Pacific Northwest
NATIONAL LABORATORY

PNNL-35480

Action Effectiveness Monitoring and Research of Dredged Material Placement at Woodland Islands

Final Report

December 2023

Nichole K Sather
Kailan Mackereth
Narmadha Meenu Mohankumar

U.S. DEPARTMENT OF ENERGY

Prepared for the U.S. Department of Energy
under Contract DE-AC05-76RL01830

PACIFIC NORTHWEST NATIONAL LABORATORY
operated by
BATTELLE
for the
UNITED STATES DEPARTMENT OF ENERGY
under Contract DE-AC05-76RL01830

Printed in the United States of America

Available to DOE and DOE contractors from
the Office of Scientific and Technical Information,
P.O. Box 62, Oak Ridge, TN 37831-0062
www.osti.gov
ph: (865) 576-8401
fox: (865) 576-5728
email: reports@osti.gov

Available to the public from the National Technical Information Service
5301 Shawnee Rd., Alexandria, VA 22312
ph: (800) 553-NTIS (6847)
or (703) 605-6000
email: info@ntis.gov
Online ordering: http://www.ntis.gov

PNNL-35480

Action Effectiveness Monitoring and Research of Dredged Material Placement at Woodland Islands

Final Report

December 2023

Nichole K Sather
Kailan Mackereth
Narmadha Meenu Mohankumar

Prepared for
the U.S. Army Corps of Engineers
under project EST-P-19-01

Pacific Northwest National Laboratory
Richland, Washington 99354

Abstract

Benthic invertebrates play critical ecosystem roles including the breakdown of organic matter, sediment mixing, and nutrient cycling. In the lower Columbia River and estuary, benthic invertebrates provide a foraging resource for threatened and endangered juvenile salmon. Driven by the goal to create low velocity, shallow water, and riparian shrub habitats to benefit juvenile salmon, the USACE placed 237,000 CY of dredged material, resulting in the deposition of 13.5 acres of sand on the off-channel margins at Woodland Islands. The Pacific Northwest National Laboratory (PNNL) designed and implemented an action effectiveness research study to understand how dredged material placement at Woodland Islands affected sediment conditions and benthic invertebrate assemblages. A Before-After-Control-Impact, or BACI, study design was used to evaluate the response of environmental conditions and benthic invertebrates to dredged material placement. The design included one impact site and two control sites. Sampling occurred for two years prior to dredged material placement and two years after. Our study found that spatial variation was a significant factor for both environmental and biological response variables which suggests local conditions are important considerations for mechanisms affecting benthic assemblages. We found that off-channel habitats across all locations sampled were producing benthic invertebrates, many of which are common prey items for juvenile salmon and steelhead—e.g., insects, chironomids, crustaceans, and corophium. There was a significant BACI effect (i.e., an effect at the dredged material placement site, relative to conditions at the control sites) for concentration of carbon and ammonium and percent sand in sediment, but not for phosphorous. The estimated abundance for the three invertebrate response variables—total abundance, total chironomid abundance, and total corophium abundance—was significantly lower at the impact site after dredged material placement, compared to the control sites. At Woodland Island, the estimated mean abundance for all invertebrates combined decreased 28% after dredged material placement. Estimated mean abundances of chironomid and corophium decreased by 8% and 88%, respectively. While invertebrate abundances were lower after dredged placement, the composition of benthic invertebrates was similar before and after placement suggesting that as the new habitat feature evolves, recolonization will likely follow. These findings provide a foundation for understanding potential benefits and consequences of repurposing dredged material for habitat creation in the LCRE.

Summary

Introduction

The USACE is currently evaluating the potential of restoring and expanding shallow water habitats in the lower Columbia River and estuary (LCRE) with dredged material. Benthic invertebrates play critical ecosystem roles including the breakdown of organic matter, sediment mixing, and nutrient cycling. In the LCRE, benthic invertebrates provide a foraging resource for threatened and endangered juvenile salmon. The strategic use of dredged material can create opportunities to provide ecosystem services; benefits and values derived from human-ecosystem interactions. Yet the outcomes of these activities are uncertain and while the goals of dredged material habitat creation projects are aimed at positive outcomes for aquatic and terrestrial organisms, it is necessary to understand whether there are adverse effects. In the fall of 2020, the USACE placed 237,000 CY of dredged material on the off-channel margins at Woodland Islands. The dredged material footprint increased the off-channel shoreline perimeter by 0.7 mi and resulted in the deposition of 13.5 ac of sand. The goal of this placement project included creation of low velocity, shallow water, and riparian shrub habitats to benefit juvenile salmon (USACE 2017).

Methods

The Pacific Northwest National Laboratory (PNNL) designed and implemented an action effectiveness research study to understand how dredged material placement at Woodland Islands, a complex of Islands in the mainstem of the lower Columbia River, affects sediment conditions and benthic invertebrate assemblages. The research objectives included: 1) characterizing and quantifying environmental and biological conditions in off-channel habitats in the lower Columbia River and estuary; and 2) determining the before-after effect of dredged material placement on sediment conditions and benthic invertebrates. A Before-After-Control-Impact, or BACI, study design was used to evaluate the response of environmental conditions and benthic invertebrates to dredged material placement. The design included one impact site—Woodland Islands—and two control sites—Martin Island and Bachelor Island.

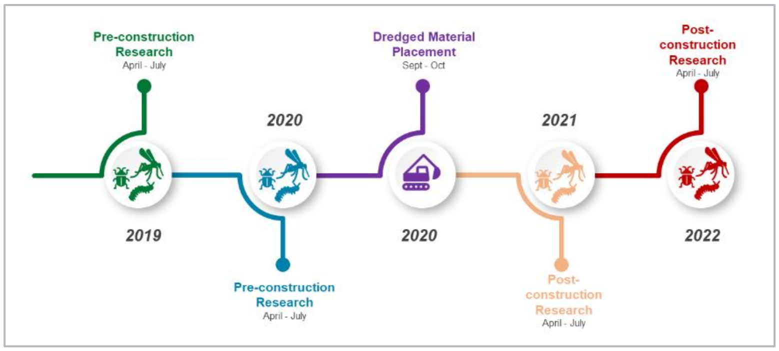

Figure 1. Research timeline of pre- and post-construction periods as well as dredged material placement.

The pre-placement sampling occurred monthly April through July during 2019 and 2020 (Figure 1). The dredged material was placed at Woodland Islands by the USACE during fall 2020. Post-placement sampling occurred monthly during April through July during 2021 and 2022. Benthic grab samples were collected to evaluate carbon and nutrient content as well as sediment grain size. Samples were also collected to understand benthic community composition and abundances in the off-channel habitats.

Environmental Results

We found evidence for significant site scale variability in carbon and ammonium concentrations as well as sediment grain size. For example, Martin Island, the location with the highest carbon and ammonium concentrations also had the highest average silt and lowest average sand content, whereas the trends in environmental conditions at Bachelor Island were opposite—lowest carbon and ammonium, lowest silt, and highest sand content. There was little evidence for interannual variation of the sediment nutrients and grain size data. However, the interactive effects of site and year suggest there are location specific responses to temporal variability. These findings indicate site-scale conditions are important considerations that likely affect environmental parameters within sediments of off-channel habitats in the region of the LCRE we investigated.

Benthic Invertebrate Results

Our study found that all off-channel habitats were producing benthic invertebrates, many of which are common prey items for juvenile salmon and steelhead—e.g., insects, chironomids, crustaceans, and corophium (Figure 2). Across all sites and years, more than 300,000 individual benthic invertebrates were collected. These invertebrates represented 88 unique taxonomic variables, including 66 invertebrate families, 11 classes, and eight phyla.

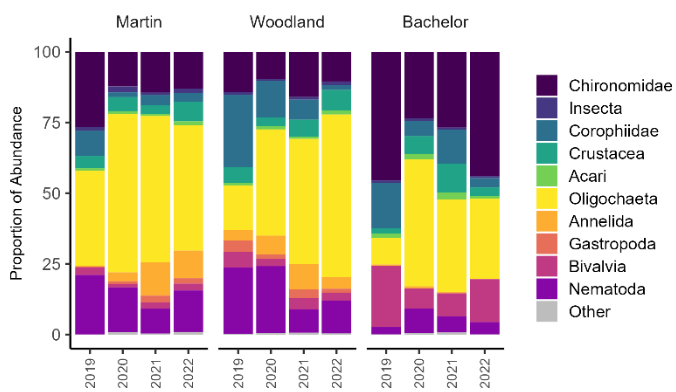

Figure 2. The proportion of the mean abundance of benthic invertebrates from 2019 – 2022 at Martin, Woodland, and Bachelor Islands.

Spatial and temporal factors were important in explaining variation in the benthic community in our study, but the sites explained more than twice the amount of variation in the data than the year. Given the predominance of taxa that are known prey items for juvenile salmon, these findings suggest that local conditions at some off-channel habitats may be important for augmenting foraging potential of migrating fish. While spatial factors appear to be predominant in governing benthic invertebrate community characteristics, we saw evidence of the influence of interannual variation on community dynamics when abundance of invertebrates declined at all sites during 2022. This year coincided with notably higher river discharge compared to the previous three years of study.

Before-After-Control-Impact Results

We evaluated the before and after effect of dredged material placement at Woodland Islands relative to conditions at the two control sites, Martin Island and Bachelor Island. There was a significant BACI effect for carbon, ammonium, and percent sand, but not for phosphorous. Reduced carbon and ammonium concentrations at the dredged placement site may reflect changes in productivity and flow that affect inputs of organic matter. The decrease in percent sand at the dredged material site may be explained by similarities between the material dredged from the Columbia mainstem and the material that existed at Woodland Island before placement. Alternatively, local effects to sediment grain size from dredged material placement may have been short-lived and were otherwise not detected when the monitoring resumed approximately six months after placement.

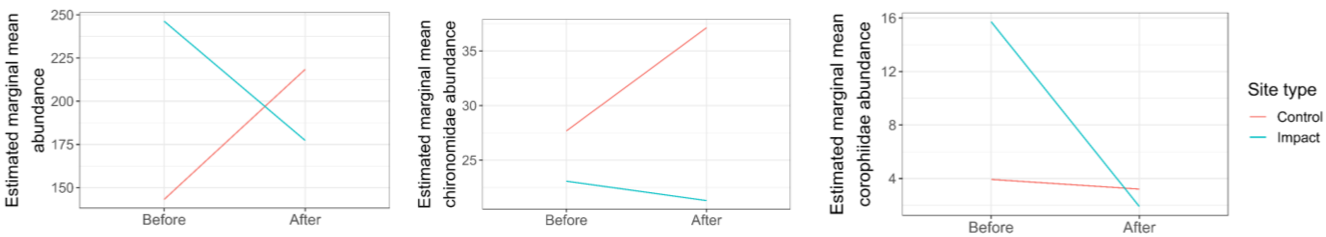

Figure 3. Total abundance for all invertebrates, chironomids, and corophium, before and after dredged material placement at the control and impact locations.

Our results indicated that the estimated abundance for the three invertebrate response variables—total abundance, total chironomid abundance, and total corophium abundance—was lower at the impact site two years after dredged material placement, compared to the control sites (Figure 3). The deposition of sediment and subsequent burial of invertebrates is a common mechanism affecting community response associated with dredged material placement. Due to the extent of dredged material placement at Woodland Islands, vertical migration by invertebrates was likely not possible for most organisms within the 13.5 acre footprint. Recolonization will likely occur via organisms that migrate from nearby habitats and may include horizontal migration, larval recruitment, or drifting/swimming organisms in the water column.

Conclusion

Benthic communities and some environmental variables were impacted by dredged material placement at Woodland Islands. Many factors interact to determine the magnitude and duration of environmental and biological responses associated with dredged material, and these factors translate into notable variation in the recovery of benthic communities. Literature on dredged material placement indicates that outcomes resulting in rapid recovery (weeks to months), or trajectories with slower responses (years) are possible. At Woodland Island, the estimated mean abundance for all invertebrates decreased 28% after dredged material placement. Estimated mean abundances of chironomid and corophium decreased by 8% and 88%, respectively. When considering the reduction in invertebrate abundances at Woodland Islands, it is important to note that the site was not completely devoid of benthic invertebrates following placement. Many of the same taxa persisted at the site following dredged material placement, compared to before placement, but we demonstrated through both univariate and multivariate analytical methods that the structure of the benthic assemblage at Woodland Islands changed. As the new habitat feature evolves, recolonization will likely follow. In addition to evaluating biological and environmental responses associated with dredged material placement, this study has demonstrated that off-channel habitats of the LCRE produce benthic invertebrates that are

important foraging resources for migrating juvenile salmon and steelhead. We have gained insights into considerations of spatial and temporal variation as well as methodologies and approaches for evaluating the effectiveness of dredged material projects focused on creating beneficial habitats. These findings provide a foundation for understanding potential benefits and consequences of repurposing dredged material for habitat creation in the LCRE.

Acknowledgments

This study was funded by the U.S. Army Corps of Engineers, Portland District. We are grateful for support and guidance from Ida M. Royer, Cynthia L. Studebaker, Mark Bierman, Chanda Littles, and Rod Moritz of the Corps. The study was coordinated through the Studies Review Work Group of the Anadromous Fish Evaluation Program and the Columbia Estuary Ecosystem Restoration Program's Expert Regional Technical Group. We appreciate the interest and constructive feedback provided by these groups during all phases of the study. We thank Adam Martin-Schwarze and Gary Johnson who contributed to the conceptualization and design of this study during the early stages of development. We are grateful to Marissa Despins, Regan Farrell, Madison Bowe, Chris Myers, and especially Eric Fischer, for their hard work and dedication in collecting and processing samples from the field. We thank Shon Zimmerman for lending his GIS expertise. Allison Myers-Pigg contributed to interpreting results of sediment carbon and nutrients and Maggie McKeon provided insights into sediment dynamics in the lower Columbia River and estuary. Benthic invertebrate samples were processed by EcoAnalysts, Inc., and sediment nutrients and grain size data were processed by Oregon State University's Central Analytical Laboratory. Thank you to Anita Waller, Kate Buenau, and David Geist who provided editorial and peer reviews of this report.

Acronyms and Abbreviations

AICc	Akaike Information Criterion corrected for small sample sizes
BACI	Before-After-Control-Impact
CEERP	Columbia Estuary Ecosystem Restoration Program
CI	confidence interval
GLMM	generalized linear mixed model
LCRE	lower Columbia River and estuary
nMDS	non-metric multidimensional scaling
NH_4	ammonium
NO_3	nitrate
PNNL	Pacific Northwest National Laboratory
PERMANOVA	permutational multivariate analysis of variance
USACE	U.S. Army Corps of Engineers

Contents

Figures

Tables

1.0 Introduction

The Columbia Estuary Ecosystem Restoration Program (CEERP) was developed to restore estuarine ecosystems in the lower Columbia River and estuary (LCRE) (Ebberts et al. 2018). A primary focus of CEERP includes restoring estuarine habitats and processes that benefit threatened and endangered salmon and steelhead, specifically juvenile life stages that use estuarine habitat for feeding, rearing, and growth before migrating to the ocean. Estuarine benthic conditions play an important role in the food web dynamics for juvenile salmonid and steelhead in the LCRE (Lott 2004, Haskell et al. 2006, Bottom et al. 2009). Factors such as seasonality (Spilseth & Simenstad, 2011), spatial differences (Roegner et al. 2010), habitat type (Sather et al. 2020), and fish size (Martin-Schwarze et al. 2018) all influence variation in juvenile fish diets. Despite these sources of variation, two recurring benthic taxa – Chironomidae and amphipods in the genus *Americorophium* – are prevalent in the diets of juvenile salmon across spatial and temporal gradients (Eaton, 2010; Haskell et al. 2006; Roegner et al. 2008; Storch & Sather, 2011; Weitkamp et al. 2022). In addition to supporting foraging conditions for juvenile salmon and steelhead, benthic invertebrates play critical ecosystem roles including the breakdown of organic matter, sediment mixing, and nutrient cycling (Merritt et al. 1984). And, due to their lesser mobility and longer life histories relative to other invertebrates, they are often studied as indicators of ecosystem health (Rosenberg & Resh, 1993).

Sediment processes in aquatic systems serve critical functions for habitat creation and protection, regulation of water quality conditions, storage and distribution of contaminants, and interactions with biological processes (Dyer, 1989; Welch et al. 2016). These processes interact with dredging operations managed by the U.S. Army Corps of Engineers (USACE), which are aimed at supporting safe vessel navigation and flood control management, providing recreation opportunities and water supply, and supporting hydropower operations (U.S. Army Corps of Engineers, 2015). Given the significance of dredging for navigation and non-navigation purposes and the importance of aquatic habitat for humans and organisms, the USACE has begun to investigate aspects of habitat development associated with using dredged material for the purposes of restoring and expanding shallow water habitats in the lower Columbia River. For example, Borde et al. (2012) found that some historic dredged material sites in the LCRE have evolved to resemble mature wetlands.

The strategic use of dredged material can create opportunities to provide ecosystem services; benefits and values derived from human-ecosystem interactions (Foran et al. 2018; Spellman, 2023). Examples of benefits associated with dredged material include creation and restoration of habitats, carbon and nutrient sequestration, navigation support, storm protection, increased recreational opportunities, and reduction of dredging placements and maintenance costs (Foran et al. 2018; Suedel et al. 2022; Welch et al. 2016). Historically, USACE navigation programs, across the country, have repurposed 30-40% of materials toward beneficial uses. New goals call for increasing these numbers to 70% by 2030 (Spellman, 2023). Terms used to describe the value of using dredged material for habitat creation and enhancement include beneficial use, Nature-based Solutions, and Engineering With Nature®. Regardless of the terminology, the potential outcomes resulting from these actions focus on benefits that span economics, the environment, and social values (Suedel et al. 2022).

Despite positive intent, there are known negative impacts associated with dredged material placement as well as uncertainties about the effectiveness of beneficial use efforts. The effects of dredged material placement includes negative consequences to benthic communities through direct burial, decreased diversity, and changes in abundance (Munari & Mistri, 2014). The size of these effects are often associated with the placement location and, driven by factors that affect succession and recolonization, can result in significant changes to benthic biota at dredged and placement sites (Bolam et al. 2006; Bolam & Whomersley, 2005). The significance of an effect is driven by the rate of recovery (Munari & Mistri, 2014). Post dredged material placement recovery rates range from months to years, where high latitudes generally have lower recovery rates compared to sub-tropical regions (Wilber & Clarke, 2007).

The USACE is exploring opportunities in the LCRE to use dredged material to create and restore shallow water habitat. The Woodland Islands were selected for a habitat creation project, using dredged material, to increase shallow water habitat available for salmonids. The Woodland Islands are located along the Washington side of the lower Columbia River at RM 84.5–86.0 (rkm 135.2–137.6). The islands were originally created by placement of dredged material in the river from the 1920s through the 1970s. The islands include wetlands and uplands and have diverse topography (USACE 2017). The goals for the Woodland Islands placement project included creating low velocity, shallow water, and riparian shrub habitats. The project was designed to complement the existing habitat matrix and to create shallow water and wetland habitats on the off-channel side of the island. In the fall of 2020, the USACE placed 237,000 CY of dredged material on the off-channel margins at Woodland Islands. The dredged material footprint increased the off-channel shoreline perimeter by 0.7 mi and resulted in the deposition of 13.5 ac of sand (USACE 2017).

In response to a critical uncertainty[1] about the effects of dredged material placement associated with beneficial use purposes, the Pacific Northwest National Laboratory (PNNL) designed and implemented an action effectiveness research study to understand how dredged material placement at Woodland Islands affects sediment conditions and benthic invertebrate assemblages.

The research objectives included characterizing and quantifying environmental and biological conditions in off-channel habitats in the lower Columbia River and estuary.

Environmental conditions:

1. Characterize the spatial and temporal variation in carbon, nutrients, and grain size in off-channel habitats.
2. Determine the before-after effect of dredged material placement on sediment conditions as measured by carbon, nutrients, and grain size at control and placement sites.

Benthic invertebrate conditions:

1. Characterize the spatial and temporal variation in benthic assemblages in off-channel habitats.
2. Determine the before-after effect of dredged material placement on sediment conditions as measured by total invertebrate abundance, total chironomid abundance, and total corophium abundance.

---

[1] The critical uncertainties associated with beneficial use of dredged material in the LCRE were identified through the Columbia Estuary Ecosystem Restoration Program and included feedback from the Expert Regional Technical Group as well as the National Marine Fisheries Service.

2.0 Methods

2.1 Study Area and Design

A Before-After-Control-Impact, or BACI, study design was used to evaluate the response of environmental conditions and benthic invertebrates to dredged material placement. The design included one impact site—Woodland Islands—and two control sites—Martin Island and Bachelor Island. The control sites helped to block for sources of non-site-level temporal variation (e.g., a high flow year) and create redundancy in the event sampling difficulties were encountered at a particular control site. The pre-placement sampling occurred monthly April through July during 2019 and 2020 (Figure 4). The dredged material was placed at Woodland Islands by the USACE during October 2020. Post-placement sampling occurred monthly during April through July during 2021 and 2022.

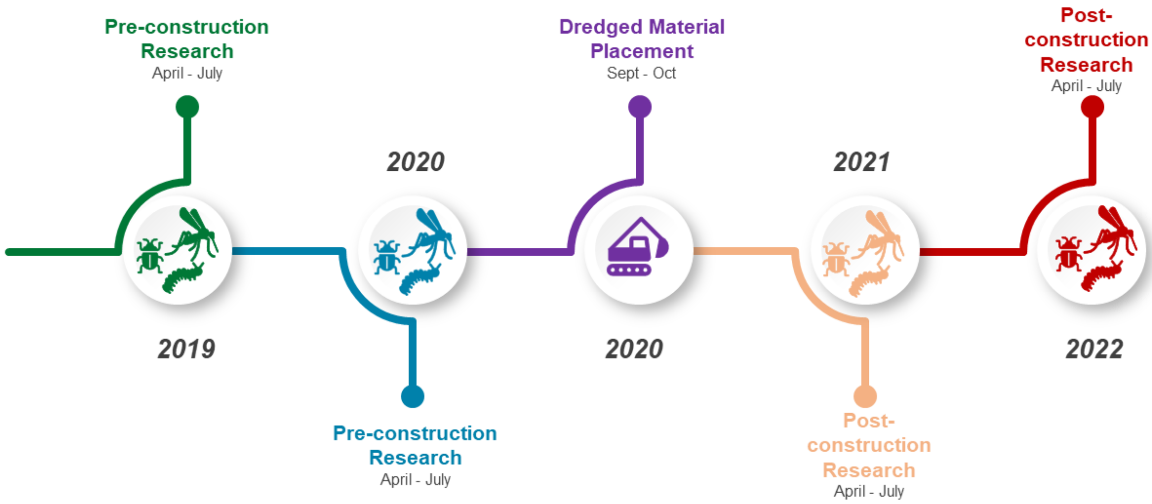

Figure 4. Research timeline of pre- and post-construction periods as well as dredged material placement.

Sampling areas (grey polygons in Figure 5) were defined at each of the three sites and stations were established to define and constrain the location for where benthic and sediment samples would be collected. Sampling area at the impact site was defined by the anticipated dredged material footprint. At the control sites, criteria for selecting the sampling area were driven by hydro-geomorphic features of the off-channel habitats that closely resembled those at the impact location. Examples included location of the sampling area relative to the island, channel width, length, and anticipated flow regime.

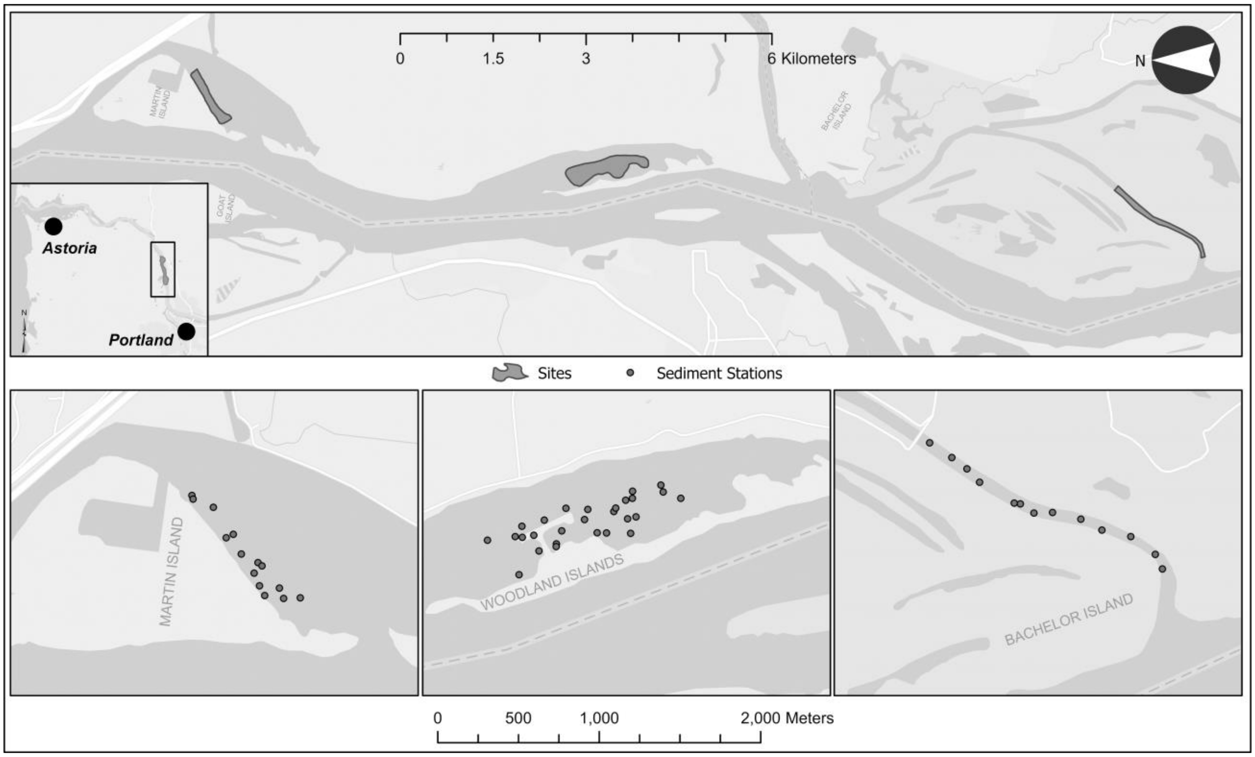

Figure 5. Sampling areas for the three sites (grey polygons). Paired benthic and sediment samples were collected from each station (grey circles) in off-channel aquatic habitats (channels behind mainstem islands). Woodland Islands (rkm 136–139) is the dredged material impact site, located across the Columbia River mainstem from St. Helens, Oregon. Martin Island (rkm 129–132) and Bachelor Island (rkm 142–148) were the control locations for the Before-After-Control-Impact study design.

Stations (green circles in Figure 5) were selected using a single randomized, spatially balanced (i.e., equal sampling density), unstratified design. The design was generated using a generalized random tessellation stratification (Stevens and Olsen, 2004) implemented through the R package spsurvey v. 4.0.0 (Kincaid et al. 2019). An unstratified design was selected for several reasons: (1) when the research was initiated, the final project design and construction footprint were uncertain; (2) while it made biological sense to consider stratification by elevation, there were inherent challenges associated with vessel navigation to underwater elevation zones and accurate bathymetry data were not available at the time the study began; and (3) the research inference focuses on the site as a whole, rather than subsets of the benthic community so sampling was allocated across the site, not according to composition across strata or zones within a site.

The design resulted in 54 stations across the three sites: 27 at Woodland, 14 at Martin, and 13 at Bachelor. Station numbers across the three locations yielded an equal sampling density based on the pre-defined sampling area at each of the three sites. The same 54 stations were sampled during each month and year of the study.

2.2 Field Data Collection

Paired benthic and sediment samples were collected monthly from April through July using a Petite Ponar® grab sampler (232 cm^2) at each of the three sites. GPS coordinates for each of the 54 stations were input into a Trimble Geo7x which was used on the vessel to navigate to the correct location for data collection. At each station, the first grab sampled served as the benthic sample. These samples were washed through a 500 µm sieve, onboard the vessel, using filtered (106 µm) river water. Samples were transferred to storage containers and preserved with 95% ethanol. A second grab was preserved for sediment analyses (grain size, carbon, nutrients). The contents of the second grab sample were emptied into a small tote where approximately 8 oz of sediment was removed, placed into a Whirl-Pak™ sampling bag, and placed on dry ice.

2.3 Laboratory Analysis

Sediment samples were processed by Oregon State University's Central Analytical Laboratory using laboratory standard operating procedures. Particle size was calculated using a hydrometer method. Nitrate and ammonium (NH_4-N) concentrations were calculated using a potassium chloride extraction method and quantified calorimetrically using a Lachat 8500 Series 2 Flow Injection Analysis System. Analysis of percent total carbon, nitrogen, and phosphorous concentrations (ppm) was conducted using inductively coupled plasma–optical emission spectrometry.

Benthic samples were enumerated to family with a target count of 300 organisms for all taxa combined, except for oligochaetes and nematodes. If subsampling was necessary (i.e., the target count was achieved before the sample had been fully analyzed), the counts were adjusted according to the proportion subsampled.

A primary interest of this research project includes understanding environmental and biological changes relevant to juvenile salmon prey resources. Nematodes and oligochaetes have been observed in diet analyses of juvenile salmonids; however, they can be numerically dominant in benthic samples. Including these taxa in the target count could have resulted in lost precision in the analysis for other taxa of interest in this study—e.g., amphipods and insects. For oligochaetes and nematodes, counts were derived by subsampling at least 25% of each sample and individuals were not included in the target count of 300 organisms. Adopting two sorting and enumerating techniques allowed us to balance the project needs and resources with research goals and objectives. All data analyses were based on adjusted counts which considered subsampling procedures.

2.4 Data Analysis

2.4.1 Environmental spatial and temporal variability

We examined the spatial and temporal variability of environmental conditions by modeling the effect of site and year on the environmental conditions (carbon, ammonium, phosphorous, sand, and silt) with generalized linear mixed models (GLMMs). The response variables involving percentages (carbon, sand, and silt) were modeled using a beta distribution and the continuous response variables (ammonium and phosphorous) were modeled using a Gaussian distribution. The fitted GLMMs included the interaction of year and site with the station as random effects and the best model among the models with various combinations of the random effects was chosen using the Akaike Information Criterion, corrected (AICc) for small sample sizes (Stoica and Selen 2004). We used the test of analysis of deviance to determine whether the interaction effect between year and site improved the model fit. Furthermore, to compare the environmental conditions across sites and years, we estimated marginal means and associated 95%

confidence intervals (CIs). Estimated marginal means are the estimated means of one variable (e.g., carbon within a given site) averaged across every level of the other variable (e.g., across all years).

2.4.2 Benthic invertebrate and community analyses

General characteristics of benthic invertebrates in off-channel habitats were evaluated using indices and summary metrics such as proportion of abundance and frequency of occurrence of taxonomic categories, as well as diversity indices (e.g., richness, evenness, Shannon, and Hill).

Benthic invertebrate communities were analyzed using PRIMER v7 (Clarke and Gorley 2015) and PERMANOVA+ (Anderson et al. 2008). The analysis included 88 variables (unique invertebrate taxa) and 857 benthic samples. Invertebrate counts were square root transformed and a Bray-Curtis distance-based resemblance matrix was calculated. Two-dimensional non-metric multidimensional scaling (nMDS) ordination was used to visualize spatial and temporal patterns in community composition and abundance. Statistical differences among spatial (site) and temporal factors (year) were determined using permutational multivariate analysis of variance (PERMANOVA). SIMPER (similarity percentage analysis) was used to evaluate the average taxa similarity within a group and the average taxa dissimilarity between groups.

2.4.3 Before-After-Control-Impact analyses

BACI analyses were used to statistically assess whether dredged material placement at the impacted site (Woodland Islands) affected the environmental conditions and the benthic invertebrate abundance compared to the two control sites (Martin and Bachelor Islands). We incorporated fixed effects for "period" (before-after) and "site type" (control-impact) to characterize the BACI effect of dredged material placement on the response variables pertaining to environmental conditions and the abundance of benthic invertebrates. We designated 2019 and 2020 as “before” and 2021 and 2022 as “after,” and the Woodland Islands site as the impact site and Martin and Bachelor Islands as the control sites. We also included random effects in our model (e.g., year, site, and station) to account for the unexplained variability in the data from the measured fixed effects.

The response variables for environmental conditions and the abundance of benthic invertebrates deviated from a normal distribution (e.g., counts, proportions), and GLMMs provide a flexible approach for analyzing such non-normal data when random effects are present. Therefore, we used a GLMM framework (Bolker et al. 2009) for the BACI analyses with fixed effects: period and site type, and their interaction (the BACI effect; (Underwood, 1991, 1992), and random effects: year, site, and station.

To investigate the BACI effect of placing dredged material on environmental conditions, we fitted GLMMs for carbon, ammonium, phosphorous, and sand. For variables measured as a percentage—carbon and sand—we converted the percentages into proportions and utilized a GLMM with a beta distribution. The choice of the beta distribution for the response variable was made after examining the skewness-kurtosis plot (Cullen & Frey, 1999; Sokal & Rohlf, 1995). For the analysis of ammonium and phosphorous concentration in sediments, we employed a GLMM with a log-normal distribution. For each response variable associated with environmental conditions, we constructed nine models that encompassed various combinations of the random effects and their interactions. The selection of the best-fitting model was determined using the AICc.

To analyze the BACI effect of dredged material placement on benthic invertebrate counts, we employed a GLMM with Poisson distribution. We fitted models for total benthic invertebrate abundance, chironomid abundance, and corophium abundance. For each response variable, we constructed nine models that encompassed various combinations of the random effects and their interactions. The selection of the best-fitting model was determined using the AICc. In Poisson GLMMs, overdispersion arises when variability in the observed data exceeds the model's assumed variability. We assessed the presence of overdispersion by using Pearson's χ^2-test for overdispersion (Dunn, 2004). We accounted for the overdispersion by adjusting our models to incorporate observation level random effects. The proportion of the total variability explained by each model was determined using Nakagawa's conditional R^2(Nakagawa and Schielzeth 2013).

In all of our models, we applied the likelihood ratio test (Neyman & Pearson, 1933) to assess the model fit with and without the BACI effect. Likelihood ratio test gauges the goodness of fit between two models, in our case, one incorporating the interaction term (i.e., BACI effect) and the other without it (i.e., no BACI effect). If the null hypothesis is rejected, it indicates that the model containing the BACI effect substantially enhances the model's fit compared to the model lacking the BACI effect. Furthermore, we reported the estimates of the BACI effect and their p-values from the fitted GLMM to determine the significance of the BACI effect.

3.0 Results

Findings related to environmental conditions and sediment characteristics, benthic assemblages, benthic invertebrate abundance, and benthic biomass are presented in the following sections.

3.1 Environmental Conditions and Sediment Characteristics

Environmental conditions described in this section include: 1) discharge at Bonneville Dam during the study period; 2) spatial and temporal characteristics of carbon, nutrients, and grain size; and 3) Before-After-Control-Impact analysis of environmental conditions.

3.1.1 Bonneville Dam Discharge

Mean discharge at Bonneville Dam during the study period differed from 2019 through 2022 with some years having lower or higher discharge, compared to the 10-year running average (Figure 6). During the first two years of the study, 2019 and 2020, the mean discharge during April – July was 229.1 and 254.6 kcfs, respectively. During the first-year post-dredged material placement, 2021, the mean discharge at Bonneville Dam during April – July was notably lower (185.6 kcfs) than the previous two years and the 10-year average (2013−2022). During the second-year post-placement, 2022, the average daily mean discharge at Bonneville Dam during April – July was higher (264.6 kcfs) than the previous three years, and higher than the 10-year running average.

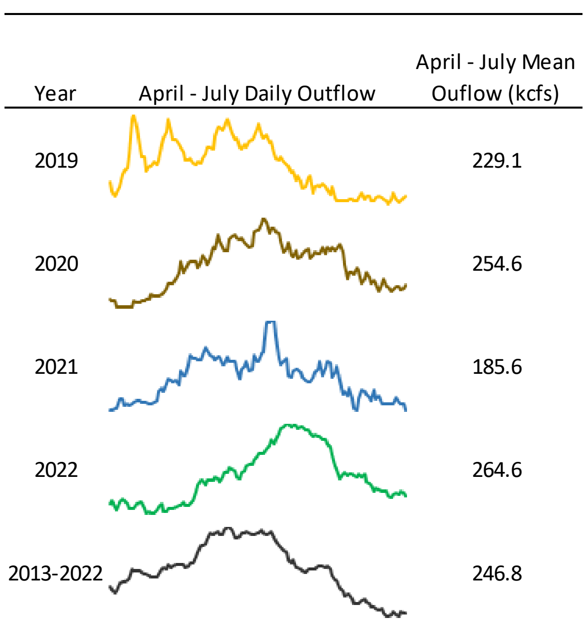

Figure 6. Discharge at Bonneville Dam during the study period: April–July 2019-2023, including the 10-year average (kcfs) corresponding to April–July 2013-2022.

3.1.2 Spatial and temporal variation of carbon, nutrients, and grain size

Percent Carbon

Bachelor Island had the lowest marginal mean carbon concentration, while Martin Island had the highest (Figure 7). The non-overlapping 95% CIs for marginal means across sites indicated that there is evidence of a significant difference in mean carbon concentration across the three sites (Figure 8). Marginal carbon concentration was greatest in 2021 and lowest in 2019 (Figure 5). The analysis of deviance showed a significant interaction between year and site (χ^2 = 24.057, P = 0.00051).

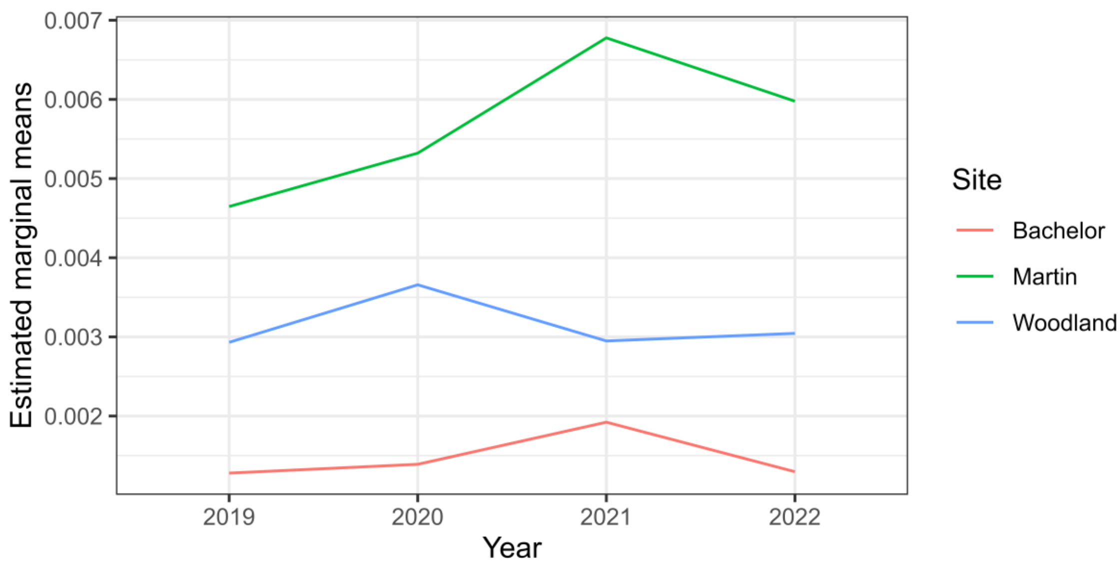

Figure 7. The estimated marginal means of total carbon (as a proportion) in sediment for factor combinations of sites (Martin, Woodland, and Bachelor Islands) and years (2019-2022).

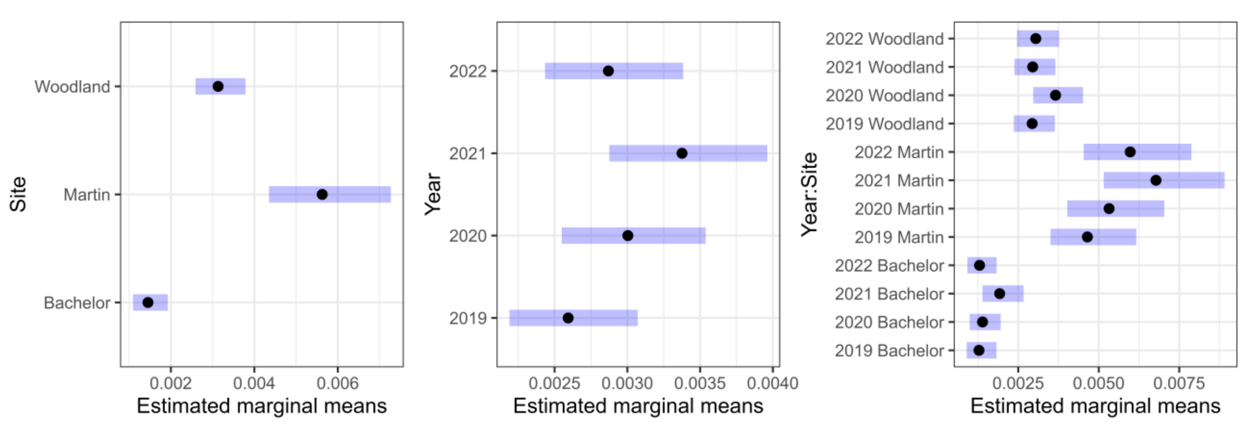

Figure 8. The estimated marginal means of proportion of total carbon in sediment for site, year, and the interaction between site and year. Error bars indicate 95% CIs.

Ammonium concentration

Bachelor had the lowest marginal mean ammonium concentration, while Martin had the highest (Figure 9). The non-overlapping 95% CIs indicate that there is a significant difference in ammonium concentrations across the three sites Marginal mean ammonium concentration in sediments was greatest in 2022 and lowest in 2019 (Figure 10). Furthermore, the interaction of year and site was significant (χ^2 = 22.76, P = 0.00088).

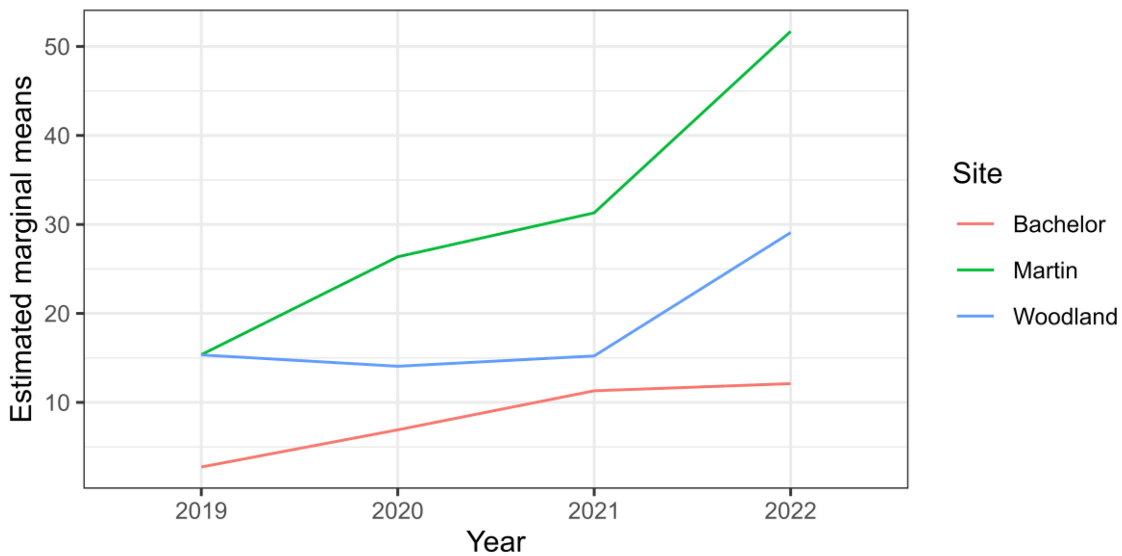

Figure 9. The estimated means of ammonium (ppm) in sediment for factor combinations of sites (Martin, Woodland, and Bachelor Islands) and years (2019-2022).

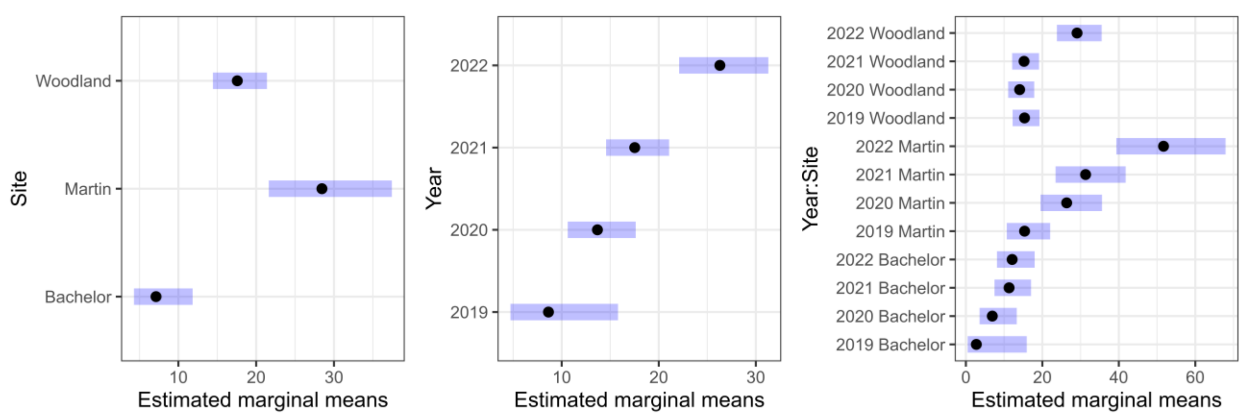

Figure 10. The estimated marginal means of ammonium (ppm) in sediment for site, year, and the interaction between site and year. Error bars indicate 95% CIs.

Phosphorous concentration

The non-overlapping 95% CI indicate that marginal mean phosphorous concentrations were significantly higher at Martin Island, compared to Woodland Islands (Figure 11). There was evidence that 2019 marginal mean phosphorus concentrations were significantly lower than the rest of the years (Figure 12). Analysis of deviance showed that the interaction term of year and site is not significant ($\chi^2 = 2.38$, $P = 0.881$).

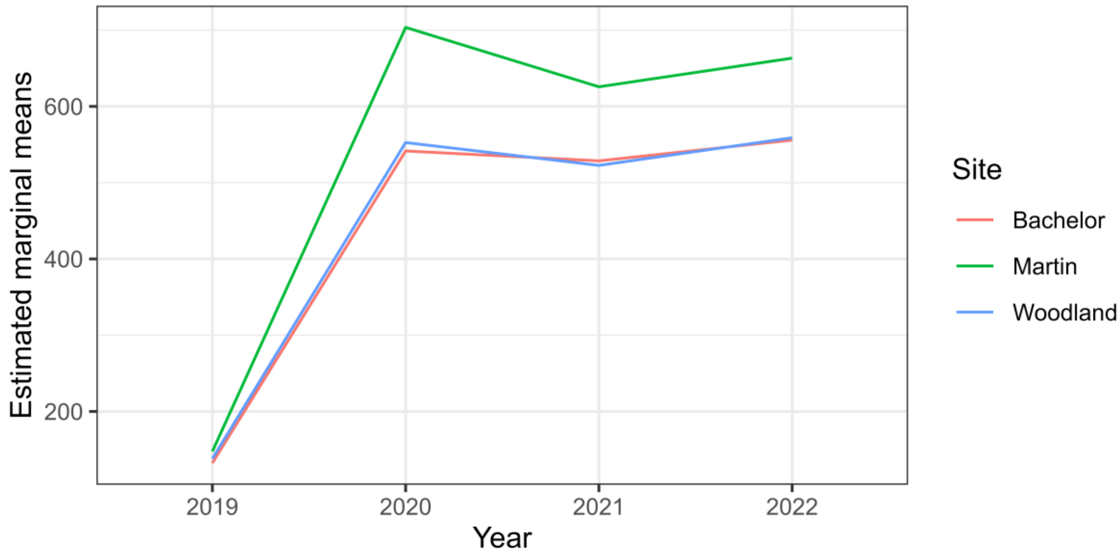

Figure 11. The estimated annual marginal means of phosphorous concentrations (ppm) in sediment for factor combinations of sites (Martin, Woodland, and Bachelor Islands) and years (2019-2022).

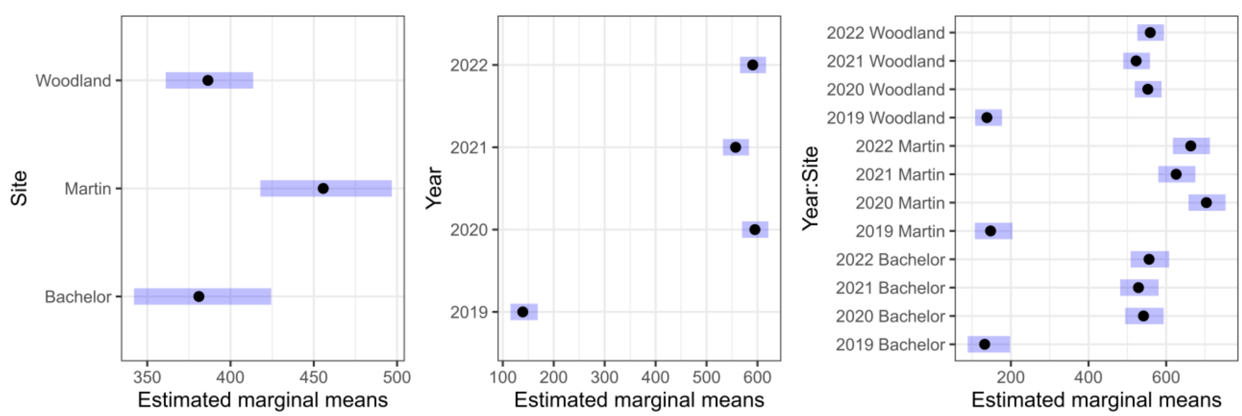

Figure 12. The estimated marginal means of phosphorous concentrations (ppm) in sediment for site, year, and the interaction between site and year. Error bars indicate 95% CIs.

Percent Sand

Bachelor Island had the highest marginal mean proportion of sand, while Martin Island had the lowest (Figure 13). There was evidence for a significant difference in the marginal mean proportion of sand across the three sites (Figure 14). The marginal mean proportion of sand at the three sites was greatest in 2019 and lowest in 2022 (Figure 14). There was evidence for significantly higher sand content in 2019, compared to in 2021 and 2022.

The analysis of deviance showed that the interaction of year and site was significant (χ^2= 58.413, $P < 0.0001$). The trends in marginal mean proportion of sand responded differently across the sites. For example, there was a notable decrease in the marginal mean proportion of sand across years at Martin whereas there was no clear trend in sand content at Bachelor and Woodland Islands across years.

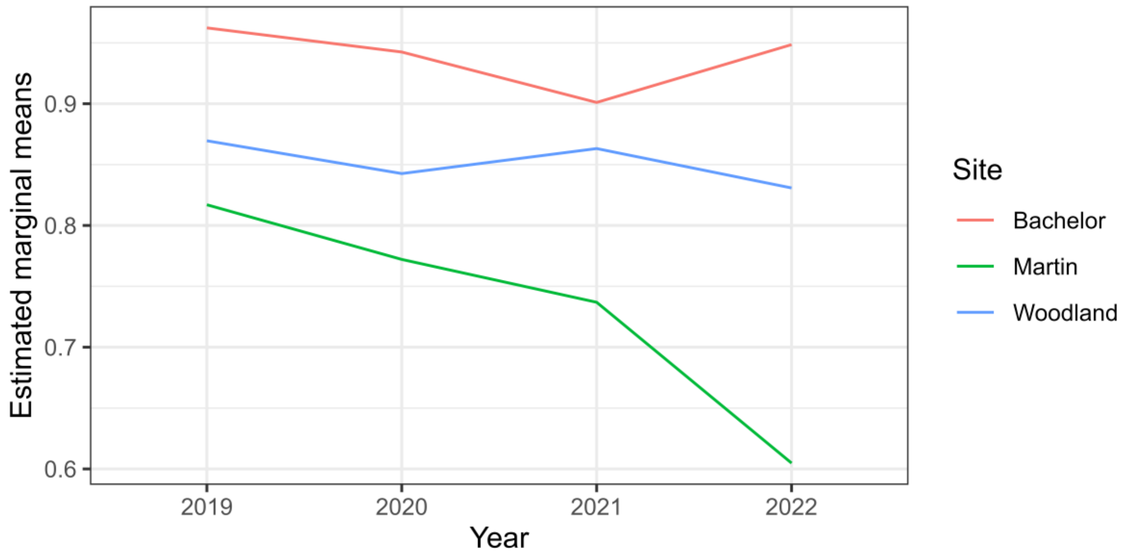

Figure 13. The estimated annual marginal means of the proportion of sand in sediment for factor combinations of sites (Martin, Woodland, and Bachelor Islands) and years (2019-2022).

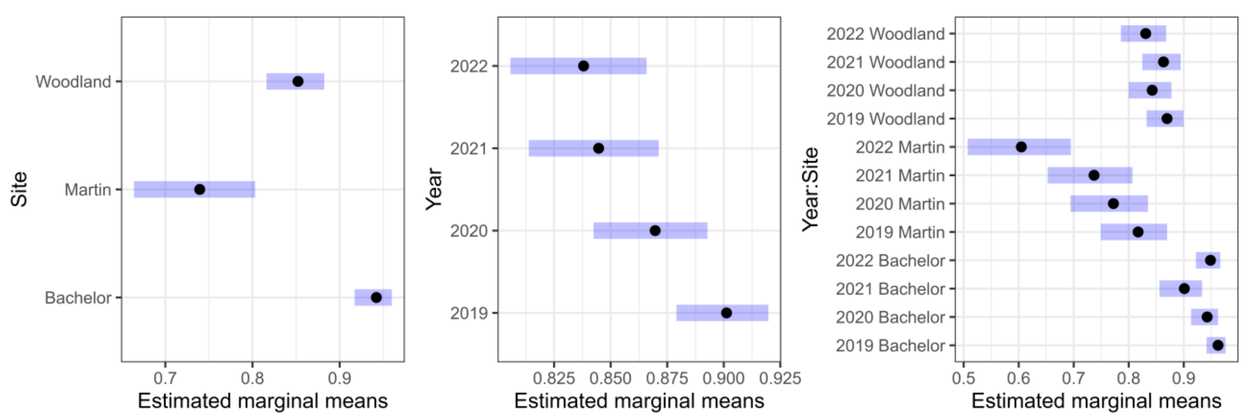

Figure 14. The estimated marginal means of the proportion of sand in sediment for site, year, and the interaction between site and year. Error bars indicate 95% CIs.

Percent Silt

The non-overlapping 95%CI indicate evidence of a significant difference of marginal mean proportion of silt among the sites. The marginal mean proportion of silt was highest at Martin Island and lowest at Bachelor Island (Figure 15). Analysis of deviance showed that the interaction of site and year was significant ($\chi^2 = 64.57$, $P < 0.001$). Annual trends in marginal silt content were different across the different sites. Martin Island showed an increase in marginal silt content across years whereas there was no increasing or decreasing trend at Woodlands Islands (Figure 16).

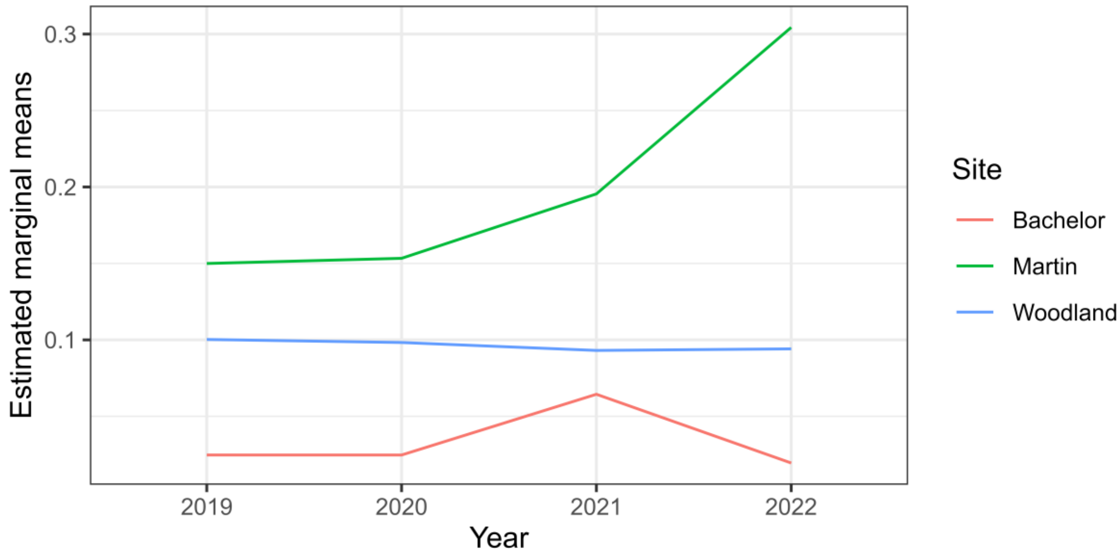

Figure 15. The estimated annual marginal means of the proportion of silt in sediment for factor combinations of sites (Martin, Woodland, and Bachelor Islands) and years (2019-2022).

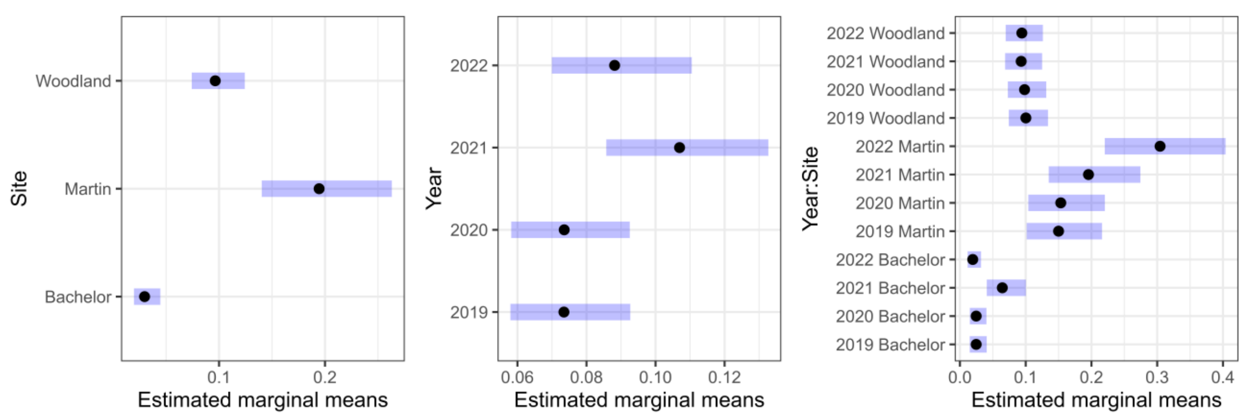

Figure 16. The estimated marginal means of the proportion of silt in sediment for site, year, and the interaction between site and year. Error bars indicate 95% CIs.

3.1.3 Before-After-Control-Impact Analysis: Environmental Conditions

The BACI effect for concentrations of percent carbon and ammonium as well as percent sand were all significant (Table 1). Results of AICc model selection are found in Appendix C.

Table 1. BACI analysis with the parameter estimates and their associated SE (standard error) and p-values from fitting a GLMM to the environmental conditions carbon (%), ammonium concentration, phosphorous concentration, sand (%). The accompanying a p-value denotes significance at the 0.05 significance level.

Response	Test	Coefficient	SE	P-value	R^2
Carbon	Intercept	-5.963	0.396	< 0.0001***	0.562
	Before vs. After (BA)	0.227	0.090	0.0119 *	
	Control vs. Impact (CI)	0.248	0.672	0.7119	
	BA x CI	-0.318	0.088	0.0003 *	
Ammonium	Intercept	2.375	0.426	< 0.0001*	0.380
	Before vs. After (BA)	0.643	0.239	0.0072 *	
	Control vs. Impact (CI)	0.304	0.673	0.6516	
	BA x CI	-0.242	0.095	0.0111 *	
Phosphorous	Intercept	5.704	0.354	< 0.0001*	0.217
	Before vs. After (BA)	0.673	0.483	0.1630	
	Control vs. Impact (CI)	-0.112	0.1276	0.3770	
	BA x CI	0.026	0.051	0.6110	
Sand	Intercept	2.303	0.546	<0.0001 *	0.836
	Before vs. After (BA)	-0.610	0.159	0.0001 *	
	Control vs. Impact (CI)	-4.392	0.914	0.6308	
	BA x CI	0.514	0.087	< 0.0001*	

Carbon percent

The likelihood ratio test indicated the GLMM model with the BACI effect (period × site type) offers a significantly better fit compared to the model with no BACI effect (χ^2=5.96, *P* =0.015). The R^2 of the model indicates that it accounts for 56% of the variability in the data. Additionally, the model exhibits a statistically significant BACI effect at the 0.05 significance level (*P* = 0.0003; Table 1) There was a decrease in the carbon percent in sediment at the impact location after the placement of dredged material (Figure 17). This decrease was relative to carbon percent at the impact site before dredged placement and relative to carbon percent at the control sites. Specifically, the estimated marginal mean percent carbon in the control sites increased from 0.256% to 0.322% and decreased from 0.328% to 0.3% at the impact site.

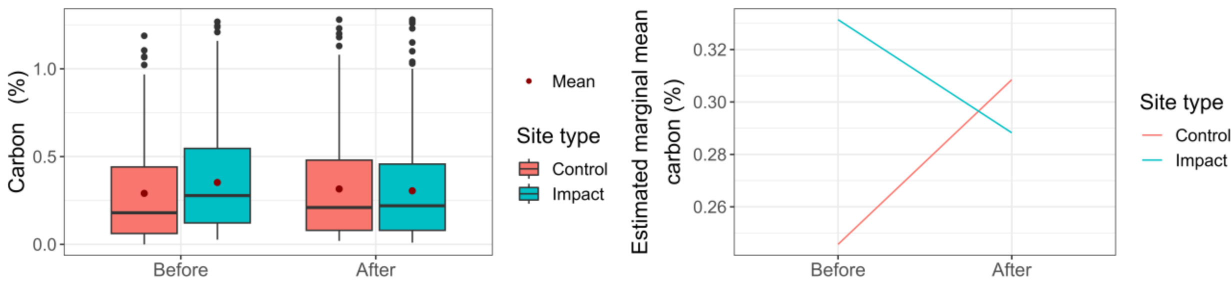

Figure 17. Percent of total carbon in sediment collected in off-channel habitats during 2019-2022, before and after impact, and the interaction plot showing the interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from the box plots for improved visualization.

Ammonium concentration

The likelihood ratio test indicated the GLMM accounting for the BACI effect significantly improved the model's fit compared to the model without the BACI effect (χ^2 =6.63, *P*=0.01). The R^2 of the model indicates that it accounts for 38% of the variability in the data. Furthermore, our model indicated that the BACI effect was significant at the 0.05 significance level (*P*=0.0111; Table.1). Ammonium concentration increased at both the control and the impact sites after the dredged material placement compared to before (Figure 18). However, this increase at the impact site was less pronounced when compared to the two control sites. For example, the estimated marginal mean ammonium concentration in the control sites increased from 10.8 to 20.5, and from 14.6 to 21.8 at the impact site.

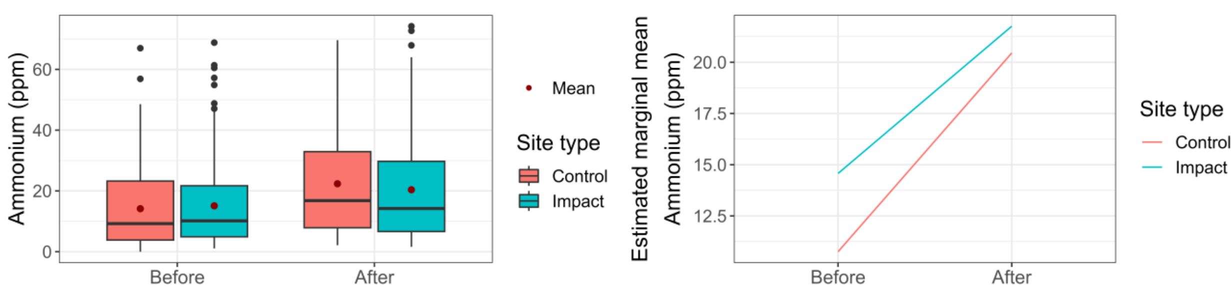

Figure 18. Ammonium (ppm) in in sediment collected in off-channel habitats during 2019-2022, before and after impact, and the interaction plot showing the interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from the box plots for improved visualization.

Phosphorous concentration

Likelihood ratio test indicated that there is no evidence suggesting that including the BACI effect improves the model fit of the GLMM (χ^2 = 0.259, *P* = 0.61). The model provided an R^2 of 21.7%. The model outcome also indicated that the BACI effect is not significant (*P* = 0.6110; Table 1). The estimated marginal mean phosphorous concentration in sediments exhibited an increase at both the impact site and control sites following the placement of dredged material (Figure 19). The estimated marginal mean concentration at the impact site increased from 269 to 541 while at the control sites, it increased from 300 to 589.

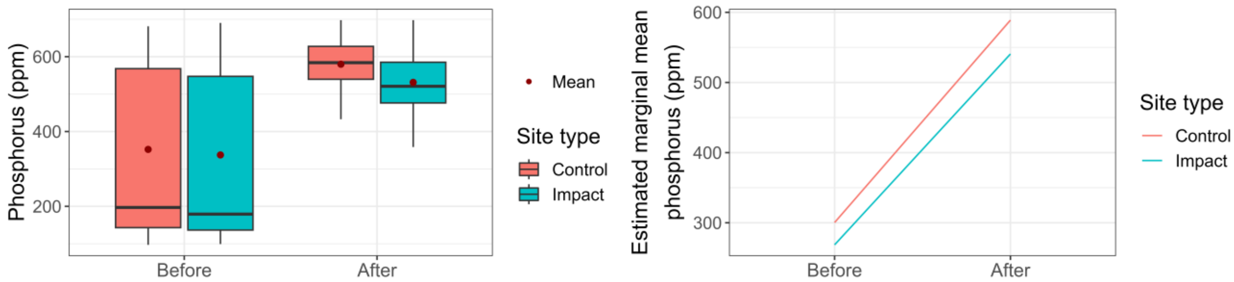

Figure 19. Phosphorous (ppm) in sediment collected in off-channel habitats during 2019-2022, before and after impact, and the interaction plot showing the interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from the box plots for improved visualization.

Sand percent

The likelihood ratio test indicated the GLMM accounting for the BACI effect significantly improved the model's fit compared to the model without the BACI effect (χ^2 =32.913, $P <$ 0.0001). The R^2 of the model indicates that it accounts for 83.6% of the variability in the data. The model results indicated that the BACI effect is significant at the 0.05 significance level ($P <$ 0.0001; Table 1). The estimated marginal mean sand percent in sediments exhibited a decrease at the impact site following the placement of dredged material, in comparison to the sand percent before placement (Figure 20). The estimated marginal mean percent at the impact site decreased from 0.866 to 0.854, while at the control sites, it dropped from 0.909 to 0.845. The model estimated a negative BACI effect which implies that there was a larger negative change in estimated marginal means in the control sites before versus after placement, as compared to the negative change observed in the impact site before versus after.

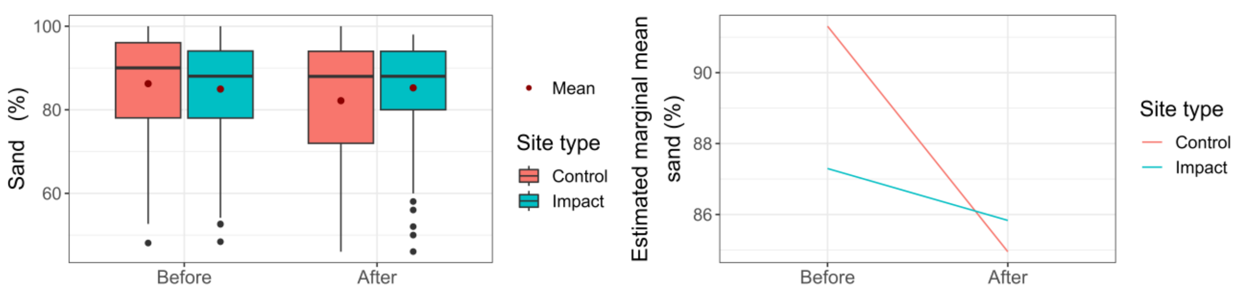

Figure 20. Percent sand in in sediment collected in off-channel habitats during 2019-2022 and the interaction plot showing the interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from the box plots for improved visualization.

3.2 Benthic Invertebrates

The following sections describe general characteristics of benthic invertebrates in off-channel habitats sampled by examining proportion of abundance and frequency of occurrence of taxonomic categories, as well as diversity indices across sites and years. In addition to general characteristics, this section also includes an analysis of benthic community composition relative to spatial and temporal factors.

3.2.1 Benthic Invertebrate Characteristics

Across all sites and years more than 300,000 individual benthic invertebrates were collected. These invertebrates represented 88 unique taxonomic variables, including 66 invertebrate families, 11 classes, and eight phyla (see Appendix A for a taxonomic list). Proportions of average abundance were calculated within each year of each site to characterize general taxonomic groups present in off-channel habitats. Common prey items for juvenile salmon – chironomids, insects, corophium, crustaceans – comprised 20–64 percent of total invertebrate abundance across a given site and year (Figure 21).

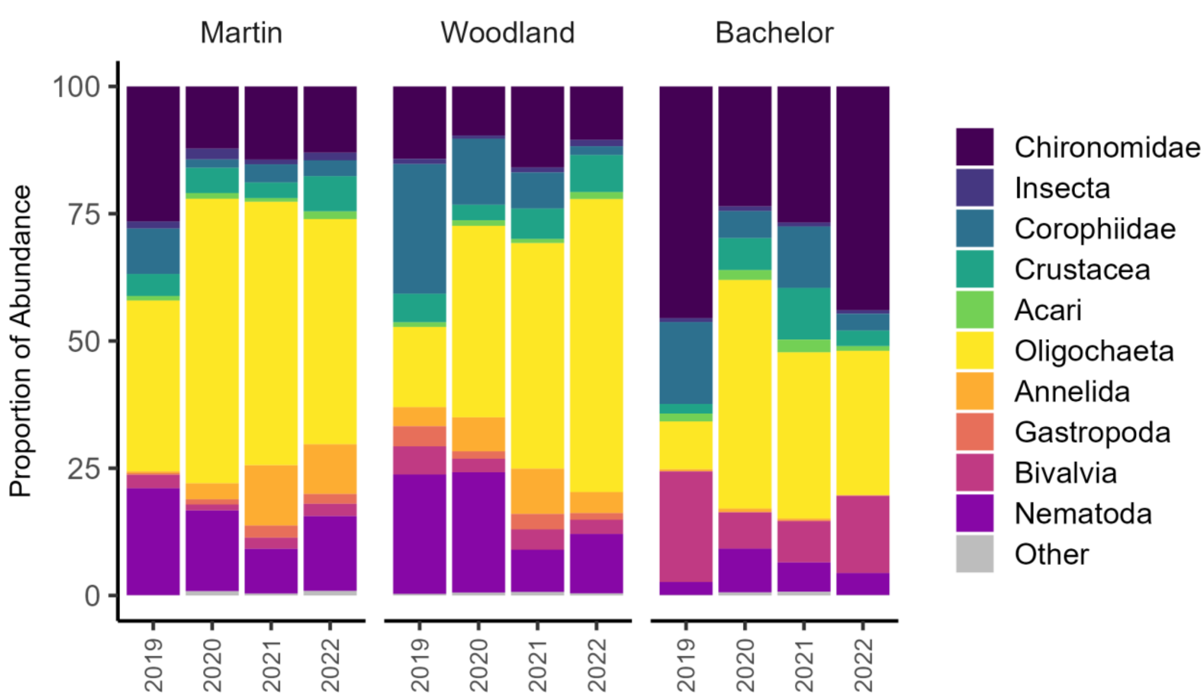

Figure 21. The proportion of the mean abundance of benthic invertebrates from 2019 – 2022 at Martin, Woodland, and Bachelor Islands.

Across all locations and years, chironomids, bivalves, oligochaetes, and nematodes occurred most frequently (most frequently (>90 percent of the time) and all other taxonomic categories occurred less than ~50 percent of the time (**Error! Not a valid bookmark self-reference.**). Table 2). At Martin Island, chironomids, insects, crustaceans, acari, oligochaetes, bivalves and nematodes all occurred greater than 70 percent of the time. At Woodland Islands, chironomids, crustaceans, oligochaetes, bivalves, and nematodes occurred greater than 70 percent of the time. At Bachelor Island only chironomids and bivalves occurred most frequently (>90 percent of the time) and all other taxonomic categories occurred less than ~50 percent of the time (**Error! Not a valid bookmark self-reference.**).

Table 2. The frequency of occurrence of benthic invertebrates, as a proportion, in sediment samples collected from Martin, Woodland, and Bachelor Islands during 2019–2022. Bolded values highlight taxa that occurred greater than 70 percent of the time.

		2019			2020			2021			2022		
Taxa	Common Name(s)	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor
Chironomidae	Non-biting midges	**0.98**	**0.99**	**1.00**	**1.00**	**0.99**	**1.00**	**1.00**	**0.94**	**1.00**	**1.00**	**0.96**	**1.00**
Insecta	Insects	**0.82**	0.60	0.15	**0.82**	0.61	0.37	**0.82**	0.67	0.52	**0.84**	0.58	0.25
Corophiidae	Scuds	**0.91**	**0.92**	**0.91**	0.68	**0.86**	0.67	**0.79**	0.69	0.94	**0.82**	0.53	0.35
Crustacea	Isopoda, copepoda, ostracoda, mysida, non-corophium amphipods	**0.70**	**0.75**	0.37	**0.91**	**0.79**	0.48	**0.96**	**0.84**	**0.94**	**0.89**	**0.76**	0.42
Acari	Mites	**0.77**	0.65	0.33	**0.84**	**0.77**	0.65	**0.86**	0.66	**0.90**	**0.77**	0.68	0.40
Oligochaeta	Segmented worms	**0.98**	**0.84**	0.37	**1.00**	**0.98**	**0.73**	**1.00**	**0.91**	**0.96**	**1.00**	**0.88**	0.62
Annelida	Leeches and Polychaeta	0.36	0.30	0.15	0.41	0.44	0.13	**0.82**	0.48	0.13	**0.70**	0.36	0.02
Gastropoda	Snails	0.41	0.60	0.02	0.54	**0.78**	0.08	**0.79**	**0.71**	0.21	0.61	0.56	0.02
Bivalvia	Clams, Mussels	**0.91**	**0.94**	**0.98**	**0.86**	**0.92**	**0.92**	**0.93**	**0.85**	**0.98**	**0.93**	**0.87**	**0.90**
Nematoda	Roundworms	**0.96**	**0.84**	0.37	**0.93**	**0.93**	0.67	**0.98**	**0.90**	**0.90**	**0.91**	**0.79**	0.35
Other	Nemertea, Cnidaria, Platyhelminthes	0.09	0.24	0.09	0.52	0.45	0.10	0.59	0.59	0.42	0.45	0.32	0.06

Across years sampled, taxa richness (at the lowest taxonomic resolution) was lowest at Bachelor Island (23-38 taxa) and highest at Woodland Islands (47-62 taxa; Table 3). When considering these diversity indices, it is important to recall that while there was an equal sampling density between the impact and control sites (which is relevant to the BACI analyses), the sample sizes (number of benthic grabs) across sites was different (see section 2.1). Despite having approximately half as many samples, taxa richness at Martin Island was comparable to Woodland Islands. Evenness values across all sites were similar and ranged from 0.431-0.520.

Table 3. Diversity measures of benthic invertebrates sampled monthly from three off-channel habitats between 2019 – 2022.

	2019			2020			2021			2022		
Diversity Measure	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor	Martin	Woodland	Bachelor
Richness	52	58	27	45	47	28	49	62	38	49	50	23
Pielous evenness (J')	0.438	0.519	0.474	0.431	0.487	0.504	0.452	0.479	0.503	0.520	0.431	0.475
Shannon (H')	1.73	2.11	1.56	1.64	1.88	1.68	1.76	1.98	1.83	2.02	1.69	1.49
Hill (exp(H'))	5.65	8.23	4.77	5.16	6.53	5.36	5.80	7.23	6.22	7.56	5.40	4.44

Trends in mean density of benthic invertebrates indicate both spatial and temporal factors variation in the data (Figure 22). Mean density was highest at Martin Island and lowest at Bachelor Island. Mean density at Martin Island increased from 2019−2021 but decreased during 2022. At Woodland Islands, mean density increased from 2019–2020, and decreased during 2021 and 2022. Mean density was similar at Bachelor Island during the first two years of the study, which was followed by an increase in 2021 and a decrease in 2022.

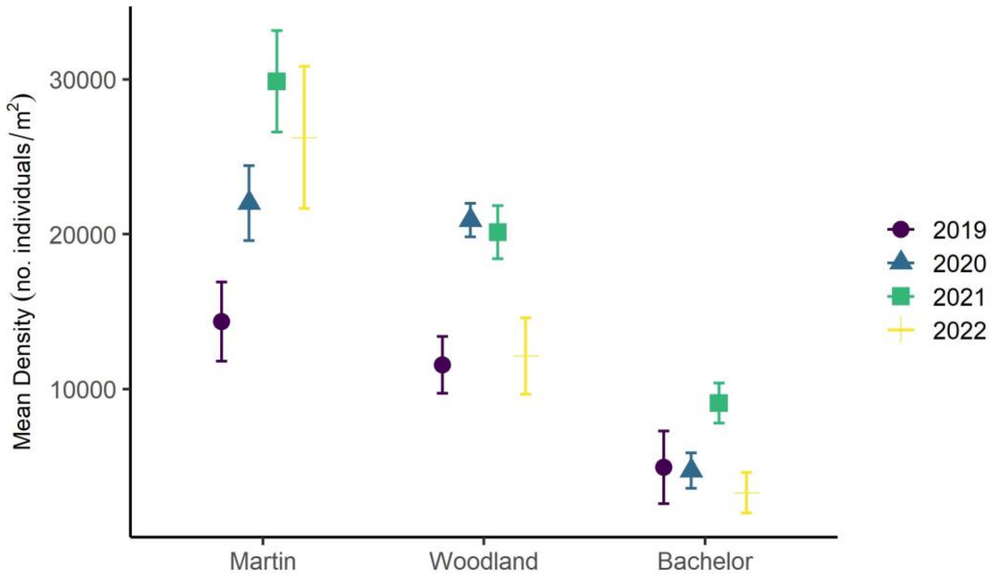

Figure 22. The mean density (no. individuals per square meter) of benthic invertebrates sampled at Martin, Woodland, and Bachelor Islands during 2019 to 2022. Error bars are the standard error of the mean.

3.2.2 Benthic Invertebrate Community

The composition and abundance of benthic invertebrates varied by site and year (Figure 23). Site explained 18.1% of the variation in the benthic communities (Table 4; Pseudo-F = 70.6, p = 0.0001) and year explained 7.4% of the variation (Pseudo-F = 21.5, p = 0.0001). Differences in benthic communities were dependent on the site and year, and the interaction of these sources was significant (Pseudo-F = 5.82, p = 0.0001). The pairwise comparisons indicate that during a given year, benthic communities were significantly different across sites (Table 5). Similarly, within a given site, communities differed across all years.

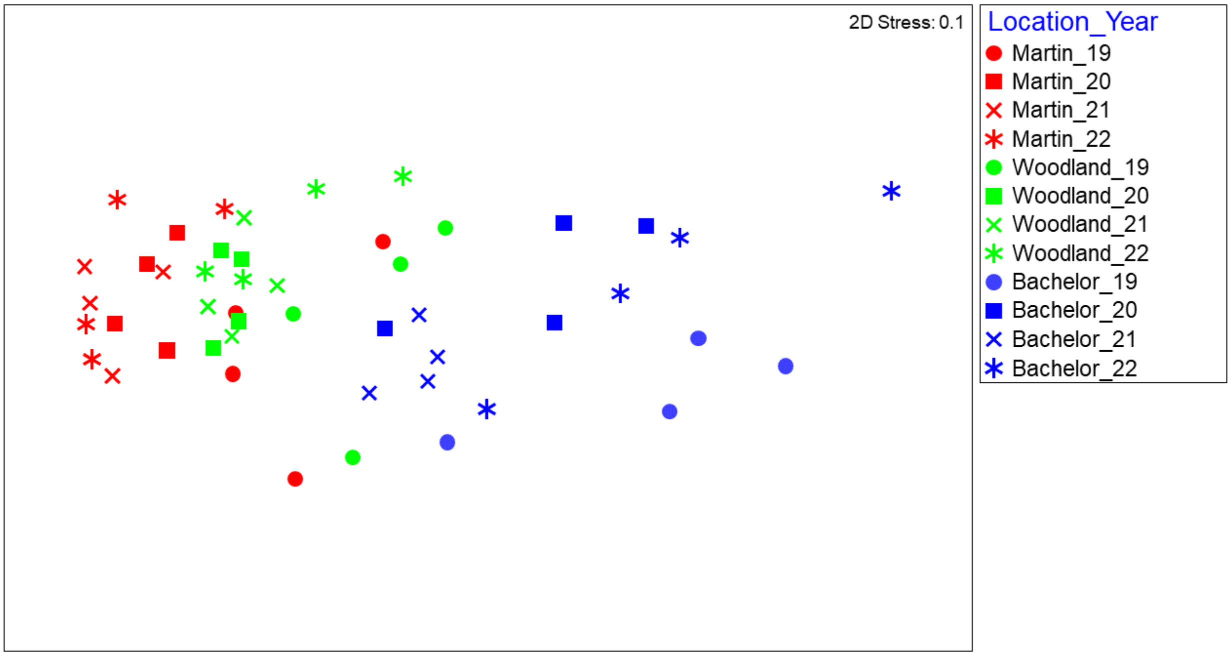

Figure 23. Non-metric multidimensional scaling (nMDS) ordination plot of fourth root transformed average counts of benthic invertebrate assemblages at Martin, Woodland, and Bachelor Islands in 2019 − 2020.

Table 4. PERMANOVA table of results for fourth root transformed benthic invertebrate counts at Martin, Woodland, and Bachelor Islands during 2019 – 2022.

Source	df	SS	MS	Pseudo *F*	*P*	Component	Var[(a)]	SD
Site	2	1.92E+05	96158	70.6	0.0001	Fixed	18.1	4.34
Year	3	8.76E+04	29213	21.5	0.0001	Fixed	7.4	3.47
Site x Year	6	47534	7922.4	5.82	0.0001	Fixed	5.0	3.15
Res	845	1.15E+06	1361.5	–	–	Random	69.4	6.07
Total	856	1.49E+06	–	–	–		–	–

(a) Estimated sizes of components of variation

Table 5. Pairwise comparisons of the site x month x year interaction terms from the PERMANOVA model

Sites	Years	t	P(perm)
Martin	2019, 2020	3.07	0.0001
Martin	2019, 2021	4.26	0.0001
Martin	2019, 2022	2.93	0.0001
Martin	2020, 2021	2.52	0.0002
Martin	2020, 2022	1.71	0.0065
Martin	2021, 2022	2.19	0.0001
Woodland	2019, 2020	4.07	0.0001
Woodland	2019, 2021	4.24	0.0001
Woodland	2019, 2022	4.69	0.0001
Woodland	2020, 2021	2.70	0.0001
Woodland	2020, 2022	3.53	0.0001
Woodland	2021, 2022	2.72	0.0001
Bachelor	2019, 2020	3.55	0.0001
Bachelor	2019, 2021	5.82	0.0001
Bachelor	2019, 2022	3.11	0.0001
Bachelor	2020, 2021	3.63	0.0001
Bachelor	2020, 2022	2.23	0.0015
Bachelor	2021, 2022	5.15	0.0001
Bachelor, Martin	2019	6.76	0.0001
Bachelor, Woodland	2019	5.36	0.0001
Martin, Woodland	2019	3.16	0.0001
Bachelor, Martin	2020	6.29	0.0001
Bachelor, Woodland	2020	5.54	0.0001
Martin, Woodland	2020	2.87	0.0001
Bachelor, Martin	2021	6.38	0.0001
Bachelor, Woodland	2021	3.49	0.0001
Martin, Woodland	2021	3.25	0.0001
Bachelor, Martin	2022	7.07	0.0001
Bachelor, Woodland	2022	4.70	0.0001
Martin, Woodland	2022	3.31	0.0001

Similarities in benthic communities within a given site were driven by five taxa—oligochaetes, chironomids, nematodes, corophium, and *Corbicula*. These taxa explained 70 percent of the cumulative contributions in similarity at each site (see Appendix B). The sites were 50-57 percent dissimilar in benthic community composition and abundance. Dissimilarities across sites were driven by 11 taxa, which individually contributed 2 − 23 percent to the total dissimilarity. While oligochaetes, nematodes, chironomids, and corophium were the primary contributors to within site similarities across years, they were also the top contributors to dissimilarities when comparing benthic communities between sites. These taxa were lower in average abundance at Bachelor Island, compared to both Martin and Woodland Islands. Comparisons of Martin and Woodland Islands indicated all but corophium were higher in abundance at Martin Island. Corbicula contributed five and seven percent of the dissimilarities between Bachelor and Martin Islands and Bachelor and Woodland Islands, respectively, and were higher in abundance at Bachelor Island compared to the other two locations (Appendix B).

3.2.3 Before-After-Control-Impact Analysis: Benthic Invertebrates

For all three BACI analyses (total abundance, chironomid abundance, and corophium abundance), the best fitted model included the random effects: year, site, station, and the interaction of station and year in addition to the fixed effects: period and site type, and their interaction (the BACI effect). Pearson's χ^2 test for overdispersion indicated that overdispersion existed, which was addressed by introducing an observation-level random effect. Benthic communities before vs. after were evaluated using ordination and second stage nMDS visualization techniques.

The total abundance, total chironomid abundance, and total corophium abundance at Woodland Islands were all significantly different at the 0.05 significance level after dredged material placement, compared to before placement and compared to the control sites (Table 6).

Table 6. BACI analysis with the parameter estimates and their associated SE (standard error) and p-values from fitting a GLMM to the total abundance, chironomidae abundance, and corophium abundance. The "*" accompanying a p-value denotes significance at the 0.05 significance level.

Response	Test	Coefficient	SE	P-value	R^2
Total abundance	Intercept	4.964	0.548	< 0.0001 *	0.654
	Before vs. After (BA)	0.423	0.345	0.2200	
	Control vs. Impact (CI)	0.543	0.855	0.5250	
	BA x CI	-0.752	0.183	< 0.0001 *	
Chironomidae abundance	Intercept	3.321	0.240	< 0.0001 *	0.309
	Before vs. After (BA)	0.294	0.219	0.1805	
	Control vs. Impact (CI)	-0.182	0.312	0.5603	
	BA x CI	-0.374	0.145	0.0099 *	
Corophium abundance	Intercept	1.364	0.571	0.0169 *	0.624
	Before vs. After (BA)	-0.201	0.726	0.7824	
	Control vs. Impact (CI)	1.392	0.410	0.0007 *	
	BA x CI	-1.913	0.299	< 0.0001 *	

Total Abundance

The likelihood ratio test showed that the BACI effect significantly improved the model fit of the GLMM (χ^2 = 16.072, P < 0.001) for total invertebrate abundance. The model yielded a R^2 of 0.654 indicating that the model explains 65.4% of variability in the data. Furthermore, our model indicated that the BACI effect is significant at the 0.05 significance level (P < 0.0001; Table 6). The estimated mean abundance showed a decrease in total invertebrate abundance after placement of dredged material, compared to the abundance before placement (Figure 24). Specifically, the estimated mean abundance at the impact site decreased from 246 to 177 individual invertebrates per grab sample, while in the control sites, estimated mean abundance increased from 143 to 218.

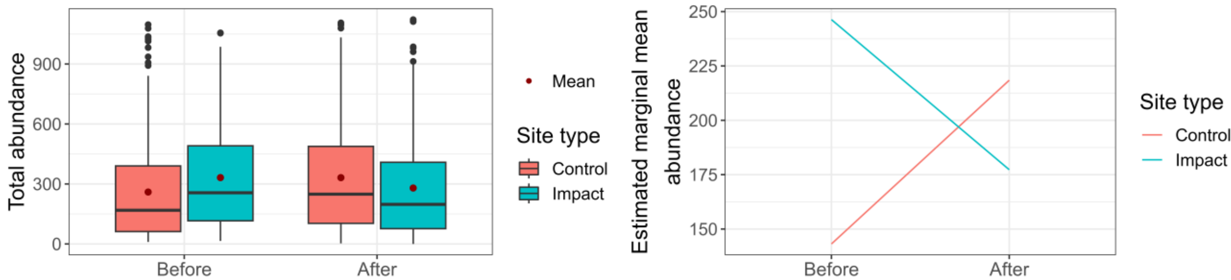

Figure 24. Total abundance of invertebrates per grab at the impact and control sites, before and after dredged material placement and the interaction plot showing the estimated interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from the box plots for improved visualization.

Chironomidae abundance

The likelihood ratio test showed that the BACI effect significantly improved the model fit of the GLMM (χ^2 =6.608, P=0.01). The R^2 of the model indicates that it explains 30.9% of the variability in the data. The estimates obtained for the BACI effect from the model indicated that the BACI effect is significant at 0.05 significance level (P < 0.0099; Table 6). According to interaction plots, the estimated means indicated a decrease in Chironomidae abundance after the placement of dredged material at the impact site (Figure 25). Specifically, the estimated mean Chironomidae abundance at the impact site decreased from 23 to 21 individuals per ponar grab, while in the control sites, it increased from 27 to 37.

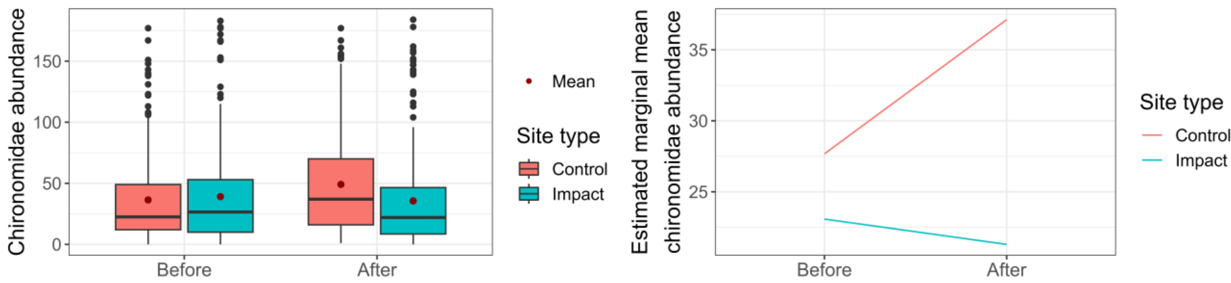

Figure 25. Total chironomidae abundance at the impact and control sites, before and after dredged material placement and the interaction plot showing the estimated interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from box plots for improved visualization.

Corophium abundance

The likelihood ratio test indicated that the BACI effect significantly improved the model fit of the GLMM ($\chi^2 = 35.552, P = 2.48 \times 10^{-9}$). The R^2 of the model indicates that it explains 62.4% of the variability in the data. The estimates obtained for the BACI effect from the model indicated that the BACI effect is significant at the 0.05 significance level ($P < 0.0001$; Table 6). Dredged material placement significantly decreased the estimated mean corophium abundance compared to before placement (Figure 26). The estimated mean corophium abundance in the control sites decreased from 4 to 3 individuals per ponar grab, whereas in the impact site, it decreased from 16 to 2.

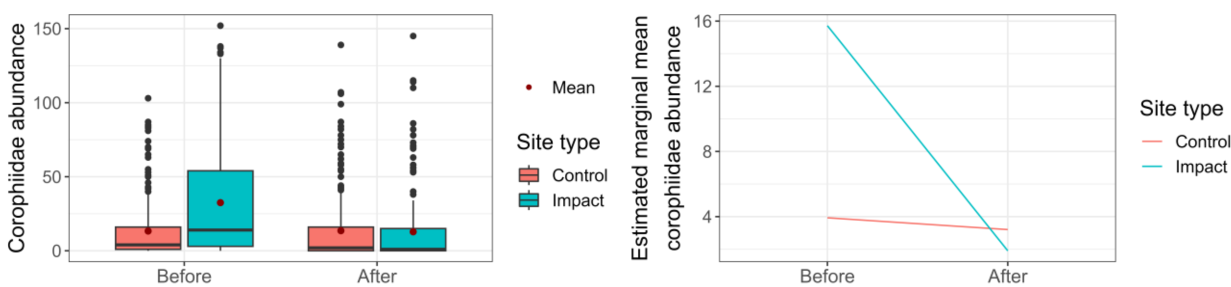

Figure 26. Total corophium abundance at the impact and control sites, before and after dredged material placement and the interaction plot showing the estimated interaction between "period" (before-after) and "site type" (control-impact). Data beyond the upper 95th percentile is excluded from box plots for improved visualization.

Benthic Invertebrate Community

In addition to univariate approaches, we wanted to consider multivariate approaches that considered the entire community relative to before-after responses in community structure (e.g., composition and abundance). The ordination plots (Figure 27A) demonstrate a notable amount variation in community structure and high species turn-over across months. In the time periods that followed dredged material placement, 2021 and 2022, the community structure at Woodland Islands collapsed, as demonstrated by the few data points which were dissimilar from the collapsed group. The collapsed structure is driven by a few samples that are notably different in community composition and/or abundance. This is most apparent during April–July 2021, April and May 2022, and July 2022 (Figure 27A). Further evidence for differences in community structure was provided by the second stage nMDS plot (Figure 27B), which demonstrates little correlation in community structure in the before and after groups. With the exception of June 2022, the after group had higher correlation in the resemblance matrices, and community structure, as indicated by the tight clustering of data points on the left of the second stage nMDS plot. The community structure and associated resemblance matrices of the before group were more similar to each other than to the after group, but there was also more variation within this group, compared to the after group. This is seen in both plots in Figure 27.

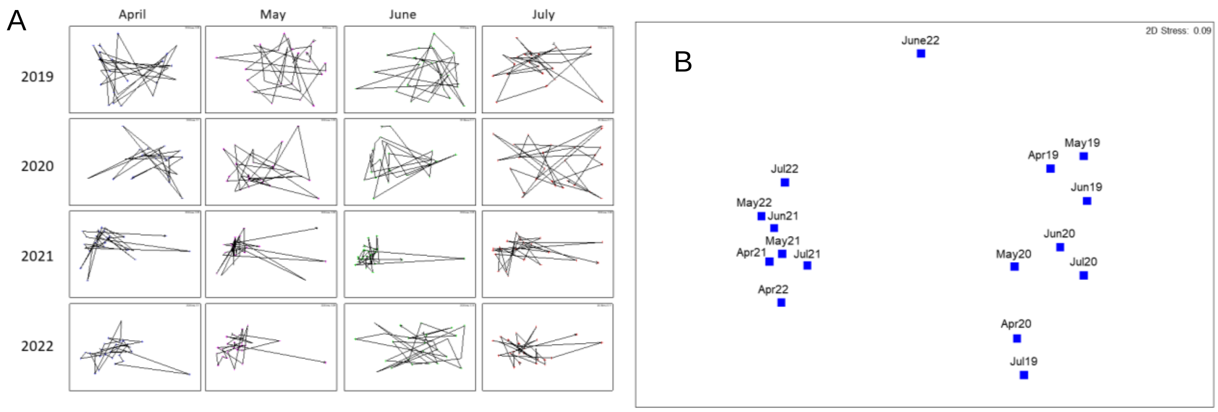

Figure 27. nMDS plots based on Bray-Curtis similarities of square root transformed abundance of benthic invertebrates at Woodland Islands April–July, 2019–2022 (A). Second stage nMDS plot (B) based on Pearson rank correlations between pairs of the 16 resemblance matrices corresponding to benthic community assemblages during each month and year sampled at Woodland Islands.

Differences in community structure at Woodland Islands was driven by the absence of invertebrates at some locations following dredged material placement; five samples had no invertebrates present. In addition, changes in abundances of certain taxa between before and after dredged material placement influenced the community structure (Figure 28). Oligochaetes were the most abundant taxa and showed an increase after dredged material placement. Nematodes and corophium were abundant at the site, and both had notable decreases following dredged material placement. The remaining taxa showed marginal increases or decreases in abundance between the before and after periods (Figure 28).

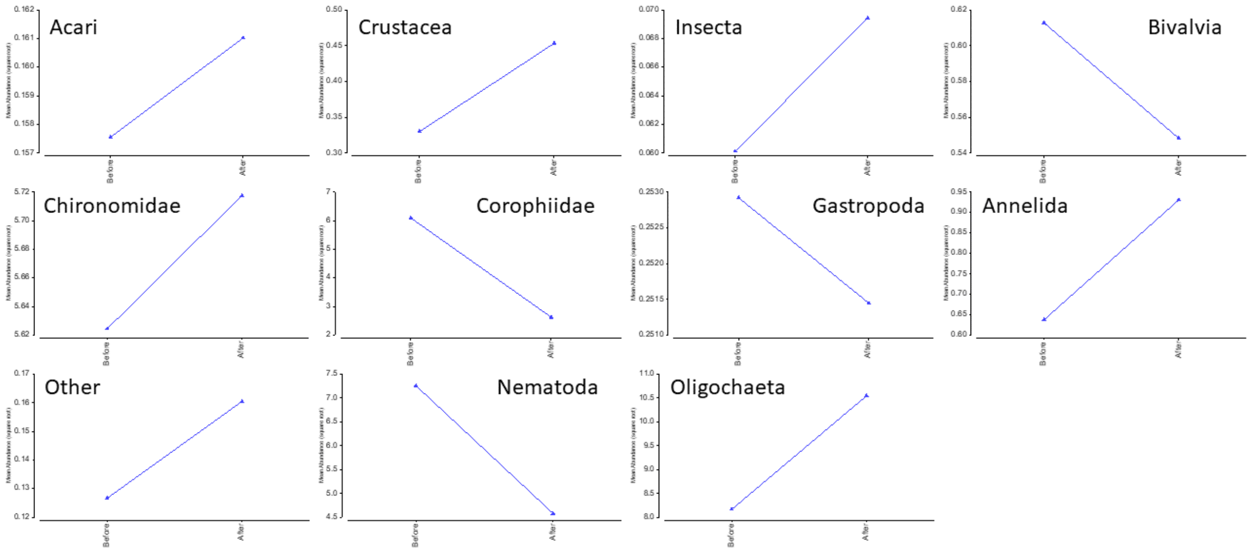

Figure 28. Summary of mean square root abundance for 11 taxonomic categories at Woodland Islands before and after dredged material placement.

Community structure at Martin and Bachelor Islands was also highly variable across months and years (Figure 29 and Figure 30; see section 3.2.2). However, unlike the collapse in community structure observed at Woodland Islands during 2021 and 2022, this pattern was not observed at Martin and Bachelor Islands. In addition, the results of the second stage nMDS analyses of the Martin (Figure 29) and Bachelor Islands (Figure 30) datasets did not suggest correlation in the community structure with “before” or “after” dredged material time periods, as was observed in the Woodland Islands dataset.

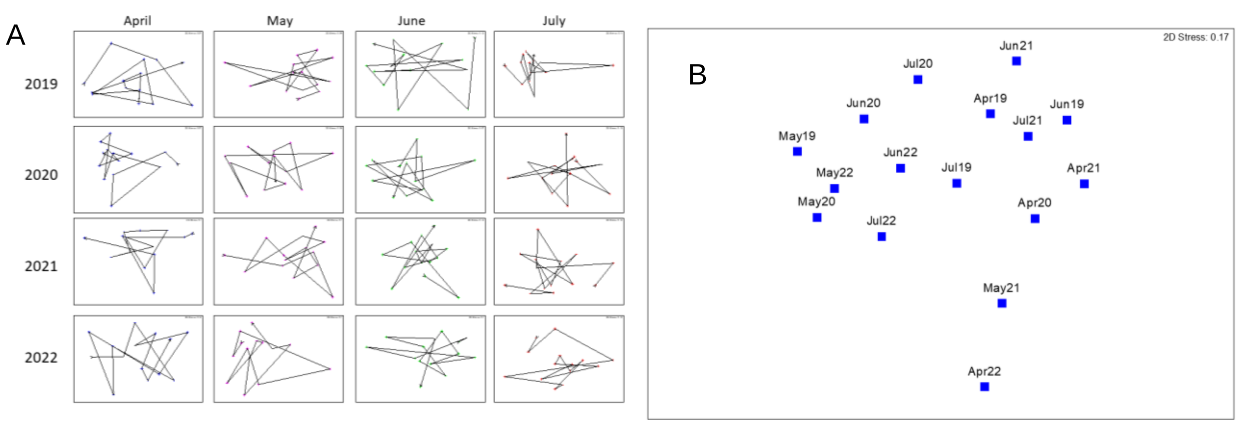

Figure 29. nMDS plots based on Bray-Curtis similarities of square root transformed abundance of benthic invertebrates at Martin Island April–July, 2019–2022 (A). Second stage nMDS plot (B) based on Pearson rank correlations between pairs of the 16 resemblance matrices corresponding to benthic community assemblages during each month and year sampled at Martin Island.

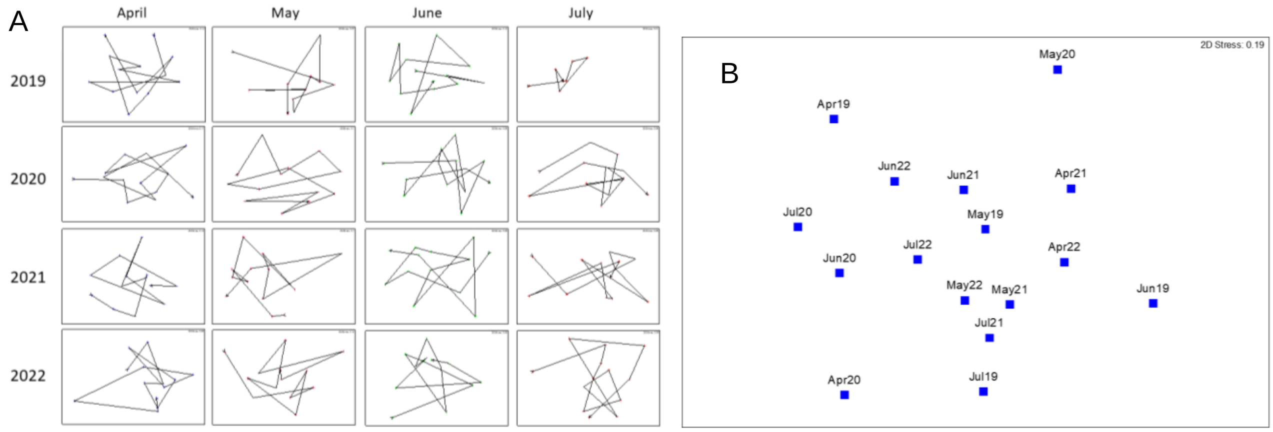

Figure 30. nMDS plots based on Bray-Curtis similarities of square root transformed abundance of benthic invertebrates at Bachelor Island April–July, 2019–022 (A). Second stage nMDS plot (B) based on Pearson rank correlations between pairs of the 16 resemblance matrices corresponding to benthic community assemblages during each month and year sampled at Bachelor Island.

4.0 Discussion

This research evaluated the environmental conditions and benthic invertebrate assemblages in off-channel habitats in the lower Columbia River and estuary. In addition to providing new insights into spatial and temporal factors related to environmental and biological conditions in the benthos, this study evaluated the effect of dredged material placement.

4.1 Environmental Conditions

4.1.1 Sediment Nutrients and Grain size

All sites had low absolute amounts of total carbon and total nitrogen. This may be expected for surficial sediments in higher energy environments, such as river sediments and areas of higher biological activity. Bulk surficial sediments from the Columbia River Wells Dam have reported percent carbon values of 0.5, whereas the Columbia River mouth percent carbon has been reported at 1.34 (Meyers, 1997; Keil et al. 1994; Prahl et al.1994). Thus, the sites sampled in our study, both before and after dredged material placement, were within the range of previously reported carbon content in the region.

Martin Island, the location with the highest carbon and ammonium concentrations also had the highest average silt and lowest average sand content, whereas the trends in environmental conditions at Bachelor Island were opposite—lowest carbon and ammonium, lowest silt, and highest sand content. The environmental conditions at Woodland Islands were intermediate to those at the other locations. These findings are generally consistent with the understanding that nutrients bind more with sediments of finer particle sizes (Meyers, 1997; Garvey and Whiles, 2017). Stations at each of the three locations were clustered in the upper-most segments of the channels behind Columbia River mainstem islands, and yet the position of sites within the overall landscape likely influenced differences in hydrologic conditions and connectivity to the mainstem, that in turn affected retention of nutrients in sediments. For example, the upstream entrance to the channel behind Bachelor Island experienced notable shoaling. While we never observed the channel to be completely cut off, this meant the entrance was sometimes not passable via boat, suggesting hydrologic connectivity was constrained at this location compared to the other sites sampled. While shoaling sometimes occurred at the upstream ends of the Woodland Islands complex, we noticed this site experienced a greater degree of exposure to wind and waves, with similar frequency and intensity to those we observed in the mainstem, compared to the other locations. These nuances among locations, coupled with the results of our study, indicate that local, site-scale conditions are important considerations that likely affect environmental conditions in sediments of off-channel habitats in the region of the LCRE we investigated.

There was a trend of increasing silt and decreasing sand at Martin Island during the four-year study period that was not observed at the other two locations. Grain size distribution in off-channel sediments is influenced by local hydrogeomorphic conditions and by site location within landscape-scale processes such as tidal-fluvial forcing, mainstem flow reversal, and proximity to tributaries. Sediment transport in the lower Columbia River and estuary is typically driven by winter storms or the spring freshet. Possible sources that may have contributed to sediment conditions observed in our study include the Columbia River mainstem, and tributaries such as the Lewis and Willamette Rivers. Given the site-scale variability in sediment conditions, coupled with the USACE’s goals on beneficial use of dredged material, it may be worth investigating the influence of sediment loads from the Columbia and relevant tributaries. This may help to elucidate factors governing site-scale sediment conditions and provide better insights into landscape-scale planning for future dredged material placement sites.

4.1.2 Spatial and Temporal Variability

All environmental response variables, except for phosphorous, showed significant spatial variability among sites. Spatial variation in environmental response variables have been observed in stream/river (Farrell et al. 2018), wetland (Lu et al. 2018), and estuarine habitats (Rauch & Denis, 2008). In these studies, variation in nutrients—carbon, nitrogen, phosphorus—was associated with differences in particle size of sediment (e.g., sand, silt, clay) and organic matter (e.g., coarse, fine). Other factors governing variation in nutrients in these studies included distribution of sediment and organic matter, influence of riparian and instream vegetation communities, microbial and invertebrate communities, and position within the hydrological system (Farrell et al. 2018, Lu et al. 2018, Rauch and Denis, 2008). Sources of nutrient content in tidal wetlands are also related to sediment delivery from upstream locations (Whigham, 2009). While all three of our sites were in the Lower Tidal River hydrogeomorphic zone of the LCRE (Jay et al. 2016), and the areas sampled included the upper reaches of off-channel habitats, it is likely that local conditions relative to hydraulic connectivity with the mainstem, local influence of wetland and riparian vegetation, and differences in grain size and invertebrate assemblages all influenced sediment nutrient values.

Typically, amounts of carbon and nitrogen vary temporally in response to the growing season and changes in flow (Allan and Castillo 2007). For example, water samples collected within the Columbia River estuary have demonstrated fluctuations in nutrient concentrations associated with seasonal freshets, which transport nutrients through the river, and phytoplankton blooms, which utilize nutrients for growth (Gilbert et al. 2013). When assessed as a whole, there is little evidence of a temporal effect within this study across the years, except for phosphorous values in 2019 and ammonium values in 2022; however, there is a lot of variation around the marginal means estimates and minor trends in temporal variation appear within the site-scale level, possibly masked by the strength of the spatial variability. Specifically, ammonium levels appear to increase across the years within each site and at Bachelor and Martin the increase was consistent through the study period while there was no change at Woodland until a significant increase occurred in 2022. Lack of a significant temporal signal may be due to tidal inundation which results in daily changes in duration of inundation as well as fluctuations of water depth, possibly resulting in resuspension and dispersion of organic matter, potentially masking the longer-term temporal effect.

4.1.3 Before-After-Control-Impact

We evaluated the before and after effect of dredged material placement at Woodland Islands relative to conditions at the two control sites, Martin Island and Bachelor Island. There was a statistically significant BACI effect for carbon, sand, and ammonium, but not for phosphorous.

While the percent sand decreased at both the control and impact sites before vs. after dredged material placement, the reduction at the control sites was greater than what was modeled at the impact site. This minimal reduction in sand content at a dredged material site is counter to what might be expected. In an Illinois River study of characteristics associated with different dredged material conditions, Koel and Stevenson (2002) found sites that received dredged material were predominantly characterized by greater sand content, compared to sites that never received dredged material. Similarly, in the lower Columbia River Estuary, Hinton et al. (1992a, 1992b) characterized sediments from dredge placements at Miller Sands and Rice Island as being predominantly median grain to fine sand. In our study, the estimated marginal means of sand content across all three sites ranged from 60-90% and silt content was less than ~15% for most times sampled. The absence of an overall increase in sand may be explained by similarities between the material dredged from the Columbia mainstem and the material that existed at Woodland Island before placement. The sediment conditions we encountered have likely been influenced by a combination of historic and present-day conditions related to operation of the Federal Columbia River Hydropower System, dredging, pile dikes, and hydrodynamics. The Woodland Islands site has evolved due in part to historic dredged placement and installation of a pile dike system that began in the 1920s (USACE 2017). Likewise, any effects to sediment grain size from dredged material placement may have been short-lived and were otherwise not detected when the monitoring resumed approximately six months after placement.

The absence of an increase in percent sand may also be explained by the distribution of stations relative to the placement of dredged material at Woodland Islands. Only seven of the 27 sites at Woodland Islands ended up being on or very close to the dredged material placement zone. It is possible that if there were changes in sand content from dredged material placement, these were localized to the placement area—i.e., changes in sand did not occur at the majority of stations sampled at Woodland Islands or were not significant enough to produce a signal. Recall the BACI study uses an equal sampling density across impact (Woodland Islands) and control (Martin and Bachelor Islands) locations and was not designed to distinguish changes between individual stations.

Percent carbon also decreased at the impact site, relative to before and after dredged material placement, and compared to the control sites. The amount of carbon in a depositional environment (e.g., sediments) is related to system productivity (e.g., vegetation, benthic invertebrate community) and the energy of the environment. On a local scale, microbial and invertebrate communities process carbon sources into particulate (coarse or fine) or dissolved forms. Changes in flow mobilize and disperse carbon sources throughout the fluvial system (Allan and Castillo, 2007). Reductions in total carbon could be attributed to direct burial of aquatic vegetation, algal mats, and other carbon sources by the dredge placement. Another factor could be that the placement altered the instream structure of the site, which could affect localized patterns in flow altering the existing mosaic of carbon deposits across the site. Also, it is possible that the placement of the dredged material caused resuspension and redistribution of lightweight, surficial sources of carbon at the site.

While ammonium concentrations increased at both the control and impact sites after dredged material placement, the BACI effect is explained by the increase in ammonium at the control locations, relative to the increase at the impact site. Production of ammonium requires microorganisms to complete nitrogen fixation, and concentrations in aerobic environments are typically low because ammonium is quickly oxidized to nitrate or assimilated by organisms (McClain et al. 1998). Lower ammonium production at the impact site in comparison to the control sites could, again, be attributed to the changes in instream structure impacting flow and sediment accretion, which may impact the microbial communities responsible for fixing nitrogen. While our study did not collect samples of the dredged material prior to placement, the source of the material came from the mainstem on the opposite side of Woodland Islands. Due to this proximity, perhaps there were similarities in the microbial community that were able to persist following the impact of dredge placement.

4.2 Benthic Invertebrates

4.2.1 Benthic assemblages in off-channel habitats

Our study found that off-channel habitats were producing benthic invertebrates, many of which are common prey items for juvenile salmon and steelhead, e.g., insects, chironomids, crustaceans, and corophium. Weitkamp et al. (2022) found that 60% of all prey consumed by juvenile Chinook salmon and steelhead migrating in the mainstem of the lower Columbia River and estuary were chironomid insects and corophiid amphipods. In a meta-analysis of sub-yearling Chinook salmon foraging characteristics across the gradient of the lower Columbia River and estuary, Martin-Schwarze et al. (2018) found that 60-80% of the non-fish biomass in gut contents of salmon captured in the tidal river zones came from insects, predominantly Diptera (the order to which the Chironomidae family belong). This study also found that larger fish tended to consume more benthic-orienting prey (e.g., amphipods, annelids, molluscs), a pattern that has been observed in other estuarine diet studies of juvenile salmon (Bottom et al. 2011, Macdonald et al. 1987).

Out of the 88 taxonomic designations from our study, approximately five taxa—oligochaetes, chironomids, nematodes, corophium, and *Corbicula fluminea*—were responsible for explaining similarities within sites. These same taxa were also responsible for explaining dissimilarities of assemblages across sites with dissimilarities being explained by lower abundances at Bachelor Island and higher abundances at Martin Island. The commonality of taxa that describe both the similarities within and dissimilarities within and between sites indicates the taxonomic composition was similar across the range of sites we sampled, and that the same five taxa are driving benthic community patterns in this region of the LCRE. The predominance of these taxa is consistent with findings from other tidal freshwater systems in the LCRE, where oligochaetes and chironomid larvae tend to be numerically dominant contributors to macrobenthic assemblages (Jones et al. 1990, McCabe and Hinton 1998). Amphipods, specifically *Americorophium* sp., also contribute notable proportions to the abundance in these systems (Jones et al. 1990, Hinton et al. 1992a, Findlay et al. 2009).

Taxonomic dissimilarities among sites were largely explained by higher abundances of the top five taxa at Martin Island, compared to other sites; recall Martin Island had the lowest sand and highest silt content. In a study of beneficial use of dredged material, Bolam et al. (2006b) noted abundance and diversity of invertebrates was positively associated with finer sediments. The exception of higher abundances at Martin Island was for *Corbicula,* whose dissimilarities among benthic assemblages across sites were explained by higher abundances at Bachelor Island. *Corbicula fluminea* is a non-native clam that has been established in waterways of the U.S., including the LCRE, for nearly a century. *Corbicula* is tolerant of a broad range of sediment compositions, but it has an affinity for high sand content and low slopes in Columbia River habitats (McCabe et al. 1997, Robb-Chavez et al. 2023). These characteristics are most consistent with the habitat conditions found at Bachelor Island. While densities of *Corbicula* were 3.8 times greater at Bachelor compared to Martin Island, Bachelor Island densities were 1.3 times that of Woodland, over the course of the study. In the two years following dredged material placement, the densities of *Corbicula* at Woodland Islands decreased 9.7%, while the density at Bachelor Island increased 3.2%.

While chironomids were the most abundant insect observed across the three sites, twenty-nine other families of insects representing eight different orders were also sampled. Benthic substrates support a diverse richness of insect taxonomies, as was observed by Sather et al. (2020), where benthic invertebrate samples in LCRE wetland channels were dominated by insects (~50% of the invertebrate composition). In our study, the contribution of insects to the benthic community was lower compared to the Sather et al. (2020) study and is likely explained by the predominance of oligochaetes. Oligochaetes were 2.8 times more abundant than the next most abundant taxa, the chironomids, and when their contribution is excluded, total insect contribution across all sites increases from 16.3% to 28.2%. Larval and pupal dipterans tend to be the most common insect observed in benthic samples, which is typically driven by large proportions of larval chironomids (Storch and Sather 2011, Haskell et al. 2006). Benthic organisms provide foraging resources for numerous aquatic organisms, and yet the connection between the benthos and juvenile salmon may be somewhat indirect. Holometabolous invertebrates, those that transition to an adult form following a distinct larval and pupal stage (e.g., diptera, trichoptera, megaloptera), may spend a season or years growing as larva in the benthos and are often consumed at the water surface during adult emergence, or when they return to the water surface to deposit eggs (Bottom et al. 2009, Spilseth and Simestad 2011, Sather et al. 2020). Regardless of the mechanisms associated with the transfer of energy between benthos and juvenile salmon, the off-channel habitats in our study produce beneficial prey resources for these species.

4.2.2 Spatial and Temporal Variability

Spatial and temporal factors were important in explaining variation in the benthic community in our study; site explained more than twice the amount of variation in the data than year. In wetlands within the LCRE Upper Estuary Zone, Sather et al. (2020) found the variation in benthic community assemblages was primarily explained by site (60%) and season (21%), and predominant taxa included larval insects, amphipods, and nematodes. These taxa, in addition to many others, were also found in the benthos of our study in off-channel habitats further upriver. In a study of Chinook salmon feeding ecology in off-channel habitats near LCRE’s Sandy River delta region, Storch and Sather (2011) noted there was considerable spatial and temporal variation in the selection of prey by juvenile salmon. Despite occurring in the diets of fish intermittently, amphipods were noted to be a preferred prey item, relative to the abundance of other taxa found in the diets of fish and taxa in the benthos. These findings, along with our findings, suggest that local conditions at some off-channel habitats may be important for augmenting the foraging potential of migrating juvenile salmon.

We saw evidence of the influence of environmental conditions on benthic assemblages during 2022 when the densities of invertebrates declined at all sites. This may be related to river discharge and the higher-than-average conditions during 2022, compared to the previous three years. Changes in flow regimes, particularly high flow events, can have a significant adverse impact on benthic invertebrates (Bruno et al. 2016). In a study across multiple habitat types within the tidal fluvial zone of the Columbia River, Jones et al. (1990) observed that community structure and distribution of invertebrates was most impacted by seasonal variations in flow, and flows impacted habitats differently due to differences in structure and connectivity. Spatial variability in benthic communities is driven by environmental factors—habitat connectivity, hydrodynamics, physiochemical conditions—as well as biological conditions such as organism mobility and foraging resources (Ptatscheck et al. 2020; Li et al. 2018). Yet, dynamic environmental conditions in estuaries associated with seasonal and interannual variation can influence benthic invertebrate recruitment (Chainho et al. 2007).

4.2.3 Before-After-Control-Impact

Our results indicated that the estimated mean abundance for the three invertebrate response variables—total abundance, total chironomid abundance, and total corophium abundance—was lower at the impact site after dredged material placement, compared to the control sites. The largest effect was seen for total invertebrate abundance, and while the abundance of many taxa decreased following dredged material placement at Woodland Islands, oligochaetes showed a more remarkable decline. In aquatic ecosystems, oligochaetes are associated with habitats ranging from sand to mud, and several species are reportedly adapted to conditions that include high organic matter and low dissolved oxygen (Thorp and Covich 2010). The burrowing behavior of oligochaetes, coupled with their algal and detrital feeding habits, means these organisms affect sediment mixing and microbial activity in aquatic environments (Van Regteren et al. 2017). There are several plausible mechanisms associated with dredged material placement and impacts to oligochaetes, as well as other benthic invertebrates. The deposition of sediment and subsequent burial of invertebrates is a common mechanism affecting community response (Bolam and Rees 2003). In a field experiment designed to understand vertical migration of invertebrates following dredged material, Bolam (2011) found that oligochaetes were able to migrate with 6 cm of sediment placed over experimental plots, but no migration occurred in plots with 16 cm of sediment. The deposition of sediment at Woodland Islands ranged from 0-6 m and included a 13.5 acre footprint (USACE 2023). Due to the extent of dredged material placement at Woodland Islands, vertical migration by invertebrates was an unlikely possibility for organisms. Recolonization of dredged material placement sites typically comes from organisms that migrate from nearby habitats, which may include horizontal migration, larval recruitment, or drifting and swimming organisms in the water column (Bolam et al. 2010, Bolam and Rees 2003).

At Woodland Islands, corophium were dominated by *Americorophium salmonis* and estimated mean abundances decreased 88% across the site after placement of dredged material. The decrease of corophium was especially notable at the stations directly associated with the dredged placement, which experienced a 90% loss of corophium. Although a trend of decreased abundance of corophium at the control sites was also observed, the loss of corophium at Woodland Islands was significantly greater. While the overall trend in estimated mean abundance for chironomids at Woodland Islands indicated an 8% decline after placement, results were variable. The seven stations most closely associated with placement experienced a 9% decline in chironomids, while the remaining stations at Woodland Islands experienced a 29% increase after placement. These findings suggest the placement of dredged material may have indirectly affected juvenile salmon foraging opportunity at Woodland Islands and that effect varied by station across the site.

While juvenile salmon consume a diversity of prey items, chironomid insects and corophiid amphipods are notably prevalent in the diets of migrating juveniles in wetland and mainstem habitats (Haskell et al. 2006, Bottom et al. 2009, Eaton 2010). In the lower estuary, multiple surveys have been conducted to assess benthic conditions prior to instream construction (pile dike replacement) and dredge activities (excavation and placement) (Hinton et al. 1992a, 1992b; McCabe and Hinton, 1990, 1998; McCabe et al. 1990, 1997). These studies identified the tube-dwelling amphipod *A. salmonis* as a dominant benthic invertebrate in the LCRE benthos, and found greater densities on shallow demersal slopes and tidal flats with low flow rates (Jones et al. 1990, McCabe and Hinton 1998). In the Illinois River, Koel and Stevenson (2002) found chironomids were less abundant at sites that had received dredged material compared to those that had not. The immediate effects of dredged material are likely explained by burial, but our findings of differences two-years post-placement suggests that other factors, such as recruitment or changes in hydrodynamics due to the new habitat feature, may be affecting recovery. In addition, the deposition of material has resulted in the conversion of deep to shallow habitats, and the change in elevation may mean that pre-placement conditions, in terms of structure and function, are not viable, at least at some stations.

There are numerous factors governing the magnitude and duration of environmental and biological responses associated with dredged material, and these factors translate into notable variation in the recovery of benthic communities. Rapid recovery rates (e.g., weeks to months) have been reported in association with habitats that are already disturbed, stressed, located in high-energy environments, or associated with natural high-disturbance regimes (Bolam and Rees 2003, Wilber and Clarke 2007). Much of the variation in response and recovery has been attributed to differences among habitats, local disturbance regimes, and differences in volumes of dredged material (Bolam and Rees 2003). Notably, the depth and volume of dredged material placement has been negatively correlated with species richness and abundance in the benthos (Dauvin et al. 2022). Expectations for recovery rates of a given project and location must be tempered with these conditions in mind, and care should be taken when attempting to translate outcomes of projects from different locations, habitats, ecosystems, and climates. Our review of the literature on dredged material placement did not find an ideal comparison to the Woodland Islands project, making it difficult to say whether the results we found are expected or similar to other studies. It is also challenging to address questions about the expected trajectory of benthic community response given our post-placement monitoring showed an effect two years after monitoring, and we didn’t see clear trends in the community response to the impact. As the new habitat feature evolves, recolonization will likely follow. Additional monitoring will provide insights into the trajectory and outcome of the project relative to supporting foraging potential for juvenile salmon.

When considering the reduction in invertebrate abundances at Woodland Islands, it is important to note that the site was not completely devoid of benthic invertebrates. Many of the same taxa persisted at the site following dredged material placement, compared to before placement. However, we demonstrated through both univariate and multivariate analytical methods that the structure of the benthic assemblage had changed, and while we noted a decrease in observed mean abundance for several taxa at Woodland Islands, some increased. The research we conducted was designed to evaluate the impact at Woodland Islands with what had originally been planned as two distinct placement locations within the site and twice as much dredged material (~400,000 cy) than was placed. Our pre-placement monitoring had already begun when the placement design changed, and it was not possible to alter the location of our randomly selected stations without compromising the study design. The impacts detected through our analyses may have been driven by the influence of the seven stations most closely associated with the placement. For example, out of the 27 stations sampled across the Woodland Islands site, the impact of dredged material placement may have been limited to the vicinity of placement and great enough that it influenced the outcome of the results. Alternatively, it is possible the placement had an impact on benthic communities across the Woodland Islands site, regardless of whether a station was near or far from the placement location. This seems unlikely because the observed mean abundances revealed that not all taxa were adversely affected (i.e., some increased after placement), and community characteristics and diversity measures at Woodland Islands were similar before versus after placement. We recommend further investigation into the influence of distance relative to location of dredged material placement and hypothesize that benthic invertebrates nearest to the dredged material placement will be most affected.

There are seemingly endless ways to measure the impact of a dredged material project, and disturbance is frequently described via changes in community structure (e.g., diversity, abundance, biomass; Bolam and Rees 2003) with few studies including functional measures of response (e.g., biological traits, trophic measures, ecological modeling; Bolam 2012). In studies where investigators used multiple response variables, results can be contradictory; for example, some variables may indicate an impact from dredged placement, whereas others suggest no impact. Pezy et al. (2018) used a combination of structural and functional variables to evaluate the impact of disposed dredged material on benthic communities in the Bay of Seine, France. Diversity, evenness, and other biotic indices yielded ambiguous results, whereas some structural measures—e.g., abundance, richness, biomass—and ecological modeling approaches helped to elucidate the effects of dredged material placement. Bolam (2012) argues that the key function of benthic invertebrates is to provide a foraging base for secondary consumers, and that quantifying secondary production and energy transfer is an ecologically relevant measure that should be incorporated into dredged material projects. It is possible that future efforts to track the biological response at Woodland Islands could use secondary productivity, with archived samples from this study as a response variable. The inclusion of functional response variables may help to inform trajectories, as well as the uncertainties of beneficial use sites in the LCRE. By using a more holistic ecological approach, studies that consider functional responses may be best poised to translate potential benefits of dredged material projects into terms that are more directly relevant to the management of target species of concern, such as threatened and endangered salmon populations.

Benthic invertebrates have been used as indicators of ecosystem health as well as disturbance. For example, many genera and species in the Chironomidae family are used to make assessments about ecological health because they are the most widespread insect family, they occur in most wet habitats (e.g., river, lake, puddles), and they demonstrate evolutionary plasticity (Rosenberg and Resh, 1993; Ferrington et al. 2008). Because of this, chironomids are generally perceived as tolerant of disturbance and are often the first colonizers after a disturbance event (Serra et al. 2017). Our modeled results indicated a negative response by chironomids to dredged placement, despite only 25% of the Woodland Islands stations being located on or in the direct vicinity of the dredged material placement. Similarly, Koel and Stevenson (2002) found chironomids were less abundant in an Illinois River at sites that had received dredged material compared to those that had not. In another example using taxa as indicators of disturbance, Bolam and Rees (2003) described oligochaetes as opportunistic and noted these can be one of the first colonizers after a dredged material placement. However, our study found this group of invertebrates was one of the most negatively impacted after dredged material placement. Discrepancies between studies on taxa-specific responses to disturbance may relate to a combination of environmental and biological conditions, as well to the taxonomic resolution among datasets. For example, while chironomids may generally be thought of as tolerant of disturbance, the Chironomidae family includes approximately 10,000 species (Cranston, 1995) and inherent variation among life history characteristics likely translates into a range of responses, making generalizations challenging (Lencoioni et al. 2012, Serra et al. 2017).

It is relevant to acknowledge that our study did not identify taxa beyond the family level. Depending on the research questions and the availability of taxa-specific literature to reference, finer taxonomic resolutions may help to elucidate taxa-based responses to disturbance. However, we were interested in the response of select salmon prey taxa, as well as the response of invertebrate abundance and community structure. These response measures helped to provide a broader picture of the ecological response associated with dredged material placement. Additionally, salmon diet data are not typically reported at taxonomic resolutions lower than family levels (i.e., partially digested gut contents are difficult to speciate) and evaluating chironomid and corophium at lower taxonomic levels would not have helped to provide additional insights into implications of the dredged material project relative to juvenile salmon foraging potential. Pires et al. (2021) found little evidence that finer taxonomic resolutions (genus level) provided different insights into wetland invertebrate communities compared with higher resolutions (family level). With this in mind, we suggest that when planning for future studies, care should be taken when considering taxonomic resolution, as this will help to prioritize response variables relative to research questions and available resources.

4.3 Model performance

A BACI model incorporated two fixed effects: one for Before-After (BA) and another for Control-Impact (CI) or 'treatment' effects. On their own, these primary effect terms lack informative value in impact assessment. A significant BA effect merely signifies a change from before to after the impact, potentially stemming from natural temporal variations across all locations. Similarly, a significant CI effect may result from inherent spatial differences between control and impact locations across all time periods. In the BACI model, the crucial factor lies in the whether the interaction term between the BA and CI terms are significant, which indicates a significant treatment effect. Additional considerations include the presence of spatiotemporal variability, which can obscure the actual treatment effects (Coleman et al. 2013), leading to high Type II error (i.e., failure to detect a significant effect when one exists) and subsequently low statistical power (Fisher et al. 2019). Consequently, it becomes imperative to scrutinize spatiotemporal variability and address it by incorporating random effects into the model. In our approach, we included random effects for site, year, and station in the model to account for such potential variability.

The optimal models, chosen based on AICc for carbon percentage and sand percentage, revealed the significance of the random effect of the interaction between station and year, in addition to the random effects of site, station, and year. The significance of the interaction term "station and year" suggests the presence of unobserved variability resulting from the joint influence of station and year. This accounts for variability across stations and years that isn't captured by only grouping years for BA and grouping stations for CI. In the case of sand percentage, the optimal models identified from AICc indicated a significant interaction between station and site, in addition to the random effects of site, station, and year. This points to unobserved heterogeneity or variability in sand content arising from the joint influence of stations and sites. It's worth considering that because stations are nested within sites, the impact of sand content may vary across each site and station combination. For ammonium content, the BACI model selected from AICc included site, station, and year as random effects, highlighting unexplained variability from these variables not accounted for by BA and CI in fixed effects. In the case of phosphorus content, the AICc-selected model identified only site and year as significant random effects.

In our analyses for benthic abundance, the optimal models for total abundance, chironomid abundance, and corophium abundance chosen by using AICc incorporated random effects for site, station, year, and the interaction of station and year. The statistically significant random effect representing the interaction between station and year indicate variations in abundance across different stations and years. The models also incorporated observation-level random effects, to satisfy the distributional assumptions of the fitted GLMMs using Poisson distribution. The random effects address individual variability within site, station, and year, adjusting the estimates to provide a more generalized representation of the population means.

We noted the estimated marginal mean abundances appear lower than the sample means. It is important to recognize that sample means reflect the actual group averages without considering the underlying structure or variation due to random effects. Estimated marginal means, derived from the model, incorporate both fixed and random effects, and present adjusted means accounting for the overall population mean (fixed effects: group means) and individual variability within groups (random effects: station: year). This adjustment results in a more generalized representation of population means. The inclusion of random effects induces "shrinkage," pulling estimates toward the overall mean to enhance model generalization and mitigate undue influence from rare or extreme observations (Bell et al. 2019).

The visual representations of the distribution of benthic abundance data for control-impact vs before-after groups within the box plots was similar, and yet the GLMMs resulted in a significant BACI effect. Several factors could contribute to this. It is crucial to recognize that box plots primarily serve as visual summaries, highlighting central tendency, spread, and potential outliers and deviations from normality and skewness in distributions can obscure differences in box plots. Unlike box plots, our GLMM accommodates these distributional deviations, yielding valid results even when the data do not adhere strictly to normality. Additionally, box plots might lack sensitivity to subtle mean differences, particularly when there is high variability within groups. Furthermore, the visualizations may not adequately consider random effects, leading to an incomplete portrayal of the data. In contrast, the GLMM explicitly addresses overall variability within and between groups, incorporating both fixed and random effects, enhancing sensitivity to mean differences. We also determined that including the interaction of year and station as random effects substantially increased the variability explained by the model. Such interaction effects may contribute to differences that are challenging to discern in simple visualizations such as box plots. While box plots serve a valuable purpose in offering an overview of the distribution of data, our GLMM provides a more robust and thorough analysis, incorporating both visual patterns and the underlying statistical significance, making it a more powerful tool for discerning nuanced relationships within the data.

Contending with the inherent variability in ecosystems is a challenging task, especially when attempting to explain and project effects governed by complex biological and environmental interactions (Peek et al. 2003). Our models for total abundance and corophium abundance explained 60% of the variability in the data and the model for chironomid abundance explained 30% of the variability. In a meta-analysis designed to evaluate trends in benthic diversity across large spatial scales, Renaud et al. (2009) reported multiple statistically significant results with coefficients of determinations less than 0.41. In that study, “weak but significant” trends were defined by $R^2 < 0.1$. Our R^2 values suggest that our models explain a similar proportion of variability in the data, contributing to a representation of underlying population trends, while minimizing the impact of noise or outliers associated with benthic communities.

5.0 Conclusion

Benthic communities and some environmental variables were impacted by dredged material placement at Woodland Islands. Many factors interact to determine the magnitude and duration of environmental and biological responses associated with dredged material, and these factors translate into notable variation in the recovery of benthic communities. Literature on dredged material placement indicates outcomes may result in rapid recovery (weeks to months), or trajectories with slower responses (years) are also possible. At Woodland Island, the estimated mean abundance for all invertebrates decreased 28% after dredged material placement. Estimated mean abundances of chironomid and corophium decreased by 8% and 88%, respectively. When considering the reduction in invertebrate abundances at Woodland Islands, it is important to note that the site was not completely devoid of benthic invertebrates. Many of the same taxa persisted at the site following dredged material placement, compared to before placement, but we demonstrated through both univariate and multivariate analytical methods that the structure of the benthic assemblage at Woodland Islands changed. As the new habitat feature evolves, recolonization will likely follow. In addition to evaluating biological and environmental response associated with dredged material placement, this study has demonstrated that off-channel habitats of the LCRE produce benthic invertebrates that are important foraging resources for migrating juvenile salmon and steelhead. We have gained insights into considerations of spatial and temporal variation as well as methodologies and approaches for evaluating the effectiveness of dredged material projects focused on creating beneficial habitats. These findings provide a foundation for understanding potential benefits and consequences of repurposing dredged material for habitat creation in the LCRE.

6.0 References

Allan, J. D., and M. M. Castillo. 2007. *Stream Ecology: Structure and Function of Running Waters*. 2nd ed. New York: Chapman and Hall.

Anderson, M. J., R. N. Gorley, and K. R. Clarke. 2008. *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*. PRIMER-E, Massey University, Albany, Auckland 0632, New Zealand.

Bell, A., M. Fairbrother, and K. Jones. 2019. “Fixed and Random Effects Models: Making an Informed Choice.” *Quality & Quantity* 53 (2): 1051-1074.

Bolam, S. G. 2011. "Burial survival of benthic macrofauna following deposition of simulated dredged material." *Environmental Monitoring and Assessment* 181(1-4): 13-27.

Bolam, S. G. 2012. "Impacts of dredged material disposal on macrobenthic invertebrate communities: A comparison of structural and functional (secondary production) changes at disposal sites around England and Wales." *Marine Pollution Bulletin* 64(10): 2199-2210.

Bolam, S. G., et al. 2010. "Macrofaunal recolonisation following the intertidal placement of fine-grained dredged material." *Environmental Monitoring and Assessment* 168(1): 499-510.

Bolam, S. G. and H. L. Rees 2003. "Minimizing impacts of maintenance dredged material disposal in the coastal environment: A habitat approach." *Environmental Management* 32(2): 171-188.

Bolam, S. G., H. L. Rees, P. Somerfield, R. Smith, K. R. Clarke, R. M. Warwick, M. Atkins, and E. Garnacho. 2006. “Ecological consequences of dredged material disposal in the marine environment: A holistic assessment of activities around the England and Wales coastline.” *Marine Pollution Bulletin* 52(4), 415-426. https://doi.org/https://doi.org/10.1016/j.marpolbul.2005.09.028

Bolam, S. G., M. Schratzberger, and P. Whomersley. 2006. “Macro-and meiofaunal recolonisation of dredged material used for habitat enhancement: temporal patterns in community development.” *Marine Pollution Bulletin* 52(12), 1746-1755.

Bolam, S. G., and P. Whomersley. 2005. “Development of macrofaunal communities on dredged material used for mudflat enhancement: a comparison of three beneficial use schemes after one year.” *Marine Pollution Bulletin* 50(1), 40-47. https://doi.org/https://doi.org/10.1016/j.marpolbul.2004.08.006

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J.-S. S. White. 2009. “Generalized Linear Mixed Models: A Practical Guide for Ecology and Evolution.” *Trends in ecology & evolution (Amsterdam)* 24 (3): 127-135. https://doi.org/10.1016/j.tree.2008.10.008.

Borde, A.B., V.I. Cullinan, H.L. Diefenderfer, R.M. Kaufmann, S.A. Zimmerman, J. Sagar, K.E. Buenau and C. Corbett. 2012. Lower Columbia River and Estuary Ecosystem Restoration Program Reference Site Study: 2011 Restoration Analysis. Richland, Washington, Pacific Northwest National Laboratory

Bottom, D. L., A. Baptista, J. Burke, Campbell, E. Casillas, S. Hinton, D. A. Jay, M. A. Lott, G. McCabe, R. McNatt, M. Ramirex, G. Roegner, C. Simenstad, S. Spilseth, L. Stamatiou, D Teel, and J. E. Zamon. 2011. *Estuarine Habitat and Juvenile Salmon: Current and Historical Linkages in the Lower Columbia River and Estuary Final Report 2002-2008*. Fish Ecology Division, Northwest Fisheries Science Center, U.S. National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, WA.

Bottom, D., C. Simenstad, L. Campbell, J. Burke, E. Casillas, B. Craig, C. Eaton, S. Hinton, R. McNatt, G. Roegner, S. Schroder, L. Stamatiou, and J. Zamon. 2009. *Historic Habitat Opportunities and Food-Web Linkages of Juvenile Salmon in the Columbia River Estuary*. Annual Report of Research prepared for the U.S. Department of Energy, Bonneville Power Administration, Environment, Fish, and Wildlife Division, Portland, Oregon, by the Fish Ecology Division, NOAA Fisheries, Northwest Fisheries Science Center, Seattle, WA.

Bottom, D., G. Anderson, A. Baptista, J. Burke, M. Burla, M. Bhuthimethee, L. Campbell, E. Casillas, S. Hinton, K. Jacobson, D. Jay, R. McNatt, P. Moran, G. Roegner, C. Simenstad, V. Stamatiou, D. Teel, and J. E. Zamon. 2008. *Salmon Life Histories, Habitat, and Food Webs in the Columbia River Estuary: An Overview of Research Results, 2002-2006*. Report prepared for the U.S. Army Corps of Engineers, Portland District, Northwest Division, Portland, Oregon, and Bonneville Power Administration, Environment, Fish, and Wildlife Division, Portland, OR, by the Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle WA.

Bruno, M. C., M. J. Cashman, B. Maiolini, S. Biffi, and G. Zolezzi. 2016. “Responses of Benthic Invertebrates to Repeated Hydropeaking in Semi-Natural Flume Simulations.” *Ecohydrology* 9 (1): 68-82

Coleman, M. A., A. Palmer-Brodie, and B. P. Kelaher. 2013. “Conservation Benefits of a Network of Marine Reserves and Partially Protected Areas.” *Biological Conservation* 167: 257-264

Chainho, P., J. L. Costa, M. L. Chaves, D. M. Dauer, and M. J. Costa. 2007. “Influence of seasonal variability in benthic invertebrate community structure on the use of biotic indices to assess the ecological status of a Portuguese estuary.” *Marine Pollution Bulletin* 54 (10): 1586-1597.

Clarke, K. R. and R. N. Gorley. 2015. *PRIMER v7: User Manual/Tutorial.* PRIMER-E Led, Plymouth, UK.

Cranston, P. S. 1995. “Introduction.” In *The Chironomidae: Biology and Ecology of Non-Biting Midges*, edited by P. D. Armitage, P. S. Cranston, and L. C. V. Pinder, 1–7. London: Chapman & Hall.

Cullen, A., and H. Frey. 1999. *Probabilistic techniques in exposure assessment: A Handbook for Dealing with Variability and Uncertainty in Models and Inputs*. New York: Springer Science and Business.

Dauvin, J.-C., N. Bauz, S. Lesourd. 2022. "Benthic impact assessment of a dredge sediment disposal in a dynamic resilient environment." *Marine Pollution Bulletin* 179: 113668.

Dunn, P. 2004. “Log-linear models.” In *Encyclopedia of Social Measurement* Vol. 2, edited by K. Kempf-Leonard, p. 585-589. Dallas, TX: Elsevier.

Dyer, K. 1989. “Sediment Processes in Estuaries: Future Research Requirements.” *Journal of Geophysical Research-Oceans* 94 (C10): 14,327-314,339.

Ebberts, B. D., B. D. Zelinsky, J. P. Karnezis, C. A. Studebaker, S. Lopez-Johnston, A. M. Creason, L. Krasnow, G. E. Johnson, and R. M. Thom. 2018. “Estuary Ecosystem Restoration: Implementing and Institutionalizing Adaptive Management.” *Restoration Ecology* 26 (2): 360-369. https://doi.org/10.1111/rec.12562.

Eaton, C. D. 2010. Resource Partitioning, Habitat Connectivity, and Resulting Foraging Variation Among Salmonids in the Estuarine Habiat Mosaic School of Aquatic and Fisheries Sciences Seattle, University of Washington M.S.

Edwards, R. T. 1998. “The Hyporheic Zone.” In *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*, edited by R. J. Naiman and R. E. Bilby, 399-429. New York: Springer–Verlag.

Farrell, K. J., A. D. Rosemond, J. S. Kominoski, S. M. Bonjour, J. Rüegg, L. E. Koenig, C. L. Baker, M. T. Trentman, T. K. Harms, and W. H. McDowell. 2018. “Variation in Detrital Resource Stoichiometry Signals Differential Carbon to Nutrient Limitation for Stream Consumers Across Biomes.” *Ecosystems* 21(8): 1676-1691.

Ferrington, L. C., W. P. Coffman, and M. B. Berg. 2008. “Chironomidae.” In *An Introduction to the Aquatic Insects of North America*, 4^{th} ed., edited by R. W. Merritt, K. W. Cummins, and M. B. Berg, 847-989. Dubuque, IA: Kendall Hunt Publishing Company.

Findlay, S. E. G., W. C. Nieder, and S. Ciparis. 2009. “Carbon flows, nutrient cycling, and food webs in tidal freshwater wetlands.” In *Tidal Freshwater Wetlands,* edited by A. Barendregt, D. F. Whingham, and A. H. Baldwin 137-144. Netherlands: Backhuys Publishers.

Fisher, R., G. R. Shiell, R. J. Sadler, K. Inostroza, G. Shedrawi, T. H. Holmes, J. M. McGree, and R. Freckleton. 2019. “Epower: An R Package for Power Analysis of before-after-Control-Impact (Baci) Designs.” Methods in ecology and evolution 10 (11): 1843-1853

Foran, C. M., K. A. Burks-Copes, J. Berkowitz, J. Corbino, and B. C. Suedel. 2018. “Quantifying Wildlife and Navigation Benefits of a Dredging Beneficial-Use Project in the Lower Atchafalaya River: A Demonstration of Engineering with Nature.” *Integrated Environmental Assessment and Management* 14 (6): 759-768. https://doi.org/10.1002/ieam.4084.

Garvey, J. E., and M. R. Whiles. 2017. *Trophic Ecology*. Boca Raton, FL: CRC Press.

Gilbert, M., Needoba, J., Koch, C., Barnard, A., and A. Baptista. 2013. "Nutrient loading and transformations in the Columbia River estuary determined by high-resolution in situ sensors." *Estuaries and Coasts* 36 (4): 708-727.

Goldsmith, A. M., F. H. Jaber, H. Ahmari, and C. R. Randklev. 2021. “Clearing up cloudy waters: a review of sediment impacts to unionid freshwater mussels.” *Environmental Reviews* 29 (1): 100-108. https://doi.org/10.1139/er-2020-0080

Haskell, C. A., K. F. Tiffan, R. C. Koch, and D.W. Rondorf. 2006. *Crims Island Habitat Restoration in the Columbia River Estuary – Fisheries Monitoring and Evaluation*, 2004. U.S. Army Corps of Engineers, Portland, OR.

Haskell, C.A, K.F. Tiffan, and J.O. Olson. 2007. *Crims Island Habitat Restoration in the Columbia River Estuary – Fisheries Monitoring and Evaluation, 2006*. U.S. Army Corps of Engineers, Portland, OR.

Hinton, S. A., R. L. Emmett, and G. T. McCabe. 1992a. *Benthic Invertebrates and Sediment Characteristics in Subtidal Habitat at Rice Island, Columbia River Estuary, December 1991 and March 1992*. Coastal Zone and Estuarine Studies Division, Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, WA.

Hinton, S. A., R. L. Emmett, and G. T. McCabe. 1992b. *Fishes, Shrimp, Benthic Invertebrates, and Sediment Characteristics in Intertidal and Subtidal Habitats at Rice Island and Miller Sands, Columbia River Estuary, 1991*. Coastal Zone and Estuarine Studies Division, Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, WA.

Jay, D. A., Borde, A. B., & Diefenderfer, H. L. 2016. "Tidal-Fluvial and Estuarine Processes in the Lower Columbia River: II. Water Level Models, Floodplain Wetland Inundation, and System Zones." *Estuaries and Coasts* 39 (5): 1299-1324. https://doi.org/10.1007/s12237-016-0082-4

Jones, K. K., Simenstad, C. A., Higley, D. L., and D. L. Bottom. 1990. "Community Structure, Distribution, and Standing Stock of Benthos, Epibenthos, and Plankton in the Columbia River Estuary." *Progress in Oceanography* 25(1-4): 211-241.

Keil, R. G., E. Tsamakis, C. B. Fuh, J. C. Giddings, and J. I. Hedges. 1994. Mineralogic and textural controls on the organic composition of coastal marine sediments. Hydrodynamic separation using SPLITT-fractionation. *Geochimica et Cosmochimica Acta* 58: 879-893.

Kincaid TM, AR Olsen, MH Weber. 2019. Spsurvey: Spatial Sampling Design and Analysis. R package version 4.1.0.

Koel, T. M., and K. E. Stevenson. 2002. "Effects of Dredge Material Placement on Benthic Macroinvertebrates of the Illinois River." *Hydrobiologia* 474 (1): 229-238. https://doi.org/10.1023/A:1016500324467

Lencioni, V., L. Marziali, and B. Rossaro. 2012. "Chironomids as bioindicators of Environmental Quality in Mountain Springs." *Freshwater Science* 31 (2): 525-541.

Li, F., J. D.Tonkin, and P. Haase, P. 2018. "Dispersal Capacity and Broad-Scale Landscape Structure Shape Benthic Invertebrate Communities Along Stream Networks." *Limnologica 71:* 68-74.

Lott M. 2004. *Habitat-Specific Feeding Ecology of Ocean-Type Juvenile Chinook Salmon in the Lower Columbia River Estuary, School of Aquatic and Fishery Sciences*. University of Washington, Seattle, WA.

Lu, Q., J. Bai, G. Zhang, Q. Zhao, and J. Wu. 2018. "Spatial and seasonal distribution of carbon, nitrogen, phosphorus, and sulfur and their ecological stoichiometry in wetland soils along a water and salt gradient in the Yellow River Delta, China." *Physics and Chemistry of the Earth, Parts A/B/*C 104: 9-17.

Macdonald J. S., I. K. Birtwell, and G. M. Kruzynski. 1987. "Food Habitat Utilization by Juvenile Salmonids in the Campbell River Estuary." *Canadian Journal of Fisheries and Aquatic Science* 44:1233–1246.

Martin-Schwarze A., N.K. Sather, J.E. Zamon, M.F. Ramirez, and J.R. Cordell. 2018. "Landscape-scale Analysis of Juvenile Salmon Diets in the Lower Columbia River and Estuary." Appendix F in: Johnson GE, KL Fresh, and NK Sather (eds.), *Columbia Estuary Ecosystem Restoration Program, 2018 Synthesis Memorandum*. PNNL-27617, Final report submitted by Pacific Northwest National Laboratory to U.S. Army Corps of Engineers, Portland District, Portland, Oregon. Available at: https://www.cbfish.org/EstuaryAction.mvc/Index.

McCabe, G. T. and S. A. Hinton. 1998. Benthic Invertebrates and Sediment Characteristics in and Adjacent to a Proposed Channel Widening Area (River Mile 31-34), Columbia River Estuary. Seattle, WA, Fish Ecology Division, Northwest Fisheries Science Center, National Marine Fisheries Service: 28.

McCabe, G. T. and S. Hinton. 1990. Benthic Infauna and Sediment Characteristics in the Columbia River near Westport, Oregon, August 1989. Portland, OR: 14.

McCabe, G. T., S. A. Hinton, R. L. Emmett and R. J. McConnell. 1990. Benthic Invertebrates, Sediment Characteristics, and Demersal Fishes Off Cottonwood Island, Columbia River, Before and After Rock Groin Construction, 1987-1988 Coastal Zone and Estuarine Studies Division, Northwest Fisheries Center, National Marine Fisheries Service 64.

McCabe, G. T., S. A. Hinton, R. L. Emmett, and B. P. Sandford. 1997. Benthic invertebrates and sediment characteristics in main channel habitats in the lower Columbia River. Northwest Science 71(1): 45-55.

McCabe, G. T. and S. A. Hinton. 1998. Benthic Invertebrates and Sediment Characteristics in and Adjacent to a Proposed Channel Widening Area (River Mile 31-34), Columbia River Estuary. Seattle, WA, Fish Ecology Division, Northwest Fisheries Science Center, National Marine Fisheries Service: 28.

McClain, M. E., R. E. Bilby, and F. J. Triska. 1998. Nutrient Cycles and Responses to Disturbance. In R. J. Naiman and R. E. Bilby (Eds.), *River Ecology and Management,* pgs.347-372. Springer-Verlag, New York, New York.

Merritt R.W., K.W. Cummins, and T.M. Burton. 1984. Processing and Cycling of Nutrients. In V. H. Resh and D. M. Rosenberg (Eds.), *The Ecology of Aquatic Insects*, pgs. 134-163. Praeger Publishers, New York, NY.

Meyers, P. A. 1997. Organic geochemical proxies of paleoceanographic, paleolimnologic, and paleoclimatic processes. Organic Geochemistry 27(5): 213-250.

Minshall, G.W. 1984. Aquatic Insect-Substratum Relationships. In V. H. Resh and D. M. Rosenberg (Eds.), *The Ecology of Aquatic Insects*, pgs. 358-400. Praeger Publishers, New York, NY.

Montes, MF. 2016. Nutrients. In MJ Kennish (Ed.), *Encyclopedia of Estuaries*. Pgs. 464-466. Springer Netherlands.

Munari, C., and M. Mistri. 2014. Spatio-temporal pattern of community development in dredged material used for habitat enhancement: A study case in a brackish lagoon. Marine Pollution Bulletin, 89(1), 340-347. https://doi.org/https://doi.org/10.1016/j.marpolbul.2014.09.039

Nakagawa, S. and H. Schielzeth. 2013. A general and simple method for obtaining R^2 from generalized linear mixed-effects models. Methods in Ecology and Evolution, 4(2), 133–142. doi:10.1111/j.2041-210x.2012.00261.x

Neyman, J., & Pearson, E. S. 1933. IX. On the problem of the most efficient tests of statistical hypotheses. Philosophical Transactions of the Royal Society of London. Series A, Containing Papers of a Mathematical or Physical Character, 231(694-706), 289-337.

Peek, M. S., Leffler, A. J., Flint, S. D., & Ryel, R. J. (2003). How much variance is explained by ecologists? Additional perspectives. Oecologia, 137, 161-170.

Pezy, J.-P., A. Raoux, S. Marmin, P. Balay and J.-C. Dauvin. 2018. "What are the most suitable indices to detect the structural and functional changes of benthic community after a local and short-term disturbance?" Ecological Indicators 91: 232-240.

Pires, M. M., Grech, M. G., Stenert, C., Maltchik, L., Epele, L. B., McLean, K. I., ... & Batzer, D. P. (2021). Does taxonomic and numerical resolution affect the assessment of invertebrate community structure in New World freshwater wetlands? Ecological indicators, 125, 107437.

Prahl, F. G. and P. G. Coble. 1994. "Input and behavior of dissolved organic carbon in the Columbia River Estuary." Changes in Fluxes in Estuaries: Implications from Science to Management: 451-457.

Ptatscheck, C., Gansfort, B., Majdi, N., & Traunspurger, W. 2020. The influence of environmental and spatial factors on benthic invertebrate metacommunities differing in size and dispersal mode. *Aquatic Ecology*, *54*, 447-461.

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Raftery AR. 1995. Bayesian model selection in social research. In *Sociological methodology*, pages 111–163.

Rauch, M. and L. Denis. 2008. Spatio-temporal variability in benthic mineralization processes in the eastern English Channel. *Biogeochemistry,* 89(2): 163-180

Renaud, P. E., Webb, T. J., Bjørgesæter, A., Karakassis, I., Kedra, M., Kendall, M. A., ... & Zettler, M. L. 2009. Continental-scale patterns in benthic invertebrate diversity: insights from the MacroBen database. *Marine Ecology Progress Series*, *382*, 239-252.

Resh V. H. 1979. Sampling Variability and Life History Features: Basic Considerations in the Design of Aquatic Insect Studies. *Journal of the Fisheries Board of Canada*, 36(3), 290-311.

Robb-Chavez, Salvador B., et al. 2022. "Broadscale distribution, abundance, and habitat associations of the invasive Asian clam (Corbicula fluminea) in the lower Columbia River, USA." *International Review of Hydrobiology* 107.5-6: 179-195.

Roegner, G. C., Baptista, A., Bottom, D. L., Burke, J., Campbell, L., Elliot, C., Hinton, S., Jay, D., Lott, M.-A., Lundrigan, T., McNatt, R., Moran, P., Simenstad, C. A., Teel, D., Volk, E., Zamon, J., & Casillas, E. 2008. Estuarine Habitat and Juvenile Salmon: Current and Historical Linkages in the Lower Columbia River and Estuary, 2002-2004.

Roegner, G. C., Dawley, E. W., Russell, M., Whiting, A., & Teel, D. J. 2010. Juvenile Salmonid Use of Reconnected Tidal Freshwater Wetlands in Grays River, Lower Columbia River Basin. Transactions of the American Fisheries Society, 139(4), 1211 - 1232. http://www.informaworld.com/10.1577/T09-082.1

Rosenberg D.M. and V.H. Resh. 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, Inc. New York, NY.

Sather N. K., A. M. Martin-Schwarze, K. F. Mackereth, and G. J. Johnson. 2020. "Foraging opportunity and diet composition of juvenile Chinook salmon in restoring tidal wetlands." Chapter 3 in *Restoration Action Effectiveness Monitoring and Research in the Lower Columbia River and Estuary, 2016-2017*. Final technical report submitted by PNNL and NMFS to the U.S. Army Corps of Engineers, Portland District, Portland, OR.

Serra, S. R., Graça, M. A., Dolédec, S., and M. J. Feio. 2017. Chironomidae traits and life history strategies as indicators of anthropogenic disturbance. Environmental Monitoring and Assessment, 189, 1-16.

Shousha, S., R. Maranger, and J-F. Lapierre. 2023. Decadal Changes in Anthropogenic Inputs and Precipitation Influence Riverine Exports of Carbon, Nitrogen, and Phosphorus, and Alter Ecosystem Level Stoichiometry. *Global Geochemical Cycles* 37: e2023GB007820. https://doi.org/10.1029/2023GB007820.

Sokal, R. R., & Rohlf, F. J. 1995. Biometry. W.H. Freeman and Company.

Spellman, S. A. 2023. *Beneficial Use of Dredged Material Command Pholosophy Notice*. Department of the Army Headquarters, US Army Corps of Engineers.

Spilseth, S., and C. Simenstad. 2011. Seasonal, Diel, and Landscape Effects on Resource Partitioning between Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) and Threespine Stickleback (*Gasterosteus aculeatus*) in the Columbia River Estuary. *Estuaries and coasts*, *34*(1), 159-171. https://doi.org/10.1007/s12237-010-9349-3

Stevens, D. and A. Olsen. 2004. Spatially Balanced Sampling of Natural Resources. *Journal of the American Statistical Association* 99: 262-278.

Storch, A.J. and N.K. Sather. 2011. Feeding Ecology. In: *Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats of the Lower Columbia River, 2007–2010.* Johnson et al., pp. 4.1–4.20, PNNL-20083, Pacific Northwest National Laboratory, Richland, WA.

Suedel, B. C., McQueen, A. D., Wilkens, J. L., Saltus, C. L., Bourne, S. G., Gailani, J. Z., King, J. K., & Corbino, J. M. 2022. Beneficial use of dredged sediment as a sustainable practice for restoring coastal marsh habitat. *Integrated Environmental Assessment and Management* 18(5), 1162-1173. https://doi.org/https://doi.org/10.1002/ieam.4501

Thorp, J. H., & Covich, A. P. (Eds.). 2010. *Ecology and Classification of North American Freshwater Invertebrates*. Academic press.

Underwood, A. J. 1991. Beyond BACI: Experimental Designs for detecting Human Environmental Impacts on Temporal Variations in Natural Populations. *Marine and Freshwater Research* 42(5), 569-587. https://doi.org/10.1071/MF9910569

Underwood, A. J. 1992. "Beyond BACI: The Detection of Environmental Impacts on Populations in the Real, but Variable, World." *Journal of Experimental Marine Biology and Ecology* 161(2): 145-178.

USACE (U.S. Army Corps of Engineers). 2015. Dredging and Dredged Material Management Engineer Manual, Issue.

USACE (U.S. Army Corps of Engineers). 2017. Woodland Island Beneficial Use of Dredged Material. ERTG template. 33pp

USACE (U.S. Army Corps of Engineers). 2020. IFR & FEA: Section 204 Studies: Beneficial Use of Dredged Material for Ecosystem Restoration, Woodland Islands, Lower Columbia River Estuary. USACE - Portland District, Portland, OR. 127 pp. https://usace.contentdm.oclc.org/digital/collection/p16021coll7/id/14217

USACE 2023. Summary Design Memo – Woodland Islands Restoration Project Contingency and Trade-off Considerations for Project Implementation. USACE - Portland District, Portland, OR. 5 pp.

Van Regteren, M., Ten Boer, R., Meesters, E. H., & De Groot, A. V. 2017. Biogeomorphic impact of oligochaetes (Annelida) on sediment properties and Salicornia spp. seedling establishment. *Ecosphere*, *8*(7), e01872.

Weitkamp LA, BR Beckman, DM VanDoornik, A Munguia, ME Hunsicker, KC Jacobson, and ML Journey. 2020. "Juvenile Salmon in Mainstem Habitats of the Lower Columbia River and Estuary: The Indirect Benefits of Tidal Wetland Restoration." Chapter X in *Restoration Action Effectiveness Monitoring and Research in the Lower Columbia River and Estuary, 2016-2017*.

Weitkamp, L. A., Beckman, B. R., Van Doornik, D. M., Munguia, A., Hunsicker, M., and M. Journey. 2022. "Life in the Fast Lane: Feeding and Growth of Juvenile Steelhead and Chinook Salmon in Main-Stem Habitats of the Columbia River Estuary." *Transactions of the American Fisheries Society* 151(5), 587-610. https://doi.org/10.1002/tafs.10376

Welch, M., E. T. Mogren, and L. Beeney. 2016. A Literature Review of the Beneficial Use of Dredged Material and Sediment Management Plans and Strategies https://pdxscholar.library.pdx.edu/cgi/viewcontent.cgi?article=1034&context=publicservice_pub

Whigham, D. F. 2009. Primary Production in Tidal Freshwater Wetlands. In A. Barendregt, D. F. Whingham, & A. H. Baldwin (Eds.), *Tidal Freshwater Wetlands* (pp. 115-122). Backhuys Publishers.

Wilber, D., and D. Clarke. 2007. “Defining and assessing benthic recovery following dredging and dredged material disposal.” Proceedings XXVII World Dredging Congress Final technical report submitted by PNNL and NMFS to the U.S. Army Corps of Engineers, Portland District, Portland, OR.

Appendix A

The following table describes the taxonomies of the benthic invertebrates encountered in off-channel habitats during 2019−2022.

Table 7. Benthic Invertebrate Taxonomies

Phylum	Class	Order	Family	Unique Taxa*
Annelida	Clitellata	Rhynchobdellida	Glossiphoniidae	Glossiphoniidae
Annelida	Clitellata	NA	NA	Oligochaeta
Annelida	Polychaeta	Aciculata	Nereididae	Nereididae
Annelida	Polychaeta	Sabellida	Sabellidae	Sabellidae
Arthropoda	Arachnida	Trombidiformes	Arrenuridae	Arrenuridae
Arthropoda	Arachnida	Trombidiformes	Hydryphantidae	Hydryphantidae
Arthropoda	Arachnida	Trombidiformes	Hygrobatidae	Hygrobatidae
Arthropoda	Arachnida	Trombidiformes	Lebertiidae	Lebertia sp.
Arthropoda	Arachnida	Trombidiformes	Lebertiidae	Lebertiidae
Arthropoda	Arachnida	Trombidiformes	Limnesiidae	Limnesiidae
Arthropoda	Arachnida	Trombidiformes	Limnocharidae	Limnocharidae
Arthropoda	Arachnida	Trombidiformes	Mideopsidae	Mideopsidae
Arthropoda	Arachnida	Trombidiformes	Pionidae	Pionidae
Arthropoda	Arachnida	Trombidiformes	Sperchontidae	Sperchontidae
Arthropoda	Arachnida	Trombidiformes	Torrenticolidae	Torrenticolidae
Arthropoda	Arachnida	Trombidiformes	Unionicolidae	Unionicolidae
Arthropoda	Arachnida	NA	NA	Acari
Arthropoda	Arachnida	Sarcoptiformes	NA	Oribatei
Arthropoda	Insecta	Coleoptera	Elmidae	Elmidae
Arthropoda	Insecta	Coleoptera	Haliplidae	Haliplidae
Arthropoda	Insecta	Coleoptera	Hydraenidae	Hydraenidae
Arthropoda	Insecta	Coleoptera	NA	Coleoptera
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogonidae
Arthropoda	Insecta	Diptera	Ceratopogonidae	Ceratopogoninae
Arthropoda	Insecta	Diptera	Chironomidae	Chironomidae
Arthropoda	Insecta	Diptera	Dolichopodidae	Dolichopodidae
Arthropoda	Insecta	Diptera	Empididae	Empididae
Arthropoda	Insecta	Diptera	Ephydridae	Ephydridae
Arthropoda	Insecta	Diptera	Muscidae	Muscidae

Phylum	Class	Order	Family	Unique Taxa*
Arthropoda	Insecta	Diptera	Psychodidae	Psychodidae
Arthropoda	Insecta	Diptera	Simuliidae	Simuliidae
Arthropoda	Insecta	Diptera	Tipulidae	Tipulidae
Arthropoda	Insecta	Diptera	NA	Diptera
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetidae
Arthropoda	Insecta	Ephemeroptera	Baetidae	Baetis sp.
Arthropoda	Insecta	Ephemeroptera	Caenidae	Caenidae
Arthropoda	Insecta	Ephemeroptera	Ephemeridae	Ephemeridae
Arthropoda	Insecta	Ephemeroptera	Heptageniidae	Heptageniidae
Arthropoda	Insecta	Ephemeroptera	Leptohyphidae	Leptohyphidae
Arthropoda	Insecta	Hemiptera	Corixidae	Corixidae
Arthropoda	Insecta	Megaloptera	Sialidae	Sialidae
Arthropoda	Insecta	Odonata	Coenagrionidae	Coenagrionidae
Arthropoda	Insecta	Odonata	Gomphidae	Gomphidae
Arthropoda	Insecta	Plecoptera	Nemouridae	Nemouridae
Arthropoda	Insecta	Plecoptera	Perlidae	Perlidae
Arthropoda	Insecta	Plecoptera	Perlodidae	Perlodidae
Arthropoda	Insecta	Trichoptera	Hydropsychidae	Hydropsychidae
Arthropoda	Insecta	Trichoptera	Hydroptilidae	Hydroptilidae
Arthropoda	Insecta	Trichoptera	Leptoceridae	Leptoceridae
Arthropoda	Insecta	Trichoptera	Polycentropodidae	Polycentropodidae
Arthropoda	Insecta	Trichoptera	Rhyacophilidae	Rhyacophilidae
Arthropoda	Insecta	Trichoptera	NA	Trichoptera
Arthropoda	Malacostraca	Amphipoda	Anisogammaridae	Anisogammaridae
Arthropoda	Malacostraca	Amphipoda	Corophiidae	Corophiidae
Arthropoda	Malacostraca	Amphipoda	Crangonyctidae	Crangonyctidae
Arthropoda	Malacostraca	Amphipoda	Gammaridae	Gammaridae
Arthropoda	Malacostraca	Amphipoda	Hyalellidae	Hyalellidae
Arthropoda	Malacostraca	Amphipoda	NA	Amphipoda
Arthropoda	Malacostraca	Isopoda	Asellidae	Asellidae
Arthropoda	Malacostraca	Isopoda	Asellidae	Caecidotea sp.
Arthropoda	Malacostraca	Mysida	Mysidae	Mysidae
Arthropoda	Malacostraca	Tanaidacea	NA	Tanaidacea
Arthropoda	Ostracoda	NA	NA	Ostracoda
Cnidaria	Hydrozoa	Anthoathecatae	Hydridae	Hydra sp.

Phylum	Class	Order	Family	Unique Taxa*
Cnidaria	Hydrozoa	Anthoathecatae	Hydridae	Hydridae
Mollusca	Bivalvia	Unionoida	Unionidae	Unionidae
Mollusca	Bivalvia	Venerida	Cyrenidae	Corbicula
Mollusca	Bivalvia	Sphaeriida	Sphaeriidae	Sphaeriidae
Mollusca	Bivalvia	NA	NA	Bivalvia
Mollusca	Gastropoda	Heterostropha	Murchisonellidae	Murchisonellidae
Mollusca	Gastropoda	Hygrophila	Ancylidae	Ancylidae
Mollusca	Gastropoda	Hygrophila	Lymnaeidae	Lymnaeidae
Mollusca	Gastropoda	Hygrophila	Physidae	Physidae
Mollusca	Gastropoda	Hygrophila	Planorbidae	Planorbidae
Mollusca	Gastropoda	Littoridinomorpha	Hydrobiidae	Hydrobiidae
Mollusca	Gastropoda	Littoridinomorpha	Lithoglyphidae	Lithoglyphidae
Mollusca	Gastropoda	Littoridinomorpha	Semisulcospiridae	Semisulcospiridae
Mollusca	Gastropoda	Littoridinomorpha	Thiaridae	Thiaridae
Mollusca	Gastropoda	Littoridinomorpha	NA	Truncatelloidea
Mollusca	Gastropoda	NA	NA	Gastropoda
Nematoda	NA	NA	NA	Nematoda
Nemertea	Enopla	Hoplonemertea	Tetrastemmatidae	Prostoma sp.
Nemertea	Enopla	Hoplonemertea	Tetrastemmatidae	Tetrastemmatidae
Platyhelminthes	Turbellaria	Tricladida	Planariidae	Planariidae
Platyhelminthes	Turbellaria	Tricladida	Planariidae	Turbellaria
Sipuncula	NA	NA	NA	Sipuncula

*Unique Taxa is the lowest taxonomic resolution obtained for the given taxa; most are at the family level. Some (e.g., Acari) correspond to higher resolutions, and others (e.g., Lebertia sp.) correspond to lower resolutions.

Appendix B

Results of SIMPER (similarity percentages) of benthic communities. SIMPER is used to understand which taxa contribute to similarities within groups or which taxa contribute to differences between groups (Clarke and Gorley 2015). Average abundance values are based on the square root transformed counts of invertebrates.

Table 8. SIMPER results of average taxa similarity within sites across all years.

Site	Average Dissimilarity %	Taxa	Common Name(s)	Mean Abundance	Mean Similarity	Similarity/Standard deviation	Contribution %	Cumulative Contribution %
Bachelor	53.02	Chironomidae	Insect; non-biting midges	5.49	20.75	2.29	39.13	39.13
Bachelor	53.02	Corbiculidae	bivalve, clam	3.08	11.35	1.12	21.41	60.54
Bachelor	53.02	Oligochaeta	segmented worms	4.09	7.72	0.82	14.56	75.1
Martin	53	Oligochaeta	segmented worms	14.1	18.92	2.08	35.7	35.7
Martin	53	Chironomidae	Insect; non-biting midges	7.81	10.28	1.95	19.39	55.1
Martin	53	Nematoda	roundworms	7.34	8.77	1.71	16.54	71.64
Woodland	44.25	Oligochaeta	segmented worms	9.36	11.87	1.35	26.84	26.84
Woodland	44.25	Chironomidae	Insect; non-biting midges	5.67	9.3	1.62	21.02	47.86
Woodland	44.25	Nematoda	roundworms	5.92	7.12	1.2	16.1	63.96
Woodland	44.25	Corophiidae	crustacean, scuds	4.36	5.51	0.74	12.46	76.41

Table 9. SIMPER results of average taxa dissimilarity between site groups across all years

Site	Average Dissimilarity %	Taxa	Common Name(s)	Mean Abundance Bachelor Island	Mean Abundance Martin Island	Mean Dissimilarity	Dissimilarity/Standard deviation	Contribution %	Cumulative Contribution %
Bachelor and Martin	57.54	Oligochaeta	segmented worms	4.09	14.1	14.86	1.52	23.32	23.32
Bachelor and Martin	57.54	Nematoda	roundworms	1.69	7.34	8.24	1.42	12.93	36.25
Bachelor and Martin	57.54	Chironomidae	Insect; non-biting midges	5.49	7.81	6.08	1.13	9.55	45.79
Bachelor and Martin	57.54	Corophiidae	crustacean, scuds	2.38	3.16	4.29	0.96	6.74	52.53
Bachelor and Martin	57.54	Corbiculidae	bivalve, clam	3.08	1.49	3.24	1.09	5.09	57.62
Bachelor and Martin	57.54	Ostracoda	crustacean, seed shrimp	1.69	2.3	3.05	1.08	4.78	62.4

Bachelor and Martin	57.54	Sphaeriidae	bivalve, clam	0.12	1.97	2.6	1	4.08	66.47
Bachelor and Martin	57.54	Sabellidae	polychaete worms	0.01	2.1	2.26	0.46	3.55	70.03
Bachelor and Woodland	54.24	Oligochaeta	segmented worms	4.09	9.36	11.74	1.32	19.46	19.46
Bachelor and Woodland	54.24	Nematoda	roundworms	1.69	5.92	7.74	1.21	12.83	32.29
Bachelor and Woodland	54.24	Chironomidae	Insect; non-biting midges	5.49	5.67	6.53	1.01	10.82	43.11
Bachelor and Woodland	54.24	Corophiidae	crustacean, scuds	2.38	4.36	6.5	0.96	10.78	53.89
Bachelor and Woodland	54.24	Corbiculidae	bivalve, clam	3.08	2.4	4.38	0.91	7.26	61.15
Bachelor and Woodland	54.24	Ostracoda	crustacean, seed shrimp	1.69	2.46	3.89	1.1	6.45	67.6
Bachelor and Woodland	54.24	Sabellidae	polychaete worms	0.01	1.68	1.95	0.45	3.23	70.83

Martin and Woodland	50.1	Oligochaeta	segmented worms	14.1	9.36	9.76	1.2	17.81	17.81
Martin and Woodland	50.1	Nematoda	roundworms	7.34	5.92	6.04	1.23	11.03	28.83
Martin and Woodland	50.1	Chironomidae	Insect; non-biting midges	7.81	5.67	5.34	1.09	9.75	38.58
Martin and Woodland	50.1	Corophiidae	crustacean, scuds	3.16	4.36	4.77	0.99	8.71	47.29
Martin and Woodland	50.1	Sabellidae	polychaete worms	2.1	1.68	2.7	0.57	4.93	52.22
Martin and Woodland	50.1	Ostracoda	crustacean, seed shrimp	2.3	2.46	2.69	1.12	4.91	57.13
Martin and Woodland	50.1	Corbiculidae	bivalve, clam	1.49	2.4	2.13	0.94	3.89	61.02
Martin and Woodland	50.1	Sphaeriidae	bivalve, clam	1.97	1.07	2.08	1	3.8	64.82
Martin and Woodland	50.1	Asellidae	crustacean, isopods	1.27	0.49	1.35	0.63	2.47	67.29
Martin and Woodland	50.1	Hyalellidae	crustacean, amphipod	1.01	0.65	1.25	0.67	2.28	69.57
Martin and Woodland	50.1	Ceratopogonidae	Insect; biting midges	1.24	0.57	1.25	1.05	2.28	71.86

Appendix C

Akaike Information Criterion corrected (AICc) Model Selection: The table below displays a subset of the models fitted for benthic invertebrates, along with their corresponding AICc values. The model with the lowest AICc is considered the best-fitting model for the data.

Table 10. Benthic Invertebrates AICc Model Selection

Response	Fixed effects	Random effects	AICc
Total abundance	Before After (BA) Control Impact (CI) BAxCI	No random effect	276821.9
		Site, Year, Station	149713.5
		Site, Year, Station, StationxYear, Observation level ID	11412.52 (best fitted)
Chironomidae abundance	Before After (BA) Control Impact (CI) BAxCI	No random effect	65030.49
		Site, Year, Station	42169.16
		Site, Year, Station, StationxYear, Observation level ID	8314.198 (best fitted)
Corophiidae abundance	Before After (BA) Control Impact (CI) BAxCI	No random effect	54902.16
		Site, Year, Station	30544.19
		Site, Year, Station, StationxYear, Observation level ID	6128.053 (best fitted)

The table below displays a subset of the models fitted for environmental conditions, along with their corresponding AICc values. The model with the lowest AICc is considered the best-fitting model for the data.

Table 11. Environmental Conditions AICc Model Selection

Response	Fixed effects	Random effects	AICc
Carbon (%)	Before After (BA) Control Impact (CI) BAxCI	No random effect	-7797.676
		Site, Year, Station	-8324.452
		Site, Year, Station, StationxYear	-8341.838 (best fitted)
Ammonium (ppm)	Before After (BA) Control Impact (CI) BAxCI	No random effect	7976.25
		Site, Year, Station	7784.768 (best fitted)
Phosphorous (ppm)	Before After (BA) Control Impact (CI) BAxCI	No random effect	11855.45
		Site, Year, Station	11455.94
		Site, Year	11453.94 (best fitted)
Sand (%)	Before After (BA) Control Impact (CI) BAxCI	No random effect	-1243.664
		Site, Year, Station	-2081.774
		Site, Year, Station, StationxYear	-2210.387 (best fitted)

Pacific Northwest National Laboratory

902 Battelle Boulevard
P.O. Box 999
Richland, WA 99354
1-888-375-PNNL (7665)

www.pnnl.gov