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Assessing Stream Channel Restoration: The Phased Recovery Framework

Jacob M. Dyste
University of Montana, Missoula

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ASSESSING STREAM CHANNEL RESTORATION:

THE PHASED RECOVERY FRAMEWORK

By

JACOB MATTHEW DYSTE

Bachelor of Science, University of Oregon, Eugene, Oregon, 2009

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Approved by:

Scott Whittenburg, Dean of The Graduate School
Graduate School

H. Maurice Valett, Chair
Division of Biological Sciences

Diana Six
College of Forestry and Conservation

Winsor Lowe
Division of Biological Sciences

Dyste, Jacob, M.S., Summer 2017 Systems Ecology

Phased Recovery Framework for Channel Reconfiguration Stream Restoration

Chairperson: H. Maurice Valett

Channel reconfiguration projects command a large portion of stream restoration resources, while long-term monitoring and research is severely underfunded and rarely implemented. This has led to a limited knowledge base about ecological response and efficacy. Although channel reconfiguration projects are being implemented to restore biological function to lotic systems, the document responses are highly variable and little evidence has shown these projects are reaching their target goals. I predicted the inconsistent response to these projects is the result of disturbance-induced successional processes and catchment-scale water quality impairment. To address how these endogenous and exogenous factors influence stream response to channel reconfiguration, I developed the phased recovery framework and tested it by assessing nine channel reconfiguration sites in western Montana. Each site was composed of a restored reach ranging in age from 1 to 18 years and reference reach representing a minimally disturbed target condition. Five sites were located in unimpaired forested watersheds, while four sites were located in a human-developed watershed experiencing nutrient enrichment, increased fine sediment loads, and elevated stream temperatures. At each site, I assessed macroinvertebrates and associated habitat metrics (physical habitat, canopy cover, stream temperature, dissolved oxygen, and organic matter) and evaluated their response to endogenous and exogenous controls. While physical habitat appears to reach restoration targets immediately following channel reconfiguration, other metrics do not. Channel reconfiguration projects appear to undergo a reorganization phase where organic matter standings stocks and macroinvertebrate densities are significantly reduced for one to two years. Following the reorganization phase, watershed condition drives the recovery trajectory of restored reaches. In unimpaired watersheds, endogenous control, particularly the successional sere of the riparian zone drives ecosystem recovery. Over time, organic matter standing stocks, based on benthic chlorophyll *a*, become more similar to reference condition ($r^2 = 0.46$, $p = 0.05$). This corresponds with shredder abundance increasing as projects get older ($r^2 = 0.78$, $p = 0.05$). Additionally, macroinvertebrate communities become more similar with project age based on Bray-Curtis Dissimilarity ($r^2 = 0.59$, $p = 0.13$). At impaired sites, oldest restoration reaches were closest to point-source nutrient enrichment and this exogenous control dictated ecosystem recovery. Specifically, benthic chlorophyll *a* ($r^2 = 0.98$, $p = 0.01$) and collector-gatherer abundance ($r^2 = 0.84$, $p = 0.09$) increased as sites got older, the opposite trend of unimpaired sites. These results suggest that free of watershed-scale impairment, restoration reaches likely take upwards of two decades to recover to reference condition. If impairment is present, it can exert a strong endogenous control on recovery that overwhelms the influence of channel reconfiguration restoration.

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Introduction

Extensive degradation of lotic ecosystems has created a sense of urgency for stream restoration that overshadows efforts to understand the long-term ecological implications of these actions through monitoring and research (Roni et al. 2013). While recognized as important, post-restoration monitoring is underfunded, irregularly implemented, and carried out over limited timescales (Bash & Ryan 2002). Monitoring occurs in less than 20% of stream restoration projects and when it does occur it typically lasts for five years or less (Roni et al. 2002, Bernhardt et al. 2005). Without monitoring, it is impossible to determine how well ecological restoration projects are meeting stated goals (Palmer et al. 2005). Monitoring is also critical because it allows practitioners to refine techniques, adaptively manage projects, and leverage future funding sources (Kondolf & Micheli 1995).

In the realm of stream restoration, construction of an entirely new channel, a technique called ‘channel reconfiguration,’ now accounts for approximately 32% of all projects implemented, and an even larger percentage of financial resources (Palmer et al. 2014). Channel reconfiguration has proliferated in the past 20 years with the rise of the natural channel design approach to restoration (Rosgen 1996). While controversial within the ecological restoration community (Lave 2009), natural channel design approaches provide practitioners with a systematic process for restoring the geomorphic form of degraded stream channels (Kondolf 2006). This method of restoration seeks to create the physical template for proper biological function through the construction of geomorphically stable channel forms and complex habitat (Palmer et al. 2010). In part because of its geomorphic basis, restoration practitioners commonly use assessment of physical structure alone to evaluate success of channel reconfiguration projects (Kondolf and Micheli 1995). With this approach, a project may be considered successful based strictly on proper implementation of the engineering design, a protocol that does little to enhance our limited understanding of ecological responses to restoration of this type (Palmer et al. 1997, Sudduth et al. 2011).

Unlike most other forms of stream restoration, channel reconfiguration acts as a severe disturbance to benthic and riparian zones of lotic systems, further emphasizing the need for monitoring and assessment (Tullos et al. 2009). Channel construction and floodplain grading creates stream reaches initially devoid of riparian plants, stream autotrophs, and macroinvertebrates, which must then be reestablished through natural recolonization or human intervention. While macroinvertebrates and stream autotrophs can recover to pre-restoration levels rapidly (Pederson et al. 2007), disturbance to the riparian zone can be long-lasting and alters critical aquatic-terrestrial linkages (Likens and Bormann 1974, Lake 2000). Following disturbance, canopy cover can take at least fifteen years to reach pre-disturbance levels, in the interm promoting elevated insolation and reduced allochthonous inputs for more than a decade (Lennox et al. 2011, Wallace et al. 1997). Reduction in canopy cover also affects trophic structure, primary production, and thermal regime within the benthic zone (Johnson 2004, Kiffney et al. 2004). Consequently, it is likely that restored stream reaches undergo major structural and functional changes over successional timescales comparable to those observed following other forms of disturbance (e.g., Fisher et al. 1982, Molles 1990, Valett et al. 2002). Therefore, temporal changes in stream-riparian corridor condition resulting from channel reconfiguration would be an endogenous control on ecosystem recovery because within system processes drive the trajectory.

Odum (1969) called this temporal interaction of complex successional processes ‘ecosystem development’ and identified phases representing how the structure and function of ecosystems change as they move towards a steady-state climax condition. These phases are characterized, in part, by differences in food web complexity, species diversity, and organic matter (OM) standing stocks. In the realm of stream restoration, when researchers seek to address recovery following channel reconfiguration they often employ similar timescales to post-project monitoring (i.e., 5 - 7 years), which is likely not enough time to allow restoration-induced ecosystem development processes to approach or reach steady-state (Lake et al. 2007). Instead, efforts to address response, particularly of macroinvertebrates, likely capture early phases of recovery and fail to address later phases influenced by the successional seres of the riparian zone.

Reduction in canopy cover associated with channel reconfiguration increases gross primary production (Lamberti & Steinman 1997), which can significantly alter macroinvertebrate abundance and community composition (Behmer & Hawkins 1986). These changes in canopy cover can continue to influence the macroinvertebrate community for nearly two decades (Stone and Wallace 1998). This potentially explains why channel reconfiguration is shown to increase macroinvertebrate density (Moerke et al. 2004) and select for disturbance-related functional traits such as multivoltinism and collector-gathering feeding modes (Rios-Touma et al. 2014) in recently completed (<5 years) projects.

While using an ecosystem development-based perspective to understand restoration recovery is logical, exogenous influences (i.e., force generated from outside the stream-riparian corridor) often complicate application of classical successional theory to lotic systems (Fisher 1990). Increasing agricultural and urban development within a watersheds is a documented exogenous influences and is associated with decreased water quality (Allan et al. 1997), increased sedimentation (Harding et al. 1998), and erratic flow regimes (Allan 2004). For macroinvertebrate communities, human induced impairment is associated with increased abundance in opportunistic taxa, such as Chironomidae, and decreases in Ephemeroptera and Plecoptera families (Jones & Clark 1987).

Restoration occurs across a broad range of environmental conditions and strong evidence exists showing that human influences can exert stronger controls on ecological recovery than reach-scale restoration, such as channel reconfiguration (Bernhardt & Palmer 2011). Specifically, a long-term study of macroinvertebrate response to restoration in a developed European watershed showed that macroinvertebrate communities did not change based on recovery time, but more so in response to watershed-scale urban development (Leps et al. 2016). This suggests exogenous control in impaired watersheds (i.e., characterized by anthropogenically degraded water quality, Walsh et al. 2005) can act as a press disturbance on ecosystem condition in restored channels that is likely to exist until managers address watershed-level degradation (Lake 2007).

Macroinvertebrates are often monitored to evaluate ecosystem response to restoration because they rapidly reflect changes in ecological condition (Barbour et al. 1999), but their sensitivity response to channel reconfiguration appears to be highly variable (Miller et al. 2009). Macroinvertebrate habitat controls, such as benthic stability, food resources, and stream temperature act as local filters that determine community composition (Poff et al. 2006). To address the role of these controls in community structure, past efforts applied functional-trait analysis to understand how communities shift in response to environmental change (Mouillot et al. 2006) with promise for application to restoration assessment. However, a potential issue with using this approach to generalize responses to restoration across systems is that local influences vary greatly from stream to stream, complicating interpretation of environmental change across watersheds (Menezes et al. 2010). Studies that address macroinvertebrate response to restoration across stream systems generally fail to show substantial functional changes, suggesting that site-specific habitat controls might complicate efforts to generalize response across systems (Ernst et al. 2012).

Embracing the perspective that ecosystem recovery following channel reconfiguration is a function of endogenous and exogenous controls provides restoration practitioners with a framework for restoration monitoring and assessment. Here, I introduce the ‘Phased Recovery Framework (PRF)’ as an approach to understanding how channel reconfiguration alters ecosystem structure over time, and how interaction with watershed condition organizes temporal trajectory. Based on succession, disturbance, and restoration literature, I predicted that ecosystems would recover in three distinct phases: 1) Reorganization (0 - 2 yeas) - In-stream habitat is heavily altered and characterized by low OM standing stocks; macroinvertebrates must recolonize from outside the restoration project area and patterns in community composition merely reflect rapidly colonizing species, 2) Developmental (2 - 15 years) - macroinvertebrate richness and density reach or exceed reference levels; community composition and OM standing stocks reflect continued disturbance as the result of increased insolation, 3) Mature (15+ years) - macroinvertebrate communities and OM standing stocks are indistinguishable from the reference condition indicating the system is approaching steady-state (Fig. 1).

To address the role of endogenous and exogenous controls, I assessed channel reconfiguration projects in western Montana (USA) streams chosen to represent a range of implementation dates and environmental impairment. At paired restored and reference reaches, I sampled macroinvertebrates and metrics representing features known to act as community filters including physical habitat, benthic OM standing stocks, canopy cover, and physicochemical variables. To evaluate the endogenous and exogenous recovery trajectories predicted by the PRF, I asked two primary research questions: 1) How do relevant community filters and associate macroinvertebrate communities change over time? and 2) How do these recovery trajectories differ in the larger context of watershed condition? These inquires prompted a two-step data analysis procedure where I first compared condition of community filters and macroinvertebrate communities between restored and reference reaches, how changes in these metrics occurred over time, and whether temporal change occurred in phases. I then accounted for watershed condition, separated sites based on the presence or absence of evident factors promoting impairment at the catchment scale, and repeated a similar analysis process. The results from these analyses were used to discuss the role of time in structuring macroinvertebrate response to channel reconfiguration and how watershed condition in influences characteristic temporal response.

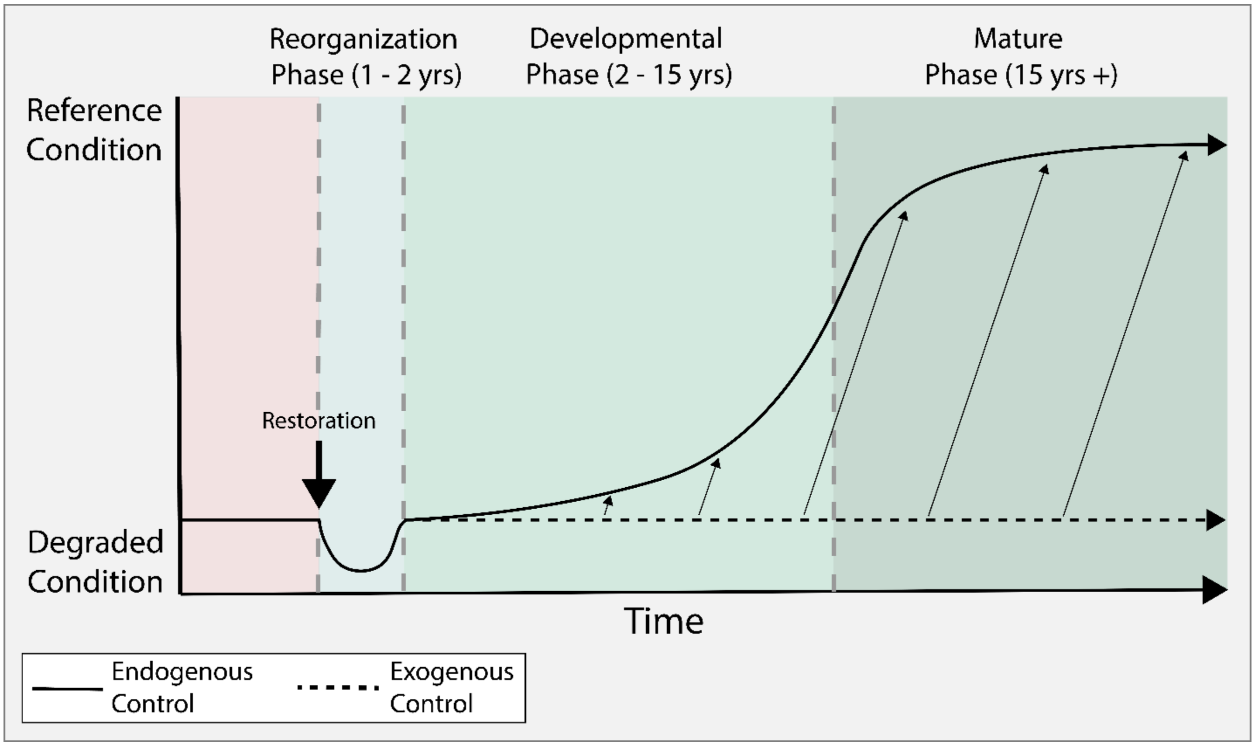

Figure 1. Conceptual model of the Phased Recovery Framework for stream ecosystem recovery following channel reconfiguration restoration projects. Under endogenous control, ecosystems move through a reorganization, developmental, and mature phase over time (solid line) as they proceed towards reference condition. In the presence of exogenous human impairment, sites exist in a press disturbance state that will persist (dashed line) until further human intervention removes the influence and promotes convergence (arrows) with endogenous trajectory.

Methods

Study Sites

I initially identified all channel reconfiguration projects within three watersheds in western Montana - the Upper Clark Fork (UCF), Middle Clark Fork (MCF), and Blackfoot (BF). Restoration projects were considered potential sites only if they were implemented on 1st – 3rd-order streams because of the strong aquatic-terrestrial linkage in these systems (Vannote et al. 1980). I separated sites implemented on snowmelt-driven streams from groundwater-induced spring creeks, which have comparatively stable flows and temperature regimes (Pierce et al. 2014). In addition, I found that cattle grazing continues to influence riparian growth along many of the spring creeks identified. Because of these factors, I restricted sampling sites to snowmelt-driven systems. This selection process identified nine suitable sites located on five streams across the three watersheds (Fig. 2).

The character of initial disturbance that prompted restoration, project implementation date, and predicted recovery phase differed among sites (Table 1). Past placer mining disturbed three sites (S1, S3, S5), creating channelized streams with reduced habitat complexity, increased sediment loads, and restricted floodplain interaction (Hilmes and Wohl 1995). Logging (S7) and cattle grazing (S9) at two other sites created similar downcut and channelized conditions. Copper mining initially disturbed sites in the UCF watershed (S2, S4, S6, S8), generating floodplain soils contaminated with copper, zinc, lead, cadmium, mercury, and arsenic (Moore et al. 1989), which were removed prior to channel reconfiguration (MNRDP 2005). Restoration actions implemented on all sites were consistent with the natural channel design principles and typically included grading to produce an active floodplain, cutting an entirely new channel, placement of

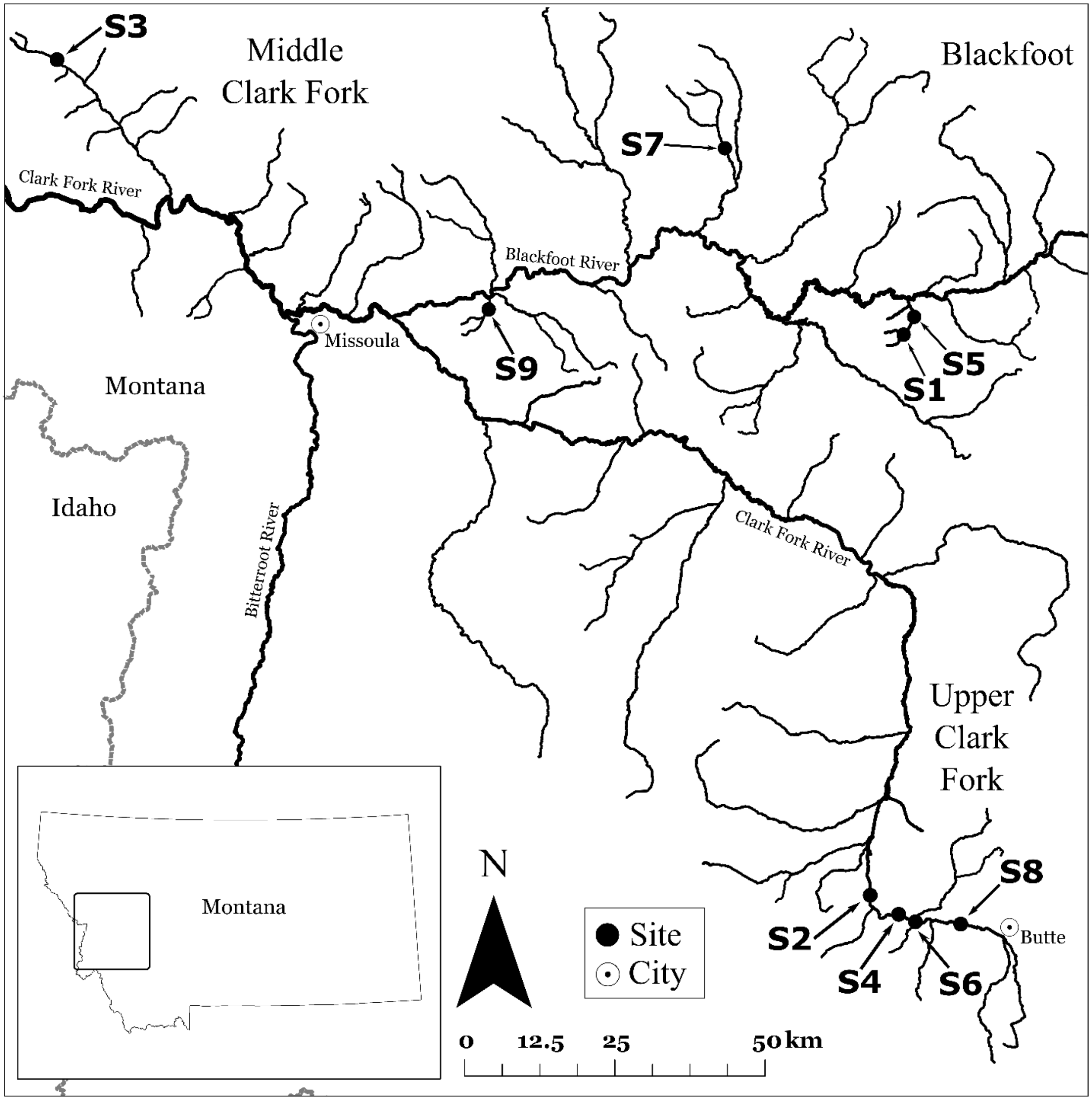

Figure 2. Map of the nine sites that were assessed for this study. Sites were located on five low-order streams in three watersheds (HUC8) - Upper Clark Fork (S2, S4, S5, S8), Blackfoot (S1, S5, S7, S9), and Middle Clark Fork (S3).

Table 1. Site characteristics including identification number, stream name, watershed, year restored, recovery phase, and initial disturbance of the nine sites assessed. Recovery phase is based on the number of years between project implementation and assessment in summer 2016.

Site	*Stream Name*	*Watershed (HUC 8)*	*Year Restored*	*Predicted Recovery Phase*	*Initial Disturbance*
1	Sauerkraut	Blackfoot	2015	Reogranization	Placer Mining
2	Silver Bow	Upper Clark Fork	2015	Reogranization	Copper Mining
3	Ninemile	Middle Clark Fork	2014	Reorganization	Placer Mining
4	Silver Bow	Upper Clark Fork	2011	Developmental	Copper Mining
5	Sauerkraut	Blackfoot	2009	Developmental	Placer Mining
6	Silver Bow	Upper Clark Fork	2007	Developmental	Copper Mining
7	Dunham	Blackfoot	2001	Mature	Logging
8	Silver Bow	Upper Clark Fork	2001	Mature	Copper Mining
9	Bear	Blackfoot	1998	Mature	Grazing

instream habitat structures, and planting of native riparian species along the newly formed channel. The nine sites chosen for this study ranged in time since implementation (age) from 1 to 18 years with three sites occupying each of the predicted phases associated with the recovery framework.

Study sites varied in elevation, catchment size, land use, and geology (Table 2). Elevation ranged among sites from 1084 to 1644 m and watershed area spanned more than an order of magnitude from 21 to 936 km^2. Sites in the MCF and BF watersheds (S1, S3, S5, S7, S9) drain predominantly dry-mesic montane landscapes, with mixed conifer cover and no developed land use upstream of the sites. Sites in the UCF watershed (S2, S4, S6, S8) have mixed land use dominated by lower montane and valley grasslands with significantly more upstream development (Montana Natural Heritage Program Land Cover Database 2017). In the context of the need for restoration in response to initial disturbance (Table 1), MCF and BF sites are considered relatively unimpaired by continued human influence. In contrast, external factors beyond those that motivated restoration actions can be seen as additional sources of impairment in the UCF watershed. These sites are impaired by excess nutrients from the Butte wastewater treatment plant, increased sediment transport, and reduced instream flows (Montana DEQ 2014).

Riparian plant communities in the region are a mosaic of tree and shrub species dominated by black cottonwood (*Populus balsamifera)*, Douglas fir (*Pseudotsuga menziesii)*, Rocky Mountain maple (*Acer glabrum)*, thinleaf alder (*Alnus incana),* and various willow species (*Salix*). Conifer abundance decreases and cottonwood and willow become more dominant as sites increase in elevation and proximity to the continental divide (Vance et al. 2010). Geology of the sites varies greatly within watersheds and along stream systems based on stratigraphic unit, lithology, and geologic age (Lewis 1998).

Table 2. Watershed characteristics of each site including elevation, watershed area, land cover classification (Montana Natural Heritage Program Land Cover Database 2017), and geology (Vuke et al. 2007). Land cover classification is similar among Middle Clark Fork and Blackfoot sites, while Lower Clark Fork sites have more developed land cover. Geology varies greatly between each site.

Site	*Elevation (m)*	*Watershed Area (km^2)*	*Land Cover Classification*	*Geology (Unit – Lithology – Age)*
1	1544	34	Dry-mesic montane mixed conifer	Newland Limestone – Carbonate - Mesoproterozoic
2	1556	936	Lower montane, foothill, valley grassland/ Developed human use	Tertiary Volcanic - Aphanitic -Tertiary/ Alluvium - Quaternary
3	1207	480	Dry-mesic montane mixed conifer	Glacial Drift - Quaternary/ Prichard Formation - Meta-argillite - Mesopro.
4	1607	936	Lower montane, foothill, valley grassland/ Developed human use	Undifferentiated Clastic - Medium-grained - Tertiary/ Alluvium - Quaternary
5	1451	34	Dry-mesic montane mixed conifer	Newland Limestone - Carbonate - Mesoproterozoic
6	1616	936	Lower montane, foothill, valley grassland/ Developed human use	Undifferentiated Clastic - Medium-grained - Tertiary/ Alluvium - Quaternary
7	1310	86	Dry-mesic montane mixed conifer	Missoula Group - Quartzite - Mesoproterozoic/ Alluvium - Quaternary
8	1644	936	Lower montane, foothill, valley grassland/ Developed human use	Boulder batholith - Quartz Monzonite - Cretaceous/ Alluvium - Quaternary
9	1084	21	Dry-mesic montane mixed conifer	Missoula Group - Quartzite - Mesoproterozoic/ Alluvium - Quaternary

Lack of pre-restoration data prompted the use of a reference-treatment experimental design, where each site consists of a paired reference and restored (treatment) reach (Roni et al. 2005). The reference reach represents a minimally disturbed condition, which serves as a target goal for the paired restored reach. At each of the five sites in the MCF and BF watersheds, I sampled independent reference reaches located in close proximity to restored reaches on the same stream. Restoration project managers identified these reference reaches before implementation to assist with project planning and monitoring. Reference systems were selected when they reflected little or no human-induced disturbance and were located close enough to restored reaches to experience similar watershed conditions. Mining history of the UCF represents a watershed-scale disturbance that precluded the possibility of using within-stream reference reaches. Reference data for these sites was generated by employing nearby minimally disturbed streams and using data acquired directly from the Montana Department of Environmental Quality's Stream Reference Project, which samples undisturbed streams across the state (Appendix B).

Site Assessment and Sampling

At each site, I delineated a representative reach with length equal to twenty times the mean of five bankfull width measurements (Bouwes et al. 2011). Physical habitat, canopy cover, OM stocks, and physicochemical characteristics were assessed at both restored and reference reaches over a two-week period in summer 2016. I sampled macroinvertebrates at each site during a one-week period in early autumn 2016 before leaf fall. Reference and restored reaches at each site were assessed on the same day to minimize influences of temporal variation.

Physical Habitat

In each reach, I classified and measured habitat types as either run, riffle, or pool following Bission et al. (2011) and calculated relative abundance of each habitat type along the length of the reach. I characterized streambed sediment composition using granulometry to determine size class distribution (Bevenger and King 1995). Sinuosity, calculated as channel length divided by valley floor distance in meters (Fitzpatrick et al. 1998), and gradient (i.e., slope) were measured using geographical information system software (ArcGIS 10.5, ESRI, Redlands, CA). In three riffles within each reach, I randomly toe-point sampled streambed sediment in a zigzag pattern and assigned each particle to a size class using a gravelometer (Rickly Hydrological, SAH-97, Columbus, OH) until 200 data points were collected. Median size class (D50) and the relative abundance of fine sediment (< 2mm) were determined from these data.

Canopy Cover

To quantify canopy cover, a convex densiometer (Ben Meadows, Model A, Janesville, WI) with 17 line intersections, was held 0.3 m over the water surface and the number of line intersections covered by riparian vegetation was counted (Kaufmann and Robinson 1998). The number of cover line intersections was recorded while facing upstream, downstream, left, and right, at 11 equally spaced transects along each reach. Percent canopy cover at each transect was calculated as the number of covered intersections divided by the total possible number of intersections (n = 68). Total percent canopy cover per reach was then determined as the mean of the 11 transect values.

Physicochemical Metrics

Upon arriving at a site, I placed PME MiniDOT loggers (PME, Inc. Vista, CA) in riffles downstream of each reach to measure stream temperature (°C) and dissolved oxygen (DO) concentration (mg/l). Loggers were collected after 24 hours and these data were used to calculate diel mean, minimum, and maximum values of both physicochemical metrics.

Organic Matter

OM standing stocks and algal standing crops were determined to evaluate OM (as ash-free dry mass, AFDM) and photosynthetic pigment as chlorophyll *a (*chl *a).* Autochthonous OM included filamentous algae, macrophyte, or bryophyte. At each sites, I isolated an area of streambed using an open-ended cylinder (area = 0.22 m^2) and collected all coarse organic matter (> 1 mm) from it. I repeated this procedure three times in a riffle and stored each sample on ice in the field (n = 3 per reach). In the laboratory, each OM sample was thawed and total wet weight recorded. From the thawed sample a subsample was removed for chl *a* analysis. The subsample was placed in 90% buffered acetone solution for 24 hours, after which the extractant was centrifuged for 10 minutes. Immediately after centrifuging, I measured chl *a* using spectrophotometry (at 664, 665, and 750 nm) on a Jasco V-550 spectrophotometer following McIntire et al. (1996), and reported as chl *a* standing crop (mg/m^2). The remaining sample was weighed again, dried for 48 hours, and reweighed. A dry subsample was then taken, weighed, combusted, and reweighed. AFDM and sample area were used to calculate OM standing stock (g/m^2).

Macroinvertebrates

Macroinvertebrates were collected from three riffles in each reach using a Surber sampler (0.093 m^2, 500 µm net). At three points across each riffle, I placed the base of the surber sampler firmly against the streambed, disturbed the substrate for 10 seconds and cleaned all large cobbles individually to create a composite sample of all collected benthic material. This generated three representative samples per reach. Samples were placed in 95% ethanol in the field and transferred to 70% ethanol in the laboratory for storage. I spread each sample evenly across a tray (Caton 1991), randomly isolated an area representing a known percent of the tray using a metal square, and removed all macroinvertebrates from the subsample. I continued randomly isolating subsamples and removing macroinvertebrates until 300 organisms were collected for identification. The 300 organisms collected were identified to family level using the taxonomic guide of Merritt and Cummins (1996). After 300 organisms were collected for identification, I finished counting macroinvertebrates from the remaining subsample to estimate density.

These data were used to address macroinvertebrate abundance, diversity, functional response, and community similarity. From an initial pool of 24 community metrics, I selected 10 frequently observed to be different between restored and reference reaches (two-tailed *t*-tests for each site). Calculated diversity metrics were taxa richness, Shannon-Weiner index, and percent Ephemeroptera, Plecoptera, and Trichoptera (EPT). Functional traits were related to either functional feeding groups (FFG) or habit, and were applied based on Merritt and Cummins (1996) and Poff et al. (2006). Functional feeding groups are based on morphological adaptations for food acquisition and habits are based on mechanisms used to move around the benthic zone (Voshell 2002). Selected feeding traits were percent shredder, collector-gatherer, and scraper. Selected habit traits were percent sprawler, clinger, and swimmer.

I addressed differences in community composition using Bray-Curtis dissimilarity at different levels of resolution (Bray & Curtis 1957). Bray-Curtis dissimilarity was used to address compositional similarity between reaches based on the abundance of distinct taxa at each (Beals 1984). To address community similarity within sites, I calculated Bray-Curtis dissimilarity between restored and reference reaches based on the mean abundance of each taxon from the three samples. Because variation in macroinvertebrate community composition is natural within stream systems (Lammert & Allan 1999), I characterized expected dissimilarity by calculating Bray-Curtis values for within-reach communities at all reaches based on the three collected samples. I then used the standard deviation of the mean (n = 45) to serve as the range of expected variability within any given system.

Data Analysis

I used a two-step procedure to address endogenous recovery following the PRF and exogenous influences on trajectories using the previously described habitat and macroinvertebrate metrics as response variables (Table 3). All statistical analysis was conducted with SigmaPlot v. 13 (Systat Software, San Jose, CA). In these analyses, I used a combination of absolute measures and response ratios. I used response ratios to normalize habitat and macroinvertebrate responses to their respective reference systems. These ratios were determined following Benayas et al. (2009) as ln(restored/reference) where 'restored' and 'reference' are the mean of absolute measures for a given response variable. For values of zero, I calculated response ratios as ln(restored+1/reference+1). Response ratios were evaluated using two-tailed, one-sample t-tests to determine if response ratios were significantly different from zero (i.e., restored condition = reference condition, $p \leq 0.05$). Additionally, two-tailed, t-tests were used to evaluate differences between paired groups of data (i.e., restored vs reference, impaired vs unimpaired, $p \leq 0.05$).

In order to test the influence of recovery phase on habitat and macroinvertebrate response following restoration, I used either one-way analysis of variance (ANOVA) or Kruskal-Wallis ANOVA with phase as the main factor (three levels: reorganization, developmental, mature). Following a significant assessment, I used Student-Newman-Keuls post-hoc tests for multiple comparisons. I then used simple linear regression to address recovery as a continuous process. I used age as the independent variable and macroinvertebrate or habitat data as response variables. Analyses were conducted using both absolute data and response ratios. To address exogenous influences, I first designated sites as impaired (S2, S4, S6, S8) or unimpaired (S1, S3,

Table 3. Habitat and macroinvertebrate metrics that were used as response variables in statistical analysis.

Metrics					
Habitat	Physical	Canopy Cover	BOM	Physicochemical	
	Sinuosity	% Canopy Cover	AFDM	Temperature	
	% Pool		Chlorophyll *a*	DO	
	D50				
	% Fines				
Macroinvertebrates	Abundance	Diversity	FFG	Habit	Community
	Density	Taxa Richness	% Shredder	% Sprawler	Bray-Curtis
		Shannon-Weiner	% Collector-Gatherer	% Clinger	
		% EPT	% Scraper	% Swimmer	

S5, S7, S9) based on the EPA's 303(d) listing of UCF sites for elevated nutrient loads (Montana DEQ 2016) . I then used linear regression to address how project age influenced habitat and macroinvertebrate recovery at impaired and unimpaired sites.

Results

Recovery trends across all sites

Measures of physical habitat structure were similar among phases (ANOVA, $p > 0.05$), indicating consistent channel form (i.e., sinuosity, Fig. 3a and % pool; Fig. 3b) as well as streambed composition at restored and reference reaches (i.e., D50, Fig. 3c and % fines; Fig 3d). Likewise, there was no relationship between age and physical habitat structure (linear regression, $r^2 < 0.2$, $p > 0.05$). These trends were consistent whether data were analyzed as absolute or response measures. Across all sites, slight increases in percent pool and decreases in sinuosity, D50, and percent fines in restored reaches were not significant compared to reference reaches (one-sample t-test, $p > 0.05$; Fig. 4).

Stream temperature across all reaches ranged from a low of 6.5°C (S7 restored) to a high of 23.6°C (S4 restored; Fig. 4a). Lowest mean (±1 SE) temperature recorded was 7.9 ± 0.06°C (S7 restored) and highest average was 18.3 ± 0.15 (S4 restored; Fig. 5a), which were recorded at the same sites where maximum and minimum values were observed. Across all sites, average temperature in restored reaches (14.04 ± 1.33°C) was significantly greater than in reference reaches (10.24 ± .37°C, Mann-Whitney Rank Sum Test, $p = .04$). DO concentrations ranged from 4.57 mg/l (S4 restored) to 13.94 mg/l (S6 restored; Fig. 5b). Lowest mean DO concertation was 7.36 ± 0.16 mg/l (S4 restored) and highest was 9.29 ± 0.02 mg/l (S7 reference; Fig. 4b). Average DO concentrations were lower in restored reaches (8.3 ± 0.25 mg/l) than reference reaches (9.09 ± 0.07 mg/l, Mann-Whitney Rank Sum Test, $p = 0.03$). Across all sites, minimum,

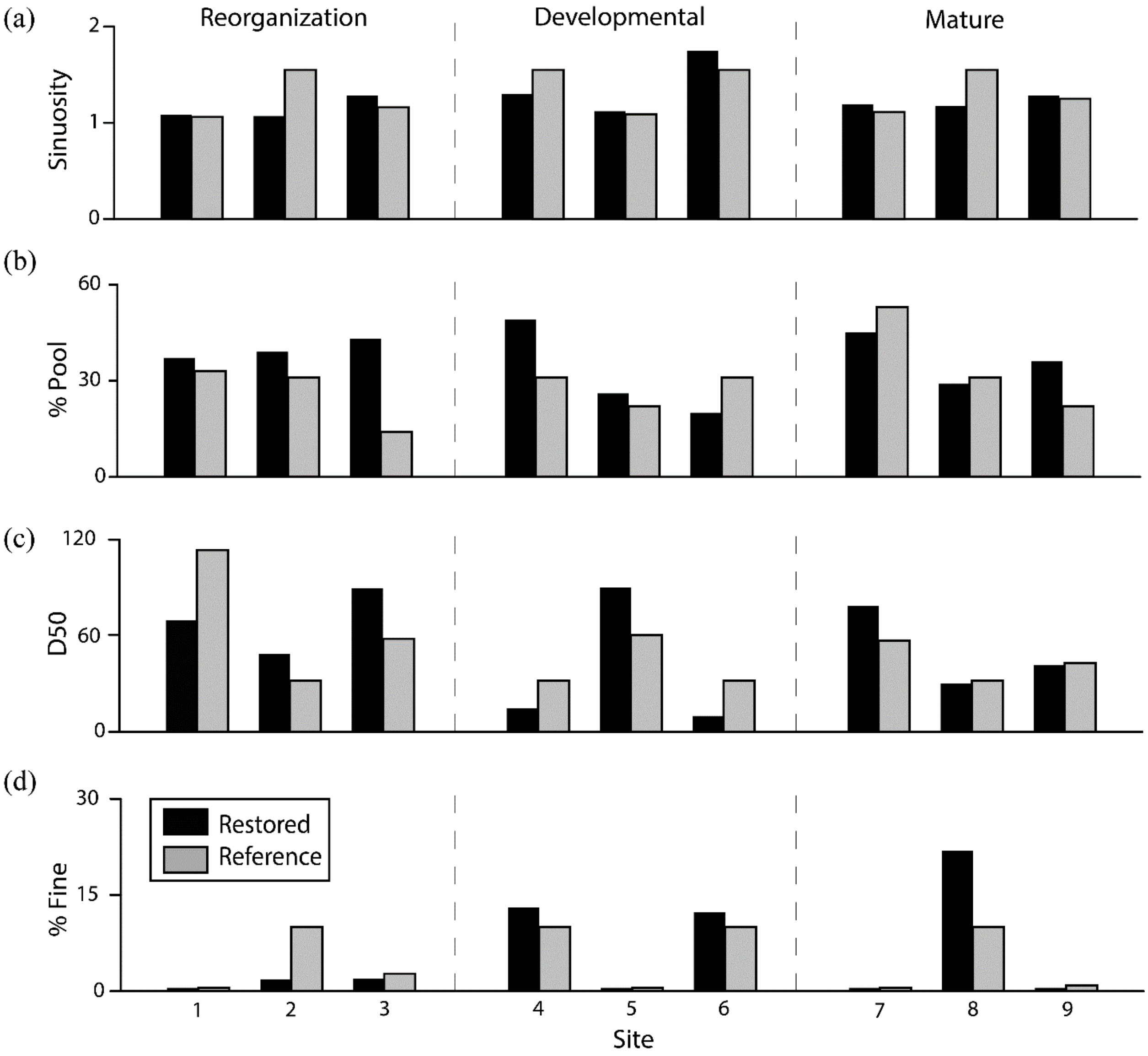

Figure 3. Absolute data for physical habitat metrics - sinuosity (a), % pool (b), D50 (c), and % fines (d) at restored (black bars) and reference (grey bars) reaches. Across sites differences between restored and reference reaches were not significant ($p < 0.05$). Vertical dashed lines group data by predicted recovery phase.

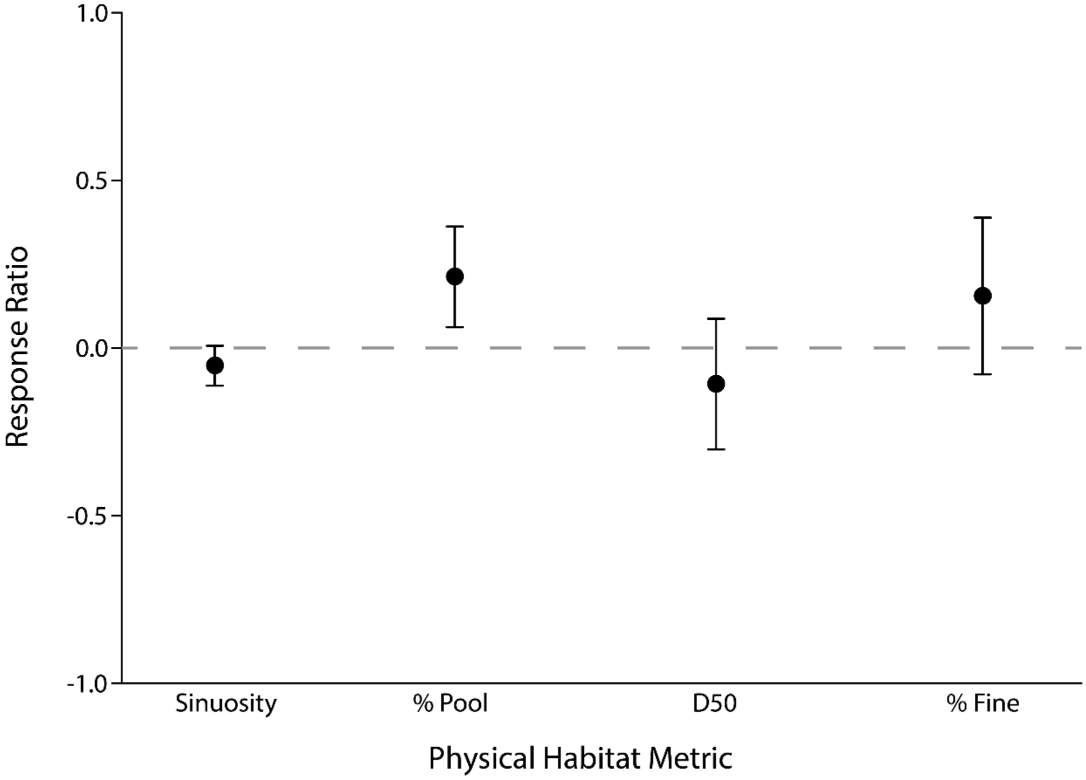

Figure 4. Mean (± 1 SE) response ratio of each physical habitat metric at all sites. A response ratio of zero indicates that restored reaches are identical to reference reaches. The direction of the difference in restored and reference reaches indicates either an increase or decrease in the value of each metric. Based on one-sample t-tests, none of the values were significantly different from zero ($p > 0.05$).

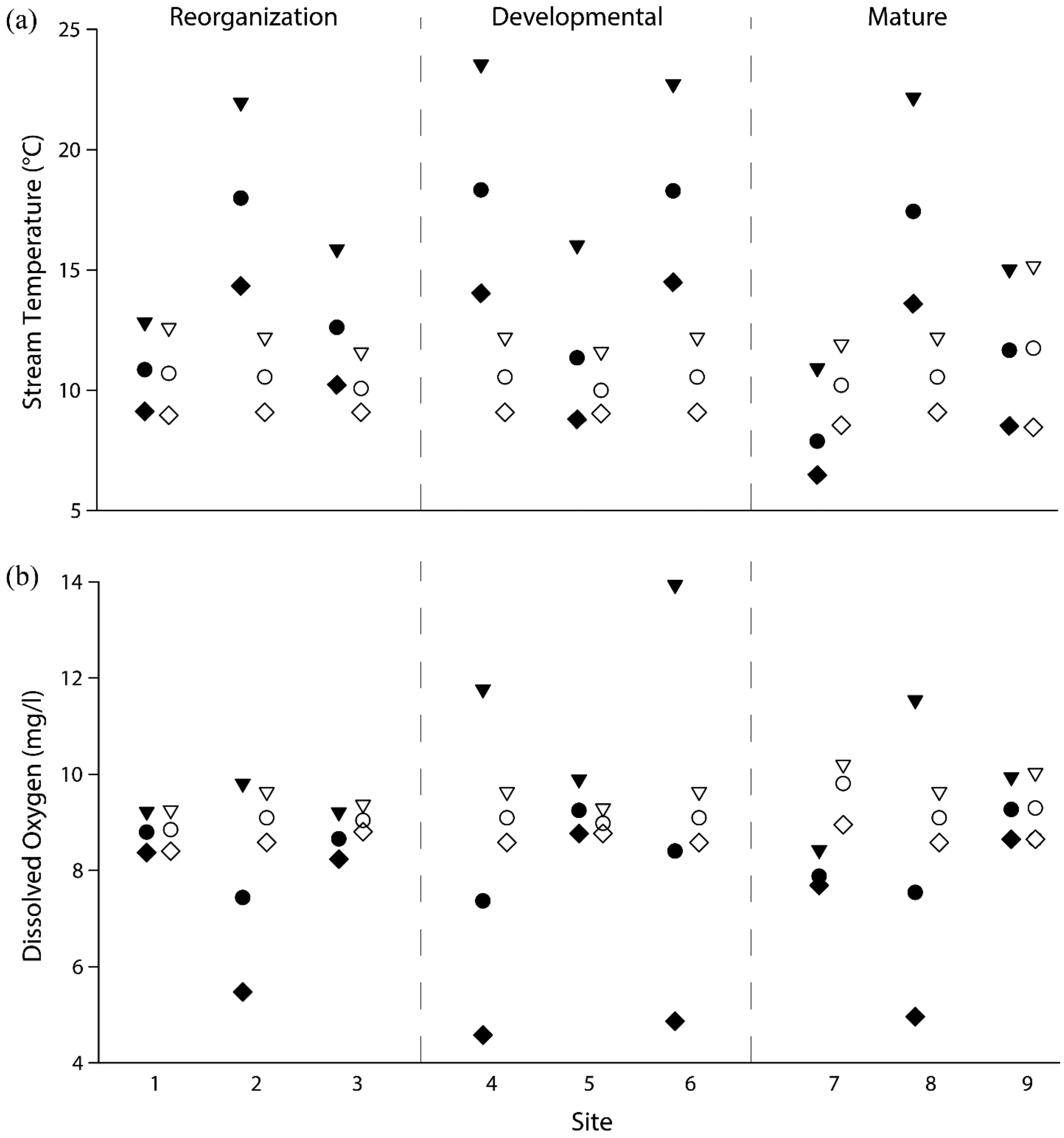

Figure 5. Maximum (▼), mean (●), and minimum (♦) diel stream temperature (a) and diel dissolved oxygen concentration (b) for each restored (black) and reference (white) reach. Both measurements were recorded on the same day at restored and reference reaches.

mean, and maximum values for any physicochemical metrics did not differ significantly among phases (ANOVA, $p > 0.05$), nor was there a relationship between the metrics and age (linear regression, $r^2 < 0.1$, $p > 0.05$) whether data were analyzed as absolute or response measures. However, mean diel range was significantly greater at restored reaches (6.8 ± 1.9°C) compared to reference reaches (3.4 ± 1.3°C, t-test, $p < 0.05$).

Canopy cover was drastically less well developed along restored reaches compared to reference streams for all phases of recovery (t-test, $p < 0.05$; Fig. 6). Canopy cover in reference reaches ranged from 59.02% to 81.3%, with a mean of 67.1 ± 3.33%. Restored reaches in reorganization and developmental phases ranged from 0.1% to 8.1% canopy cover, while reaches in the mature phases were associated with riparian cover ranging from 8.7% to 38.0%. These data suggest greater riparian development in later stages of recovery, but canopy cover did not differ significantly among phases (Kruskal-Wallis ANOVA, $p = 0.61$). However, when riparian development is based on absolute measures, a strong relationship exists between age and percent canopy cover ($r^2 = 0.57$, $p = 0.02$; Fig. 7), although much of the relationship is influenced by two older sites with cover in excess of 25%.

Across all reaches, OM stocks and chl *a* standing crops were greatest at S8, restored (AFDM = 59.5 ± 14.9 g/m^2, Chl *a* = 143.8 ± 94.6 mg/m^2, Table 4). No measurable OM or chl *a* was found at S9 or S7 reference (Table 4). Based on absolute data alone, OM standing stocks and chl *a* standing crops did not differ among phases (ANOVA, $p > 0.05$). While OM and chl *a* standing crops were reduced in the reorganization phase and increased in both developmental and mature phases based on response ratios, only chl *a* in the reorganization phase differed from reference condition (one-sample t-test, $p \leq 0.05$, Fig. 8). However, viewed as response ratios, chl *a* did show a strong relationship with age (linear regression, $r^2 = 0.46$, $p = 0.05$; Fig. 9). No such

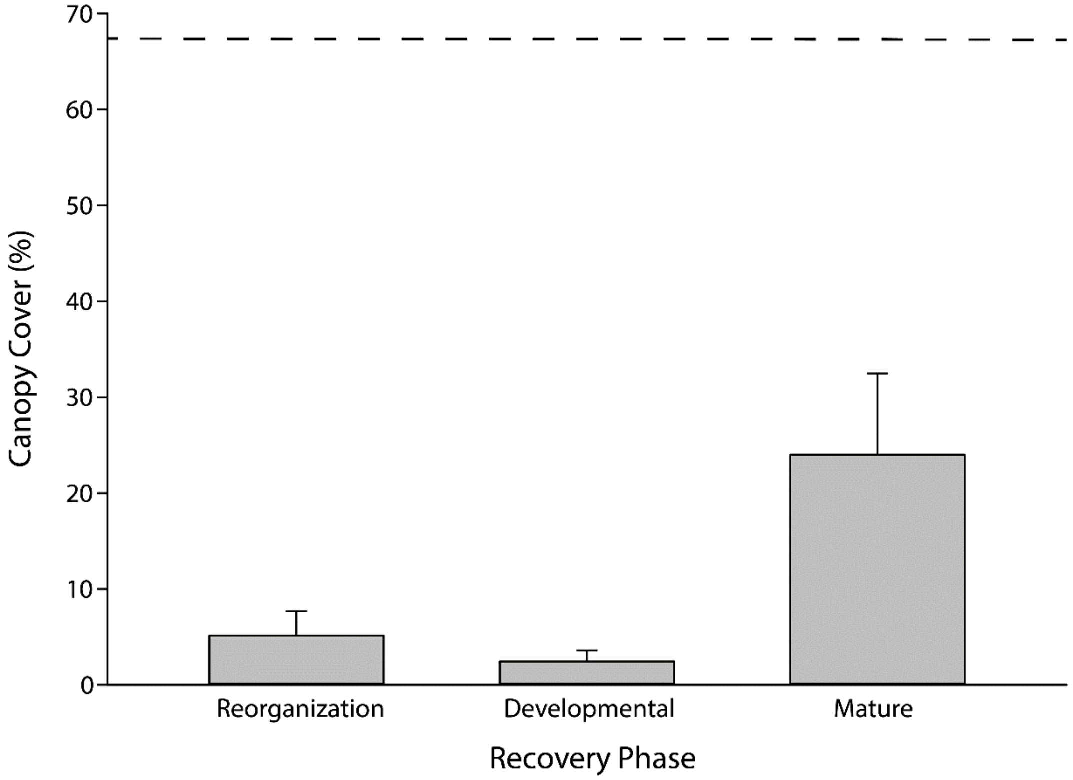

Figure 6. Mean (±1 SE) percent canopy cover of the three restoration reaches in each recovery phase. Dashed line is the mean percent canopy cover of the reference reaches (67.1 ± 3.33 SE). Percent canopy cover was significantly lower at each phase than reference condition (t-test, $p \leq 0.05$), however none of the phases were significantly different from each other (ANOVA, $p > 0.05$).

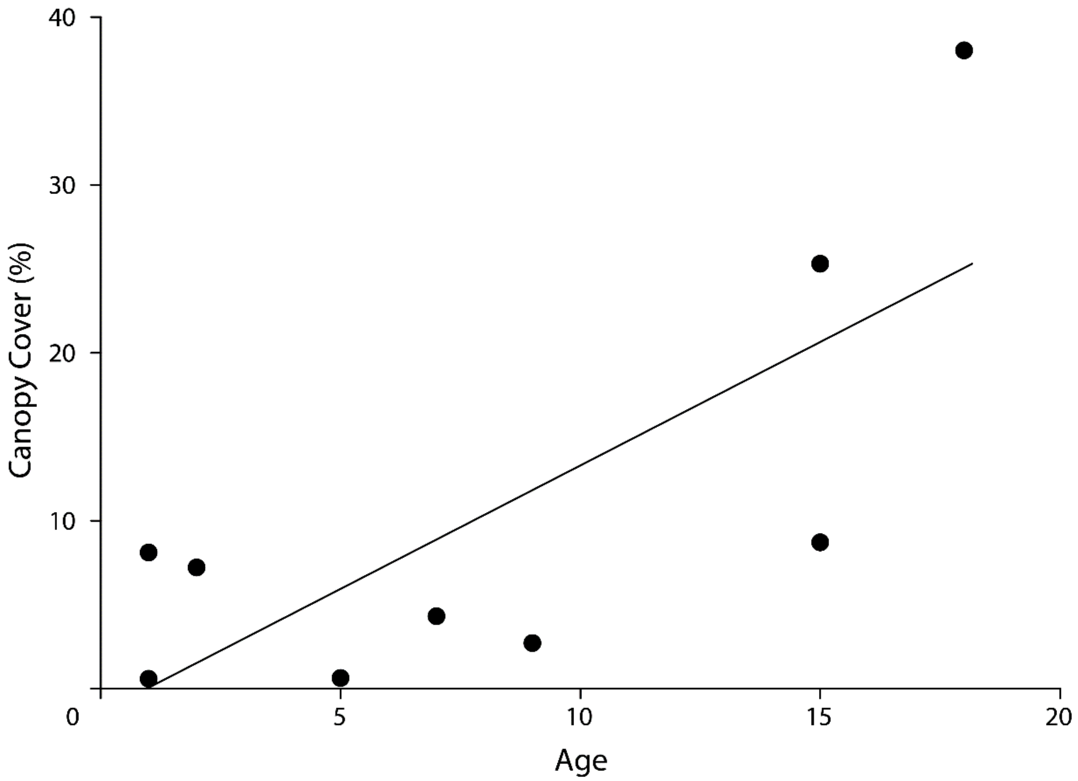

Figure 7. Relationship between percent canopy cover and years since restoration project implementation at all restored reaches ($r^2 = 0.57$, $p = 0.02$).

Table 4. OM and chlorophyll *a* values for restored and reference reaches at each site

	Benthic Organic Matter			
	AFDM (g/m^2)		Chlorophyll *a* (mg/m^2)	
Site	Restored	Reference	Restored	Reference
1	0.00 ± 0.00	12.9 ± 2.4	0.00	3.5 ± 0.9
2	4.6 ± 1.44	5.7 ± 0.4	1.2 ± 1.4	7.3 ± 2.0
3	1.4 ± 0.2	5.1 ± 1.7	0.2 ± 0.1	3.1 ± 2.7
4	49.0 ± 20.7	5.7 ± 0.4	10.9 ± 10.7	7.3 ± 2.0
5	4.3 ± 0.2	7.6 ± 2.7	2.0 ± 1.0	0.8 ± 0.4
6	18.1 ± 13.4	5.7 ± 0.4	22.1 ± 14.2	7.3 ± 2.0
7	1.1 ± 0.8	0.00 ± 0.00	0.3 ± 0.1	0.00 ± 0.00
8	59.5 ± 14.9	5.7 ± 0.4	143.8 ± 94.6	7.3 ± 2.0
9	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00

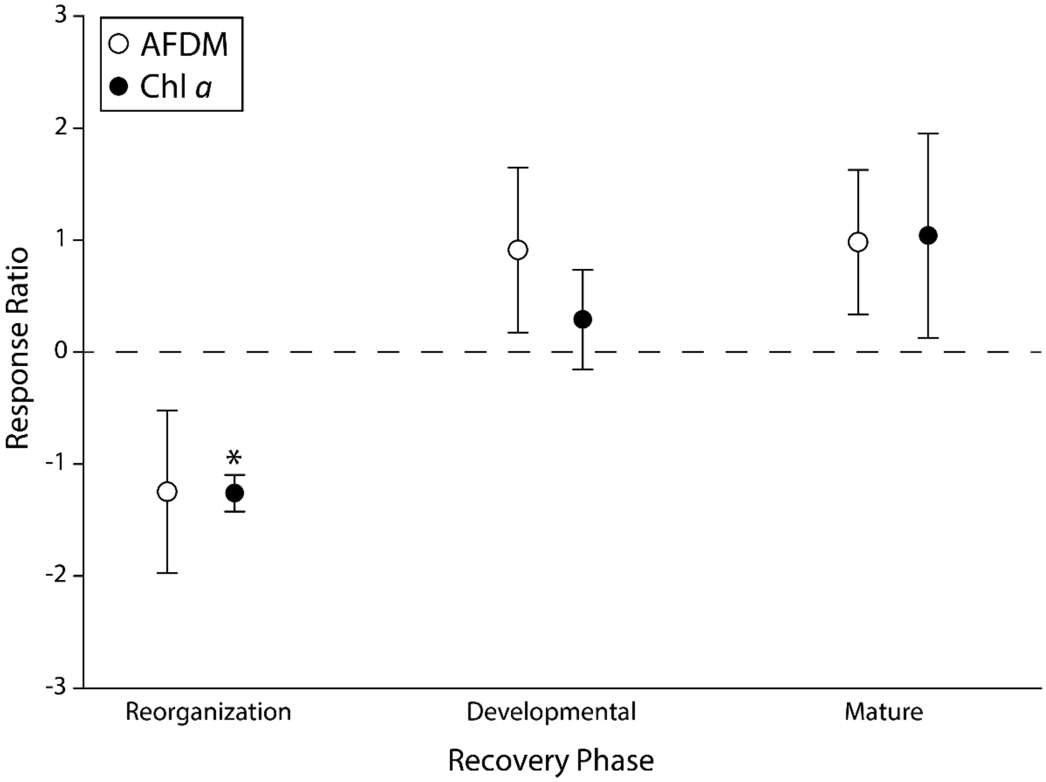

Figure 8. Mean (± 1 SE) response ratios of AFDM (open circle) and chl *a* (closed circle) at each recovery phases. Asterisks indicates a statistically significant difference from zero (one-sample t-test, $p < 0.05$).

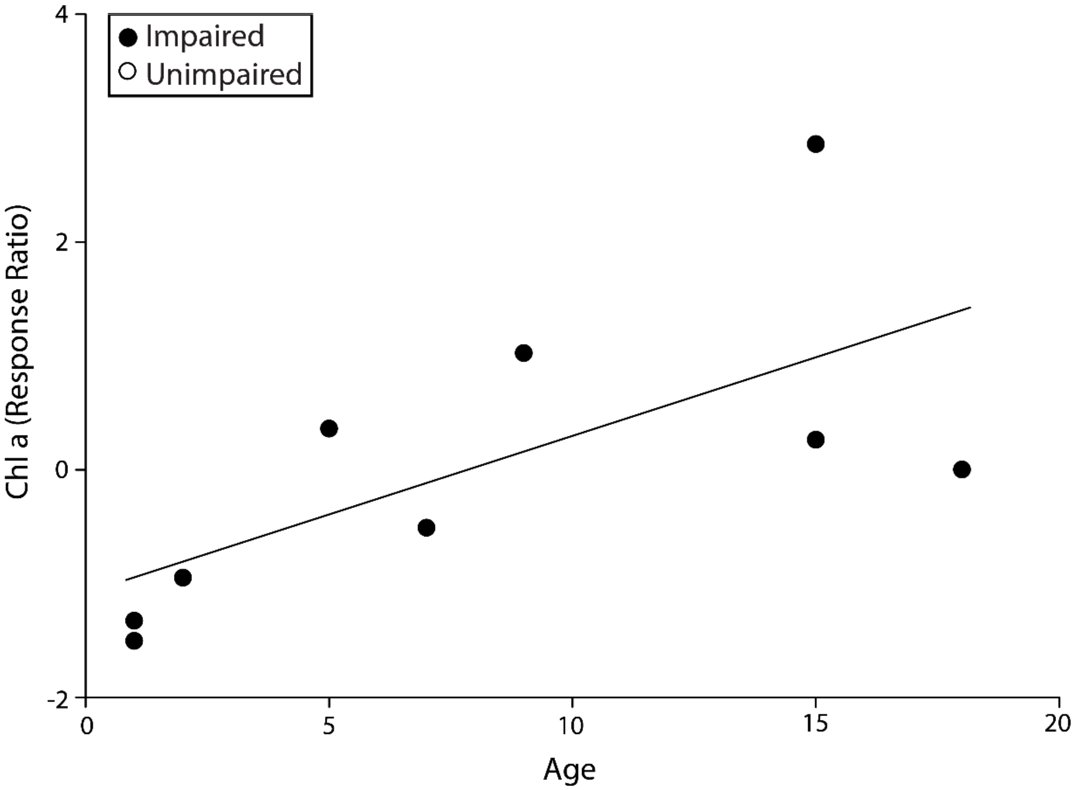

Figure 9. Relationship between benthic chlorophyll *a* response ratios and project age at all sites (r^2 = 0.46, p = 0.05).

relationship was observed between age and OM standing stock (linear regression, $r^2 = 0.27$, $p = 0.15$; Appendix C).

Across all sites, taxon richness, Shannon-Weiner index values, and shredder abundance were significantly different among restored and reference reaches (t-test, $p \leq 0.05$; Table 5). Taxa richness was significantly lower at restored reaches (13.59 ± 1.24) than reference reaches (18.04 ± .26). Based on Shannon-Weiner Index values, references reaches (2.22 ± 0.04) had greater diversity than restored reaches (1.73 ± 0.10). Finally, shredder abundance was also greater at reference reaches (14.59 ± 1.45%) than at restored reaches (9.73 ± 4.31). While restored reaches had greater variation in macroinvertebrate density (6,115 ind./m^2, S2 to 51,240 ind./m^2, S5) than at reference reaches (7,049 ind./m^2, S7 to 30,866 ind./m^2, S9), difference in density was not significant across all sites (t-test, $p > 0.05$). One-year old projects were the only sites with significantly decreased macroinvertebrate densities and increased percent EPT at restored compared to reference reaches (t-test, $p \leq 0.05$; Appendix D). Overall, percent EPT, collector-gatherer abundance, and habit metrics were similar among restored and reference reaches across all sites (t-test, $p > 0.05$).

Values for community similarity between restored and reference reaches are expected to fall within 0.12 to 0.24, if fully recovered. In contrast, site community similarity ranged from a minimum of 0.21 ± 0.02 (S9) to a maximum of 0.78 ± 0.01 (S8) across all paired study reaches (Appendix E). Only one site, S9 (0.21 ± .07 SE), fell within the expected range of variability. While macroinvertebrate community structure is clearly different among restored and reference reaches, no trends in recovery were identified for any community metric based on phase (ANOVA, $p > 0.05$) or age (linear regression, $r^2 < 0.3$, $p > 0.05$).

Table 5. Mean for each of the 10 macroinvertebrate response variables across all restored and reference reaches based on absolute counts. Bold values indicate a significant difference between reaches (t-test, $p < 0.05$)

	Reach Type	
Metric	Restored	Reference
Density	24,152 ± 4,291	20,827 ± 2,433
Taxa Richness	**13.59 ± 1.24**	**18.04 ± 0.26**
S-W Index	**1.73 ± 0.10**	**2.22 ± 0.04**
% EPT	59.31 ± 7.42	64.26 ± 3.56
% Shredder	**9.73 ± 4.31**	**14.59 ± 4.93**
% Collector-Gatherer	45.75 ± 4.93	46.51 ± 3.32
% Scraper	16.58 ± 2.97	20.63 ± 3.45
% Sprawler	39.18 ± 6.29	26.69 ± 4.02
% Clinger	45.75 ± 5.24	53.58 ± 4.45
% Swimmer	8.21 ± 2.44	12.09 ± 1.60

Recovery trends at impaired vs unimpaired sites

Following restoration, impaired sites had smaller D50 (25.8 ± 8.7) and greater percent fines (12.3 ± 4.1) than unimpaired sites (D50 = 73.9 ± 8.9, % Fines = 0.6 ± 0.3), but neither reach type differed significantly from reference condition (one-sample t-test, $p > 0.05$; Fig. 10). Additionally, impaired and unimpaired sites had sinuosity and percent pool similar to reference condition (one-sample t-test, $p > 0.05$) suggesting comparable geomorphic form (Fig. 10). Among impaired sites, percent fines decreased with age (linear regression, $r^2 = 0.88$, $p = 0.06$, absolute data; Appendix E). Physical habitat at unimpaired sites was similar across ages (linear regression, $r^2 < 0.4$, $p > 0.2$, absolute data and response ratios). Average values and associated ranges for diel variation in temperature were greater in impaired than unimpaired reaches (t-test, $p < 0.05$; Fig. 11a). DO also displayed greater diel variation, but decreased mean concentrations at impaired sites (t-test, $p > 0.05$; Fig. 11b). At impaired and unimpaired sites, physicochemical metrics were not correlated with age (linear regression, $r^2 < 0.4$, $p > 0.2$; data not shown).

Percent canopy cover at impaired sites (7.1 ± 6.09%) was lower compared to unimpaired sites (13.26 ± 6.23%), but did not differ significantly (Mann-Whitney Rank Sum Test, $p > 0.05$). However, the relationship between percent canopy cover and age was stronger at impaired sites ($r^2 = 0.78$, $p = 0.12$) than unimpaired sites ($r^2 = 0.49$, $p = 0.19$; Fig. 12), while small sample size resulted in relationships that were not statistically significant. In contrast to canopy cover results, impaired sites had greater (Mann-Whitney Rank Sum Test, $p < 0.05$), OM stocks (25.73 ± 12.87 g AFDM/m^2) and chl *a* standing crops (44.5 ± 33.37 mg/ m^2) compared to unimpaired sites (1.36 ± 0.79 g AFDM/m^2, 0.5 ± 0.38 mg/m^2 chl *a*; Table 4). Significant relationships between chl *a* and age existed at both impaired ($r^2 = 0.98$, $p = 0.01$) and unimpaired sites ($r^2 = 0.86$, $p = 0.02$; Fig. 13). However, the relationships were fundamentally different since chl *a* standings crops in

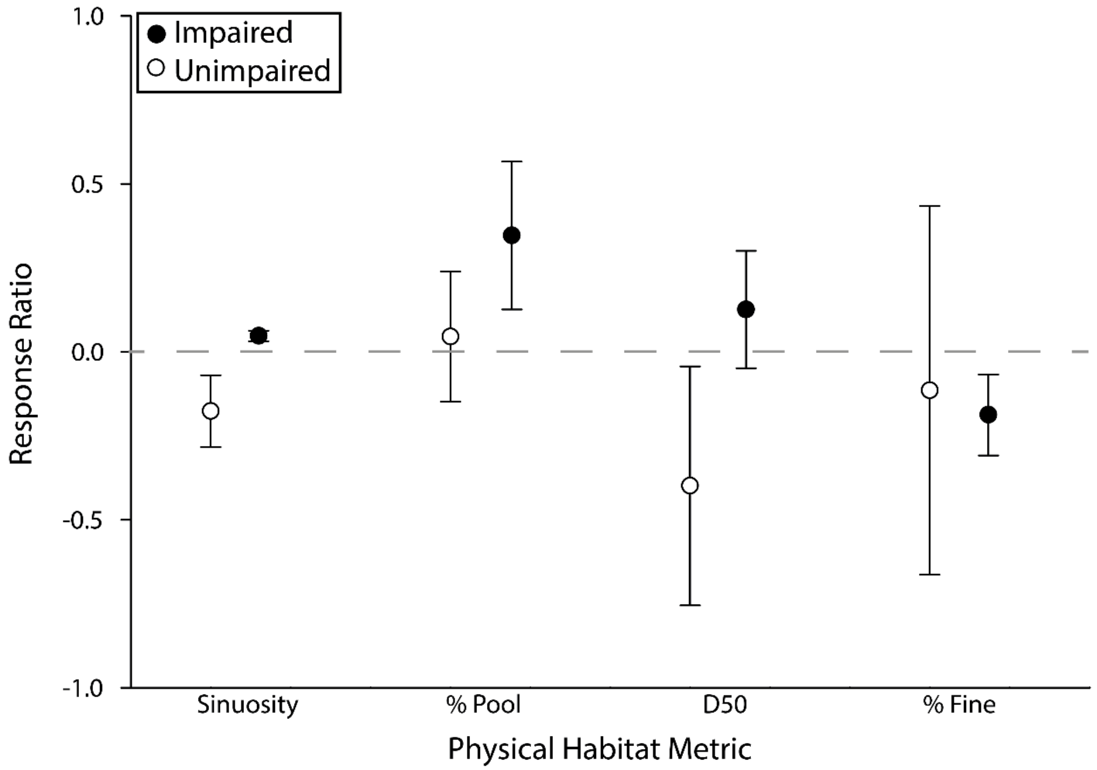

Figure 10. Mean (± 1 SE) response ratio of each physical habitat metric at impaired and unimpaired sites. Based on one-sample t-tests, none of the values were significantly different from zero (*p* > 0.05).

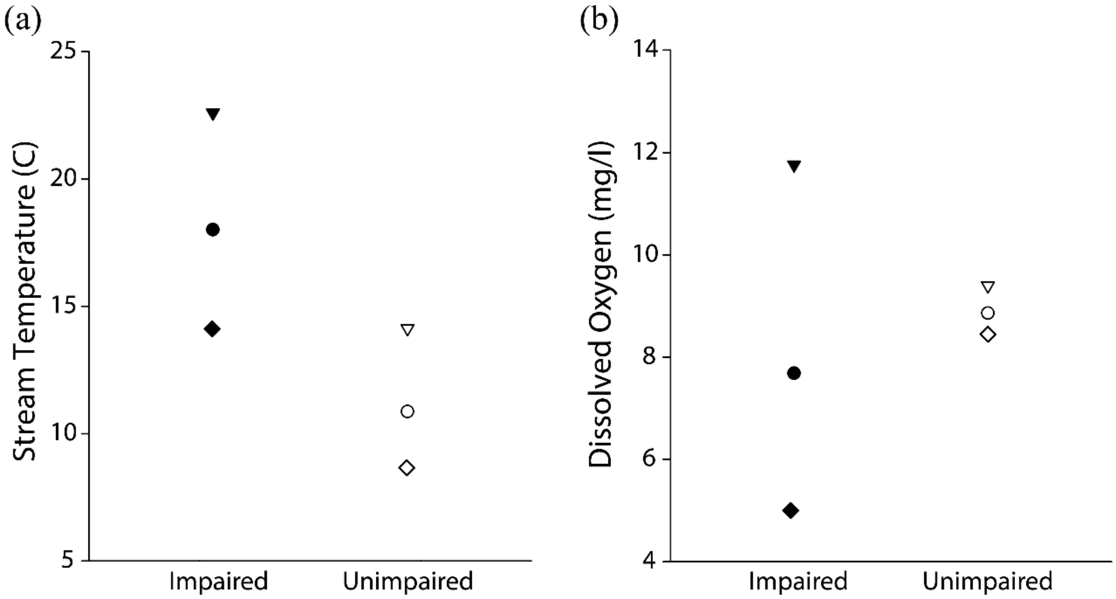

Figure 11. Overall maximum (▼), mean (●), and minimum (♦) stream temperature (a) and dissolved oxygen concentration at (b) impaired (black) and unimpaired(white) restored reaches.

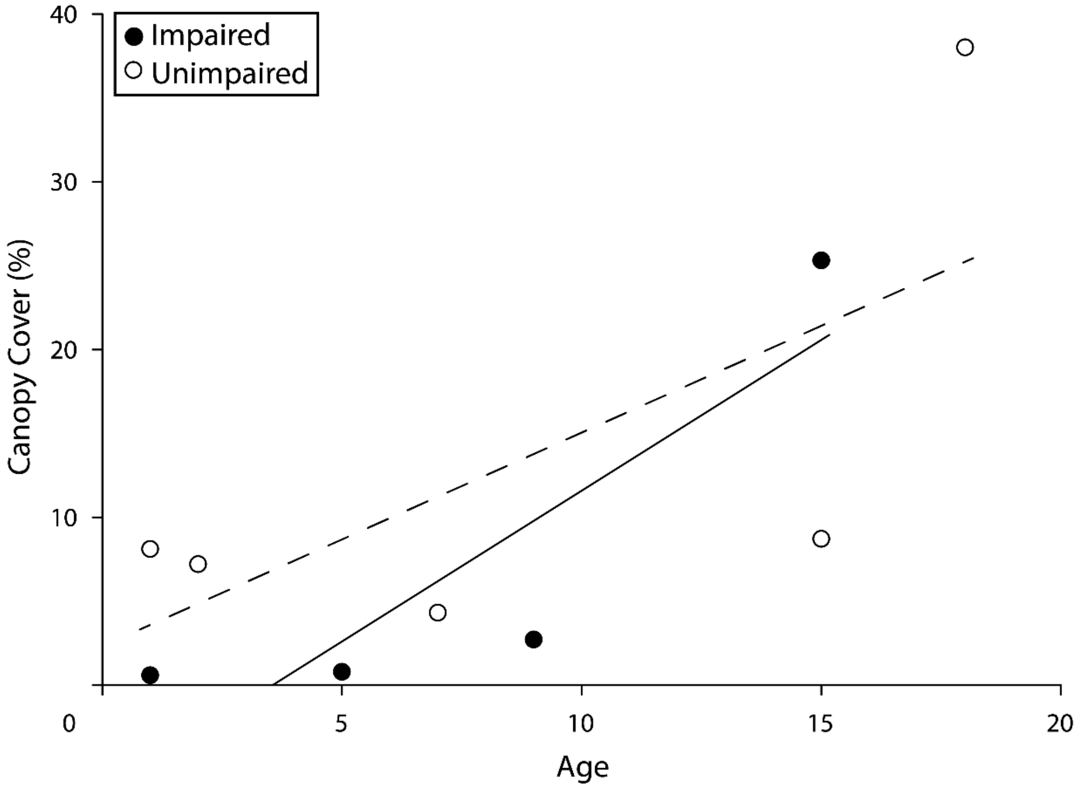

Figure 12. Relationship between percent canopy cover and project age at impaired (solid line, $r^2 = 0.78$, $p = 0.12$) and unimpaired (dashed line, $r^2 = 0.49$, $p = 0.19$) reaches separately.

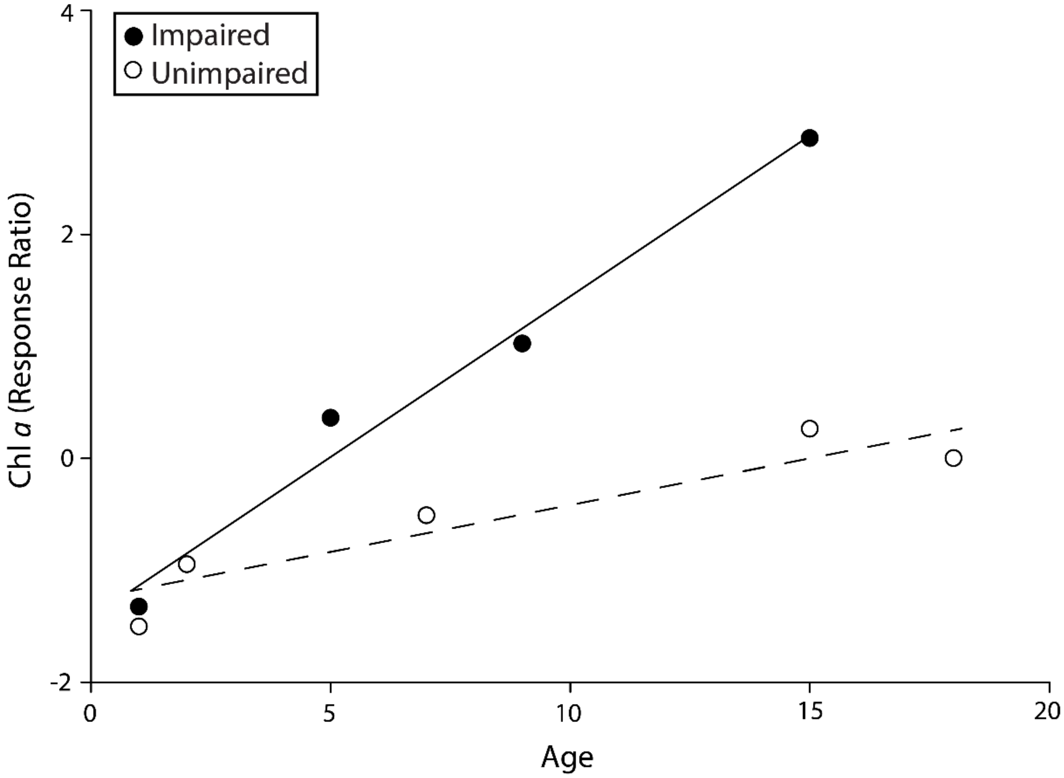

Figure 13. Relationship between benthic chlorophyll *a* response ratios and project age at impaired (solid line, $r^2 = 0.98$, $p = 0.01$) and unimpaired (dashed line, $r^2 = 0.86$, $p = 0.02$) reaches separately.

impaired reaches became increasingly greater compared to reference condition (i.e., increasingly positive values for response ratios) while chl *a* abundance in restored unimpaired sites converge on those characteristic of reference condition based on response ratios approaching zero.

Macroinvertebrate communities at restored reaches were significantly more similar (t-test, $p \leq 0.05$) to their reference condition at unimpaired sites (BC = 0.39 ± 0.07) compared to impaired sites (BC = 0.71 ± 0.04; Fig. 14). Additionally, community similarity formed clusters (ANOVA, SNK, $p < 0.05$) based on a combination of phase and watershed condition where sites in the reorganization phase (BC = 0.56 ± 0.10; Fig. 14a) are significantly different than sites older than two years which are impaired (BC = 0.75 ± 0.02; Fig. 14b) and unimpaired sites (0.29 ± 0.07; Fig. 14c). Trends in macroinvertebrate community similarity appeared related to time since restoration in impaired and unimpaired sites, but a small sample size may have contributed to a lack of statistical significance ($p > 0.05$). At impaired sites, macroinvertebrate communities become less similar to reference condition over time ($r^2 = 0.75$, $p = 0.13$, b = 0.01), while at unimpaired sites they become more similar ($r^2 = 0.59$, $p = 0.13$, b = - 0.02) over comparable timescales (Fig. 15).

FFGs also responded differently over time at impaired and unimpaired sites. At impaired sites, collector-gatherer abundance increased as sites got older ($r^2 = 0.86$, $p = 0.07$; Fig 16a), while at unimpaired sites, shredder abundance increased with age ($r^2 = 0.78$, $p = 0.05$; Fig 16b). These FFG responses correspond with Diptera taxa (e.g., Chironomidae) occurring at greatest abundance at older impaired sites and Plecoptera taxa (e.g., Nemouridae) becoming more abundant as unimpaired sites get older. The only significant habit response was in unimpaired sites older than one-year with <10% canopy cover (S3, S5, S7). Each of these sites had significantly increased sprawler abundance and decreased clinger abundance at restored compared to reference reaches (t-test, $p < 0.05$; Appendix F).

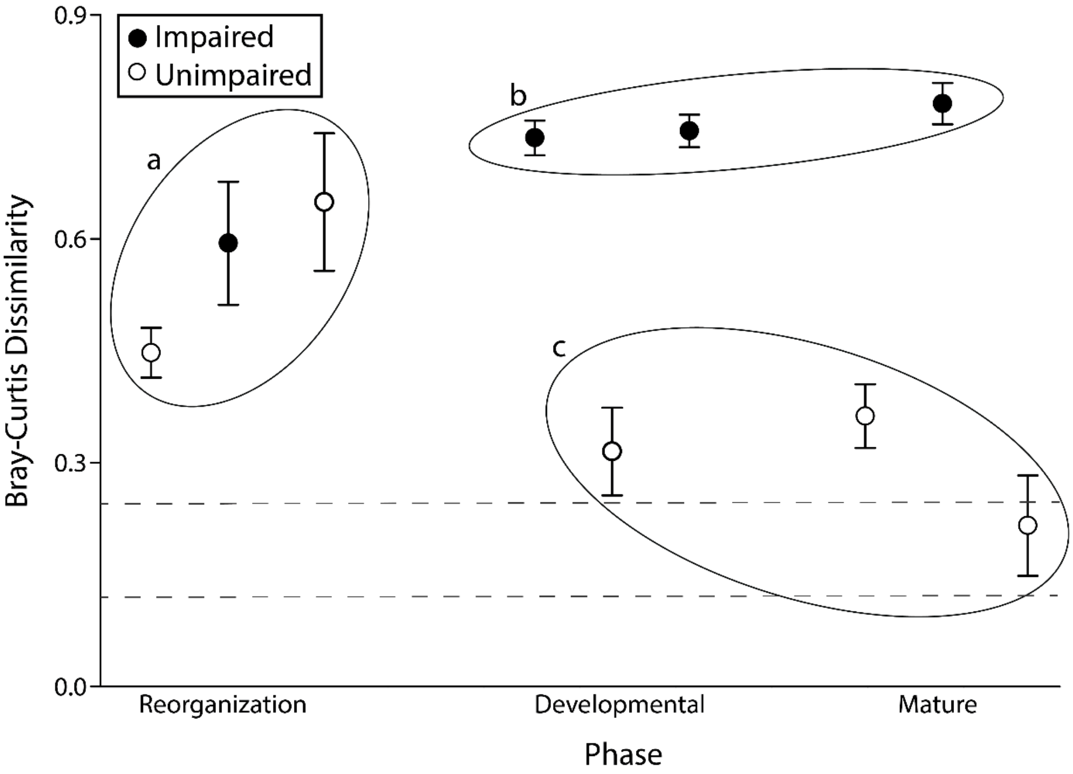

Figure 14. Mean (± SE) Bray-Curtis dissimilarity between each sites restored and reference reach grouped by phase. Dashed lines represent the expected range of dissimilarity among reaches in close proximity and experiencing similar watershed conditions (0.12 to 0.24). There were no significant difference among phases alone (ANOVA, $p > 0.05$). However, there was a significant difference between impaired sites (BC = 0.71 ± 0.04) and unimpaired sites (BC = 0.39 ± 0.07). Additional dissimilarity at (a) reorganization phase (BC = 0.56 ± 0.10) was significantly different than (b) impaired (BC = 0.75 ± 0.02) and (c) unimpaired (0.29 ± 0.07) sites (ANOVA, SNK, $p < 0.05$)

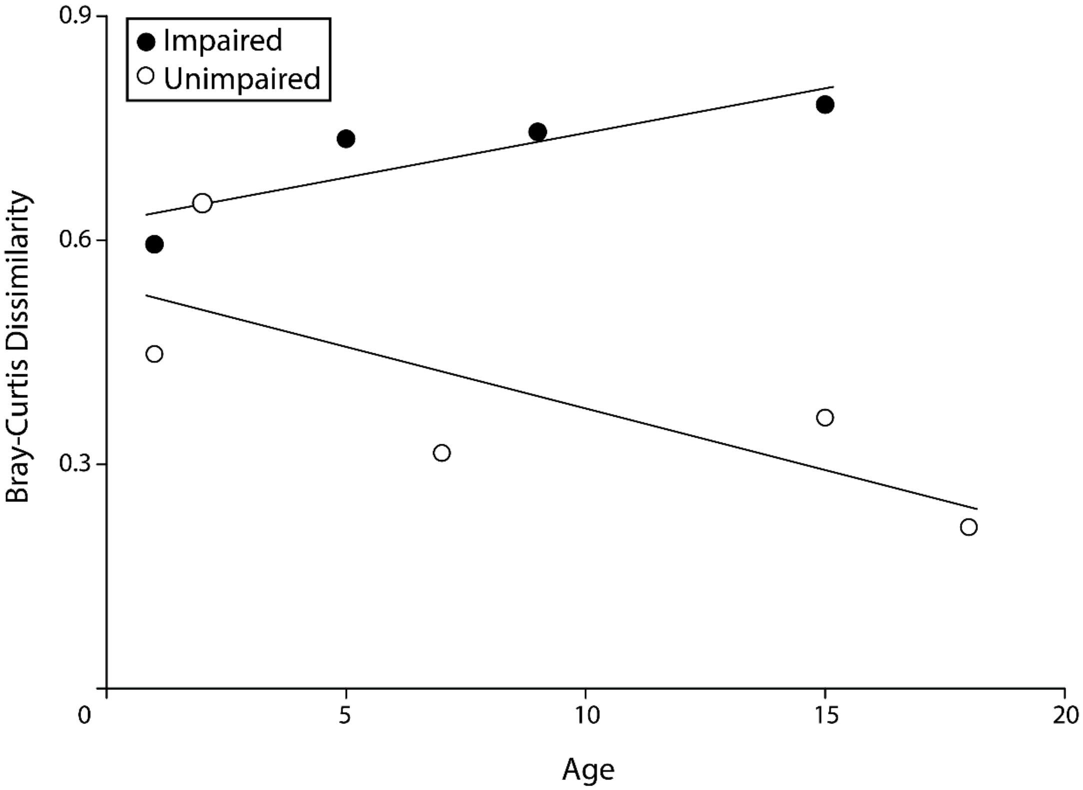

Figure 15. Relationship between Bray-Curtis Dissimilarity and age at impaired ($r^2 = 0.75$, $p = 0.13$) and unimpaired ($r^2 = 0.59$, $p = 0.13$) sites. Macroinvertebrate communities become less similar to reference condition over time at impaired sites and more similar over time at unimpaired sites.

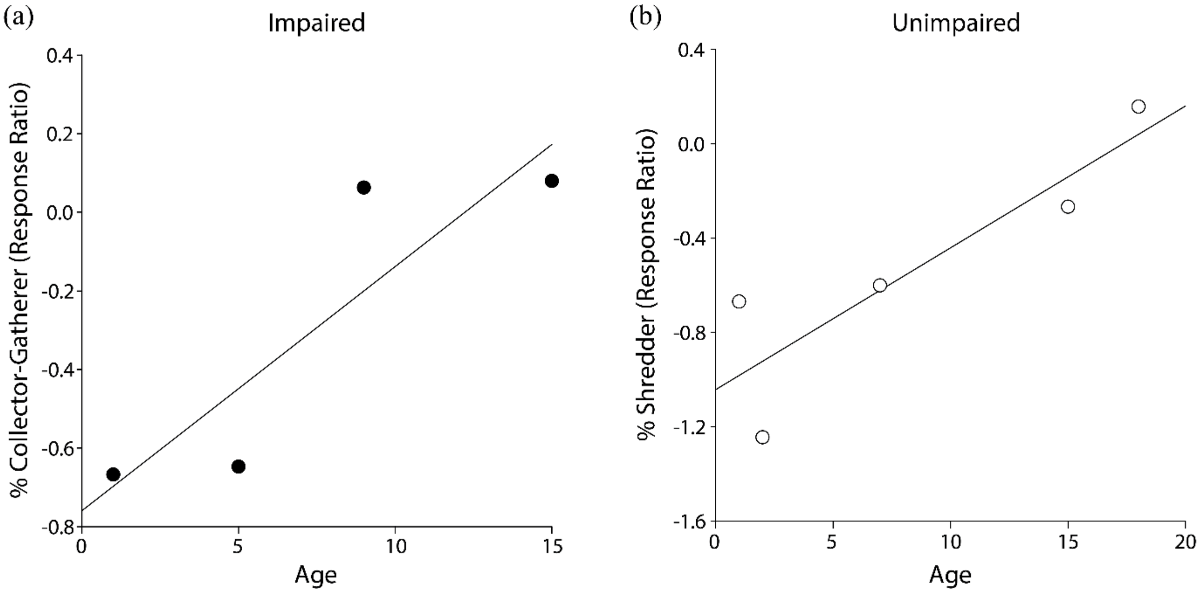

Figure 16. Primary functional feeding group response at impaired and unimpaired sites. At impaired sites, collector-gatherer abundance increases over time ($r^2 = 0.84$, $p = 0.09$), while at unimpaired sites shredder abundance increases as restoration projects get older ($r^2 = 0.78$, $p = 0.05$).

Discussion

While channel reconfiguration was generally successful at constructing desired physical habitat, conditions in riparian and benthic zones failed to reach reference conditions at the majority of restored reaches despite as much as 15 years of recovery. Although canopy cover increases as channel reconfiguration projects age, restoration reduced canopy cover below reference condition at all sites and riparian development provided less than half of the desired cover over the course of recovery. In the benthic zone, macroinvertebrate communities and OM standing stocks were similar in restored and reference reaches only at the oldest site sampled in a minimally disturbed watershed. This discrepancy between physical habitat and biological response provides further evidence for the need to reconsider how stream ecosystems respond and recover following channel reconfiguration projects.

Endogenous control alone, specifically age-based phase, was not a strong indicator of ecosystem recovery following channel reconfiguration when aggregating across all nine sites. Predictable temporal behavior only appeared to drive recovery at all channel reconfiguration projects for one to two years following channel reconfiguration. I observed an overall reduction in chl *a* standing crop in the reorganization phase sites. Macroinvertebrate community patterns in this phase were most evident in the one-year old projects. These restored reaches were characterized by reductions in macroinvertebrate density and increased EPT abundance. The density response is consistent with Biggs et al. (1998) who showed a rapid recolonization approximately one-year after restoration. In my study, projects older than two years had macroinvertebrate densities greater than or equal to reference condition. Increases in EPT abundance at these restored reaches indicates that mechanisms of recolonization occurring following channel reconfiguration initially favors these taxa (Wallace 1990). Additionally, EPT abundance decreased and Chironomidae abundance increased after two years at unimpaired sites. Tullos et al. (2009) found a similar result, showing increased Chironomidae abundance was common at restoration projects considered to be beyond the reorganization phase.

At sites without anthropogenic influence, reduction in canopy cover appears to exert a strong influence on recovery in the benthic zone after initial recolonization, a feature characteristic of the developmental phase. Developmental conditions following reorganization are likely what the majority of monitoring and research efforts capture. In particular, three recent studies on channel reconfiguration showed that restoration projects had either no influence or a disturbance effect (i.e., changes that favored generalist taxa) on macroinvertebrate communities (Tullos et al. 2009, Ernst et al. 2012, Rios-Tuoma et al. 2015). Specifically, Tullos et al. (2009) showed a disturbance effect in projects ~ 4 years old in rural catchments, which was characterized by increased Chironomidae abundance. This translated to increased abundance of generalist traits such as rapid development and dominance by collector-gatherer FFGs. Additionally the authors showed a decrease in shredder abundance in restored reaches. This is similar to results in this study that documented an increase in generalist traits and a decrease in specialist traits in unimpaired sites with reduced canopy cover (< 10%). This increase in generalist taxa (i.e., Chironomidae) corresponded to a sharp reduction is specialized families (i.e., Heptageniidae and Peltoperlidae).

Three primary filters to macroinvertebrate communities include thermal regime, food resources, and habitat stability (Poff et al. 2006). Changes in the latter two, which occur as the result of altered OM dynamics, appeared to play the largest role in macroinvertebrate response to channel reconfiguration. This change in OM form appears to result directly from canopy cover reduction. In western Montana, the dominate forms of OM in unimpaired headwater streams with dense canopy cover were biofilm and bryophyte. These forms of OM, particularly bryophyte, provide relatively stable habitat refugia for macroinvertebrates during periods of increased discharge (Suren and Duncan 1999). At restored reaches in this study, filamentous algae, typically associated with open canopy cover, was the dominate form of OM. Filamentous algae is more vulnerable to high flows and therefore is a less stable form of habitat for macroinvertebrates (Grimm and Fisher 1989, Biggs et al. 1999). Furthermore, bryophytes trap fine particulate OM providing a critical food resource to macroinvertebrates, while filamentous algae is a poor food resource (Stream Bryophyte Group 1999, Cummins and Klug 1979). Additionally, reduction in allochthonous inputs caused by slow development of riparian cover at restored reaches represents an additional food web change. This change is illustrated in this research by shredder abundance increasing with riparian growth. Therefore, if a restored reach has decreased riparian cover and increased filamentous algae growth, food resource and habitat stability changes are influencing the condition of macroinvertebrate communities.

Based on these data, it is difficult to determine how long this developmental phase may last before a reach transitions into the mature phase. I based recovery phases on project age because restoration practitioners and funders typical plan projects in time-based cycles. However, this is likely not appropriate for sites under endogenous control where canopy cover appears to be the primary driver of ecosystem recovery. Sites in the predicted age range of the mature phase had the largest variation in canopy cover and ecosystem condition. This variation could be the result of natural processes, riparian planting failure, or relic impacts from channel construction. For example, Laub et al. (2013) found that riparian soils have high bulk density and low root biomass at channel reconfiguration projects for more than 10 years after channel construction. This is critical because soil characteristics exert a strong influence on how well riparian plants establish and grow (Bhattacharjee et al. 2008). Based on the slow riparian establishment observed following restoration, it is possible that riparian zones recover slower following channel reconfiguration than after natural disturbances. It would inform monitoring practices to be able to predict a density threshold for canopy cover or a recovery rate where riparian conditions re-exert influence over benthic zone conditions. However, these processes are likely a function of stream size, riparian planting care, soil compaction, and natural growing season, meaning any prediction would be site-dependent (Sweeney et al. 2002).

In the current study, human-induced impairment exerted a strong exogenous influence on the condition of the benthic zone at two sites in the developmental phase (S4 and S6) and one in the mature phase (S8). The level of impairment at these sites is severe; sites included in this study are known by the State of Montana to exceed total maximum daily loads for nitrogen, sediment, and temperature (Montana DEQ 2014). Specifically, point-source impairment from the wastewater treatment plant created recovery trends in the benthic zone exactly the opposite of what I would have expected based on endogenous recovery patterns seen at unimpaired sites. At impaired sites, macroinvertebrate communities and OM standing stocks in restored reaches became less similar to reference condition over time. This trend is the result of older sites being in closer proximity to the wastewater discharge point. Nutrient enrichment supports large standing stocks of macrophytes, which are generally a poor resource for EPT taxa (Voshell 2002, Pederson et al. 2007). Our results indicate that the level of impairment present at these sites is great enough to completely reverse expected recovery trends. This adds to the body of literature showing that catchment-level impairment has a stronger effect on ecosystem structure and function than reach-scale restoration (Palmer et al. 2010). Specifically, Louhi et al. (2011) and Leps et al. (2016), studied macroinvertebrate response in developed (i.e., impaired) watershed over longer time-periods than typically addressed (~20 years) and showed little impact of restoration on community structure.

My research suggests that exogenous controls generated at the watershed scale act as a press disturbance (Lake 2000) on restored reaches, which is likely to determine ecosystem conditions in spite of restoration efforts until continued catchment-level degradation is addressed. If human intervention removed all watershed-scale impairment affecting a site, it is logical to assume a restored reach would transition to its corresponding endogenous phase of recovery based on project age. However, the rate and magnitude of this transition is more likely a function of the target reference condition, level of human intervention, and system resiliency (Folke et al. 2004). I used minimally disturbed reference sites in this study because of their availability in Montana. It is unlikely in developed watersheds that human intervention will occur at a scale large enough to achieve a minimally disturbed target (Ehrenfeld 2000). Therefore, “best attainable condition” maybe a more suitable target in order to acknowledge the difficulty of removing all human influence (Stoddard et al. 2006). Best attainable condition must be defined based on a system’s resiliency (i.e., the magnitude of disturbance a system can endure before a regime shift occurs, Holling 1973). Specifically for macroinvertebrates, availability of species to recolonize a disturbed system plays a critical role in ecosystem resiliency and therefore likely determines recovery capacity for channel reconfiguration projects in impaired watersheds (Peterson et al. 1998).

Generalizing macroinvertebrate response to restoration has produced highly variable results, particularly when assessing common response metrics such as richness and density (Miller et al. 2009). This issue is compounded when using a space-for-time substitution due to natural variation within and among stream systems (Heino et al. 2004). Understanding this difficultly is critical when setting restoration goals and evaluating recovery. Among the metrics I used to evaluate macroinvertebrate response to channel reconfiguration, very few showed a statistically significant change between restored and reference condition, even though differences in community structure were clear through similarity metrics. In particular, macroinvertebrate metrics most commonly used to evaluate restoration success - richness, density, and diversity - failed to identify discernable recovery trends across sites based on age or phase. This is consistent with results from Ernst et al. 2012, who showed little difference in macroinvertebrate communities between restored and reference reaches based on species richness, abundance, and diversity metrics. As others have suggested, functional group metrics appeared to generate more consistent results when comparing across systems (Tullos et al. 2009).

Conclusion

The PRF provides a construct for evaluating stream restoration projects that recognizes the inherent ecosystem development that will accompany channel reconfiguration. In that context, this research determined ecosystem recovery following channel reconfiguration restoration to be an interconnected function of endogenous and exogenous controls. The primary endogenous control on ecosystem recovery in watersheds free of anthropogenic influences appears to be the successional sere of the riparian zone. Channel reconfiguration significantly reduces riparian canopy cover for a significant amount of time, which influences conditions in the benthic zone of streams. Specifically, increased insolation favors growth of filamentous algae, not the common form of OM in most headwater systems. This change in OM form, combined with the reduction in allochthonous inputs from the riparian zone, act as a filter on the macroinvertebrate community that favors generalist taxa.

In the presence of anthropogenic influence like nutrient enrichment, exogenous controls appear capable of superseding any recovery that may occur as the result of channel reconfiguration. At impaired site included in this study, point-source nutrient enrichment from a wastewater treatment plant was the dominant factor in determining OM standing stocks and structuring macroinvertebrate communities. This impairment occurred at a level capable of producing trends exactly opposite of those observed in unimpaired watersheds. This research adds to the growing body of literature questioning the need for expensive reach-scale restoration projects when degraded conditions occur at the watershed-scale (Bernhardt and Palmer 2011).

Conducting this research in Montana allowed me to investigate extreme cases of endogenous and exogenous recovery trajectories. As human populations continue to expand and influence ecosystems, the threshold between impaired and unimpaired ecosystems will likely become more ambiguous. This ambiguity makes setting appropriate goals of ecosystem recovery and devising monitoring schemes following restoration more challenging and increasingly critical in situations where the two recovery trajectories cannot be considered entirely independent. Furthermore, current monitoring time-periods and methods appear inadequate to capture the changes lotic ecosystem undergo following channel reconfiguration. More resources need to be allocated to determining appropriate restoration goals and ensuring projects are reaching those targets.

The field of stream restoration currently focuses on reestablishing the geomorphic form of degraded lotic ecosystems in order to achieve ecological goals. Despite the fact that practitioners are now adept at constructing new stream channels, there is sparse evidence indicating that these project are achieving all of their goals. Obscured in the effort to build new channels is the fundamental ecological principle that streams change following disturbance and continue to be a reflection of their watershed (Hynes 1975, Stevens and Cummins 1999). In pristine headwater systems, where restoration is common, we must acknowledge that channel reconfiguration projects sever aquatic-terrestrial linkages critical to the function of lotic systems. While these projects typically involve riparian planting post-construction, time allocated for monitoring and adaptive management is insufficient to make sure recovery is occurring. If riparian plantings fail to become established, channel reconfiguration projects risk becoming a persistent disturbance on the landscape. In watersheds with extensive anthropogenic influence, channel reconfiguration projects may represent a misallocation of resource because they likely fail to address the root cause of degradation.

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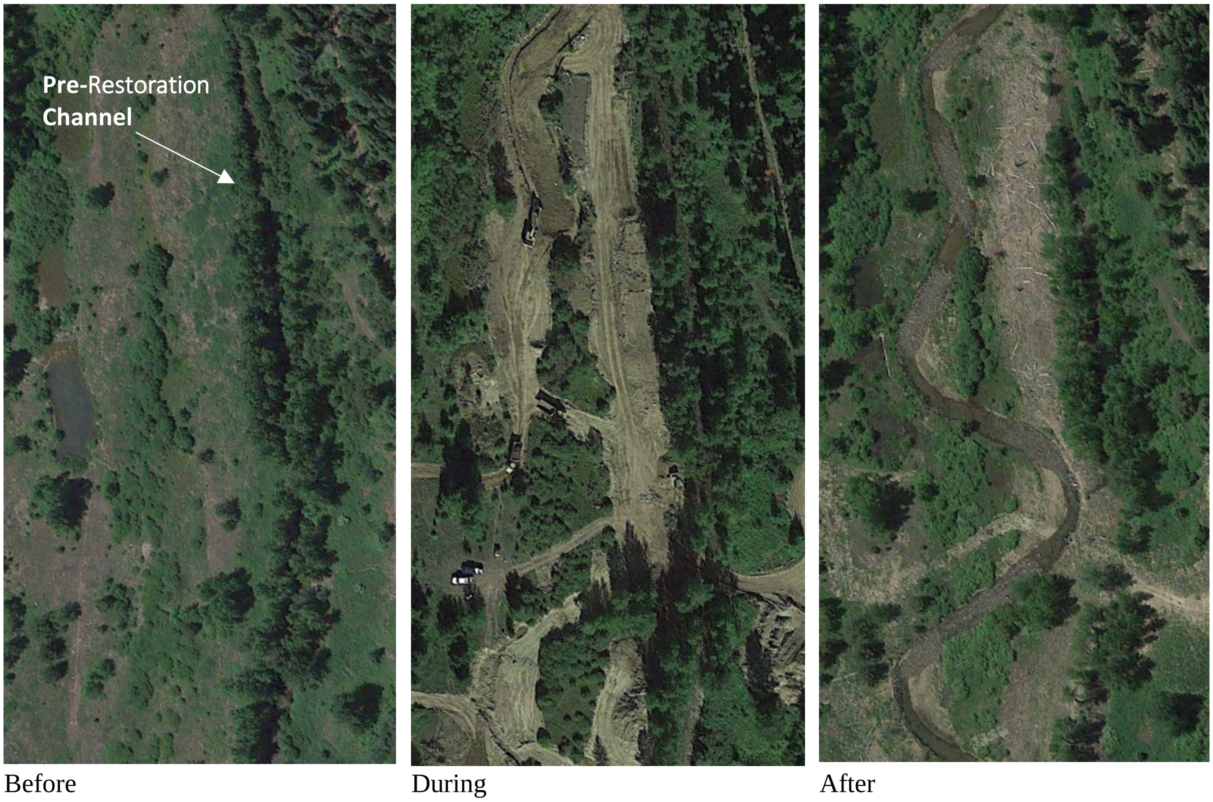
Appendicies

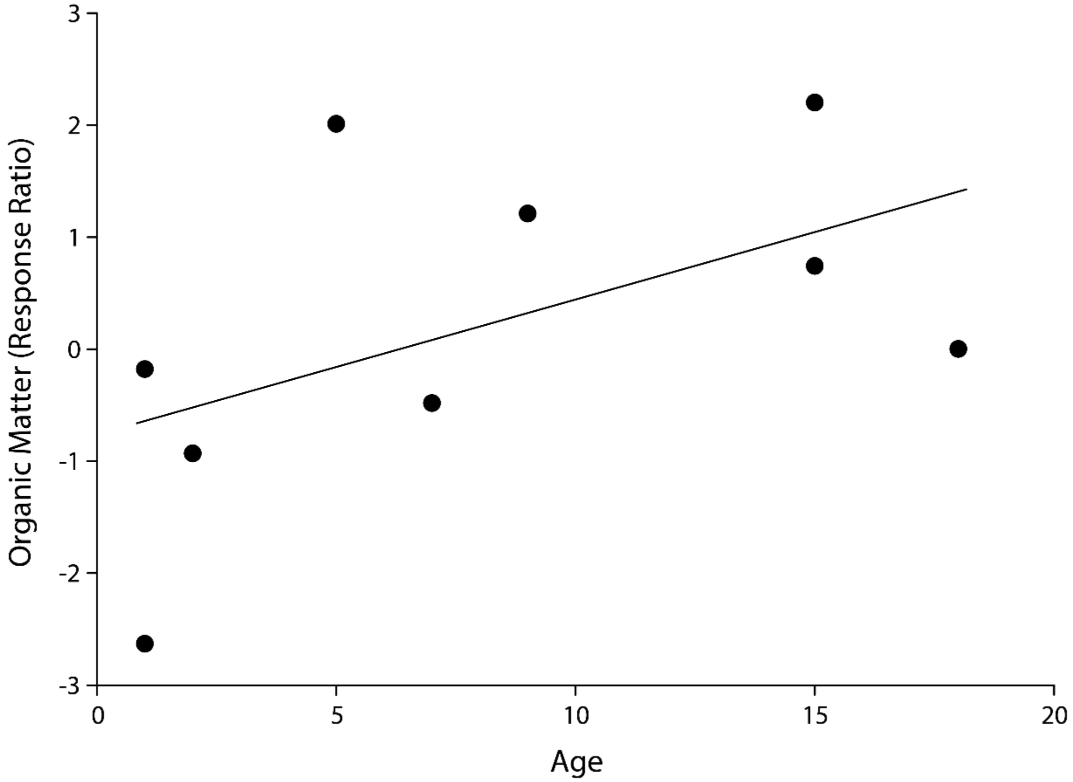


Before During After

Appendix A. Aerial photographs of a channel reconfiguration in process (Ninemile Creek, MT)

Appendix B. Source of reference data for impaired sites

Variable	Reference System	Source
Physical Habitat	French Creek, Mill Creek, Willow Creek, German Gulch	Measured
Water Quality	Willow Creek, Baggs Creek	Montana DEQ
Canopy Cover	Mill Creek, Willow Creek	Measured
Organic Matter	German Gulch, Baggs Creek	Montana DEQ
Macroinvertebrates	German Gulch, Baggs Creek	Montana DEQ



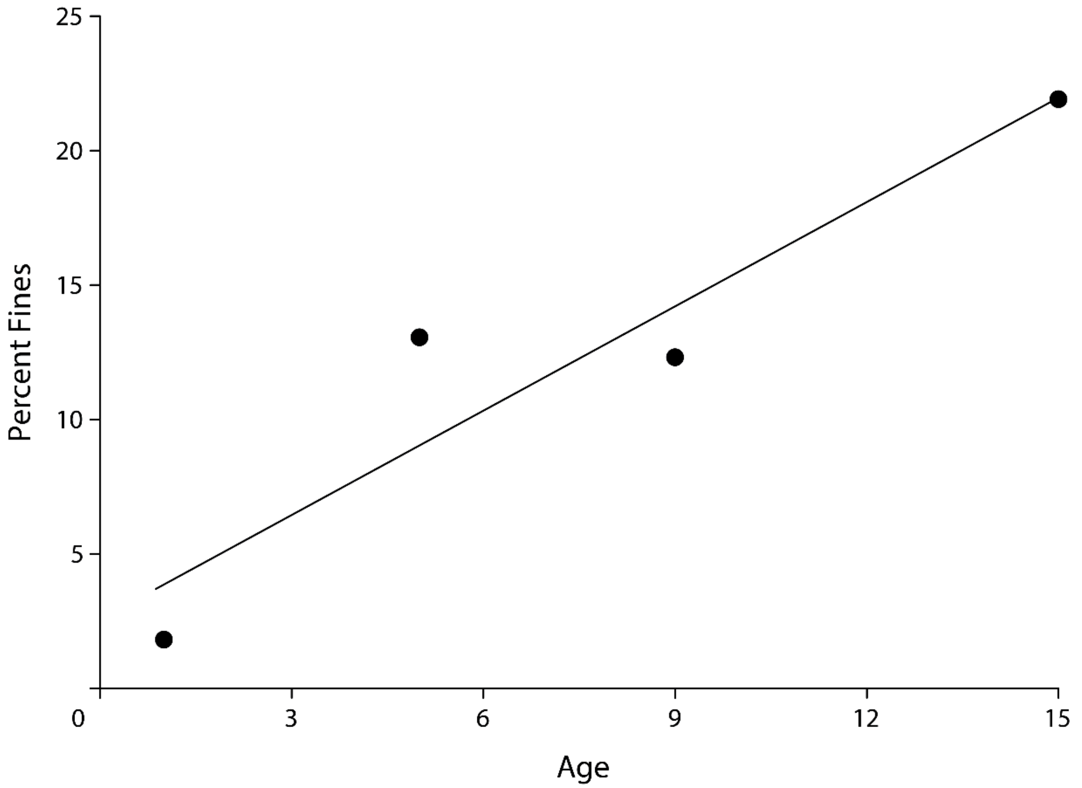
Appendix C. Relationship between organic matter response ratio and age for all sites (r^2 = 0.27, *p* = 0.15).

Appendix D. Mean absolute values for each of the 10 macroinvertebrate response variables. Bold values indicate a significant difference between restored and reference values based on the three samples collected at each reach (t-test, $p < 0.05$). Shaded sites are impaired.

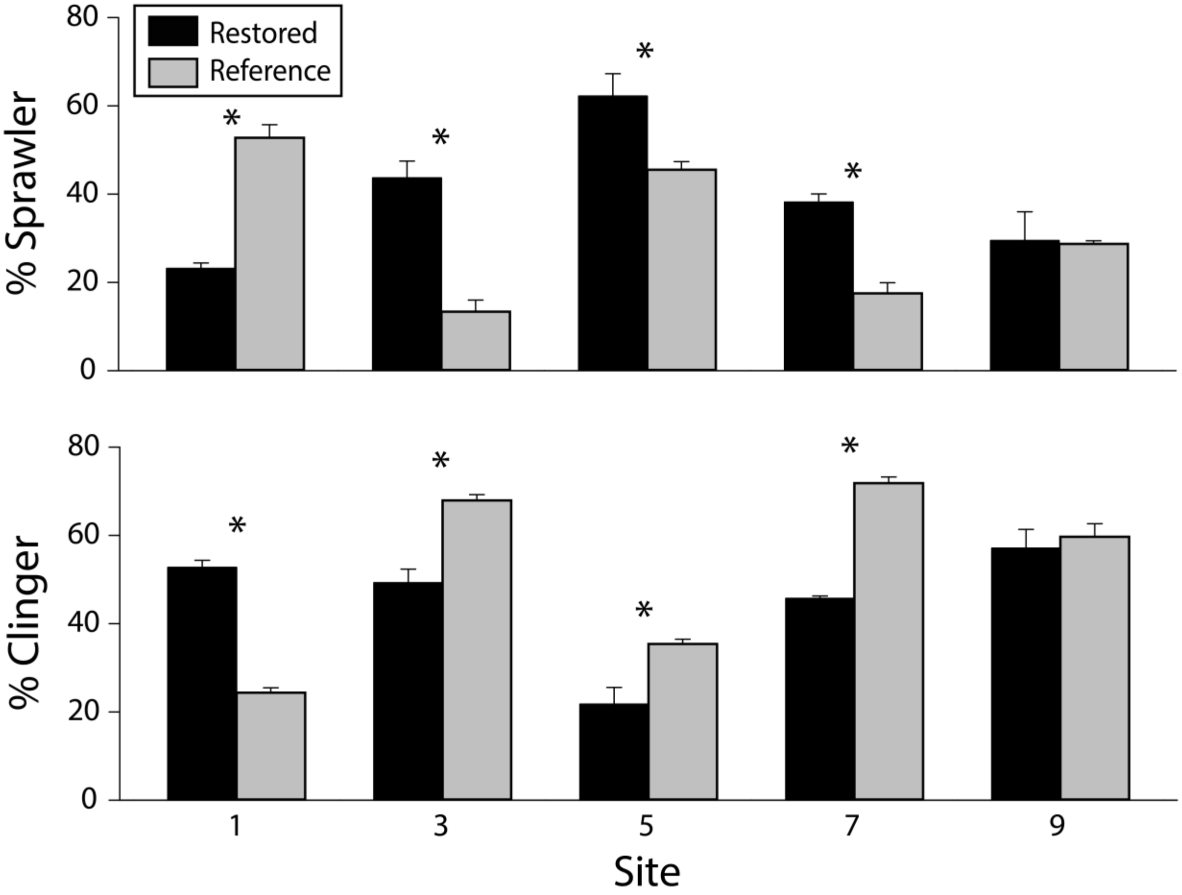
Site-Reach	Density (per m^2)	Taxa Richness	S-W Index	% EPT	% SR	% C-G	% SC	% Sprawl	% Cling	% Swim
1-Rest	**11,927**	17	2.02	**81.3**	**9.2**	**41.0**	**35.4**	**23.2**	**52.7**	23.3
1-Ref	**20,964**	19	2.05	**53.7**	**18.0**	**67.4**	**3.0**	**52.8**	**24.3**	20.72
2-Rest	**6,155**	**12**	**1.71**	**83.6**	**46.9**	**22.1**	9.9	**10.0**	**37.8**	**5.2**
2-Ref	**20,828**	**18**	**2.32**	**65.7**	**18.7**	**43.1**	19.3	**20.5**	**55.8**	**10.5**
3-Rest	17,916	19	2.06	**54.7**	**2.1**	**61.2**	**13.3**	**43.7**	**49.3**	**7.0**
3-Ref	11,773	19	1.96	**90.1**	**7.3**	**34.2**	**38.6**	**13.4**	**67.9**	**18.4**
4-Rest	22,509	**9**	**1.46**	76.3	**1.9**	**22.6**	**13.9**	**22.1**	**75.9**	**0.1**
4-Ref	20,828	**18**	**2.32**	65.7	**18.7**	**43.1**	**19.3**	**20.5**	**55.8**	**10.5**
5-Rest	51,240	**13**	**1.46**	**43.2**	6.9	**73.3**	13.9	**62.2**	**21.7**	15.6
5-Ref	33,484	**19**	**2.20**	**61.0**	12.6	**60.8**	7.8	**45.5**	**35.4**	16.5
6-Rest	**34,477**	**10**	**1.33**	65.7	**1.1**	45.9	**6.7**	**46.2**	52.2	0.4
6-Ref	**20,828**	**18**	**2.32**	49.2	**18.7**	43.1	**19.3**	**20.5**	55.8	10.5
7-Rest	36,842	**11**	**1.76**	67.9	6.9	**54.1**	**23.7**	**38.2**	**45.7**	**15.7**
7-Ref	7,049	**17**	**2.10**	71.3	9.0	**34.4**	**37.0**	**17.5**	**71.8**	**4.7**
8-Rest	12,269	**12**	**1.53**	**2.8**	**1.1**	46.7	**5.8**	**77.5**	**19.4**	**0.9**
8-Ref	20,828	**18**	**2.32**	**65.7**	**18.7**	43.1	**19.3**	**20.5**	**55.8**	**10.5**
9-Rest	24,032	20	2.27	58.3	11.4	44.9	26.7	29.5	57.1	5.7
9-Ref	30,866	19	2.39	56.0	9.8	49.2	22.0	28.7	59.7	6.4

Appendix E. Bray-Curtis Dissimilarity for each site. Expected range is 0.12 to 0.24.

Site	Bray-Curtis Dissimilarity
1	0.44 ± 0.01
2	0.59 ± 0.02
3	0.65 ± 0.03
4	0.74 ± 0.01
5	0.32 ± 0.02
6	0.74 ± 0.01
7	0.36 ± 0.01
8	0.78 ± 0.01
9	0.21 ± 0.02



Appendix F. Relationship between percent fines and age at impaired sites ($r^2 = 0.88$, $p = 0.06$)



Appendix G. Habit response (% sprawler and % clinger) at unimpaired restored and reference reaches. Sprawlers are associated with generalist taxa like Chironomidae and clingers are a associated with sensitive taxa like Heptageniidae.