

RESEARCH ARTICLE

Assessing stream channel restoration: the phased recovery framework

Jacob M. Dyste^{1,2} , H. Maurice Valett¹

Channel reconfiguration is one of the most common and costly stream restoration techniques, though its effectiveness is frequently questioned. Project monitoring often tracks changes in macroinvertebrate communities and other responses for a 5-year period. However, channel reconfiguration is a documented disturbance to stream ecosystems, suggesting that this form of restoration initiates succession over longer time frames than monitoring typically captures. To address the role of succession in stream ecosystem recovery, we developed the Phased Recovery Framework (PRF) which proposes benchmarks represented by predictable habitat structure and community composition based on project age. The PRF was tested across nine stream restoration projects in western Montana, ranging in age from 1 to 18 years, each paired with an established reference system. We tested for differences in channel form, habitat character, and macroinvertebrate community composition. While restoration established desired channel form, most biotic variables had not recovered to reference condition even for the oldest projects. Across all sites, phases of the PRF were poor predictors of response. However, analyzing responses to reconfiguration independently for sites in watersheds with unimpaired water quality versus those experiencing excessive nutrient enrichment (i.e. impaired sites) indicated that biological variables converged on reference conditions at unimpaired sites, but diverged across impaired reaches. These large-scale anthropogenic influences may play a stronger role in recovery than do changes to channel form and need to be incorporated into project design and success criteria. Assessment of the PRF suggests that short-term monitoring is not likely to produce reliable indicators of effectiveness without incorporating locally appropriate change associated with watershed impairment and successional progression.

Key words: disturbance, effectiveness, macroinvertebrates, monitoring, stream restoration, succession

Implications for Practice

- Channel reconfiguration successfully reestablishes geomorphic form upon completion, but stream habitat and communities undergo protracted successional change following restoration.
- Biological recovery should proceed through predictable phases that are associated with riparian canopy closure in streams of forested landscapes.
- Reconfiguration of channels in systems impaired at the watershed scale by exogenous influences like waste-water effluent may be driven to alternate states of degraded ecological integrity.
- Benchmarks for restoration success should anticipate successional change and recognize the potential for large-scale constraints on project design and success criteria.

Introduction

Restoration of degraded lotic ecosystems has outpaced efforts to understand its long-term ecological implications through monitoring and research (Bernhardt et al. 2005; Roni et al. 2005). Monitoring is particularly important when restoration involves “channel reconfiguration,” construction of an entirely new channel, an approach that severely disturbs both benthic

and riparian zones (Tullos et al. 2009). While channel reconfiguration, particularly the natural channel design approach (Rosgen 1996) has drawn criticism (Lave 2009), the method has proliferated because it provides practitioners with a systematic process for restoring the geomorphic form (Kondolf 2006). The method seeks to create the physical template for proper biological function through construction of geomorphically stable channel forms and complex habitat (Palmer et al. 2010). At the same time, strictly geomorphic approaches to monitoring limit understanding of ecological responses to channel reconfiguration (Kondolf & Micheli 1995; Sudduth et al. 2011).

Channel construction and floodplain grading creates stream reaches initially devoid of riparian plants, stream autotrophs, and macroinvertebrates, which are reestablished through natural

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¹Division of Biological Sciences, The University of Montana, Missoula, MT 59812, U.S.A.

²Address correspondence to J. M. Dyste, email jdyste@gmail.com

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recolonization or human intervention. While macroinvertebrates and stream autotrophs can recover to prerestoration levels rapidly (Pedersen et al. 2007), disturbance to the riparian zone can be long-lasting, altering aquatic-terrestrial linkages (Lake 2000) through elevated insolation and reduced allochthonous inputs for over a decade (Wallace et al. 1997). Hence, it is likely that restored reaches undergo major structural and functional changes over successional timescales following reconfiguration comparable to those observed following other forms of disturbance (e.g. Molles 1982; Valett et al. 2002). Therefore, temporal changes in stream-riparian corridor condition resulting from channel reconfiguration may act as endogenous drivers (i.e. forces generated from inside the stream-riparian corridor) of ecosystem recovery.

Odum (1969) called this temporal interaction of complex successional processes “ecosystem development” and identified phases representing how structure and function change as ecosystems move toward steady-state climax conditions. Using an ecosystem development-based perspective to understand restoration recovery is logical, but exogenous influences (i.e. change generated from outside the stream-riparian corridor) often complicate application of classical successional theory to lotic systems (Fisher 1990). By its very nature, restoration occurs within a broader anthropogenic setting and evidence shows that human influences such as urban and agricultural development at the watershed scale can exert stronger controls on ecological recovery than reach-scale efforts such as channel reconfiguration (Bernhardt & Palmer 2011).

Macroinvertebrates and the Phased Recovery Framework

Benthic macroinvertebrates are often monitored to evaluate stream response to restoration because they rapidly reflect changes in ecological conditions such as benthic stability, food resources, and stream temperature that act as local filters and determine community composition (Poff et al. 2006). Their responses to channel reconfiguration, however, appear to be highly variable (Miller et al. 2009). Studies that address macroinvertebrate response to restoration across stream systems generally fail to show substantial changes or improvement (Ernst et al. 2012). Channel reconfiguration, however, typically removes riparian vegetation, which increases gross primary production (Lamberti & Steinman 1997), and can significantly alter macroinvertebrate abundance and community composition for nearly two decades postrestoration (Behmer & Hawkins 1986; Stone & Wallace 1998). Because efforts to address the efficacy of channel reconfiguration through macroinvertebrate responses typically address only 5 to 7 years postrestoration, they likely capture only early phases of recovery, and fail to address changes driven by the successional serotinity of the riparian vegetation.

Embracing a successional perspective for ecosystem recovery following channel reconfiguration requires recognizing both endogenous and exogenous controls over development, but provides restoration practitioners with a framework for monitoring and assessment cognizant of the importance of temporal trajectories. Here, we introduce the Phased Recovery Framework

(PRF), an approach to understanding how channel reconfiguration alters ecosystem structure over time, and how interaction with watershed condition organizes temporal trajectories. We argue that stream ecosystems should recover in three progressive phases including (1) reorganization (0–2 years) when in-stream habitat is heavily altered, characterized by low organic matter (OM) standing stocks and depauperate macroinvertebrate communities with composition reflecting rapidly colonizing species; (2) developmental (2–15 years) during which macroinvertebrate richness and density reaches or exceeds reference levels, but community composition and OM standing stocks reflect continued disturbance as the result of increased insolation; and (3) mature (15+ years) when macroinvertebrate communities and OM standing stocks are comparable to reference conditions, indicating approach to a steady-state climax condition. Associating these time intervals with phases of recovery provides a framework for site categorization that is based on knowledge of the time course of disturbance and recovery in forested systems (Bormann & Likens 1979), benthic community resilience in headwater streams (Wallace 1990; Wallace et al. 1997; Kaylor & Warren 2017), and riparian canopy recovery in restored riparian sites in the western United States (Lennox et al. 2011).

We used the PRF to address two primary research questions: (1) How do relevant community filters and associated macroinvertebrate communities change over time? and (2) How do these recovery trajectories differ in the larger context of watershed condition? Results from these inquiries are used to discuss the role of time in structuring macroinvertebrate response to channel reconfiguration, how watershed condition influences characteristic targets and trajectories, and implications for project monitoring.

Methods

Study Sites

We addressed the PRF using reconfiguration projects across nine sites ranging from 1 to 18 years postrestoration with three sites occupying each predicted phase. Streams associated with sites (first–third-order) were within three watersheds in western Montana (Upper Clark Fork, UCF; Middle Clark Fork, MCF; Blackfoot, BF; Table 1, Fig. S1, Supporting Information). Restoration at all sites was consistent with natural channel design principles including grading to produce an active floodplain, cutting a new channel, placement of in-stream habitat structures, and planting native riparian species along newly formed channels.

We employed a reference-treatment experimental design, where each site consists of paired reference and restored reaches (Roni et al. 2005). When possible, reference reaches were minimally disturbed portions of the same stream system (average distance to reference reach = 1.2 km) recognized by restoration practitioners as representative of desired status for associated restored reaches. At all sites in the MCF and BF watersheds, we selected nearby reference reaches with little or no human-induced disturbance that experienced similar watershed

Table 1. Site characteristics including stream name, watershed identity, year of channel reconfiguration, proposed phase of recovery, character of initial disturbance prompting restoration activities, and presence/absence of water quality impairment.

Site	Name	Watershed (HUC 170102)	Year Restored	Recovery Phase	Initial Disturbance	Water Quality Impairment
1	Sauerkraut	Blackfoot	2015	Reorganization	Placer Mining	No
2	Silver Bow	Upper Clark Fork	2015	Reorganization	Copper Mining	Yes
3	Ninemile	Middle Clark Fork	2014	Reorganization	Placer Mining	No
4	Silver Bow	Upper Clark Fork	2011	Developmental	Copper Mining	Yes
5	Sauerkraut	Blackfoot	2009	Developmental	Placer Mining	No
6	Silver Bow	Upper Clark Fork	2007	Developmental	Copper Mining	Yes
7	Dunham	Blackfoot	2001	Mature	Logging	No
8	Silver Bow	Upper Clark Fork	2001	Mature	Copper Mining	Yes
9	Bear	Blackfoot	1998	Mature	Grazing	No

conditions. Broadly distributed mining within the UCF watershed required use of nearby less-disturbed (i.e. not subject to flooding and mine tailing deposition associated with sites on Silver Bow Creek) and geographically appropriate data for nearby (average distance from restored reach = 24.9 km) reference systems (French Creek, Mill Creek, German Gulch, Baggs Creek) acquired from the Montana Department of Environmental Quality's Stream Reference Project (Suplee et al. 2005).

Site Assessment and Sampling

During summer 2016, physical habitat, canopy cover, OM stocks, and physicochemical characteristics were assessed at restored and reference reaches of length equal to five bank-full widths (Bouwes et al. 2011) using 11 uniformly distributed transects. Macroinvertebrates were sampled during autumn 2016 prior to leaf fall.

Physical and Physicochemical Measures

Along each reach, we measured relative abundance of run, riffle, and pool habitat following Bisson et al. (2011) and used granulometry (Bevenger & King 1995) to characterize streambed sediment size distributions. A single longitudinal transect was randomly placed in a zigzag pattern and sampled at 1 m intervals by reaching down and picking up the first sediment particle encountered on the stream bottom. Proportion of observations belonging to a given size class was calculated based on a minimum of 200 data points for each stream. Streambed composition was characterized by median size class (D50) and relative abundance of fine sediment (<2 mm). Stream sinuosity was calculated as channel length divided by valley floor distance (Fitzpatrick et al. 1998).

We monitored dissolved oxygen (DO) and temperature (°C) at 15-minute intervals over diel time frames as surrogates for system metabolism and thermal regime, respectively. PME MiniDOT loggers (PME, Inc., Vista, CA, U.S.A.) placed in a riffle were retrieved after 24 hours and data used to calculate minimum, mean, maximum, and diel ranges for both physicochemical metrics. Diel characterizations were completed during a single sunny day in the summer growing season in 2016. We recognize that potential variation in diel signals across multiple days was not captured with this approach, but employed

it for expediency and to address anticipated differences associated with differing canopy cover under similar conditions (see below).

Canopy Cover and Organic Matter

Canopy cover (mean % over 11 transects) was determined for each reach using a convex densiometer (Ben Meadows, Model A, Janesville, WI, U.S.A.) and the line intersection method (Kaufmann & Robinson 1998) at each transect.

Benthic OM (BOM) was identified as autochthonous (filamentous algae, macrophyte, or bryophyte) or allochthonous material. BOM standing stocks were determined for all types as ash-free dry mass (AFDM, g/m²) and benthic algal standing crop represented as chlorophyll *a* (mg/m²). At each site, we isolated an area of streambed ($n = 3/\text{reach}$) using an open-ended cylinder (area = 0.22 m²), collected all enclosed coarse (>1 mm) OM, and stored samples on ice in the field. In the laboratory, OM samples were thawed and total wet weight recorded. Subsamples were taken from thawed samples for chlorophyll analysis and placed in 90% buffered acetone for 24 hours. Extractant was centrifuged for 10 minutes and chlorophyll measured using spectrophotometry (Jasco V-550) following McIntire et al. (1996). The remaining sample was weighed, dried (60°C) for 48 hours, and reweighed. A dry subsample was then weighed and combusted (4 hours, 550°C) to determine AFDM.

Macroinvertebrates

To address model predictions addressing macroinvertebrate composition and abundance, and the potential to apply the PRF approach, we employed sampling and taxonomic methods broadly employed by restoration practitioners. Macroinvertebrates were collected from three riffles within each reach using a Surber sampler (0.093 m², 500 μm net). For each riffle, the single composite sample consisted of all material collected after disturbing the substrate for 10 seconds, and cleaning all large cobbles into the net at three locations across the stream. Samples were placed in 95% ethanol in the field. Invertebrates were picked from each sample in the laboratory, and a Caton (1991) tray used to randomly subsample 300 organisms that were identified to family level. Remaining macroinvertebrates

were counted and data used to determine total number of individuals in the sample as a whole and within taxonomic groups by applying relative abundance results derived from the first 300 specimens.

Data were used to calculate diversity metrics including taxa richness (S), Shannon–Weiner index (H'), and percent abundance for Ephemeroptera, Plecoptera, and Trichoptera (EPT). Taxa traits were represented as functional feeding groups (FFG; shredder, collector-gatherer, scraper) based on Merritt and Cummins (1996) and Poff et al. (2006). Bray–Curtis dissimilarity indices (BCDI) were used to address compositional similarity between reaches based on abundance of distinct taxa (Beals 1984). We calculated BCDI between restored and reference reaches based on the mean abundance of each taxon. We characterized targeted dissimilarity by calculating BCDI values for within-reach communities at all restored reaches ($n = 3/\text{reach}$) and used the standard deviation around the grand mean of reference reaches to serve as the range of expected variability within any given system.

Data Analysis

We addressed applicability of the PRF and exogenous influences on trajectories using habitat and macroinvertebrate metrics as response variables. Data are presented as means and standard error (SE) or coefficient of variation (CV). Data distributions were checked for normality and variables transformed (i.e. natural log, inverse, or arcsin-square root) when necessary. When transformation failed to generate normal distributions, we employed nonparametric assessments (Wilcoxon-signed rank test, Spearman rank correlations). All statistical data analyses were conducted with SAS V9.2 (SAS Institute, Cary, NC, U.S.A.) with $\alpha = 0.05$.

We used a combination of absolute measures and response ratios (RRs), where RRs were used to normalize habitat and macroinvertebrate responses in restored reaches to respective reference systems. RRs were calculated following Benayas et al. (2009) where $RR = \ln\left(\frac{\text{restored}}{\text{reference}}\right)$ where “restored” and “reference” are means of absolute response measures. For absolute values of zero, $RR = \ln\left(\frac{\text{restored}+1}{\text{reference}+1}\right)$.

RRs were evaluated using two-tailed paired t tests to test for deviation from zero. Unpaired t tests were used to evaluate differences in absolute measures between reach types across all nine sites. Relationships among ratios or absolute measures were also assessed with linear regression or as Pearson product-moment correlations. Influence of recovery phase on habitat and macroinvertebrate response following restoration was addressed using one-way analysis of variance (ANOVA) with phase as the main factor (three levels: reorganization, developmental, mature). Following significant main effects, we used Student–Newman–Keuls and Fisher’s least significant difference post-hoc tests for multiple comparisons. We also used simple linear regression to address recovery as a continuous process with age as the independent variable and macroinvertebrate or habitat data as response (either absolute or RR) variables.

To address exogenous influences on recovery trajectory, we first designated reaches with reconfigured channels as

impaired (sites 2, 4, 6, and 8) or unimpaired (sites 1, 3, 5, 7, 9) in terms of water quality based on the EPA’s 303(d) listing of UCF sites for elevated nutrient loads (Montana DEQ 2016). The impaired sites were all located along Silver Bow Creek downstream of waste-water treatment discharge from Butte, Montana, where historic mining practices led to channel reconfiguration and continue to stress river biota (Montana DEQ 2014). We then used linear regression to address how project age influenced habitat and macroinvertebrate recovery at impaired and unimpaired sites independently. Slope comparison was accomplished by addressing differences between regression coefficients using dummy variables and the PROC REG Solutions option in SAS. We also used two-tailed, paired t tests to evaluate differences between groups (i.e. impaired vs. unimpaired). As informed by results of community assessment (i.e. Bray–Curtis indices), we allowed community composition to identify different groupings of phase and impairment and tested how other response variables differed among groups using one-way ANOVA and multiple comparisons.

Results

Channel Form, Physicochemistry, and Canopy Responses

RRs for channel form (i.e. sinuosity and % pool) and streambed composition (i.e. D50 and % fines) did not differ from zero (Fig. S2) among phases (ANOVA, $p > 0.05$) or across all sites together (paired t test, $p > 0.05$). Similarly, measures of physical habitat structure did not change with project age (linear regression, $p > 0.05$), indicating consistent geomorphic structure in restored and reference reaches regardless of age or phase.

Differences in water temperature and DO content were evident when comparing restored and reference reaches. Compared to reference sites, average water temperature was significantly greater (paired t test, $p = 0.030$) in restored reaches as were diel ranges in temperature (6.84 ± 0.64 vs. 3.34 ± 0.44 , Wilcoxon-signed rank test, $p = 0.012$, Table S1). Conversely, both average (paired t test, $p = 0.044$) and minimum (Wilcoxon-signed rank test, $p = 0.012$) DO concentrations were lower in restored reaches. Diel DO range, however, was greater in restored reaches (3.54 vs. 0.83 mg/L, $p = 0.026$ following ln-transformation of diel DO range, Table S1), and increased with average stream temperature (Fig. S3A) and declining sediment size (Fig. S3B,C). None of these measures of stream physical or chemical condition differed among phases (ANOVA, $p > 0.05$) or changed directionally with age (linear regression, $p > 0.05$) using absolute measures or RRs.

Canopy cover was less dense (paired t test, $p = 0.002$ following ln-transformation) along restored reaches (0.1–38.0%) compared to reference streams (59.02–81.3%, Fig. 1A). Cover during reorganization and developmental phases was lower (0.1–8.1%, ANOVA, $p = 0.050$) than observed for mature systems (8.7–38.0%). Percent canopy increased with age among restored reaches ($r^2 = 0.57$, $p = 0.020$; Fig. 1B) and as RR across paired reaches ($r^2 = 0.43$, $p = 0.050$; Fig. 1C). Across all sites and reaches, significant Spearman correlations indicated

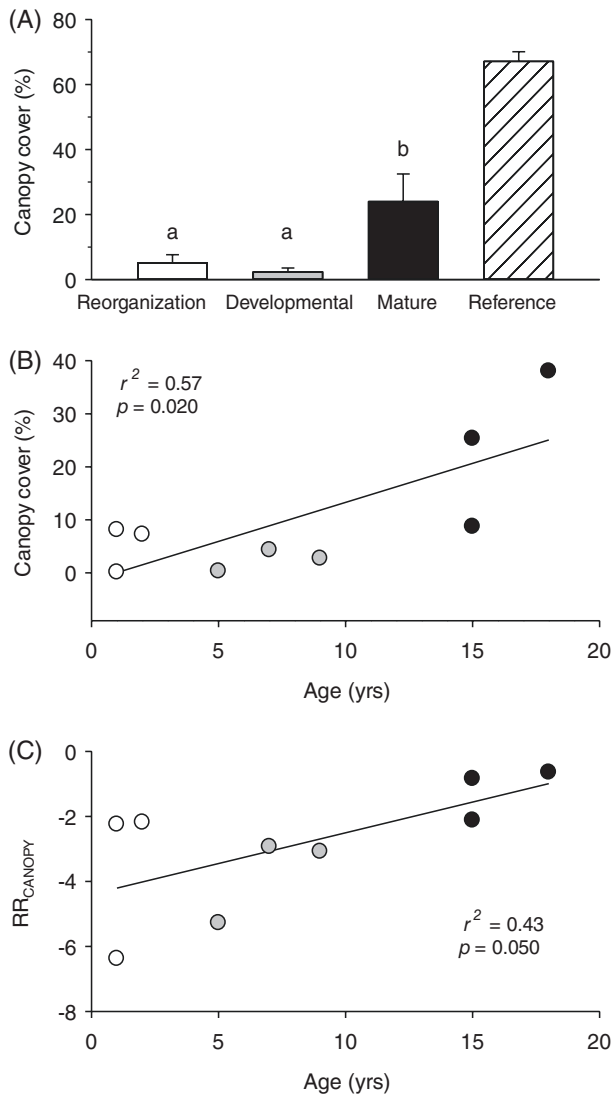


Figure 1. Percent canopy cover (a) in restored (filled bars) and reference (hatched bar) reaches. Data are means + SE ($n = 3/\text{phase}$; $n = 9$ for reference reaches). Bars with different letters are significantly different (Tukey's HSD, $\alpha = 0.05$). Relationship between project age and (B) riparian canopy cover (%) in restored reaches, and (C) canopy response ratios across all sites ($n = 9$). Fill colors represent restoration phases as: White = reorganization, gray = developmental, black = mature.

that minimum ($\rho = -0.53$, $p = 0.021$), average ($\rho = -0.65$, $p = 0.003$), maximum ($\rho = -0.65$, $p = 0.003$), and diel range for temperature ($\rho = -0.70$, $p = 0.001$) decreased with increasing canopy cover, while average DO concentration ($\rho = 0.46$, $p = 0.050$) increased (data not shown).

Benthic Organic Matter and Chlorophyll

BOM stock varied widely across reference and restored streams (Fig. S4). In restored reaches, BOM increased from 10.2 ± 7.2 g AFDM/m² during the reorganization phase, when it was similar to the mean value for reference reaches (11.56 ± 3.12), to 23.8 ± 13.2 and 37.8 ± 11.9 g AFDM/m²

during developmental and mature phases of restoration. High variability (CV = 56–120%) precluded any significant differences in stocks among phases (ANOVA, $p > 0.05$). As indicated by RRs, reaches in the reorganization phase had BOM stocks lower than associated reference reaches, while response ratios for BOM (RR_{BOM}) values greater than zero occurred during later phases (Fig. S4B). Ratios did not, however, differ significantly among phases (ANOVA, $p > 0.05$) and BOM was not related to age as absolute or ratio measures ($p > 0.05$).

Chlorophyll in reference streams averaged 15.1 ± 4.9 mg/m² (1.7 – 36.3 mg/m², data not shown). Among restored reaches, chlorophyll was more variable and extreme; maximum standing crop in restored reaches (143.8 mg/m²) was four times greater than the reference maximum. Chlorophyll response ratios (RR_{CHL}) differed significantly among phases (ANOVA, $p = 0.050$, Fig. 2A), increasing from negative values in the developmental phase to significantly greater positive values during developmental and mature phases. Similarly, chlorophyll was related to age among restored reaches ($r^2 = 0.60$, $p = 0.014$, Fig. 2B), as were RRs ($r = 0.73$, $p = 0.026$, data not shown). Chlorophyll and BOM were tightly and positively related ($r^2 = 0.84$, $p < 0.001$, Fig. 2C) as were their RRs ($r^2 = 0.75$, $p = 0.002$, Fig. 2D).

Macroinvertebrate Communities

Invertebrate abundance averaged $20,827 \pm 2,720$ ind/m² (CV = 39%) and $24,152 \pm 4,798$ ind/m² (CV = 60%) in reference and restored reaches, respectively (Table 2), and did not differ significantly between types (t test, $p > 0.05$). Densities during the reorganization phase were lower than observed for developmental or mature sites (Table 2), but no significant differences in densities occurred among phases (ANOVA, $p > 0.05$). Across all sites, S ($p = 0.005$) and H' ($p = 0.006$) were greater in reference reaches (paired t test, Table 2). Lower biotic diversity across restored reaches was reflected as negative RRs significantly less than zero for both S (-0.32 ± 0.09 , $p = 0.006$) and H' (-0.26 ± 0.07 , $p = 0.007$) across all sites (data not shown). No trends in diversity measurements were apparent among phases of recovery (ANOVA, $p > 0.05$, Table 2). Diversity in restored reaches did increase with DO content (S: $r = 0.76$, $p = 0.016$, Fig. S5A), but declined significantly with greater diel range in DO (H'; $r = -0.51$, $p = 0.030$, Fig. S5B).

The majority of invertebrates were found as EPT taxa in similar proportion (paired t test, $p > 0.05$) within reference ($64.3 \pm 4.0\%$) and restored ($59.3 \pm 8.3\%$) reaches across all sites (Table 2). EPT taxa were most abundant ($73.2 \pm 9.3\%$) during reorganization and declined in later phases of recovery, but differences were not statistically significant (ANOVA, $p > 0.05$). Shredders comprised $14.6 \pm 1.6\%$ of all individuals in reference systems (Table 2), a value significantly greater (Wilcoxon-signed rank test, $p = 0.015$) than in restored systems ($9.7 \pm 2.1\%$). Across phases, shredders were most abundant during reorganization, but highly variable (CV = 124%) and not significantly different among phases (ANOVA, $p > 0.05$). Collector-gatherers were the dominant FFG, representing more than 40% of all individuals in each site category (Table 2),

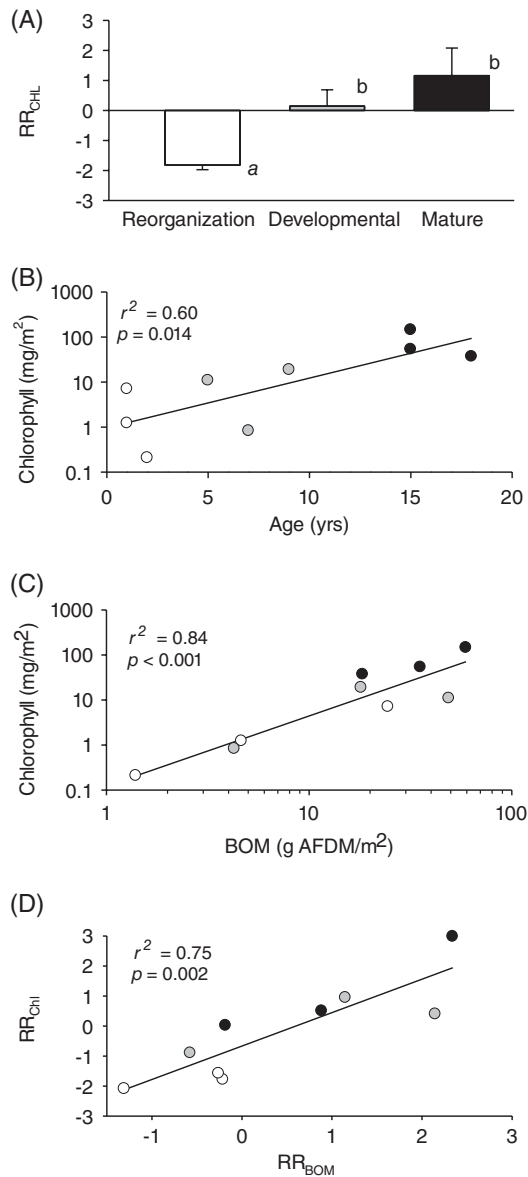


Figure 2. A) Response ratio for chlorophyll (RR_{CHL}) versus restoration phase. Bars with different letters are significantly different (Tukey's HSD, $\alpha = 0.05$). B) Chlorophyll (mg/m^2) in restored reaches versus (B) age and (C) standing stocks of BOM ($g\ AFDM/m^2$). D) RR_{CHL} versus RR_{BOM} across the nine paired study sites. Bars in (a) are means \pm SE ($n = 3$ /phase) where different letters indicate means that are significantly different (ANOVA, Fisher's least significant difference, $p < 0.05$). Fill colors are as in Figure 1.

and abundance increased with invertebrate density ($r = 0.66$, $p = 0.050$, Fig. S5C) to as much as 73% of all individuals in the restored reach with greatest invertebrate density. Collector-gatherer abundance did not differ significantly with reach type (paired t test, $p > 0.05$) or among phases (ANOVA, $p > 0.05$). Scrapers were less abundant than collector-gatherers, but generally more common than shredders (Table 2); percentage of individuals categorized as scrapers in reference ($20.6 \pm 3.9\%$) and restored ($16.6 \pm 4.2\%$) reaches (Table 2) did

not differ with reach type (paired t test, $p > 0.05$) or among recovery phases (ANOVA, $p > 0.05$).

The dissimilarity in community composition occurring between pairs of samples within reference streams (0.15 ± 0.02 , Table 2) was lower (t test, $p < 0.001$) than found among samples paired from restored and reference reaches (i.e. BCDI = 0.54 ± 0.06 , Table 2) and those values were of equivalent magnitude across all phases of restoration (ANOVA, $p > 0.05$).

Recovery Trends for Impaired and Unimpaired Sites

Physicochemical and biological characteristics of restored reaches differed between impaired and unimpaired sites. Average temperature (18.0 ± 0.21 vs. $10.87 \pm 0.80^\circ C$, $p < 0.0001$), maximum temperature (22.6 ± 0.35 vs. $14.13 \pm 0.99^\circ C$, $p < 0.0001$), and diel range in water temperature (8.49 ± 0.40 vs. $5.52 \pm 0.65^\circ C$, $p = 0.008$) were significantly greater (t test) in impaired sites than in unimpaired sites. Moreover, RRs for these metrics in impaired sites were positive (0.54–1.1, Fig. 3A) and significantly greater than zero (paired t test, $p < 0.001$), while ratios in unimpaired sites were significantly lower (t test, $p < 0.0001$) and not different from zero (Fig. 3A). Diel range in DO was 7-fold greater in impaired reaches compared to unimpaired systems (Fig. 3B). The RR for diel DO range (RR_{DORNG}) in impaired reaches averaged 2.30 ± 0.15 and was significantly greater (t test, $p < 0.001$) than in unimpaired reaches (-0.043 ± 0.26) where the mean value was not different from zero (Fig. 3B).

Measures of benthic structure also differed between impaired and unimpaired reaches. Streambeds of impaired reaches contained a greater abundance of fine sediments (12.2 ± 4.1 vs. $0.8 \pm 0.3\%$, t test, $p = 0.016$) and a smaller D50 (paired t test, $p = 0.007$). RR_{BOM} and RR_{CHL} were both positive in impaired reaches and negative in unimpaired reaches (Fig. 3B), but ratios were variable and not significantly different from zero or each other ($p > 0.05$). These differences in benthic chlorophyll content between restored and reference reaches (i.e. RR_{CHL}), however, changed with time in both unimpaired and impaired sites, but in very different ways. In both types of sites, RR_{CHL} was negative immediately following restoration and in unimpaired reaches RR_{CHL} increased with age and converged on zero in the oldest systems ($r^2 = 0.86$, $p = 0.022$, Fig. 4A). RR_{CHL} in impaired sites also increased with age ($r^2 = 0.86$, $p = 0.022$), but at a significantly greater rate (comparison of slopes, $p = 0.005$) compared to unimpaired systems, and ratios in older impaired sites reached values as great as 3.0, reflecting the much greater algal abundance in restored reaches with impaired water quality.

Impairment manifested as differing invertebrate diversity and composition, evident across phases and increasing in magnitude with project age. Percent EPT taxa declined with increasing chlorophyll in impaired reaches ($\rho = -1.0$, $p < 0.0001$, Spearman correlation) while no relationship existed between these variables in unimpaired restored reaches (data not shown). Negative RRs for taxon richness ($p = 0.029$) and Shannon–Wiener index ($p = 0.018$) were different from zero (paired t test) in impaired reaches but not in unimpaired sites

Table 2. Macroinvertebrate density and community composition in restored and reference reaches ($n=9$) and among phases of post-configuration recovery ($n=3/\text{phase}$). Data are means \pm SE. Means with unique superscripts indicated significant differences ($p < 0.05$) between reference and restored reaches. No significant differences were observed for any metric among phases of restoration.

	Macroinvertebrate Density (ind./m ²)	Taxon Richness (S)	Shannon–Wiener Index (H')	EPT (%)	Shredder Abundance (%)	Collector–Gatherer Abundance (%)	Scraper Abundance (%)	Bray–Curtis Dissimilarity Index
Reference	20,827 \pm 2,720	18.0 ^A \pm 0.3	2.22 ^A \pm 0.05	64.3 \pm 4.0	14.6 ^A \pm 1.6	46.5 \pm 3.7	20.6 \pm 3.9	0.15 ^A \pm 0.02
Restored	24,152 \pm 4,798	13.6 ^B \pm 1.4	1.73 ^B \pm 0.11	59.3 \pm 8.3	9.7 ^B \pm 2.1	45.8 \pm 3.7	16.6 \pm 4.2	0.54 ^B \pm 0.06
Phase								
Reorganization	12,000 \pm 3,395	16.1 \pm 2.2	1.93 \pm 0.11	73.2 \pm 9.3	19.4 \pm 13.9	41.4 \pm 11.3	19.6 \pm 8.0	0.52 \pm 0.04
Developmental	36,075 \pm 8,332	10.6 \pm 1.3	1.42 \pm 0.43	61.9 \pm 9.7	3.3 \pm 1.8	47.2 \pm 14.6	11.5 \pm 2.4	0.62 \pm 0.10
Mature	24,381 \pm 7,095	14.1 \pm 3.0	1.85 \pm 0.22	43.0 \pm 20.3	6.5 \pm 3.0	48.6 \pm 1.8	18.7 \pm 6.5	0.49 \pm 0.15

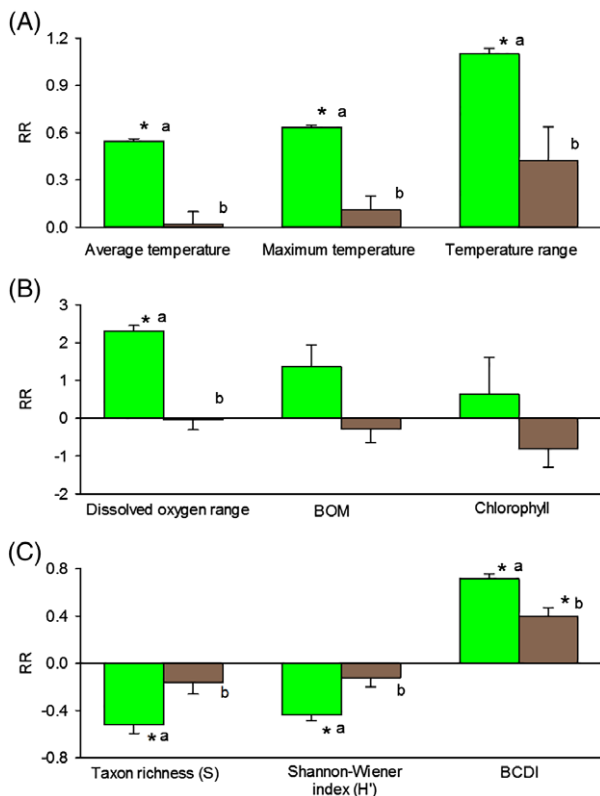


Figure 3. Mean (\pm SE) RRs in impaired (green bars) and unimpaired (brown bars) for (a) average, maximum, and diel range in water temperature, (B) diel range in dissolved oxygen, BOM, and chlorophyll, and (C) taxon richness (S), Shannon–Wiener index (H'), and BCDI (presented as absolute calculations). For each response ratio, an asterisk indicates a mean value significantly different from zero (paired t test, $p < 0.05$). For each response variable, bars with different letters are significantly different (t test, $p < 0.05$).

($p > 0.05$, Fig. 3C). Similarly, mean BCDI generated from reference–restored comparisons among impaired sites was greater than for unimpaired sites (t test, $p = 0.010$, Fig. 3C), indicating that restored invertebrate communities differed more substantially from associated reference systems when sites were impaired than they did in unimpaired systems.

Temporal trajectories of change in invertebrate composition differed between impaired and unimpaired sites. RRs for

shredder abundance in unimpaired systems increased significantly with age ($r^2 = 0.78$, $p = 0.005$, Fig. 4B) from initially negative values to nearly zero, illustrating convergence with chosen reference streams in the older systems. Among impaired sites, however, an initially elevated ratio in the 1-year-old restored site (0.92), declined dramatically and remained low (-2.3 to -2.8) despite increased project age (Fig. 4B). As broadly assessed by BCDI, macroinvertebrate community composition in the restored reaches of unimpaired systems converged on those of reference sites as projects aged ($r^2 = 0.56$, $p < 0.0001$, Fig. 4C). In contrast, BCDI values increased with age in impaired sites ($r^2 = 0.40$, $p < 0.0001$, Fig. 4C) with a slope significantly greater than in unimpaired sites (comparison of slopes, $p = 0.012$) illustrating progressive divergence from reference community structure during recovery. Differences in community composition were not related to phase when BCDI measures were assessed for impaired and unimpaired reaches together (ANOVA, $p > 0.05$, Fig. 4D).

Close association of BCDI scores illustrated the combined influences of phase and impairment as three distinct groupings (black ellipses in Fig. 4D), including reorganization, and continuing development in either impaired or unimpaired states. Using these site associations, BCDI, RR_{BOM} , RR_{CHL} , and RR_{DORNG} differed significantly (ANOVA, $p < 0.05$) among groups (Fig. S6). For all response variables, initial differences from reference conditions occurred during reorganization (i.e. RRs different from zero, paired t test, $p < 0.05$), and values converged on reference conditions in developing unimpaired systems (i.e. RRs not different from zero, paired t test, $p > 0.05$), but continued to diverge in impaired reaches responding to channel reconfiguration.

Discussion

Channel reconfiguration successfully generated desired geomorphic conditions, but most biological factors failed to reach reference conditions at the majority of restored reaches despite as much as 18 years of recovery. Although canopy extent increased as projects aged, cover was less than reference condition along restored reaches and riparian development provided less than half of desired cover at the oldest sites. Benthic chlorophyll and OM in restored and reference reaches

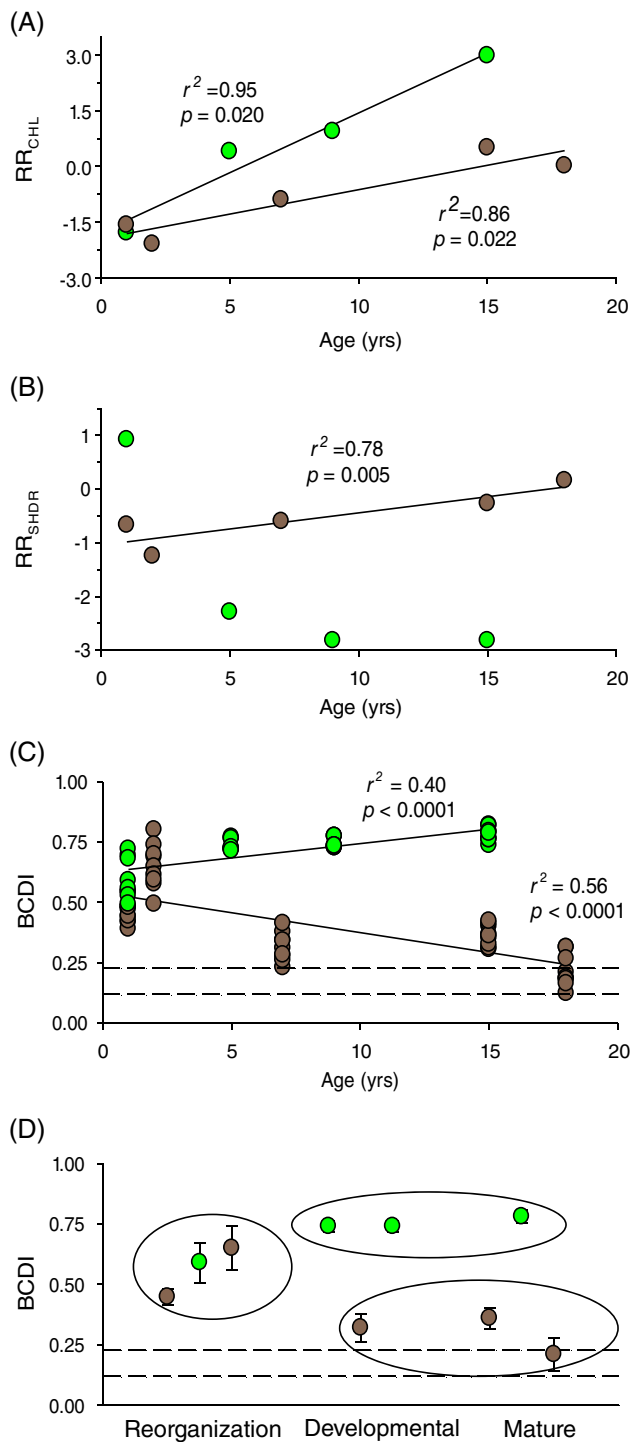


Figure 4. Characteristics of system recovery in impaired (green circles) and unimpaired (brown circles) sites. A) Chlorophyll response ratio (RR_{CHL}) and B) shredder response ratio (RR_{SHDR}) versus age. BCDI versus C) age and D) putative recovery phase. Dashed lines in C and D indicate upper and lower bounds for BCDI within reference reaches. Data in A, B, and D are site means (\pm SE in D). Panel C shows all possible paired assessments between restored and reference systems. Ellipses in D identify significantly different groups based on ANOVA assessment of community dissimilarity.

differed in ways that reflected project age and recovery phase, as well as exogenous influences generating impairment. Similarly, reference and restored macroinvertebrate community composition appeared to be fully equivalent only for the oldest project executed in an unimpaired system. Discrepancy between physical habitat and biological responses emphasizes the need for judicious assessment of how stream ecosystems respond and recover following channel reconfiguration projects.

Community Recovery Patterns: Impaired and Unimpaired Reaches

Following channel reconfiguration, macroinvertebrate taxa richness and diversity were lower in restored reaches compared to reference systems, suggesting that restoration effectively served as a disturbance from which most biotic components have yet to recover. While our study lacks pretreatment data precluding definitive assessment of the influence that reconfiguration has played in the structure and function of restored reaches, others have considered channel reconfiguration to be a disturbance to stream communities (Laasonen et al. 1998; Korsu 2004), and have documented decline in macroinvertebrate diversity (Louhi et al. 2011) and flourishing of broadly tolerant taxa following restoration (Tullos et al. 2009). Among restored reaches, the positive relationship between richness and average DO content, and the decline in diversity associated with greater diel oxygen range, suggest that minimum DO concentrations may play an important role in shaping community composition (Connolly et al. 2004). Greater range in DO, however, also co-occurred with higher temperatures and increased abundance of fine sediments, both features shown to influence invertebrate community composition following restoration (Parkyn et al. 2003; Tullos et al. 2009).

In our study, full recovery of macroinvertebrate community composition was evident only for the oldest, nonimpaired site where canopy cover was maximal, and water temperature, chlorophyll, and BOM comparable to those found in the paired reference system. While most measures of community composition did not differ among the putative phases of recovery for all sites combined, trajectories for community composition in unimpaired restored reaches illustrated directional recovery and convergence on reference conditions with time. Further evidence for recovery in unimpaired systems included changes in FFG abundance with age; while initially depressed following reconfiguration, shredder abundances recovered in older systems to proportions equal to those in reference streams. Others have documented decreased shredder abundance compared to reference conditions following channel reconfiguration (Laasonen et al. 1998; Rios-Touma et al. 2014). Rios-Touma et al. (2014) showed little recovery in shredder taxa 4 years after restoration of a Pacific Northwest headwater stream, conditions predicted by the PRF for systems transitioning from reorganization to developmental phases of recovery. In our streams, enhanced shredder recovery was more evident in older systems of unimpaired watersheds.

Recovery conditions and trajectories in impaired sites were very different from those observed in unimpaired reaches. RRs

indicated that diversity was depressed in impaired reaches compared to reference systems. Moreover, communities in impaired sites became more dissimilar from reference sites with increasing project age. These data suggest that lower diversity observed across all restored reaches resulted primarily from decline within impaired sites. Thus, community change in impaired reaches was characterized by progressive divergence with successional time, illustrating the overwhelming influences of impairment on the character of biotic recovery.

Environmental Filters and Community Recovery

Broad environmental changes such as those generated by channel reconfiguration can impose three selective filters to macroinvertebrate communities, including thermal regime, food resources, and habitat stability (Poff et al. 2006; Tullis et al. 2009). Observed changes in physicochemical conditions, organic materials, and invertebrate communities suggest potential influence from at least the first two of these filters during postrestoration recovery.

Temperatures in restored reaches were greater than in reference reaches, a feature typically related to reduced canopy cover. Parkyn et al. (2003) attributed lack of community recovery in New Zealand stream systems bordered by restored riparian buffers to elevated temperatures, reflecting incomplete canopy closure even after two decades of riparian growth. Riparian recovery following channel reconfiguration typically involves planting and growth of new vegetation, and variation in canopy development may result from natural processes, riparian planting failure, or relic impacts from channel construction. Whether maximum canopy closure observed in our study was responsible for the observed thermal similarity remains unknown, but it did co-occur with convergence of macroinvertebrate composition in restored and reference reaches.

Greatly reduced canopies along restored reaches increased both temperature and insolation resulting in warmer streams that generated greater diel change in DO. Diel range in DO has been used to address water quality and trophic status of lakes and rivers because it reflects ecosystem metabolic character (Bernhardt et al. 2017). In our study, strong links between DO range and stream temperature suggest enhanced autotrophy in warmer, more open, restored reaches that contained finer sediments and generally greater amounts of BOM. In general, allochthonous OM was sparse in restored reaches, but recovery of shredders in unimpaired reaches suggests increased autumnal inputs of allochthonous material in older systems despite relatively low canopy closure. Across all restored sites, however, chlorophyll standing crops increased with age, were closely related to BOM abundance, and were primarily represented by autochthonous filamentous green algae as compared to less abundant epilithic biofilms or bryophyte patches characteristic of reference systems. These results suggest that primary production and autochthonous BOM became an important aspect of stream recovery both in terms of benthic habitat structure and food availability.

Chlorophyll and BOM differed among phases in ways predicted by the PRF; lower abundance compared to reference

sites during the reorganization phase reflected benthic disturbance, followed by recovery to abundance levels comparable to reference systems during the developmental phase. However, contrary to the PRF, restored systems in the mature phase of response supported chlorophyll and BOM in excess of those in reference systems. These patterns, however, appear to be driven by nutrient enrichment associated with impaired watershed conditions. When viewed independently, chlorophyll standing crops in unimpaired reaches did converge on reference values as predicted by the PRF; in contrast, autotrophic biomass progressively increased in impaired reaches.

Enhanced abundance of filamentous green algae characterizes nutrient impairment in streams generally (Dodds 1991), including those in Montana (Suplee et al. 2012), and specifically in restored reaches in our study designated as impaired. In contrast, autochthonous BOM in reference streams was mostly diatoms and bryophytes. Most filamentous algae is relatively poor food for macroinvertebrates compared to diatoms (Power et al. 2015) and bryophytes trap fine particulate OM that serves as an important food resource in streams where mosses are abundant (Bowden and Stream Bryophyte Group 1999). Invertebrate community composition responded to algal proliferation as EPT abundance declined with increasing chlorophyll in impaired reaches, but not in unimpaired systems.

Endogenous and Exogenous Influences and Modifications to the PRF

When addressed across all nine sites, age-based phases were not robust predictors of recovery following channel reconfiguration. We based recovery phases on project age because restoration activities are organized by time-based business practices (Lavendel 2002). However, the time range over which transition between phases occurs is likely broad, and reflective of the time course of canopy development (Parkyn et al. 2003; Warren et al. 2016). Sites we assigned to the mature phase had the largest variation in canopy cover and varied greatly in ecosystem condition. Predicting cover thresholds associated with different phases of the PRF would inform monitoring practices. Canopy development, however, is a function of stream size, growing season length, riparian maintenance, and soil compaction (Sweeney et al. 2002) and likely restricts specific temporal benchmarks for the PRF to settings with comparable climate, biota, and restoration practices.

When application of the PRF is restricted to restored sites without impairment, biotic structure responded to reconfiguration and changed with time as predicted by the PRF. Congruence with the PRF under these conditions suggests an endogenous capacity to recover following channel reconfiguration that relies on successional processes that ultimately result in self-sustaining systems (Walker et al. 2007). A great deal of work has addressed operational and theoretical linkage between restoration and succession (Palmer et al. 1997; Walker et al. 2007; Christensen 2014) such that restoration has been frequently referred to as the manipulation of succession (Walker et al. 2007). Others have emphasized that the scale of restoration projects leaves them vulnerable to exogenous disturbances,

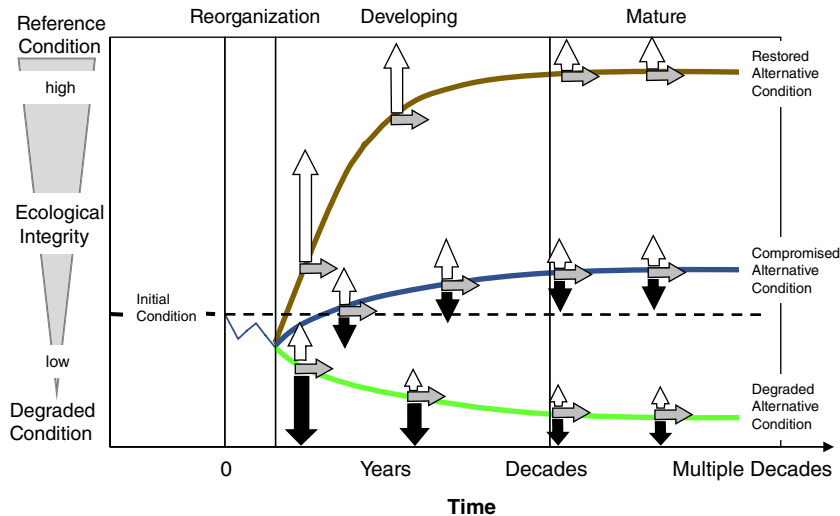


Figure 5. Schematic representation of the Phased Recovery Framework addressing changes in ecological metrics following channel reconfiguration (time = 0) design. Reconfiguration intends to direct lotic ecosystems from an initial degraded condition towards a higher degree of integrity more characteristic of reference systems, but is generally recognized as significant disturbance to lotic communities. Recovery trajectories are influenced by time (gray arrows), endogenous successional changes coupling terrestrial and aquatic form and function (white arrows), and the influence of exogenous forces responsible for impairment (black arrows). Systems experience a period of “reorganization” characterized by decreased integrity reflecting acute influences of channel disturbance, followed by a developmental phase where interaction between endogenous change and external impairment over the course of years determines the integrity of mature systems. Different trajectories (green, blue, and brown lines) therefore occur. Only when sources of external impairment are diminished or removed can systems realistically change from one trajectory to another.

materials, and energy that cross system boundaries (Parker 1997; Christensen 2014). This is particularly relevant for lotic ecosystems where external influences arrive as a consequence of downstream flow and terrestrial-aquatic interaction (Palmer et al. 1997). In our study, the exogenous influence of nutrients from waste-water effluent promoted divergence from desired reference status. As a result, recovery in restored reaches compromised by impairment differed from both reference streams and systems responding to restoration in the absence of impairment, as suggested by groups of sites derived from macroinvertebrate composition.

Accordingly, the PRF needs to incorporate exogenous influences of impairment such that that change following reconfiguration reflects the combined influences of time, endogenous successional processes, and exogenous influence (Fig. 5). Such impairment may direct systems to alternative stable states (Heffernan 2008) of lower ecological integrity depending on the degree of impairment (Müller et al. 2006) reflecting ecosystem retrogression. While typically applied to long-term declines in ecological integrity associated with prolonged successional sere (Peltzer et al. 2010), the term retrogression has also been used to describe developmental trajectories that lead to decline associated with nutrient enrichment (Müller et al. 2006) including application to successional restoration of fens in northern Europe (Schrautzer et al. 2007). Thus, the PRF incorporates a body of literature showing that catchment-level impairment often has stronger influence on stream structure and function than does reach-scale restoration (Bernhardt & Palmer 2011; Louhi et al. 2011) despite decades of recovery.

Monitoring and the PRF

This study serves the purpose of presenting the PRF as a guiding framework for assessment and provides an initial test of its application. Applying the PRF to assessment of channel reconfiguration comes with a number of inherent challenges principally related to the identity and number of sites that may serve to represent streams in different phases of restoration, availability of reference sites, and statistically appropriate distribution of larger-scale phenomena such as water quality impairment. Our study was limited by unequal distribution of impairment among streams and basins, restricting our ability to employ replicates and independently address differences among phases for impaired and unimpaired systems. While we are confident that water quality impairment driven by nutrient enrichment influenced the nature of recovery for our impaired sites, it is also possible that other unmeasured features may have influenced the recovery trajectory. As a result of these limitations, we addressed the time frame of recovery both as a discrete (i.e. phase) and continuous (i.e. age) variable. While phases can be considered snapshots along a temporally continuous sere (e.g. Bormann & Likens 1979), the efficacy of their use in terms of restoration assessment remains to be seen. We hope that by grounding the concepts presented here within decades of ecological research, the ecological restoration community will have a more practical way to evaluate and discuss how and over what timescales channel reconfiguration projects are monitored and assessed.

The majority of monitoring and research efforts likely capture the developmental phase of recovery. Our results suggest that the PRF provides a construct for evaluating stream restoration that recognizes directional ecosystem change that will

accompany channel reconfiguration and is sensitive to exogenous influences. As human populations continue to expand and influence ecosystems, the distinction between impaired and unimpaired ecosystems will likely become more ambiguous. This ambiguity makes setting appropriate goals and devising monitoring schemes more challenging and increasingly critical. Key features of planning should recognize that:

- After 1 to 2 years, macroinvertebrate communities likely exhibit disturbance characteristics due to changes associated with reduced canopy cover.
- Monitoring results for 1 to 2 years after project implementation are likely to be unreliable as longer-term assessments of recovery.
- Monitoring plans need to account for the fact that channel reconfiguration projects likely take decades to reach reference condition as the result of ecosystem development.
- Anthropogenic influences at larger spatial scales often play a stronger role in ecosystem recovery than channel reconfiguration and need to be incorporated into project design and success criteria.

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Supporting Information

The following information may be found in the online version of this article:

Table S1. Average, maximum, and minimum temperature (°C) and dissolved oxygen (mg/L) in reference and restored reaches across sites.

Figure S1. Map of study streams in northwestern Montana showing location of major watersheds, paired reference and restored study reaches (1–9), and associated geographic information.

Figure S2. Mean (\pm 1SE) response ratio (RR) of physical habitat metrics across phases (filled bars, $n = 3$ /phase) and all sites (open bars, $n = 9$).

Figure S3. Diel dissolved oxygen range (maximum – minimum, mg/L) versus (A) average water temperature, (B) % fine sediment, and (C) median substrate size (D50) among restored reaches.

Figure S4. (A) Benthic organic matter (BOM) standing stock in restored reaches by phase and in reference reaches (striped). Data are means \pm SE ($n = 3$ /phase; $n = 9$ for reference reaches). (B) Response ratio for benthic organic matter (RR_{BOM}) by phase.

Figure S5. Invertebrate diversity and composition across restored sites. (A) Taxon richness versus average dissolved oxygen concentration (mg/L); (B) Shannon-Weiner Index (H') versus dissolved oxygen range (maximum – minimum, mg/L); (C) Abundance (%) of collector-gatherers versus invertebrate density (ind/m^2).

Figure S6. (A) Bray–Curtis dissimilarity index (BCDI), and response ratios for (B) benthic organic matter (RR_{BOM}), (C) chlorophyll (RR_{CHL}), and (D) dissolved oxygen range (RR_{DORNG}) versus phases of succession suggested by community dissimilarity assessment (see Figure 5).