Downstream effects of novel water-injection dredging experiments on macroinvertebrate assemblages of a Great Plains River

by

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B.S., Louisiana State University, 2023

A THESIS

submitted in partial fulfillment of the requirements for the degree

MASTER OF SCIENCE

Division of Biology
College of Arts and Sciences

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2026

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Abstract

Sediment accumulation threatens the long-term viability of reservoirs worldwide. By trapping upstream sediments, reservoirs progressively lose water storage capacity while depriving downstream ecosystems of ecologically important sediment inputs. These challenges have driven interest in sediment management strategies that address both storage loss and downstream sediment connectivity. Water-injection dredging (WID) is an emerging reservoir-management technique that resuspends sediment from the reservoir bed and transports it downstream through dam outlets. However, the effects of reservoir sediment releases on downstream river ecosystems remain poorly understood. WID was conducted in Tuttle Creek Reservoir during fall 2025 and spring 2026, representing a novel application of WID in a large reservoir. To evaluate the effects of increased suspended and deposited sediments downstream of the reservoir, we quantified macroinvertebrate responses to WID relative to site-specific baseline conditions. We hypothesized that WID would negatively affect macroinvertebrate communities, with the strongest effects occurring near the dam attenuating with downstream distance. Macroinvertebrates were collected monthly using one-hour multihabitat D-net surveys from spring, summer, and fall 2023 to 2026 to characterize pre- and post-WID communities. Our main analytical approach was to evaluate if post-WID samples fell within site- specific baseline conditions using multivariate ordinations, log-response ratios and rarefaction analyses of taxonomic richness. WID produced a measurable downstream sediment signal, with turbidity increasing by as much as 20-fold relative to baseline conditions and elevated suspended sediment detected up to 198 km downstream. Fine sediment deposition occurred at all downstream sites but was greatest in the Big Blue River and was concentrated along shorelines and other low-velocity habitats. Macroinvertebrate responses differed between demonstrations and were most

pronounced three days after spring WID, when richness declined at most sites and taxonomic and functional composition shifted outside site-specific baseline conditions. These compositional changes were associated partly with increased representation of planktonic taxa. In contrast, responses following fall WID were weaker and limited to fewer sites. Responses coincided with substantial hydrologic and seasonal variability, limiting attribution to WID alone. Because both WID demonstrations used controlled releases and produced suspended-sediment concentrations within the river's natural range, the observed responses reflect moderate operating conditions and may not represent more intensive WID applications. Overall, WID coincided with measurable short-term responses, but macroinvertebrate effects differed between demonstrations.

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Acknowledgements

This research was made possible by the support, time, and dedication of many individuals and organizations, all of whom I am deeply grateful to acknowledge. I wish to first and foremost thank my major professor, Dr. Keith Gido, for taking a chance on hiring me as a research technician straight out of undergrad and somehow turning that decision into several years of fieldwork, bug sorting, and a master's degree. I am deeply grateful for his mentorship, patience, and continued investment in my professional growth. The field and laboratory components of this research would not have been successful without the hard work and dedication of the Fish Ecology Lab members and volunteers, who spent countless hours picking inverts under the microscope and deciding whether each speck was a bug, a rock, or simply wishful thinking. Of all the things I expected to take away from graduate school, becoming the lab's "bug girl" was not one of them, but it is a title I gladly claim. I will especially cherish the field days that blurred the line between river sampling and swimming. I am deeply grateful to Cassidy Schmidt, Brooke Nelson, Ella Huizenga, Macey Willard, and Natalie Pegg, affectionately known as the "girl bosses," who made my time at Kansas State one I will never forget. Cassidy deserves special recognition, as processing these samples, managing multiple field projects and undergraduate technicians was no easy task, and I could not have done it without her dedication, friendship, and support.

I am also grateful to my committee members, Dr. Walter Dodds and Dr. Sophia Bonjour, for sharing their passion for science, thoughtful guidance, and contributions to this research. I also thank Dr. Dave Haukos and the USGS Cooperative Research Unit at Kansas State University for generously providing access to boats that were essential to completing this research. I am equally grateful to the Kansas State University Division of Biology for providing

the research facilities and resources that made this work possible. Finally, I thank the Kansas City District of the U.S. Army Corps of Engineers and the Kansas Water Office for their collaboration and funding. I am especially grateful to Marvin Boyer (USACE-KC) for his mentorship, humor, and many shared lunches.

Dedication

This work is dedicated to my family, especially my parents, Marlene and TJ; my grandfather, Alfredo; and my younger siblings, Julian, Gabriel, Elaina, Leylani, and Thomas. Their love, encouragement, and support have carried me throughout this journey and made this accomplishment possible. As a first-generation college student, I often found myself navigating a path that was unfamiliar not only to me, but also to those closest to me. Although they may not have always understood why I chose to spend my time standing in rivers, identifying fish, picking bugs, or even moving to Kansas to pursue this career, they never stopped believing in me. They listened to explanations they did not always understand, celebrated milestones they had never experienced themselves, and have always made the long drive to remind me that I was never pursuing this journey alone. Their encouragement, sacrifices, and willingness to support a dream so different from anything our family had known gave me the confidence to keep going. I am especially proud to share this accomplishment with my younger siblings and hope that my journey reminds them that they can create paths of their own, even when those paths lead somewhere entirely new. Finally, I thank my partner, Brian Elizondo, for leaving the Louisiana bayou for the Great Plains so that I could pursue my dreams. His love, patience, unwavering support, and countless home-cooked meals carried me through every stage of this journey.

Chapter 1 -

1 | Introduction

Although dams and their associated reservoirs provide important societal benefits such as flood control and water supply, they also trap sediment and disrupt downstream transport (Morris and Fan 1998; Juracek 2015). Within reservoirs, inflowing water velocities decline, reducing its capacity to keep suspended and bedload material in transport and promoting sediment deposition (Anzani et al., 2026). As reservoirs age, sediment accumulation reduces their storage capacity, shortens operational lifespan, and creates growing management challenges globally (Schleiss et al., 2016; Shelley et al., 2016; De Vincenzo et al., 2024). These challenges are particularly relevant in Great Plains watersheds, where highly erodible soils, agricultural land use, and naturally high sediment loads can accelerate reservoir infilling (O'Neill et al., 2014). Downstream, reduced sediment supply can alter geomorphology, substrate composition, aquatic biota, and ecological processes that depend on the sediment supply and transport (Costigan and Daniels 2012; Kondolf et al., 2014; Schmutz and Moog 2018; Calapez et al., 2021).

Traditional sediment-management approaches can be costly, logistically difficult, and environmentally disruptive (Morris 2020; Dauvin et al., 2022; Panagos et al., 2024). These methods often require heavy equipment and off-site disposal areas, which can limit their feasibility for large reservoirs with substantial sediment accumulation. Future generations are challenged with creating sediment-management techniques that can reduce reservoir sedimentation while minimizing logistical and economic barriers. Water-injection dredging (WID) has emerged as an alternative sediment-management technique that may reduce some of these limitations (Welp et al., 2017; Wagner and Lewis, 2022; Fuller et al., 2024). WID operates by injecting large volumes of low-pressure water into the sediment bed through vessel-mounted

jets that resuspend and fluidize materials into a dense, sediment-laden slurry that moves downstream as a density current (Figure 1). Unlike conventional dredging, WID disturbs and resuspends sediment, relying on gravity and existing flow paths to move that material downstream, reducing the need for mechanical excavation and off-site disposal (Wagner and Lewis 2022; Fuller et al., 2024; Tyler et al., 2026). Because of these logistical and economic advantages, WID has been used in navigation channels, ports, and coastal systems across the world (Clausner et al., 1993; Welp et al., 2017). Applying WID to manage reservoir sediments is a novel approach with many potential benefits, but its downstream ecological effects need further investigation before large-scale application.

While WID may help restore sediment continuity by mobilizing accumulated reservoir sediments, the release of fine particles may also affect downstream habitats by increasing turbidity and depositing fine sediment. Fine sediment is a widespread physical stressor in stream ecosystems and an important water-quality concern in the United States (Harvey et al. 2024). This concern is reflected in national impairment reporting, where fine sediment ranked sixth among reported causes of impairment nationwide, was listed among the top five impairments in 16 of 21 EPA water-resource accounting regions and has been identified as the leading reported stressor for streams with impaired benthic macroinvertebrate communities (Governor et al., 2017; Fanelli et al., 2022).

Mechanistically, deposited fine sediment can fill interstitial spaces, reduce substrate heterogeneity and stability, limit access to oxygenated refuges within benthic and hyporheic habitats, and disrupt feeding by reducing food density or quality (Hynes, 1970; Culp et al., 1983; Minshall, 1984; Graham, 1990; Wood and Armitage, 1997). These changes can occur quickly, with effects on stream invertebrates reported within 24 hours of exposure (Culp et al., 1986). As

fine sediment accumulates, habitat suitability may decline for taxa associated with coarse, stable, and well-oxygenated substrates, while taxa that tolerate sediment or use depositional habitats may be less affected or increase in relative abundance. These responses can lead to shifts in macroinvertebrate abundance, richness, and community composition, typically described as a taxonomic shift resulting from changes in Ephemeroptera, Trichoptera, and Plecoptera taxa (EPT) towards oligochaetes or burrowing Chironomidae (Rabani et al., 2005; Jones et al., 2012; Descloux et al., 2014; Mathers et al., 2022). Macroinvertebrates are therefore useful indicators of WID-related changes because they occupy the benthic habitats most directly affected by sediment deposition, vary widely in their sensitivity to suspended and deposited sediment, and respond rapidly to physical habitat change (Resh et al. 1988; Rosenberg and Resh 1992; Henley et al. 2000).

Based on these mechanisms, short-term declines in sensitive taxa, total abundance, and overall richness would be expected immediately following WID, with recovery contingent on the rate of fine sediment dispersal, duration of exposure, and substrate recolonization (Bilotta and Brazier 2008). Consistent with these predictions, Pledger et al. (2021) documented short-term reductions in macroinvertebrate abundance, richness, and diversity following a WID operation in the tidal River Parrett, United Kingdom, with recovery to baseline conditions within five months. Macroinvertebrate communities in Great Plains rivers might differ from forested and low-sediment systems in which responses to fine sediment have been most extensively studied (Angradi, 1999; Von Bertrab et al., 2013; Bylak and Kukula, 2022). Great Plains rivers are naturally turbid, high-sediment-load systems in which macroinvertebrate communities are structured by frequent physical disturbance and elevated suspended sediment (Phillips et al. 2008; Phillips et al. 2016). Even so, riverine macroinvertebrate communities remain

comparatively understudied in the Great Plains, USA. Previous studies have described tallgrass prairie stream macroinvertebrate communities and their responses to hydrologic disturbance and land management (Miller and Golladay, 1996; Gray and Johnson, 1988; Hax and Golladay, 1998; Stagliano and Whiles 2002; Fritz and Dodds, 2004; Tow et al. 2026). Fewer studies have directly evaluated macroinvertebrate communities in large prairie rivers where high suspended-sediment loads, frequent hydrologic disturbance, and longitudinal changes in particle resources may structure assemblages differently than in smaller streams (Whiles and Dodds 2002). Large prairie rivers also support zooplankton communities that persist with high turbidity and pronounced hydrologic variability (Thorp and Mantovani 2005).

Tuttle Creek Lake, a federally operated reservoir managed by the U.S. Army Corps of Engineers, impounds the Big Blue River near Manhattan, Kansas, USA and is the largest reservoir in the Kansas River basin. Since its completion in 1962, sedimentation has reduced reservoir storage capacity by approximately 45% (Rahmani et al., 2018; Kansas Water Office, 2025) creating long-term concerns for flood-control and water-supply functions that serve a substantial portion of Kansas. To address this issue, the U.S. Army Corps of Engineers, in partnership with the Kansas Water Office, conducted WID demonstrations in fall 2025 and spring 2026 to evaluate whether accumulated reservoir sediments can be mobilized toward the bottom outlet and transported downstream through the dam. This effort was the first application of WID in a freshwater reservoir and was intended to assess both operational performance and downstream effects, including water quality, sediment transport, geomorphic responses, and ecological responses. This paper describes the macroinvertebrate responses to WID-related sediment releases across a downstream gradient. Our first objective was to quantify changes in macroinvertebrate community structure following WID and determine how responses varied

through time and with distance downstream from Tuttle Creek Dam. We hypothesized that WID would reduce taxonomic richness, abundance, and shift community composition, with the strongest responses occurring immediately downstream from the dam and weakening with distance downstream and dilution from the Kansas River. Our second objective was to evaluate taxonomic and functional responses to WID and identify the taxa and functional groups underlying shifts in community composition. We hypothesized that sediment pulses would reduce sediment-sensitive Trichoptera and Plecoptera taxa, thus favoring more sediment-tolerant taxa and/or functional groups dominated by burrowers and deposit feeders (e.g., burrowing mayflies, worms, snails, and clams). Finally, because WID occurred as intermittent sediment pulses of approximately 10 days, we predicted that responses would be strongest shortly after sediment releases, followed by partial recovery or community reorganization several weeks later.

2 | Materials and Methods

2.1 | Study Area

The Kansas River is a seventh-order stream in northeastern Kansas that originates at the confluence of the Republican and Smoky Hill rivers near Junction City, Kansas and flows east to the Missouri River (Figure 2). The river drains a watershed dominated by row-crop agriculture grassland, and pasture with the riverbed consisting primarily of sand, gravel at the edges of sandbars, and silt in the slow-flowing slack areas (O'Neill and Thorp 2011). Multiple tributaries flow into the Kansas River. The largest tributary is the Big Blue River, which drains approximately 25,000 km² across Nebraska and Kansas (Brady et al., 1998). The Big Blue River flows southward into Tuttle Creek Dam and Reservoir before entering the Kansas River in Manhattan, Kansas. Since the 1960s, Tuttle Creek Dam and other large federal dams on major tributaries have altered flow regimes, sediment transport, and channel morphology throughout

the Kansas River basin, while channelization and instream sand dredging have further modified sediment dynamics and substrate availability (Evans-White et al. 2010; Costigan and Daniels 2012; Shelley et al. 2016).

2.2 | Experimental Design

To evaluate downstream effects of WID, we repeated sampling of six sites along the Big Blue and Kansas rivers. This design emphasized within-site temporal comparisons across months and years, allowing us to establish variability in pre-WID conditions at each site to serve as the baseline for evaluating post-WID changes. Five impact sites were located at increasing river distances downstream from Tuttle Creek Dam to assess whether WID-associated changes in abiotic conditions and macroinvertebrate communities decreased with distance from the dam and following mixing with the Kansas River (Figure 2). These sites included one location on the Big Blue River approximately 5 km below the dam and four locations on the Kansas River approximately 16, 46, 105, and 198 km downstream from the dam. One control site (Above Confluence) was positioned upstream of the Kansas–Big Blue river confluence and immediately downstream of the Manhattan Wastewater Treatment Plant (WWTP) discharge. Positioning the control site downstream of the wastewater effluent ensured any effects from WWTP were incorporated into control conditions and to help distinguish effects attributable to WID-related releases.

2.3 | Macroinvertebrate sampling and identification

We sampled macroinvertebrates monthly from spring, summer, and fall 2023 to 2025 and spring 2026 to characterize pre- and post-WID communities. At each site, a single composite sample was collected using a 500-µm D-frame net across available wadable habitats (e.g., backwater, wood piles, submerged vegetation) in proportion to their occurrence. Sampling effort

was standardized to one-person-hour. Composite samples were emptied into a 10L bucket. Bucket contents were elutriated such that suspended materials were passed through a 250µm sieve and retained material was preserved in 10% formalin by volume. In the laboratory, we used a Folsom plankton splitter to produce four subsamples, or "splits", with each split reduced to a size appropriate for processing in a 90 mm diameter × 15 mm deep petri dish. Four subsamples were generally adequate to capture site taxonomic richness (Appendix A1). Macroinvertebrates were identified to the family following Merritt et al. (2019), with the following exceptions: subfamily Tanypodinae; orders Collembola, Ostracoda, and Oligochaeta; classes Gastropoda and Bivalvia; and subclass Copepoda, which were recorded at their respective taxonomic levels. Each taxon was assigned to a single functional feeding group based on assignments by the Kansas Department of Health and Environment (Table 1); collector-filterers, shredders, scrapers, predators, collector-gatherers, and piercer-herbivores. A single habit assignment was also made: climbers, clingers, burrowers, skaters, sprawlers, and swimmers. We modified assignments from KDHE to assign planktonic taxa.

2.4 | Water-injection dredging

Water-injection dredging was conducted in Tuttle Creek Reservoir during two demonstration periods: September 17–27, 2025, and March 26–April 10, 2026 (Figure 3). After each demonstration, two-day high-discharge releases were used to promote downstream transport of deposited sediment. Dredging operations occurred for 200 dredging hours in each demonstration, with daily dredging lasting approximately 20 consecutive hours per day. Pre-, during- and post-dredging mean and peak values were extracted from the United States Geological Survey (USGS) turbidity time series data from gages 06887000, 06887500, and 06892350 (U.S. Geological Survey, 2026a, 2026b, 2026c) and used to investigate effects of WID

on water physiochemistry. Continuous measurements were collected using YSI EXO multiparameter sondes. Operational conditions were adjusted as needed in response to weather, inflow, reservoir elevation, and dam management constraints, while release rates were varied to assess sediment mobilization and downstream transport under different discharge conditions.

2.5 | Statistical Analysis

Macroinvertebrate counts per sample were extrapolated to whole-sample equivalents by multiplying by the inverse of the subsampling fraction (multiplier = 2^n, where n = number of splits). Extrapolated counts were used for all subsequent abundance-based metrics. Samples were classified as pre-WID, 3 days post-WID, or 4 weeks post-WID and by demonstration (spring WID and fall WID). Impact sites were ordered by distance from Tuttle Creek Dam to evaluate whether biological responses attenuated downstream. All analyses were conducted in R version 4.5.2 (R Core Team, 2026).

2.5.1 | Community-level abundance and richness

To characterize how abundance and taxonomic richness changed following WID, community-level responses were evaluated using total counts, raw taxonomic richness, and rarefied taxonomic richness. Total count was calculated as the sum of extrapolated counts across taxa, and raw richness was calculated as the number of taxa represented by at least one individual. Rarefied richness was also used because our subsampling routine was adjusted based on the amount of debris in the sample, rather than the number of individuals. Thus, rarefaction allowed us to standardize sampling effort by the number of individuals in a sample rather than the number of splits. Values were estimated with the *rarefy* function in the vegan package (Oksanen et al. 2018) to a standardized count of 75 individuals; three samples from the control site were below this threshold and were excluded from rarefied richness analyses. Site- specific

baseline 90th percentiles were calculated from all pre-WID samples in which to compare post-WID deviations from the “typical” range of values.

2.5.2 | Log-response ratio

Log-response ratios (LRR) were calculated for total counts, raw taxonomic richness, and rarefied taxonomic richness to quantify the direction and magnitude of change relative to pre-WID (baseline) conditions. LRRs were treated as effect-size measures rather than formal hypothesis tests because samples were not replicated within sites during each sampling event.. Baseline values were calculated separately for each site as the median of all pre-WID samples pooled across baseline sampling dates. Each post-WID value was then compared with the corresponding baseline value. For richness and total count metrics, LRR was calculated as:

$$LRR = ln(Post / Pre)$$

For counts of individual taxa, LRR was calculated as:

$$LRR = ln((Post + 1) / (Pre + 1))$$

where a pseudocount of 1 was added to both terms prior to log transformation to accommodate zero counts. A LRR of zero indicated no change from baseline conditions, whereas negative and positive values indicated decreases and increases. Log-response ratio values were converted to the corresponding percent change from pre-WID conditions as:

$$\%\ Change = 100\% \times e\,(LRR - 1)$$

Where positive values represent increases and negative values represent decreases relative to pre-WID conditions. Site-level average LRR was calculated across all taxa, and 95% confidence intervals were estimated to describe variability among taxon-level responses. For visualization, the 25 taxa with the largest average LRR across sites and post-WID events were selected, allowing taxa with strong positive or negative responses to be displayed.

2.5.3 | Community composition

To test whether macroinvertebrate community composition shifted following WID relative to baseline composition, variation among samples was quantified by Bray–Curtis distances and visualized by non-metric multidimensional scaling (NMDS). Three separate NMDS ordinations were conducted to evaluate shifts in taxonomic (family level), functional feeding group, and habit composition. Treating taxonomic and trait compositions as separate response matrices allowed assessment of whether WID altered not only which taxa were present but also the functional structure of the community. Prior to all analyses, count data were log (x + 1) transformed. For the taxonomic ordination, taxa occurring in three or fewer samples were removed to reduce the influence of infrequently encountered or rare taxa (Table 1); no equivalent filter was applied to the FFG or habit compositions. Differences in community composition among WID sampling events (baseline vs post-WID (3 days post-WID, and 4 weeks post-WID)), demonstrations (fall and spring), and along the downstream gradient from Tuttle Creek Reservoir (km from dam) were tested using permutational multivariate analysis of variance (PERMANOVA) using the *adonis2* function (10,000 permutations). The primary additive model included treatment, season, and distance from dam, and separate interaction models tested whether treatment effects differed by demonstration, varied with distance, or whether each demonstration varied with distance. WID treatment, demonstrations, and distance from dam were each treated as categorical factors.

3 | Results

3.1 | WID effects on water physiochemistry

During WID operations, turbidity increased at all three USGS gages although the magnitude of turbidity varied between demonstrations and among sites (Figure 3). During fall

2025, maximum turbidity reached approximately 1,000 FNU in the Big Blue River, 550 FNU at Wamego, and 320 FNU at De Soto. The spring 2026 response was larger, with turbidity reaching approximately 1,900 FNU at Big Blue, 1,400 FNU at Wamego, and 650 FNU at De Soto. Reservoir releases ranged from approximately 57 to 113 $m^3 s^{-1}$ during the fall demonstration, whereas spring releases ranged from 42 to 283 $m^3 s^{-1}$.

3.2 | Community-level abundance and richness

Raw richness, rarefied richness, and total counts demonstrated site- and WID event-specific deviations from pre-WID conditions (Figure 4). Spring 3-day post WID samples produced the most consistent declines relative to baseline sampling, with raw and rarefied richness falling below or to the lower boundary of the 5th percentile at all impacted sites (Figure 4A–B). By 4 weeks post-spring WID, richness had returned to or exceeded the pre-WID median at four of the impact sites downstream of the Big Blue-Kansas river confluence. In the Big Blue River, raw richness remained low, but when considering sampling effort, richness had recovered to above the median. The control site showed consistently low counts and raw richness metrics relative to controls following both fall and spring WID demonstrations. Low counts at the control site prohibited the generation of rarefied richness in three of the four post-WID sampling events. Total counts showed no consistent directional response across the downstream gradient of impacted sites (Figure 4C). The control site had the most pronounced decline, with the 4-week post-fall sample falling below 5th percentile band. In contrast, 3-day post-spring WID samples at Big Blue, Downstream Confluence, Wamego, and Topeka exceeded the pre-WID median, counter to the negative richness responses observed at those same sites and sampling events.

3.3 | Log-response ratios

Log-response ratios showed that responses differed between WID events and among community metrics, sites, and taxa, with the clearest negative pattern occurring three days after spring WID. Fall responses were primarily driven by declines in total counts, whereas spring responses were characterized by initial increases in total counts and richness loss (Figure 5). Following fall WID, raw and rarefied richness generally remained near or above pre-WID conditions, whereas total counts declined across all sites during both fall post-WID sampling events. The greatest declines in richness and counts occurred at the control site four weeks after fall WID, where raw richness decreased by approximately 76% (LRR = −1.43) and total counts decreased by approximately 96% (LRR = −3.20). However, low counts prevented reliable estimation of rarefied richness at this site for all post-WID samples except the spring sample collected three days after WID. Spring WID produced a contrasting short-term response, characterized by richness loss but increased total counts at several sites. Three days after spring WID, raw and rarefied richness declined across all six sites, with reductions exceeding 50% at multiple sites. Despite these losses, total counts doubled or more at four of the five impact sites (i.e., Big Blue-Topeka). By four weeks post WID, total counts had declined below baseline at every site, whereas rarefied richness had increased above baseline at all downstream sites.

Averaged within sites, taxon-level LRRs were generally near zero with most 95% confidence intervals overlapping zero, indicating no consistent direction of response among taxa within a site (Figure 6A). Within this variability, however, mean responses differed between WID events. Three days after spring WID, mean LRRs were negative at five of the six sites, corresponding to mean reduction of approximately 30–50% from baseline; the strongest decline occurred at Wamego (~50%), while Topeka was the only site with a positive response (~28%).

By four weeks, mean responses had shifted toward zero or become positive, remaining slightly negative only at Control and Big Blue. Following fall WID, in contrast, mean LRRs relative to pre-WID baseline conditions were positive at all five downstream impact sites at both three days and four weeks. At the control site, the mean response was near baseline after three days but was approximately 65% below baseline after four weeks. Because most taxa declined following spring WID, the increases in total counts shown in Figure 5 reflected a few abundant taxa rather than a community-wide response.

Taxon-level responses were highly variable, with broad declines three days after spring WID offset by pronounced increases in a few taxa, particularly Copepoda (Figure 6B). Negative responses were concentrated among a few taxa: Corixidae was the most consistent, with mean declines exceeding 95% during all four post-WID events, and Baetidae and Caenidae showed the largest single-event declines, decreasing by more than 99% on average three days after spring WID. Ceratopogonidae, Chironomidae, Oligochaeta, and several other taxa also had negative mean LRRs in every event, though magnitudes varied. Positive responses were fewer but pronounced: Copepoda showed the strongest increase across sites, particularly three days after spring WID (mean LRR = 7.0), with Heptageniidae and Simuliidae also responding positively during several events. These results suggest that the site-level patterns reflect a mixture of taxon-specific increases and decreases rather than a uniform community response.

3.4 | Multivariate community composition

Across all three ordinations, site-specific pre-WID envelopes overlapped broadly when baseline years were pooled, indicating limited overall spatial separation (Figure 7). However, ordinations separated by year showed clearer spatial structure, particularly the distinct composition of the Big Blue River relative to the Kansas River sites (Appendix Figure B1). The

clearest compositional shift occurred three days after spring WID, when samples from all six sites fell outside their site-specific baseline envelopes in the taxonomic, functional feeding group, and habit ordinations. In comparison, fall communities that fell outside the baseline were limited to one or two sites within each ordination and were not consistent among response types. This seasonal difference was supported by PERMANOVA, with Treatment × Season interactions explaining 6.0% of the variation in taxonomic composition ($F = 2.66$, $p = 0.002$) and approximately 10% of the variation in both functional feeding-group ($F = 4.42$, $p = 0.001$) and habit composition ($F = 4.62$, $p = 0.001$; Table 2). One exception was the control site, which remained outside the overall pre-WID ordination space four weeks after fall WID.

Taxonomic composition showed the most pronounced response following spring WID. Five sites remained outside their site-specific baseline envelopes four weeks after spring WID, indicating that community composition at those sites had not yet returned to the range observed before dredging. The initial spring shift occurred primarily along NMDS1 and was associated with greater representation of Daphniidae and Copepoda (Figure 7A, D). Functional feeding-group and habit composition also changed immediately after spring WID, but these responses were persistent. Functional composition followed a similar pattern three days post spring-WID; however, most sites returned to their respective baseline envelopes within four weeks, suggesting that these functional changes were generally short-lived. In the functional feeding-group ordination, weighted averages of collector-filterers were separated most strongly along negative NMDS1 at sites Big Blue, Downstream Confluence, and Wamego. Shifts in habit groups coincided mostly with increases in planktonic taxa.

4 | Discussion

Water-injection dredging at Tuttle Creek Reservoir was associated with short-term changes in downstream physicochemical conditions. These changes were largely confined to the dredging period, although the magnitude and persistence of exposure varied among sites and between demonstrations. During WID operations, turbidity in the Big Blue River increased by as much as 20-fold relative to baseline conditions, with elevated sediment concentrations detected 198 km downstream at De Soto, Kansas. Visual observations suggest that fine sediment deposition occurred after WID at all five impact sites, with deposition most apparent in the Big Blue River, along shoreline margins, and other low-velocity habitats.

Although WID clearly increased downstream suspended and deposited sediment conditions, natural flooding and associated turbidity in the Kansas River during or following the WID demonstrations diluted some of the physicochemical effects. Moreover, biological responses were difficult to attribute solely to dredging. During the fall WID, turbidity in the Big Blue River was lower than in the Kansas River upstream of the confluence, where natural high-flow conditions contributed additional suspended sediment beyond that mobilized from WID. This coincided with deviations from baseline conditions at the control site, four weeks after fall WID. Following the spring experiment, elevated turbidity associated with high reservoir releases persisted after dredging ended, overlapping with naturally elevated suspended-sediment concentrations following flooding in the Kansas River. Unlike the fall event, however, the spring sediment signal from the Big Blue River had greater contribution to downstream conditions relative to the Kansas River upstream of the confluence. This signal was followed by prolonged reservoir releases that exceeded the range of baseline flow conditions at this site. The macroinvertebrate community shift observed three days after the spring WID therefore followed

a period of both increased sediment exposure and sustained high flows. While the timing of this shift corresponded with WID, the concurrent flow conditions make it difficult to determine whether the response reflected sediment mobilization, elevated flows, or their combined effects.

Differences in macroinvertebrate responses between the fall and spring WID events were not unexpected given the natural variability of large-river communities (Mathers et al., 2017; Mathers et al., 2023; Atristain et al., 2024). However, differences between the two WID events cannot be attributed to season alone. This variability was evident in the baseline community data, which showed that macroinvertebrate composition differed across both sites and years before dredging began (Appendix B1). Spatial differences were most pronounced in the Big Blue River, which continuously supported communities that differed from the Kansas River sites, especially in 2023, the year with the lowest mean annual discharge on record. Under the higher-flow conditions of 2024, the Big Blue assemblage showed slight overlap with the Kansas River assemblage at Wamego but remained largely distinct. This separation became less visually pronounced in 2025 with the addition of the Downstream Confluence site, whose intermediate position likely reflected the mixing of Big Blue and the mainstem Kansas River reach, upstream of the confluence, which received no WID sediment. Nevertheless, the Big Blue River remained distinct from Control and most downstream Kansas River sites, indicating that its taxonomic signature persisted across contrasting hydrologic conditions.

The temporal component of our prediction was better supported than the spatial component. Multiple community metrics declined shortly after WID and returned to or exceeded baseline conditions within several weeks. This was consistent with our prediction of a short-term response and with Pledger et al. (2021), who documented initial declines in macroinvertebrate abundance, richness, and diversity following WID, followed by recovery to control conditions

within five months. Thus, the timing of the response generally matched our expectations. However, across response metrics, the clearest signal following WID was a disproportionate increase in planktonic taxa, Copepoda and Daphniidae. The disproportionate increase in planktonic taxa following spring WID may partly reflect seasonal dynamics within the upstream reservoir. Zooplankton assemblages in Great Plains reservoirs can vary substantially, with increasing densities through spring and early summer and marked temporal shifts in *Daphnia* and copepod abundances (Wolfinbarger 1999; Bernot et al. 2004). These seasonal patterns suggest that the availability of planktonic taxa for downstream transport may have been greater during the spring WID event. Particularly since the WID pumped surface water, where zooplankton were presumably more abundant, to the bottom of the reservoir, incorporating plankton into density currents that exited the reservoir through the dam

Individual taxa showed varied responses to WID, but these responses only partly matched the sediment-related changes previously predicted. The broadest pattern of taxon-level declines occurred three days after spring WID, when Baetidae and Caenidae exhibited some of the most negative average log-response ratios (Figure 6B). Although an overall decline in Ephemeroptera was not predicted, individual mayfly families were expected to vary in their response to WID. Burrowing or interstitial taxa were expected to be less sensitive to fine-sediment deposition than taxa occupying more exposed substrate surfaces. However, this expectation was only partially supported. Ephemeridae and Potamanthidae responded positively during three of the four post-WID sampling events, but Ephemeridae declined three days after spring WID. Both families occurred primarily within the Big Blue River community, where elevated reservoir releases during spring WID produced substantial hydraulic disturbance. Thus, the negative spring response of Ephemeridae may have reflected displacement or disruption of benthic habitat

caused by increased discharge, rather than sensitivity to fine sediment alone. Although both groups were expected to tolerate or benefit from increased fine-sediment, Oligochaeta and Chironomidae declined following WID. Moreso, Corixidae showed the most consistent negative response, declining across all four post-WID sampling events. Corixids commonly occupy shallow river margins and areas of standing or slow-moving water (Karaouzas and Gritzalis, 2006). However, because Corixidae declined following both fall and spring WID, their response cannot be attributed specifically to the high spring discharge. It seems that responses likely depended on the interaction among local sediment exposure, habitat conditions, seasonal dynamics, and taxon-specific susceptibility.

The clearest example of a functional response was a shift towards greater relative representation of collector-filterers and planktonic taxa, corresponding with increases in Copepoda and Daphniidae. This response was not explicitly predicted, as WID was expected to primarily favor feeding groups associated with depositional habitats (e.g., collector-gatherers).The collector-filterer response was not limited to reservoir-associated zooplankton, as benthic collector-filterers also responded to WID, with generally positive responses by Simuliidae and more variable responses by Hydropsychidae. These increases occurred primarily following fall WID and therefore should not be interpreted as direct drivers of the spring ordination shift. Instead, they indicate that some collector-filterers tolerated or potentially benefited from conditions associated with WID. Higher current velocities can increase particle delivery and feeding opportunities for Simuliidae (Charpentier and Morin, 1994), while reservoir releases may transport fine particulate organic matter that provides additional food resources for downstream filterers (Doi et al., 2008). Hydropsychidae may also tolerate short-term increases in suspended sediment by cleaning or modifying their capture nets, and episodic sediment exposure

does not always reduce survival (Strand and Merritt, 1997; Runde and Hellenthal, 2000). Habit composition showed even less evidence of a response. One sample from the control site was instead associated with clinger taxa, but this site shifted across all ordinations despite receiving no WID-derived sediment. Its position therefore more likely reflects background temporal variation than a response to dredging.

Several limitations of our study design warrant consideration when interpreting the responses to WID. First, the upstream control site did not function as originally intended. Specifically, this site was influenced by natural flooding and responded asynchronously to the impact sites. We believe this response was driven by site-specific changes in habitat availability following high flows. These flows scoured a large sandbar, increased water depth and velocity, and washed downstream much of the previously available submerged wood. Consequently, post-flow sampling was conducted primarily in silty and vegetated margins rather than across the broader range of habitats available during baseline sampling. Hence, our total counts and raw taxonomic richness declined drastically at this site following the fall WID and limiting its usefulness as a control. Second, interannual variability was substantial across the study period and likely influenced the baseline conditions we used to evaluate WID responses. The 2023 sampling period occurred during the lowest mean annual flow on record, whereas 2025 and 2026 were comparatively high-water years. Because pre-WID baseline samples were pooled across years with different flow conditions, some apparent post-WID changes may reflect higher discharge post-WID relative to baseline conditions rather than dredging alone. Third, our sampling design relied on one composite macroinvertebrate sample per site and sampling event rather than collecting habitat-specific subsamples. Albeit we visually estimated the relative availability of major habitats within each reach and used these approximate proportions to

distribute sampling effort among habitats. This approach helped ensure that the dominant habitats within each reach were represented, but it did not allow us to quantify variation attributed to shifts in habitats availability within a site (e.g., as described above for the control site). Hence, we could not determine whether an observed change represented a consistent reach-wide response or reflected localized spatial variability. Fourth, baseline sampling was not distributed consistently across all months represented by the WID demonstrations. Specifically, baseline sampling was most consistent from June through September, whereas fewer observations were available from early spring and late fall when the WID demonstrations occurred. Because month-specific baseline data were too limited to support separate seasonal comparisons, we compared both WID demonstrations with the full multi-year baseline dataset. This approach provided a broader estimate of background variability but did not account directly for seasonal changes in macroinvertebrate communities. Hence, differences between baseline and post-WID samples may partly reflect seasonal turnover, particularly during periods when assemblages changed rapidly between adjacent months.

Future studies should more directly link sediment exposure to biological responses. Although suspended sediment was measured continuously during this study, deposited sediment was measured less frequently at transects near our study reaches. Future monitoring should measure deposition depth, particle-size composition, substrate embeddedness, and the persistence of deposited sediment at the same habitats where macroinvertebrates are sampled. These measurements would help distinguish biological responses associated with short-term exposure to the suspended sediment plume from those caused by longer-lasting changes to benthic habitat. Establishing these relationships would provide a stronger basis for developing operational thresholds and monitoring criteria for future WID projects.

5 | Conclusion

The results of this study provide an initial ecological benchmark for WID under moderate operating conditions in a naturally sediment-rich Great Plains River. Although WID released suspended sediment that subsequently deposited at our field sites, macroinvertebrate responses were limited and short-lived if present. These responses occurred during a deliberately conservative demonstration in which dredge operation and reservoir releases were closely controlled. Sediment concentrations remained within the natural total suspended solids–discharge range observed upstream of the reservoir, and no extreme water-quality deterioration or fish kills were documented. Thus, our results describe ecological responses within the natural sediment regime and may not extend to more intensive WID applications. At this point, it is not clear if WID will provide a viable approach for managing reservoir sediment and reintroducing trapped sediment to the downstream reach. Its ecological suitability, however, will likely depend on the intensity and timing of operations and on the capacity of the receiving river to transport and redistribute that sediment.

Figures

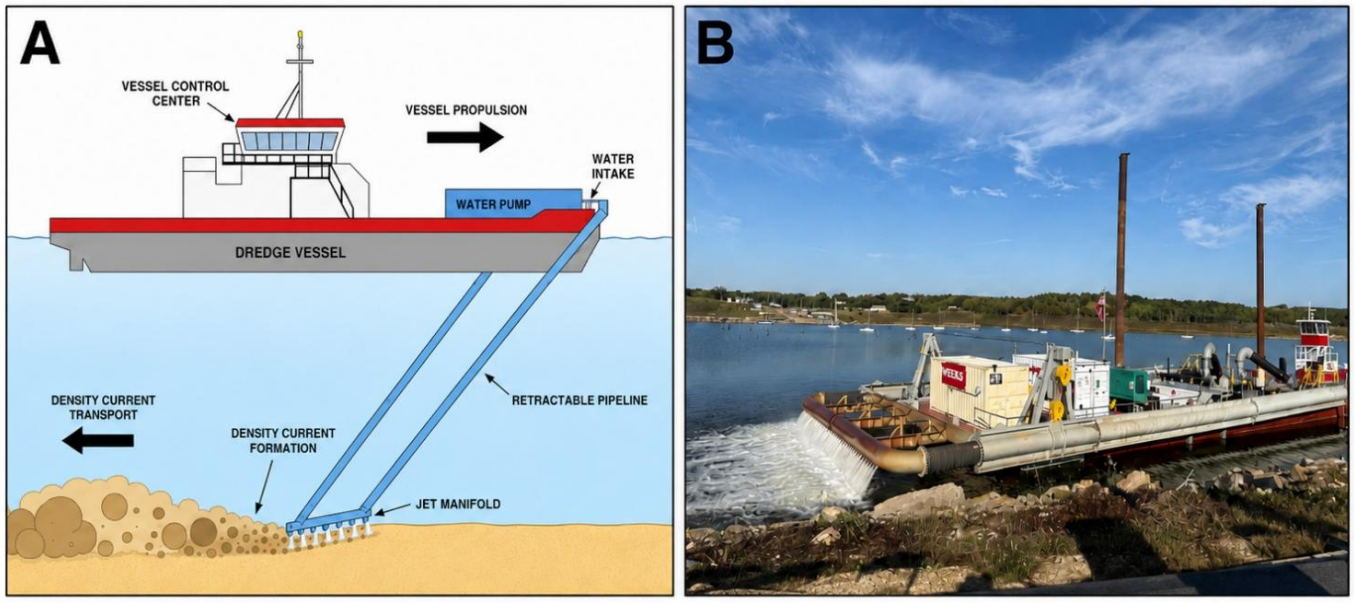

Figure 1: Conceptual and field representation of the water-injection dredging (WID) vessel. (A) Conceptual diagram of the primary components and operation of a WID vessel, modified from Wagner and Lewis (2022), showing water intake, pumping, jet manifold operation, density-current formation, and downstream density-current transport. (B) WID vessel created by Michels Construction and Custom Dredge Works shown in Tuttle Creek Reservoir. The retractable pipeline and jet manifold are shown positioned near the water surface, where low-pressure water jets are used.

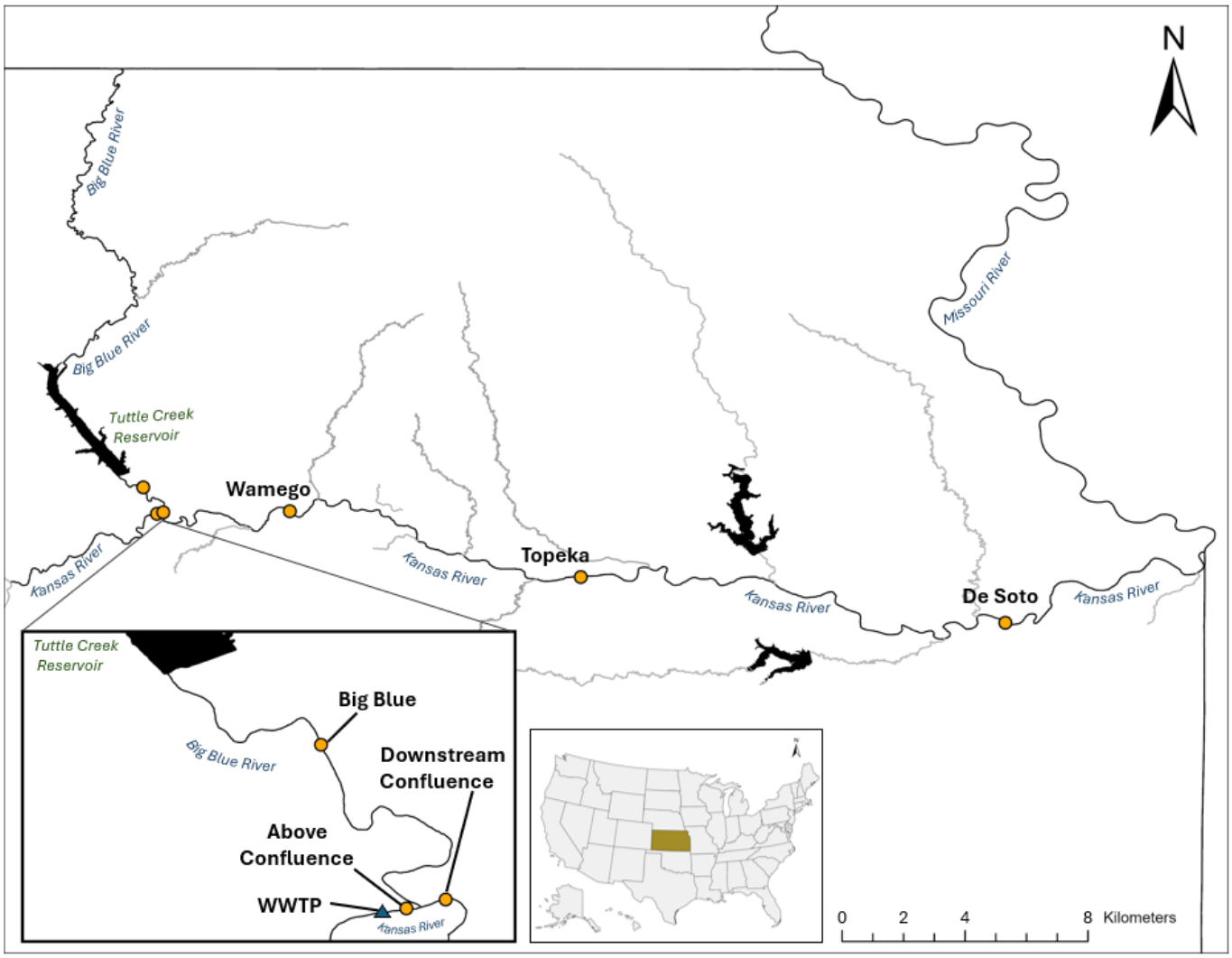

Figure 2: Map of the study area in northeastern Kansas, USA showing the location of six sampling sites along the Big Blue and Kansas rivers. Sites include one control site (Control) and five impact sites (Big Blue, Downstream Confluence, Wamego, Topeka, DeSoto). All macroinvertebrate sampling sites are indicated by orange circles, and the wastewater treatment plant (WWTP) is denoted as a blue triangle.

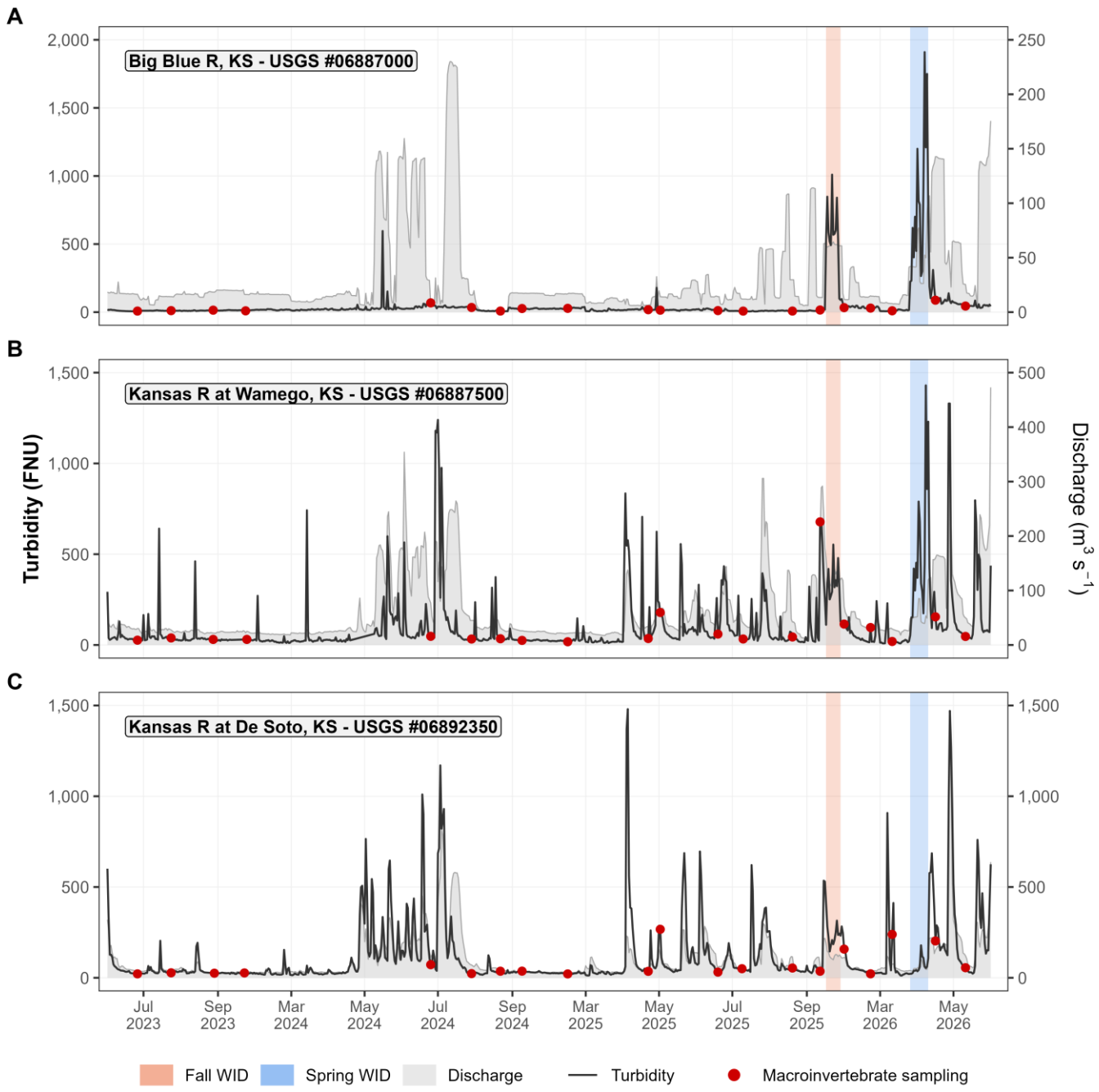

Figure 3:(A) Continuous time series of turbidity (formazin nephelometric units; FNU) and discharge at three USGS gaging stations downstream from Tuttle Creek Reservoir from June 2023 through June 2026. The time series provides hydrological and turbidity context for macroinvertebrate sampling and the timing of water-injection dredging (WID) events.

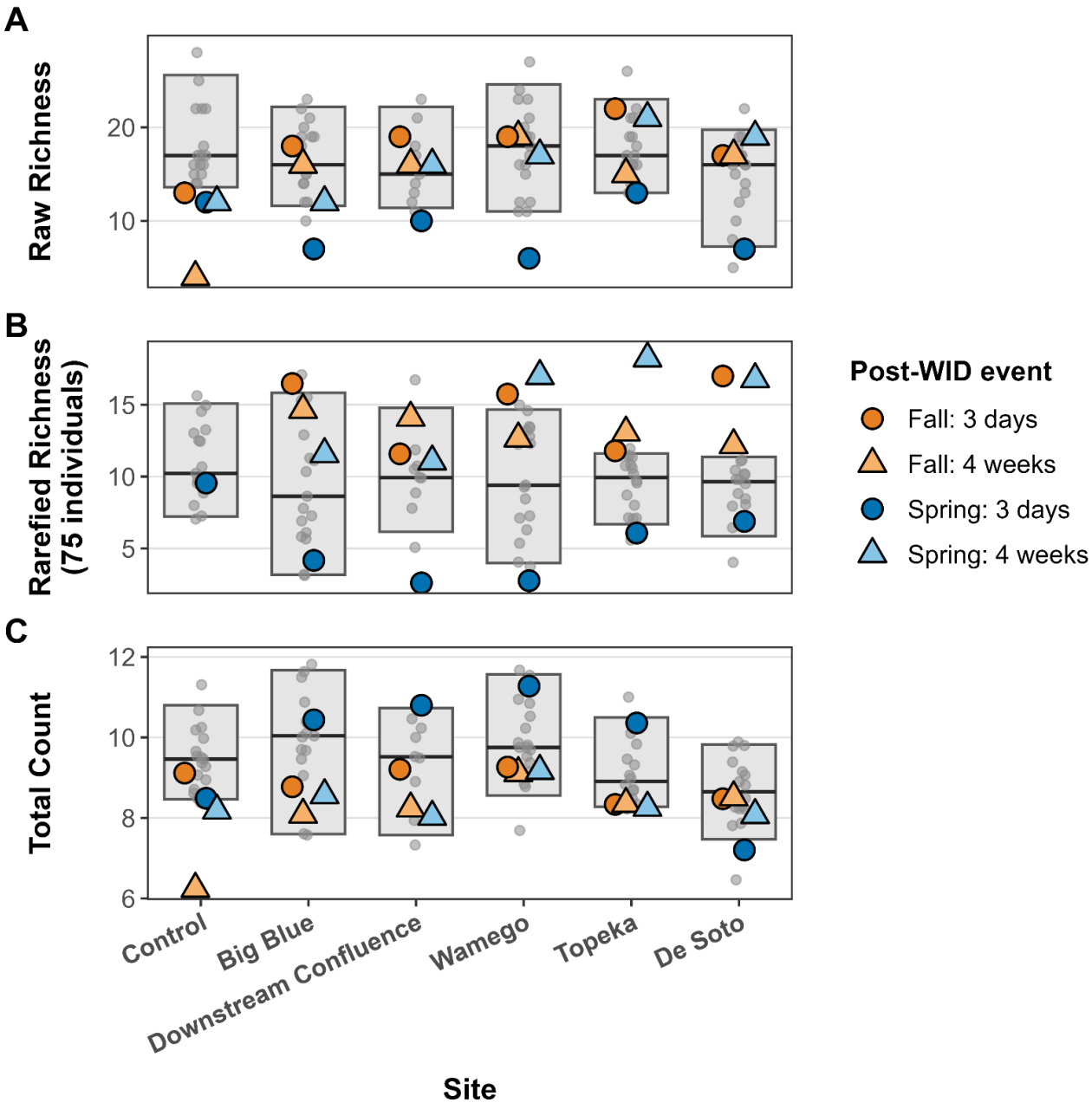

Figure 4: Variation in macroinvertebrate community metrics before and after water-injection dredging (WID) across six sampling sites. Panels show (A) raw taxonomic richness, (B) taxonomic richness rarefied to 75 individuals per sample, and (C) log-transformed total macroinvertebrate counts [ln(count + 1)]. Shaded grey band spans the 5th to 95th percentiles of the pre-WID samples at that site and the horizontal line marks the pre-WID median. Colored points are individual post-WID samples, coded by post-WID sampling event: Fall 3 days (orange circles), Fall 4 weeks (light orange triangles), Spring 3 days (blue circles), and Spring 4 weeks (light blue triangles).

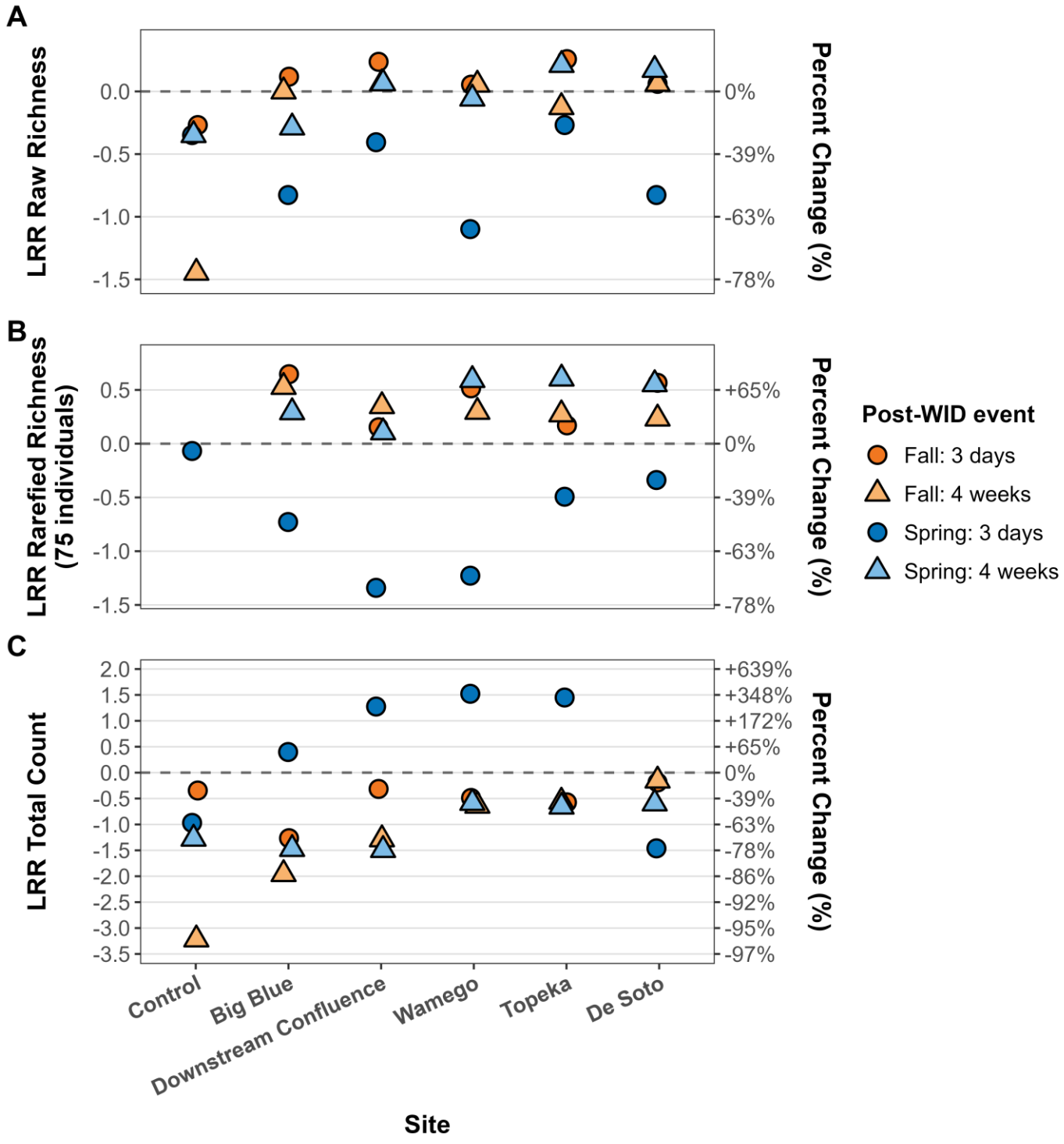

Figure 5: Log-response ratios (LRR) of macroinvertebrate community metrics following water-injection dredging (WID) across six sampling sites. Panels show (A) raw taxonomic richness, (B) taxonomic richness rarefied to 75 individuals per sample, and (C) total macroinvertebrate counts [ln(count + 1)] across six sampling sites. Low counts prevented Rarefied richness estimates on some occasions. Points represent post-WID values relative

to the site-specific pre-WID median. The first y-axis shows LRR, and the secondary y-axis shows the corresponding percent change from pre-WID conditions, calculated as (% $change = 100\% \times e\ LRR - 1$). Points represent post-WID metrics calculated relative to pre-WID conditions at each site. Colors distinguish fall and spring WID events, and shapes distinguish 3-day and 4-week post-WID sampling events.

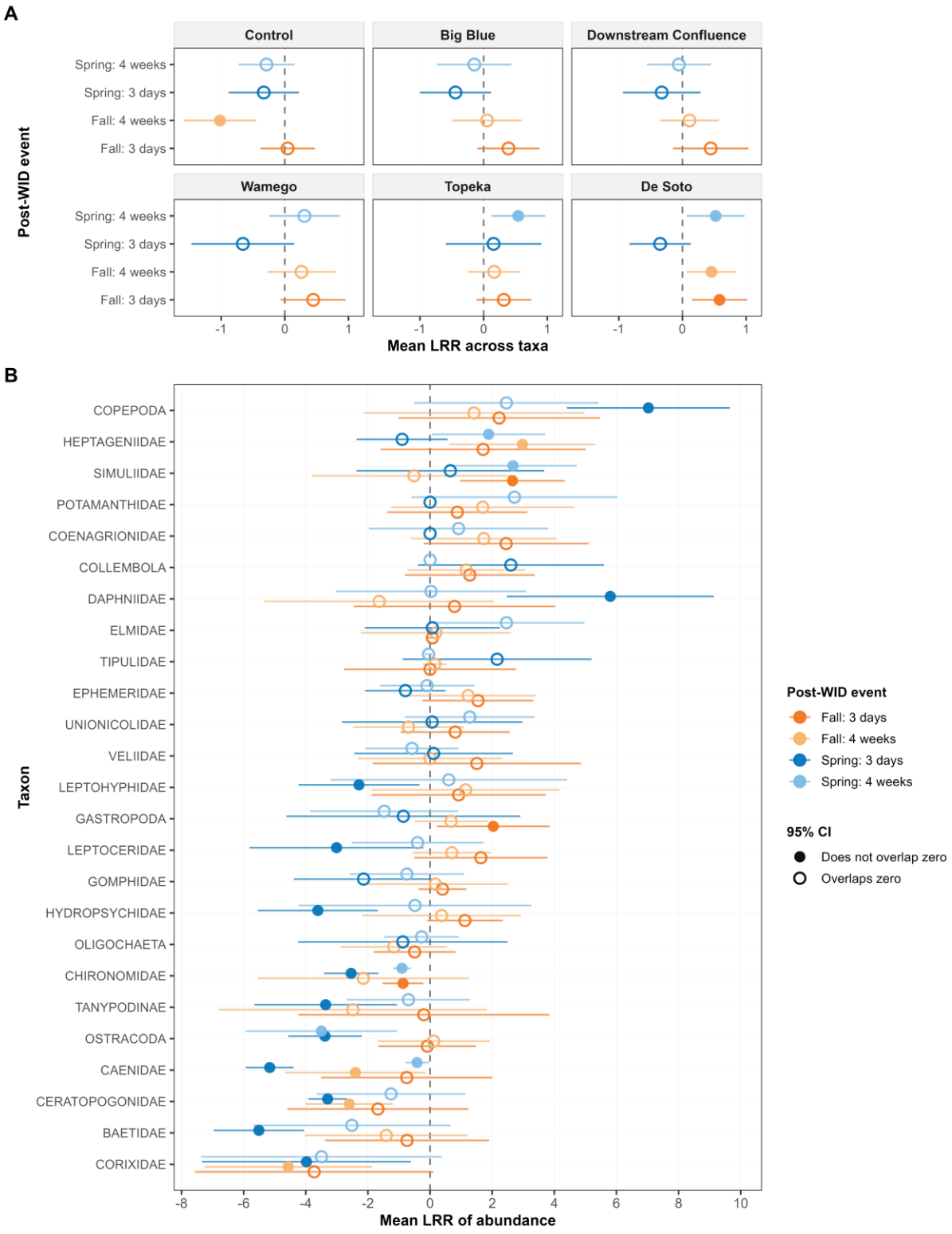

Figure 6: Log-response ratios (LRR) of macroinvertebrate counts (number per hour sampling) following water-injection dredging (WID), summarized at the site level (A) and

the taxon level (B). Colors distinguish post fall and spring WID events:: fall 3 days (dark orange), fall 4 weeks (light orange), spring 3 days (dark blue), and spring 4 weeks (light blue). Closed circles indicate that 95% confidence intervals did not overlap with 0 while open circles indicate 95% confidence intervals overlapped with zero. (A) Mean taxon response within each site. For each site and post-WID event, the LRR values of all taxa were averaged to produce one site-level mean. Points show the mean response, and horizontal bars show the 95% confidence interval across taxa. (B) Mean response for each taxon. For each taxon and post-WID event, its LRR values across the six sites were averaged to produce one taxon-level mean. Points show the mean response, and horizontal bars show the 95% confidence interval across sites. The 25 taxa with the largest mean absolute responses are shown and ordered from the most negative overall mean response at the bottom to the most positive at the top. The dashed vertical line indicates no change from pre-WID conditions.

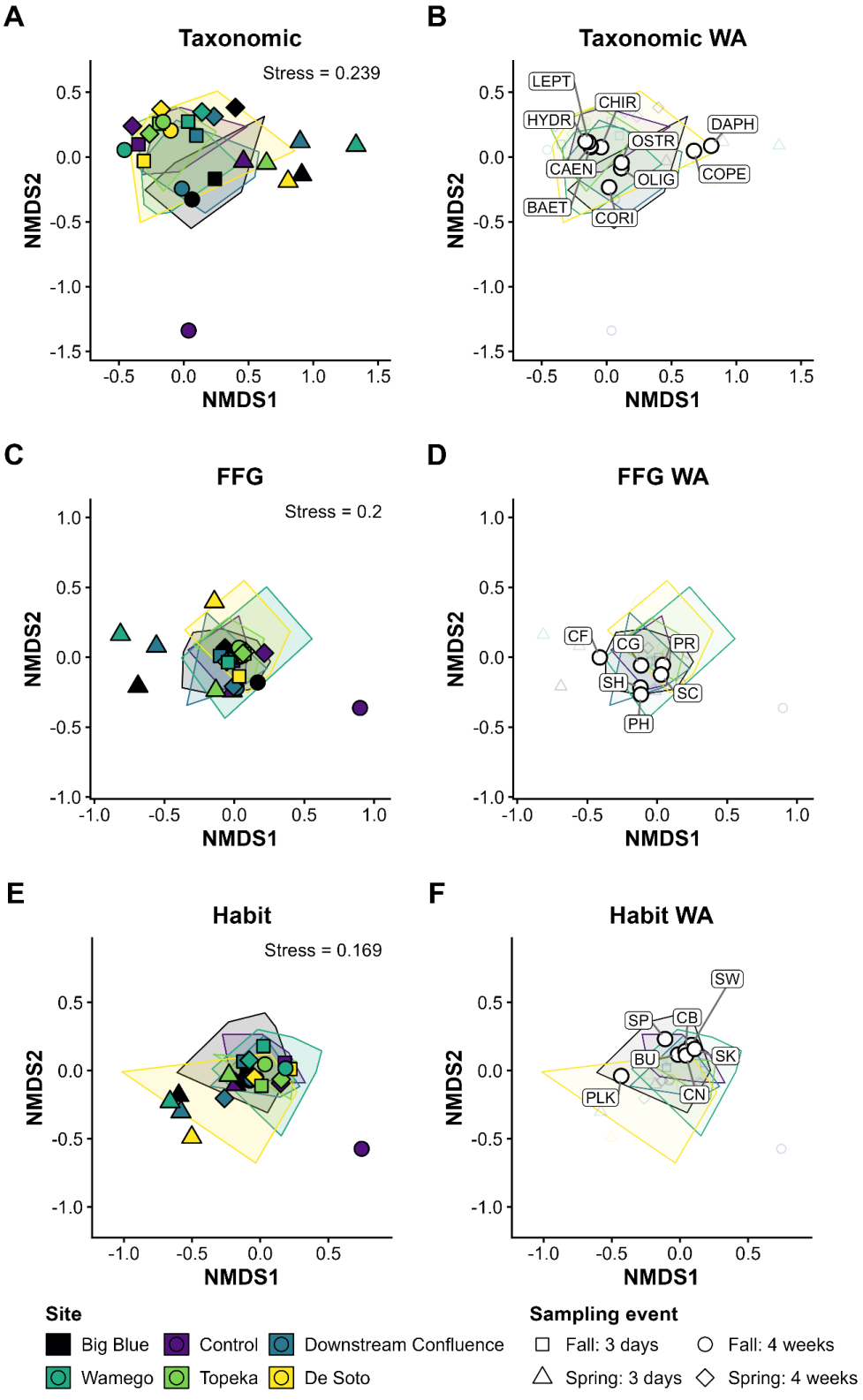
A
Taxonomic
Stress = 0.239
B
Taxonomic WA
LEPT
CHIR
HYDR
DAPH
OSTR
CAEN
COPE
OLIG
BAET
CORI
C
FFG
Stress = 0.2
D
FFG WA
CF
CG
PR
SH
SC
PH
E
Habit
Stress = 0.169
F
Habit WA
SW
SP
CB
BU
SK
PLK
CN
NMDS1
NMDS2
Site
Big Blue
Control
Downstream Confluence
Wamego
Topeka
De Soto
Sampling event
Fall: 3 days
Fall: 4 weeks
Spring: 3 days
Spring: 4 weeks

Figure 7: Nonmetric multidimensional scaling (NMDS) ordination plots of macroinvertebrate (A) taxonomic, (B) functional feeding group, and (C) habit composition before and after water-injection dredging at six sites based on a Bray-Curtis distance matrix. Semi-transparent polygons represent site-specific envelopes for pre-WID samples, providing the baseline range of community variation at each site and symbols represent post-WID samples by sampling event. Panels D–F show abundance-weighted average scores for the (D) dominant taxa, (E) functional feeding groups, and (F) habit groups in the same ordination space. Taxon labels are BAET= Baetidae, CAEN = Caenidae, CHIR = Chironomidae, COPE = Copepoda, CORI = Corixidae, DAPH = Daphniidae, HYDR = Hydropsychidae, LEPT = Leptoceridae, OLIG = Oligochaeta, and OSTR = Ostracoda. Functional feeding group codes are CF = collector-filterer, CG = collector-gatherer, PH = piercer-herbivore, PR = predator, SC = scraper, and SH = shredder. Habit codes are BU = burrower, CB = climber, CN = clinger, SK = skater, SP = sprawler, SW = swimmer, and PLK=planktonic.

Tables

Table 1: Macroinvertebrate taxa collected downstream of Tuttle Creek Reservoir, with functional feeding group (FFG) and habit assignments based on the Kansas Department of Health and Environment.

Class	Order	Family	Assignment (Taxa)	FFG	Habit
Arachnida	Acariformes	Unionicolidae	Unionicolidae	CG	CB
Bivalvia	—	—	Bivalvia	CF	BU
Branchiopoda	Cladocera	Daphniidae	Daphniidae	CF	PLK
Clitellata	Oligochaeta	—	Oligochaeta	CG	BU
Clitellata	Hirudinea	—	Hirudinea	PR	SP
Collembola	—	—	Collembola	CG	BU
Gastropoda	—	—	Gastropoda	SC	CB
Hexanauplia	Copepoda	—	Copepoda	CF	PLK
Insecta	Coleoptera	Dytiscidae	Dytiscidae	PR	SW
Insecta	Coleoptera	Elmidae	Elmidae	CG	CN
Insecta	Coleoptera	Gyrinidae	Gyrinidae	PR	SW
Insecta	Coleoptera	Haliplidae	Haliplidae	PH	CN

Class	Order	Family	Assignment (Taxa)	FFG	Habit
Insecta	Coleoptera	Hydrophilidae	Hydrophilidae	PR	SW
Insecta	Coleoptera	Noteridae	Noteridae	PR	CB
Insecta	Coleoptera	Scirtidae	Scirtidae	SC	CB
Insecta	Diptera	Ceratopogonidae	Ceratopogonidae	PR	SP
Insecta	Diptera	Chaoboridae	Chaoboridae	CG	SW
Insecta	Diptera	Chironomidae	Chironomidae	CG	BU
Insecta	Diptera	Chironomidae (Tanypodinae)	Tanypodinae	PR	SP
Insecta	Diptera	Ephydridae	Ephydridae	CG	BU
Insecta	Diptera	Muscidae	Muscidae	PR	SP
Insecta	Diptera	Psychodidae	Psychodidae	CG	BU
Insecta	Diptera	Simuliidae	Simuliidae	CF	CN
Insecta	Diptera	Tabanidae	Tabanidae	PR	SP
Insecta	Diptera	Tipulidae	Tipulidae	SH	BU
Insecta	Ephemeroptera	Baetidae	Baetidae	CG	SW
Insecta	Ephemeroptera	Caenidae	Caenidae	CG	SW
Insecta	Ephemeroptera	Ephemeridae	Ephemeridae	CG	BU
Insecta	Ephemeroptera	Heptageniidae	Heptageniidae	SC	CN
Insecta	Ephemeroptera	Isonychiidae	Isonychiidae	CF	SW
Insecta	Ephemeroptera	Leptohyphidae	Leptohyphidae	CG	SP
Insecta	Ephemeroptera	Palingeniidae	Palingeniidae	CG	BU
Insecta	Ephemeroptera	Potamanthidae	Potamanthidae	CF	BU
Insecta	Hemiptera	Belostomatidae	Belostomatidae	PR	CB
Insecta	Hemiptera	Corixidae	Corixidae	PR	SW

Class	Order	Family	Assignment (Taxa)	FFG	Habit
Insecta	Hemiptera	Gerridae	Gerridae	PR	SK
Insecta	Hemiptera	Naucoridae	Naucoridae	PR	CB
Insecta	Hemiptera	Nepidae	Nepidae	PR	CN
Insecta	Hemiptera	Notonectidae	Notonectidae	PR	SW
Insecta	Hemiptera	Pleidae	Pleidae	PR	SW
Insecta	Hemiptera	Saldidae	Saldidae	PR	CB
Insecta	Hemiptera	Veliidae	Veliidae	PR	SK
Insecta	Lepidoptera	Crambidae	Crambidae	SH	BU
Insecta	Megaloptera	Corydalidae	Corydalidae	PR	CN
Insecta	Odonata	Calopterygidae	Calopterygidae	PR	CB
Insecta	Odonata	Coenagrionidae	Coenagrionidae	PR	CB
Insecta	Odonata	Corduliidae	Corduliidae	PR	CB
Insecta	Odonata	Gomphidae	Gomphidae	PR	BU
Insecta	Odonata	Macromiidae	Macromiidae	PR	SP
Insecta	Plecoptera	Peltoperlidae	Peltoperlidae	PR	CN
Insecta	Plecoptera	Perlidae	Perlidae	PR	CN
Insecta	Trichoptera	Hydropsychidae	Hydropsychidae	CF	CN
Insecta	Trichoptera	Hydroptilidae	Hydroptilidae	SC	CB
Insecta	Trichoptera	Leptoceridae	Leptoceridae	CG	CB
Insecta	Trichoptera	Polycentropodidae	Polycentropodidae	CF	CN
Malacostraca	Amphipoda	—	Amphipoda	CG	SP
Malacostraca	Decapoda	Cambaridae	Cambaridae	CG	SP
Malacostraca	Isopoda	—	Isopoda	CG	SP
Ostracoda	—	—	Ostracoda	CG	SP

Class	Order	Family	Assignment (Taxa)	FFG	Habit
Turbellaria	Tricladida	—	Tricladida	CG	SP

Functional feeding groups (FFG): CG = collector-gatherer; CF = collector-filterer; PR = predator; SC = scraper; SH = shredder; PH = piercer-herbivore. **Habit:** SW = swimmer; BU = burrower; CB = climber; CN = clinger; SP = sprawler; SK = skater, and PLK=planktonic. **Notes:** Em dashes indicate that identification was not carried to that taxonomic level.

Table 2: Summary of PERMANOVA output testing effects of WID treatment, season, and distance from dam on taxonomic, functional feeding group, and habit composition. Additive models tested overall effects of treatment, season, and distance from dam. Interaction terms were tested in separate models to evaluate whether WID responses differed between seasons or varied along the downstream gradient. Models were based on Bray–Curtis dissimilarities of log (x + 1) transformed abundances.

Term	Taxa R^2	Taxa p	FFG R^2	FFG p	Habit R^2	Habit p
Treatment	0.083	**0.001**	0.114	**0.001**	0.100	**0.001**
Season	0.134	**0.001**	0.046	**0.010**	0.107	**0.001**
RM from dam	0.030	**0.012**	0.046	**0.014**	0.027	0.078
Treatment × Season	0.060	**0.002**	0.100	**0.001**	0.101	**0.001**
Treatment × RM from dam	0.021	0.590	0.000	0.999	0.029	0.302
Season × RM from dam	0.011	0.470	0.016	0.317	0.011	0.438
Treatment × Season × RM from dam	0.018	0.691	0.013	0.766	0.018	0.541

Note: Significant ($p < .05$) results are in bold

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Appendix A -

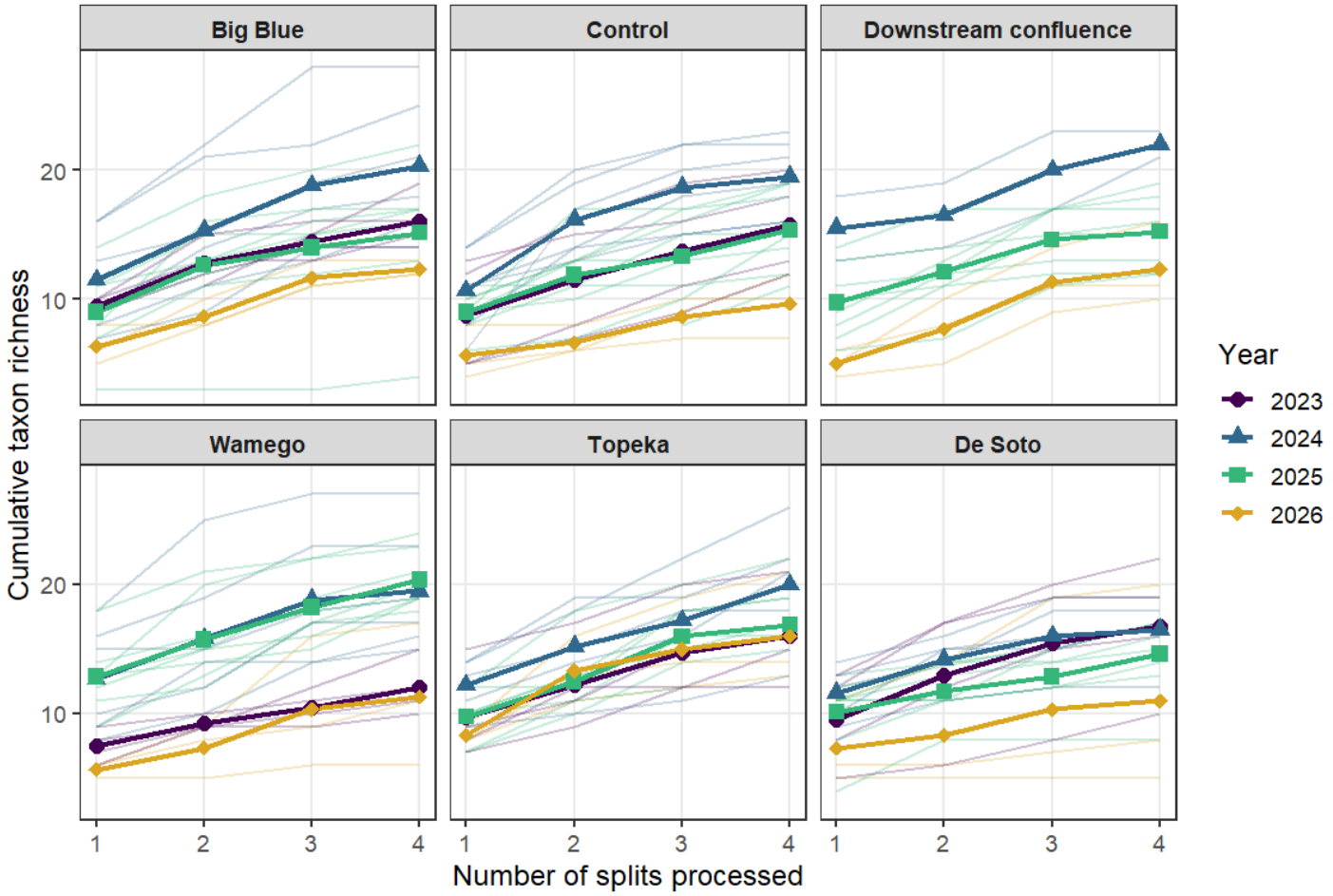

Figure A1: Cumulative taxon richness accumulation across increasing number of subsample splits for macroinvertebrate composite samples collected at six sites from 2023–2026. Each panel represents a sampling site, and each line shows the cumulative number of taxa detected as additional subsample splits were processed. Faint lines represent individual composite samples, while bold colored lines and points represent the mean cumulative richness for each year.

Appendix B -

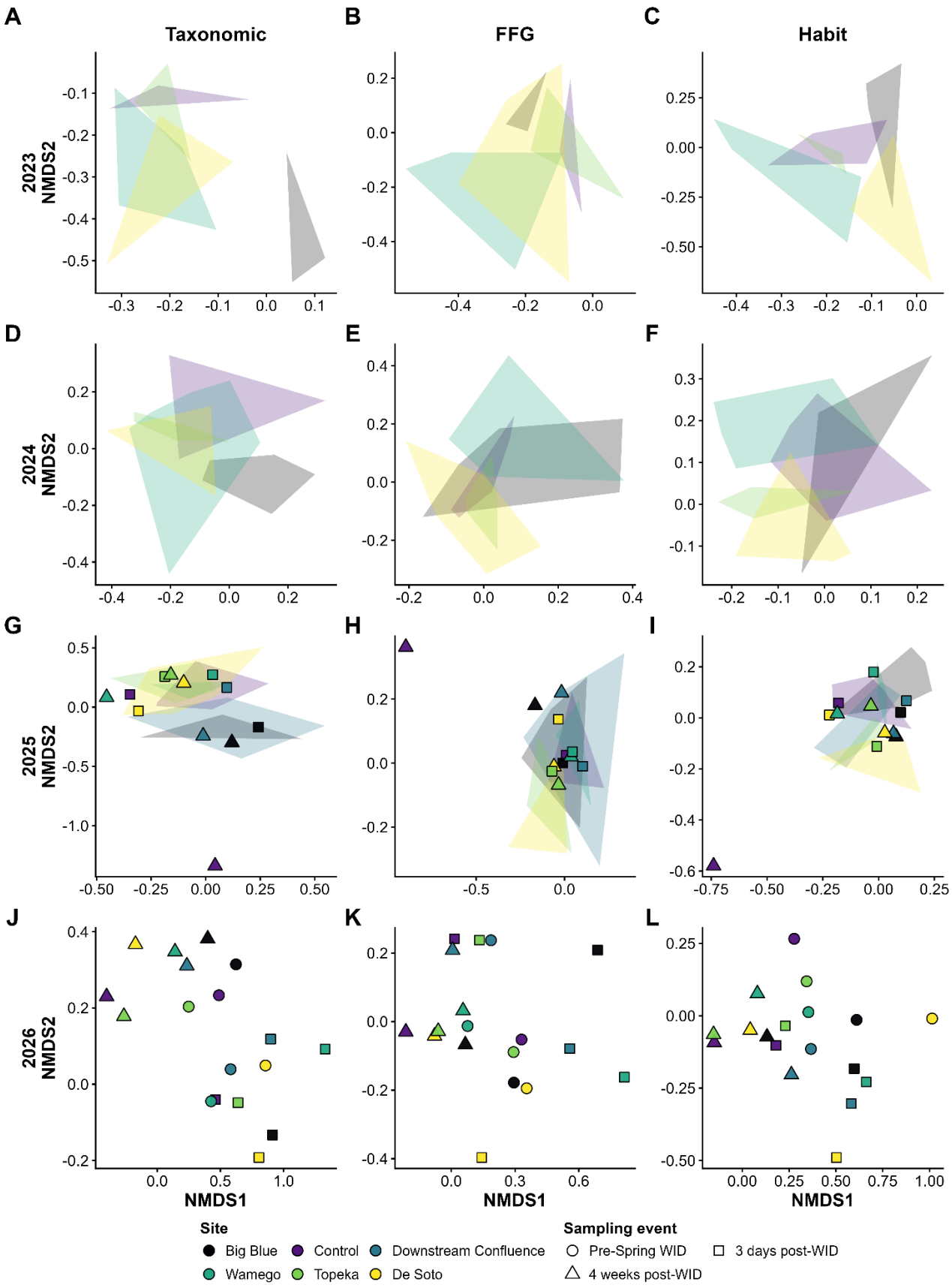
A
Taxonomic
B
FFG
C
Habit
2023 NMDS2
D
2024 NMDS2
E
F
G
2025 NMDS2
H
I
J
2026 NMDS2
K
L
NMDS1
NMDS1
NMDS1
Site
Big Blue
Control
Downstream Confluence
Wamego
Topeka
De Soto
Sampling event
Pre-Spring WID
3 days post-WID
4 weeks post-WID

Figure B1: Nonmetric multidimensional scaling (NMDS) ordination plots of macroinvertebrate taxonomic, functional feeding group, and habit composition from 2023–2026 before and after water-injection dredging (WID) at six sites based on Bray–Curtis dissimilarities. Semi-transparent polygons represent site-specific envelopes of pre-WID samples, illustrating the baseline range of community composition at each site. Symbols represent post-WID samples by sampling event, and colors indicate sites along the downstream gradient. Panels A, E, and I show taxonomic, functional feeding group, and habit composition for 2023; panels B, F, and J for 2024; panels C, G, and K for 2025; and panels D, H, and L for 2026. Only one month of pre-WID sampling was available in 2026; therefore, baseline convex envelopes are not shown for that year.