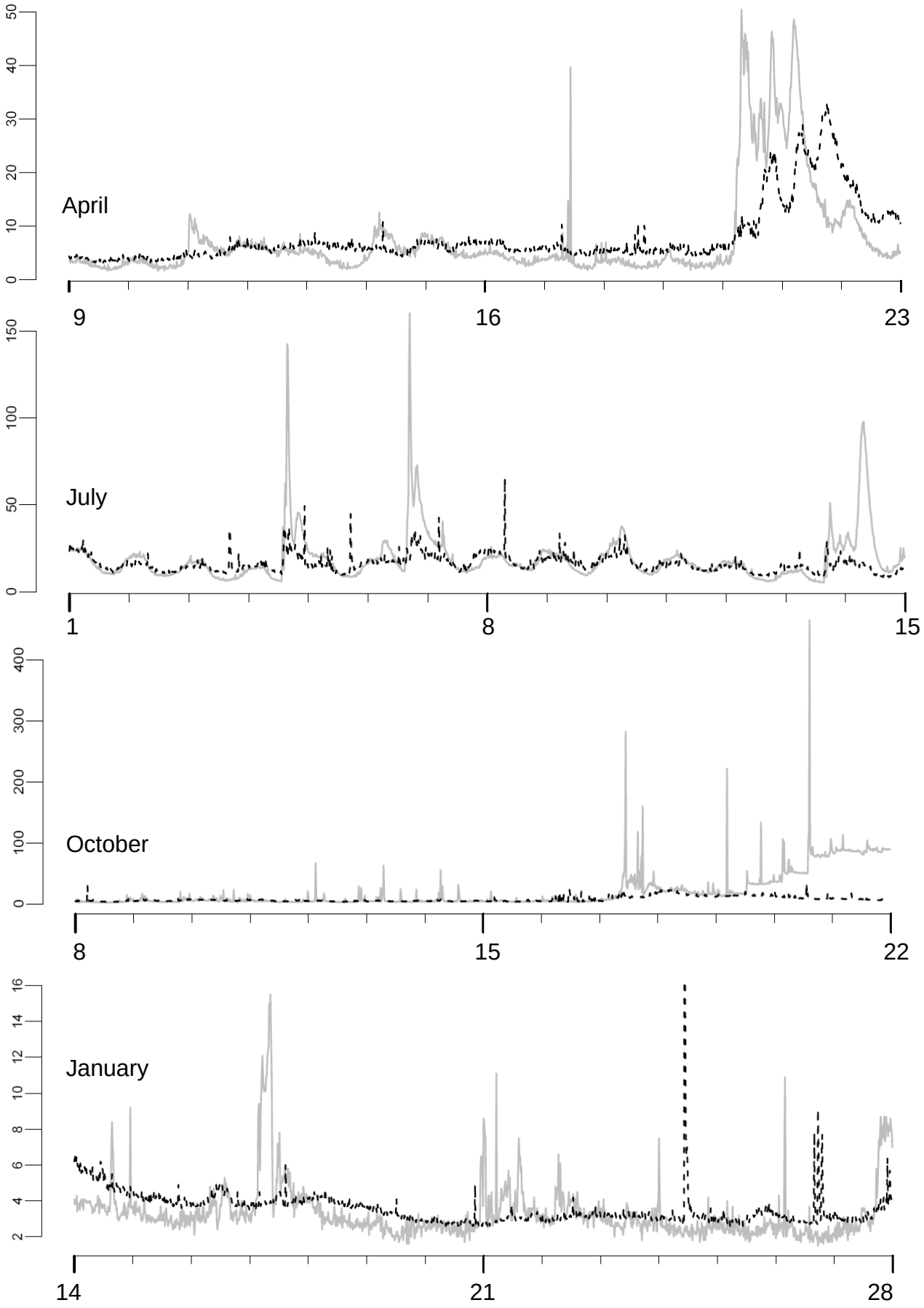

Figure 13. Turbidity data (NTUs) for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

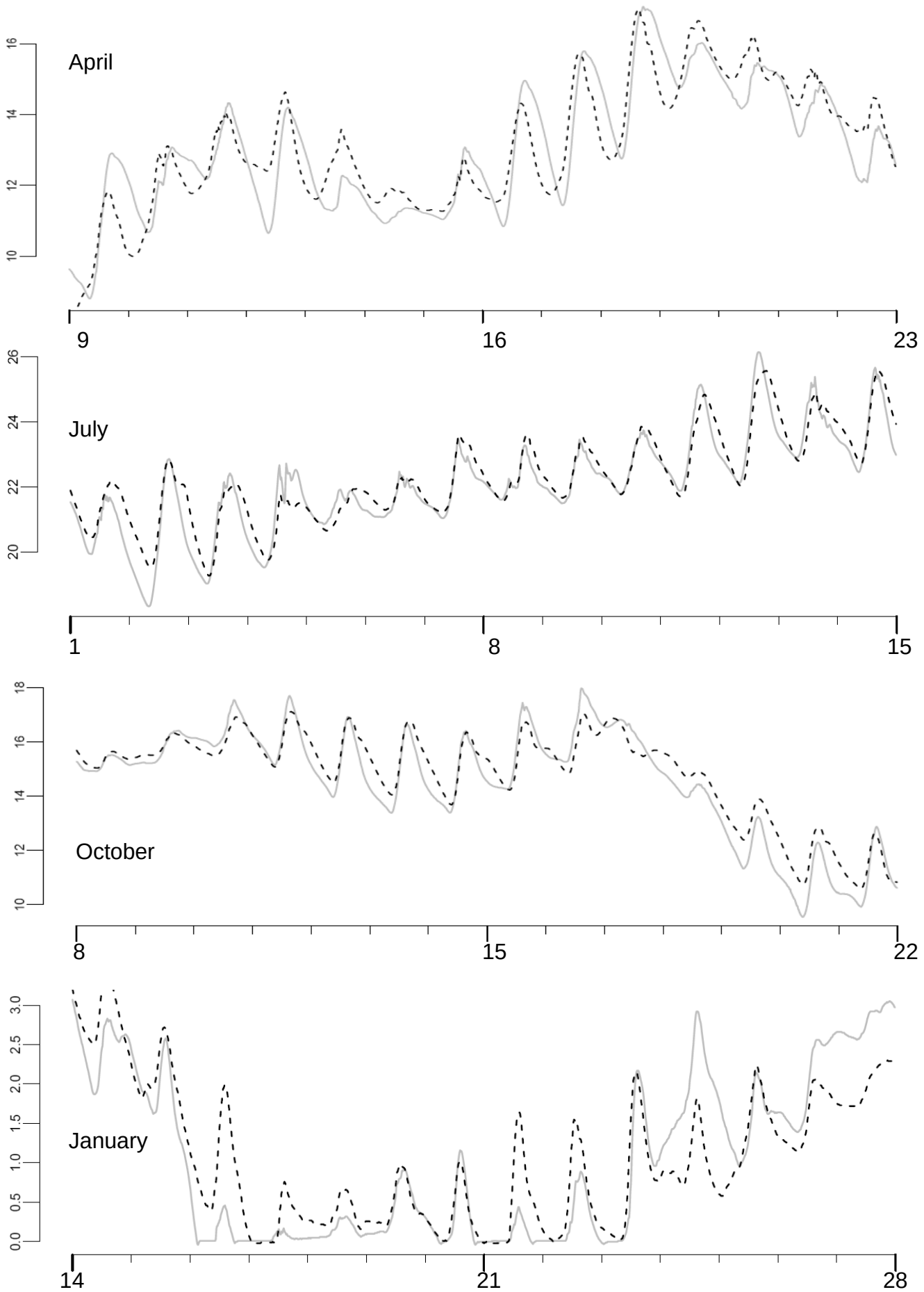

Figure 14. Water temperature data (°C) for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

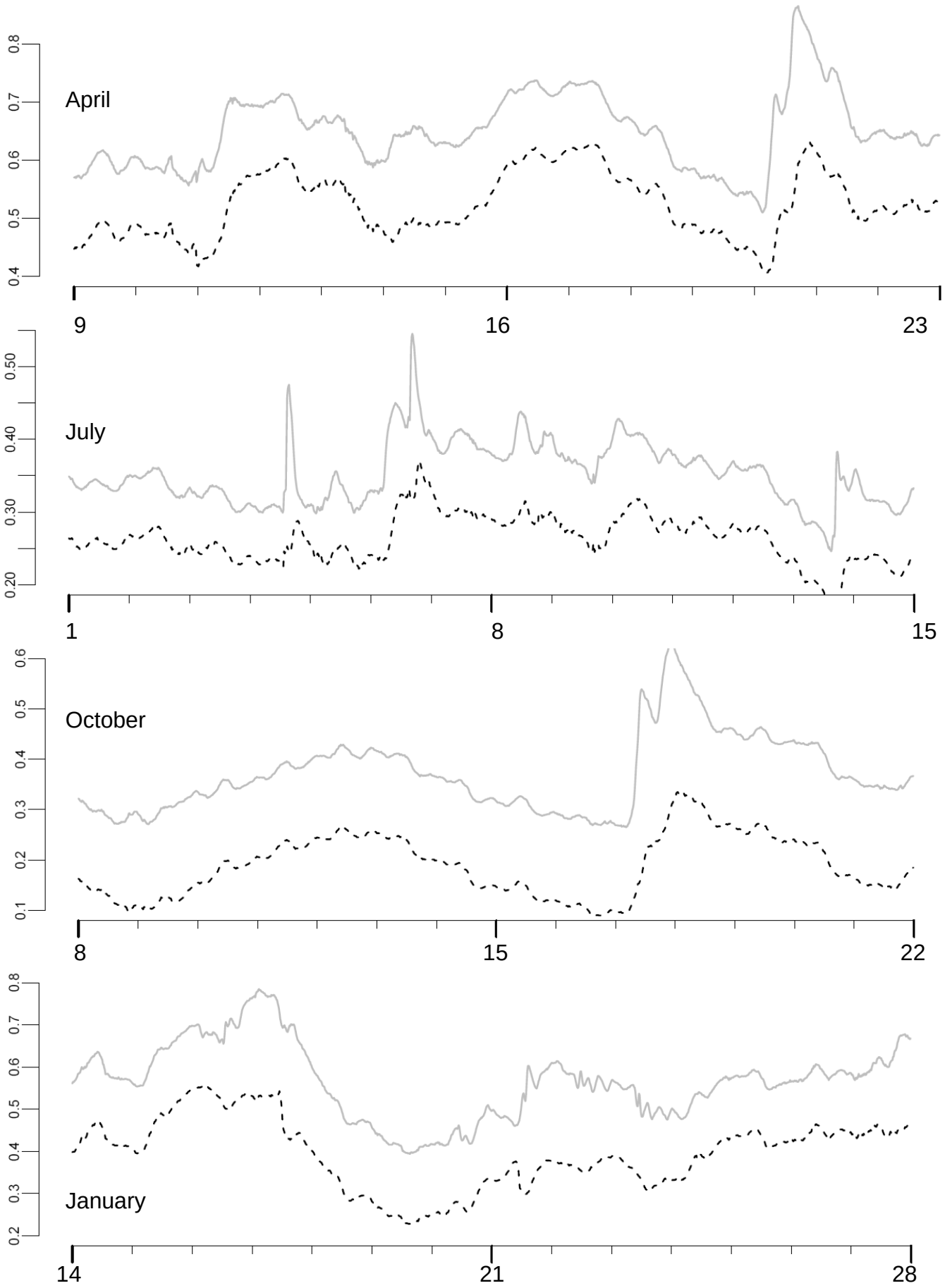

Figure 15. Relative depths (m) for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

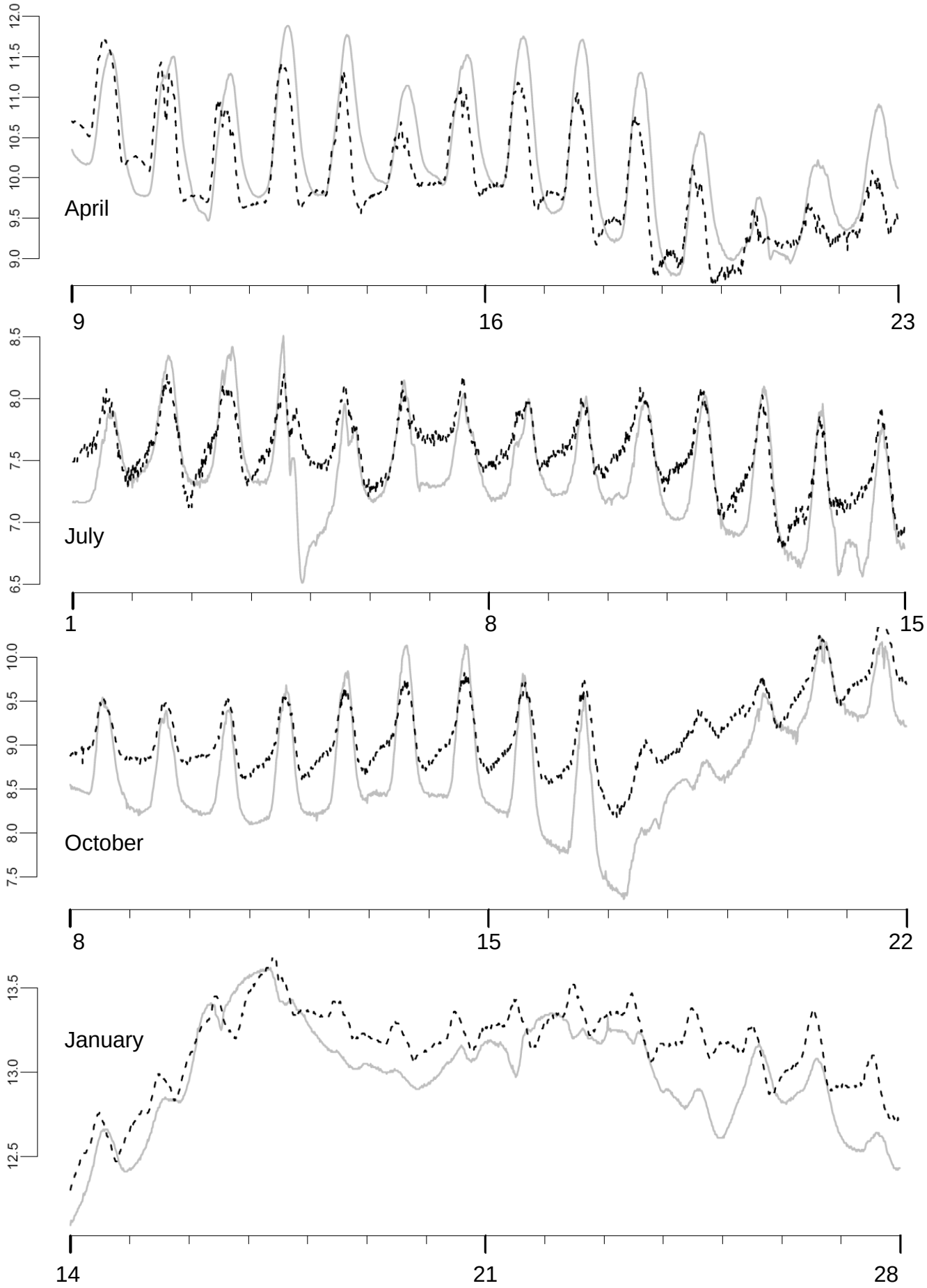

Figure 16. Dissolved oxygen (mg/L) for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

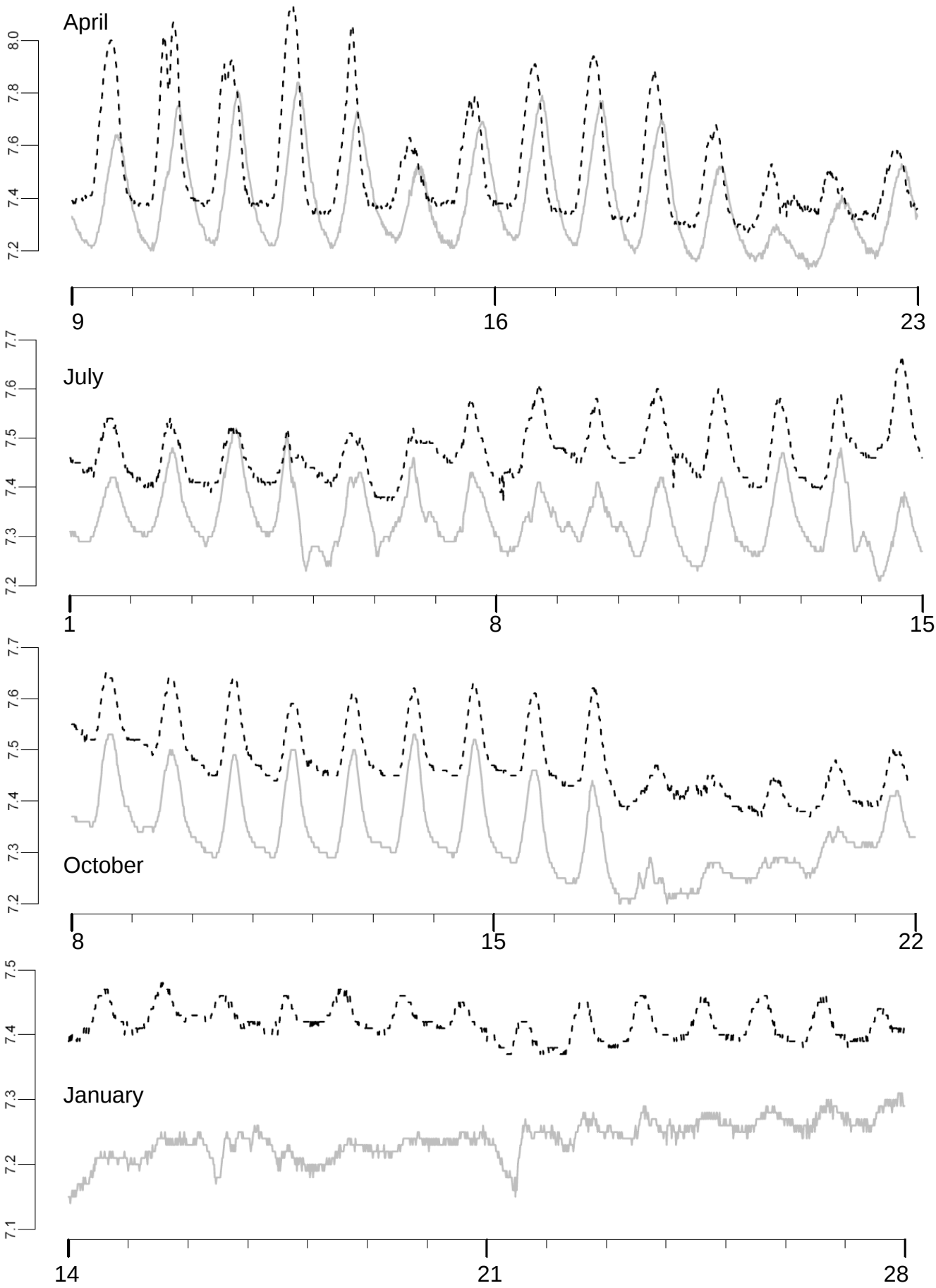

Figure 17. pH data for spring, summer, fall, and winter samples (April, July, October, and January, respectively). Data were collected at 15-minute intervals. The grey line shows values for the sample site upstream from the RMPD (Site A); the black, dashed line shows values for the sample site downstream from the RMPD (Site C).

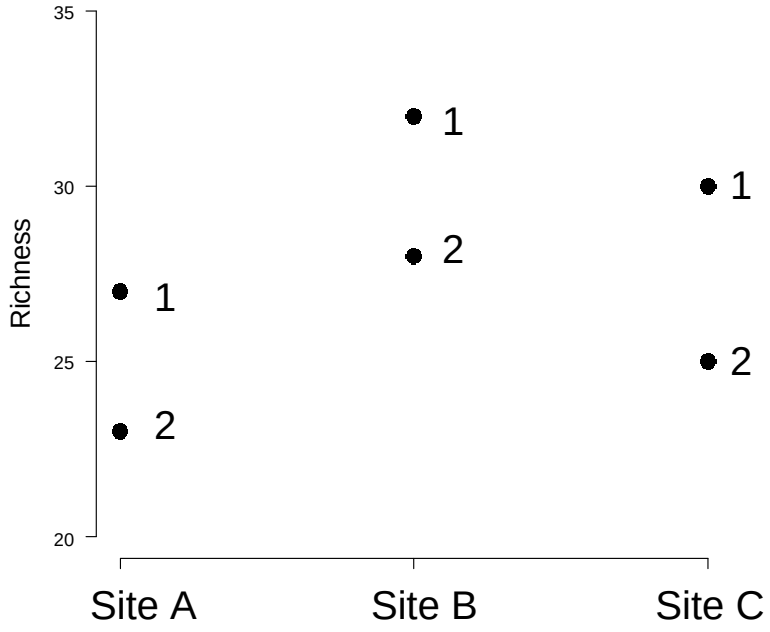

Figure 18. Observed fish species richness by site and time period. "1" indicates sampling during July; "2" indicates sampling during November.

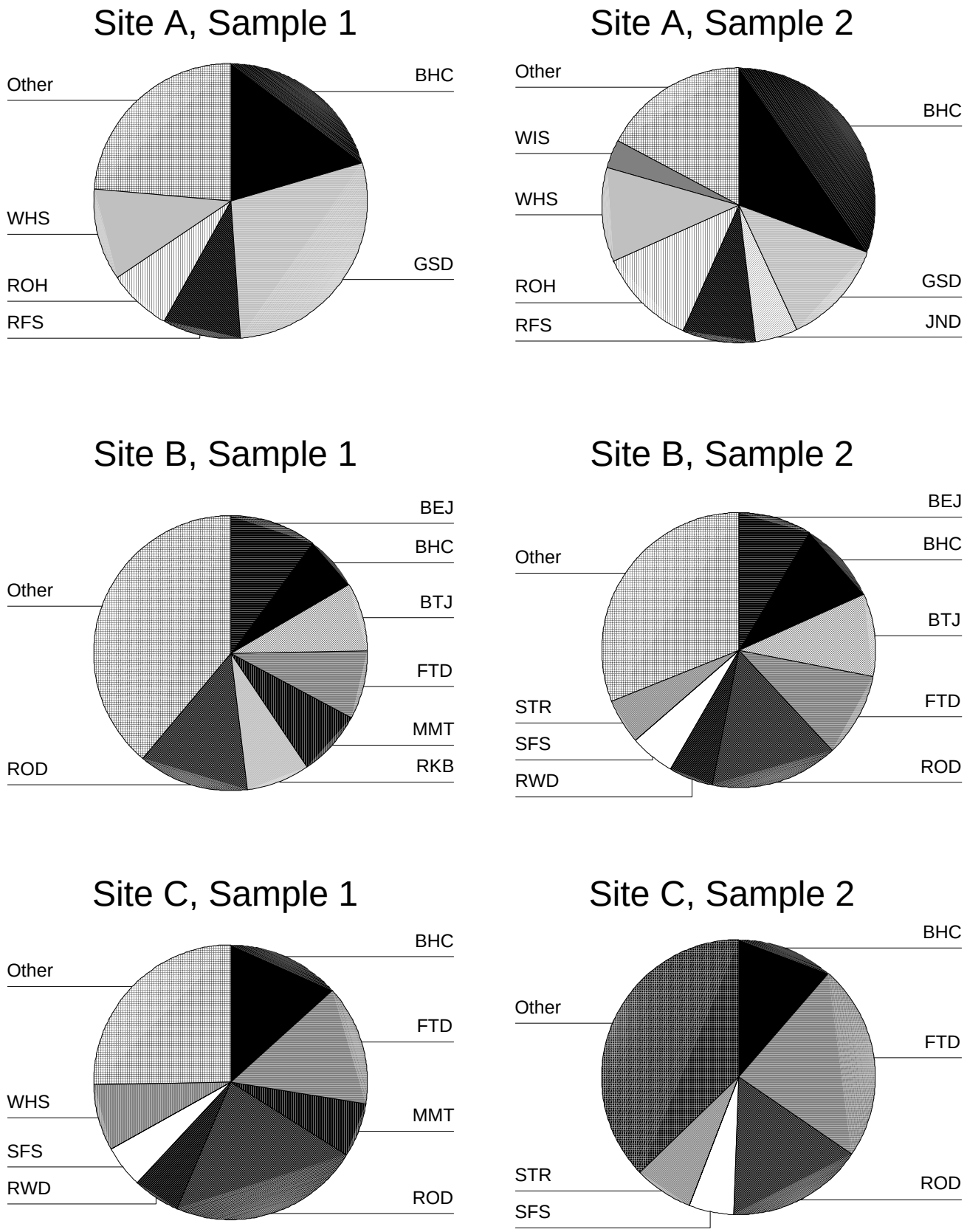

Figure 19. Relative abundances of fish species by site and time period. Species codes are presented in Table 3. Species were binned as "other" if abundances < 10% of the total.

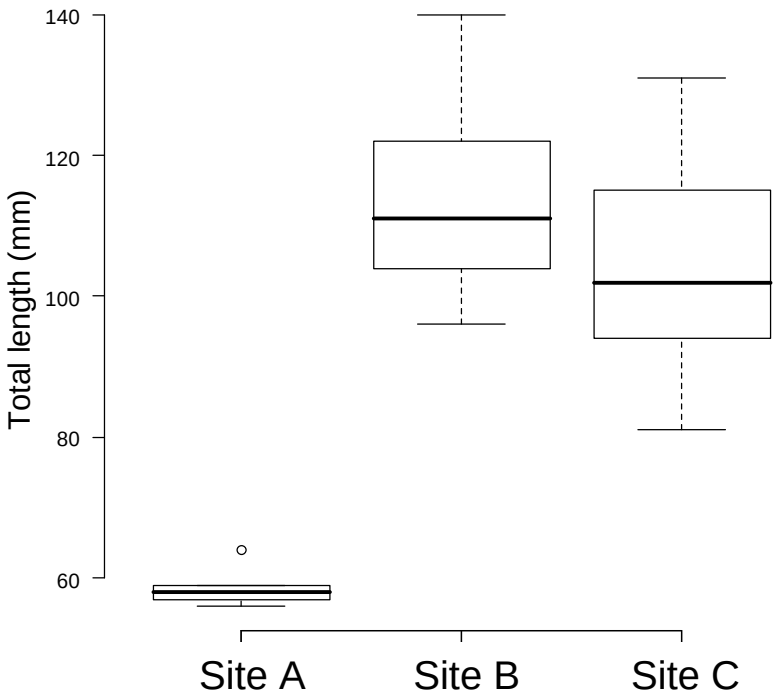

Figure 20. Roanoke logperch (P. rex) length data. Site A contained 5 samples; Site B contained 17 samples; and Site C contained 13 samples.

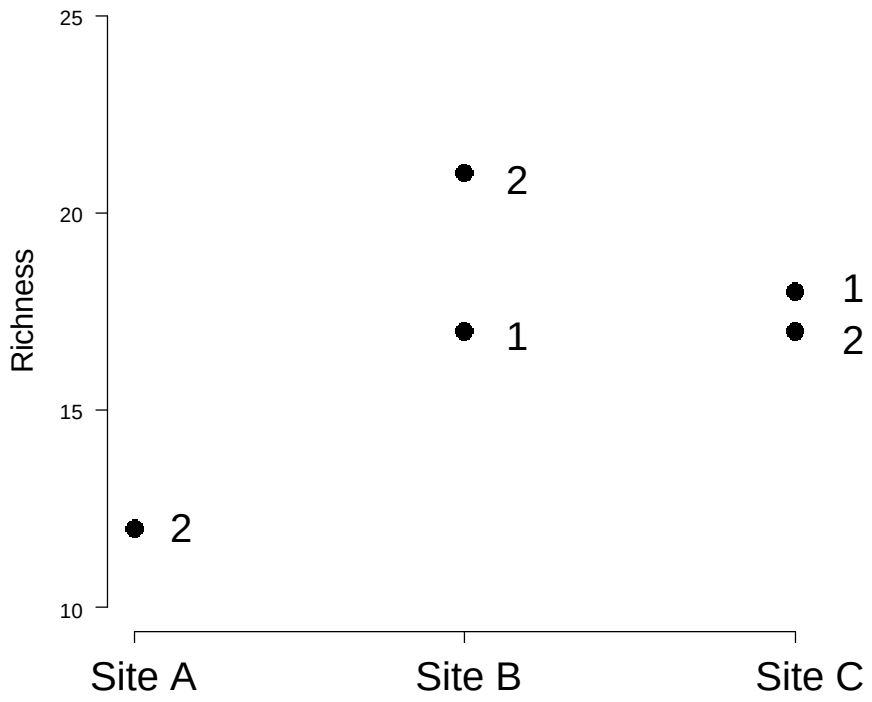

Figure 21. Taxonomic richness of benthic macroinvertebrates communities by sites and sampling period (1 = Fall 2008, 2 = Spring 2009). Site A was not sampled during Fall 2008.

Appendix A. Rainfall summary data for Roanoke, Virginia (http://climvis.ncdc.noaa.gov/). Highest ranks indicate the wettest years for the period of record (1948-2008 or 2009).

Year	Month	Mean rainfall (in)	Rank	Rank percent
2008	May	2.1	13	22
	June	4.7	41	68
	July	5.6	54	90
	August	5.2	46	77
	September	2.2	27	45
	October	1.9	21	35
	November	2.0	23	38
	December	2.3	19	32
2009	January	2.8	36	59
	February	1.2	8	13
	March	3.5	32	53
	April	3.2	36	59

Appendix B. Sonde location in Site A (upstream from RMPD). Photos: J. Jones (above) and N.P. Hitt (below).

Appendix C. Sonde location in Site C (downstream from RMPD). Photos: J. Jones (above) and N.P. Hitt (below).

Appendix D. Roanoke logperch (*Percina rex*) with visible implant elastomer marks. Photo: N. P. Hitt.

Appendix E. Bank erosion observed in Site A. Photo: N.P. Hitt.