UNIVERSITY OF CALIFORNIA
SANTA CRUZ

WHERE TO RESTORE?
INFLUENCE OF SURROUNDINGS ON STREAM RESTORATION OUTCOMES

A dissertation submitted in partial satisfaction
of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

ENVIRONMENTAL STUDIES

by

Bronwen Stanford

September 2018

The dissertation of Bronwen Stanford
is approved:

Professor Erika Zavaleta, chair

Professor Karen Holl

Professor Adam Millard-Ball

David Herbst, Ph.D.

Lori Kletzer
Vice Provost and Dean of Graduate Studies

Table of contents

List of figures and tables

Abstract

Where to restore? Influence of surroundings on stream restoration outcomes

Bronwen Stanford

Streams and rivers are both highly important for biodiversity and ecosystem function and highly sensitive to human land use change. Efforts to restore and enhance stream condition have created a booming industry; however, many restoration projects fail to achieve recovery. This dissertation explores how surrounding conditions can help explain variation in stream restoration outcomes. I assess the influence of site surroundings and land cover on stream restoration and recovery at local, watershed, and regional scales. In Chapter 1, I present results of an observational field study monitoring stream recovery following riparian restoration in rangelands in Marin County, CA, and show that greater linear lengths of riparian trees can partially buffer stream condition from grazing stresses. In Chapter 2, I present a global meta-analysis combined with land cover analysis to assess stream recovery following a disturbance to water quality. I show that most streams fail to recover to baseline conditions within the study period, and that streams with more upstream natural land cover may experience lower recovery completeness than streams in more human-dominated watersheds. Finally, in Chapter 3, I analyze the spatial distribution of stream restoration and management sites over the past 30 years on the California Central Coast using both sociopolitical and biophysical indicators. I find more restoration near more white, wealthy, and educated human communities, suggesting that current practice could better match restoration effort and need.

Together these findings reinforce calls for watershed planning to prioritize overlooked opportunities and to position restoration projects to achieve the greatest regional benefits.

Acknowledgments

I would first like to thank my committee members Erika Zavaleta, Adam Millard-Ball, Karen Holl, and David Herbst, who provided invaluable feedback, including help with editing, interpreting, and experimental design of this dissertation. I learned a great deal from each of them. I would also like to acknowledge helpful input from many colleagues including the Zavaleta lab (Sarah Skikne, Paulo Quadri, Flavia Oliveira, Elissa Olimpi, Kate Ennis, Justin Cummings, Blair McLaughlin) and the Holl lab (Alicia Calle, Josie Lesage, Andy Kulikowski, Elizabeth Howard, Leighton Reid, Tara Cornelisse) who edited and helped interpret these projects. I also benefited from the kind input of many other UCSC professors, including particularly Marm Kilpatrick and Stacy Philpott for their advice as I muddled through statistics, and Noah Finnegan for helping me think about process in streams.

For Chapter 1 I am indebted to the many landowners that welcomed me onto their properties and supported this research. In particular, I would like to thank Sally and Mike Gale, who not only allowed me to pilot the study on their ranch, but also let me stay on their property and connected me with many other landowners. This project would not have been a success without their kindness and enthusiasm. Russ Dedrick and Betsy Gardiner also generously allowed me to stay with them during fieldwork.

Chapter 1 would not have been possible without the assistance of both lab interns to help sort and identify insect samples and field assistants (and the funding to hire them provided by the EPA STAR and Heller Foundation Grants). I would like to thank UCSC undergraduates Teague Corning, Jessica Zygutis, Daniel Gomez, Jessica

Diaz, Edson Perez, Samantha Cano, Sarah Roy, and Cheryl Cary for their help in the lab and with data extraction, as well as Silvia Barbosa Rodrigues from Brazil. In particular, Teague's excitement, professionalism, attention to detail, and investment of 12 hours/week for an entire year helped me finish the sorting without losing my mind. As field assistants, Jessica Beatty and Liam Zarri cheerfully spent weeks forging on through blackberry brambles, herds of cows, and sometimes-leaking waders to collect the field data with me. I received help with methods development for this project from Bruce Medhurst, Sarah Pierce, April Robinson, Mike Peterson, Vince Resh, Michael Bogan, Matt Cover, Alison Roy, John Parodi, Cynthia Kern, Chuck Hawkins, Mike Lennox, Rafi Mazor, Eric Smith, Rob Leidy, and Allison Pfeiffer, most of whom either took me in the field with them or helped me assess my data and samples. The Marin Agricultural Land Trust and Marin County Resource Conservation District also helped with the development of this project. Mike Peterson and Dave Herbst provided invaluable help with invertebrate identification. Rob Franks helped me analyze FPOM and chlorophyll samples, and the Norris Center for Natural History allowed me to borrow many key pieces of equipment for fieldwork.

I was privileged to take part in a SESYNC-funded working group led by Holly Jones and David Moreno-Mateos that guided the early stages of Chapter 2. Involvement in this group enriched my experience, provided a template that launched the project, and also helped me learn how to complete a successful meta-analysis.

Cara Clark of the Central Coast Wetlands Group provided a database that formed the start of the restoration projects database that I analyze in Chapter 3.

Conversations with Karen Worcester, Carolyn Doehring, Mary Hamilton, Katie McNeill, Susan Keydel, Juliet Christian-Smith, Ross Clark, Kevin O'Connor, and Kristen Kittleson helped guide the development and interpretation of Chapter 3.

For the past four years of this dissertation, regular meetings with the Brain Trust, aka Elissa Olimpi and Sarah Skikne, kept me afloat. They provided moral support and endless reality checks as well as more practical help prioritizing, strategizing, and editing the mountain of work that this entailed. The chance to learn about and think through problems as a group also substantially reduced the isolation that can often be part of a dissertation.

Finally, I would like to thank my family, friends, cohort 2012 (Marcela Cely Santos, Robin Lovell, Estelí Jiménez Soto, Hamutahl Cohen, Paulo Quadri, Kate Ennis, Amber Kuss, and Jessica Hammond), Robin Lovell (aka best officemate), and particularly Chris Arends for all of the support throughout this entire process.

This dissertation was funded by a STAR Fellowship [Assistance Agreement no. FP-91762101-0] awarded by the U.S. Environmental Protection Agency, as well as by smaller grants including the Wells Fargo Coastal Sustainability Fellowship, Heller Foundation Grants, the Northern California Association of Phi Beta Kappa, Northern California Botanists, ENVS Department grants, and the UCSC Chancellor's Fellowship.

The text of this dissertation includes reprints of the following previously published material:

Stanford, B., Zavaleta, E., Millard-Ball, A., 2018. Where and why does restoration happen? Ecological and sociopolitical influences on stream restoration in coastal California. Biological Conservation 221, 219–227. doi:10.1016/j.biocon.2018.03.016

Stanford, B., Jones, H., Zavaleta, E. 2018. Meta-analysis of the effects of upstream land cover on stream recovery. Conservation Biology

The co-authors listed in these publications directed and supervised the research which forms the basis for these dissertation chapters. Erika Zavaleta and Holly Jones helped to design, interpret, and edit Chapter 2. Adam Millard-Ball and Erika Zavaleta helped to design, interpret, and edit Chapter 3. Bronwen Stanford designed all studies, collected all data, performed all analysis, and wrote all chapters.

Chapter 1 has not yet been published as of dissertation submission, but coauthors are Erika Zavaleta, Karen Holl, and David Herbst. The co-authors directed and supervised the research which forms the basis for this chapter, and helped to design, interpret, and edit the text of this chapter.

Chapter 1. Rangeland stream response to riparian corridor length

Abstract

Riparian vegetation is commonly conserved and restored in working landscapes to improve in-stream condition and water quality, with mixed results. Small-scale restoration techniques such as revegetation may benefit from increases in size, given evidence that matching the scale of the intervention and stressor can improve outcomes. We use a replicated field study to evaluate whether increasing the linear length of narrow riparian tree corridors results in measurable improvements to in-stream condition. We collected data at 41 sites with varying upstream tree cover nested in 13 rangeland streams in coastal northern California, USA. We focus on differences in benthic macroinvertebrates and their food resources, water quality, and channel form. Longer riparian corridors resulted in lower water temperatures and less fine substrate, as well as higher percentages of intolerant invertebrates. Fine particulate organic matter, invertebrate tolerance value, and invertebrate richness did not differ by tree corridor length. The covariates we included for stream effect (length of upstream gap in riparian cover, soil type) explained some of the variation in particulate organic matter and invertebrate responses: in particular, soil type had a strong effect on invertebrate community composition. Our results suggest that two of the most important aquatic stressors (high water temperatures and fine sediment) decline by ecologically meaningful amounts with increased riparian corridor length.

We conclude that longer tree corridors may provide some benefits to in-stream condition in addition to increasing habitat connectivity.

Introduction

Restoration frequently seeks to reverse the impact of land use and land cover change, whether re-planting on forestlands that have been converted to agriculture or limiting the effects of invasive species and water pollution associated with urban areas. In many cases, efforts restore ecological process are most effective at large scales, for example through removing a dam to restore connectivity and flow in a river (Holl et al. 2003; Beechie et al. 2008). However, such large-scale, process-based restoration is frequently constrained by the ongoing competing land use, particularly in mixed-use landscapes. We explore the value of increasing the size of small-scale restoration efforts to better support both ecological functioning and ongoing human land use.

Restoration and protection of riparian tree corridors is a common small-scale intervention in working landscapes and can provide multiple benefits (Naiman & Decamps 1997). Importantly, riparian revegetation can restore riparian processes with the potential for long term benefits: tree cover can provide shading and coarse organic matter inputs in the form of leaf litter, as well as slow overland water flow, reduce peak flow, limit bank erosion and soil loss, and filter fine sediment and nutrients (Naiman & Decamps 1997; Sweeney & Newbold 2014; Dixon et al. 2016). However, there are limits to the improvements that are possible without removing ongoing stressors (Roni et al. 2008), and in some cases, small-scale riparian restoration does not improve in-stream conditions (Louhi et al. 2011; Violin et al. 2011).

One factor that might improve such efforts is incremental changes in project size. Recent research has established that a minimum buffer width of 30-50 m is required to effectively filter and process nutrients and sediment before they enter the stream (Mayer et al. 2007; Sweeney & Newbold 2014), but this width is not always achievable where grazing is ongoing and wide corridors are unrealistic. Although there is some evidence that spatial positioning of riparian revegetation can influence in-stream conditions (Parkyn et al. 2003; Baker et al. 2006), little research has focused on the importance of corridor length.

Here, we tested whether longer riparian corridor length can improve stream outcomes. While narrow corridors may be ineffective at filtering fine sediment and nutrients from overland flow (Muller et al. 2016), *long* narrow corridors may be able to moderate extreme flows, store fine sediment, limit local erosion, lower stream temperatures and provide and retain coarse organic matter (Parkyn et al. 2003; Moore & Palmer 2005; Urban et al. 2006; Wohl et al. 2015a). In addition, long narrow corridors can stabilize banks and trap dead wood, supporting pool formation and channel complexity (Gurnell et al. 2016; Muller et al. 2016; Solari et al. 2016).

We performed our study in grazed grasslands of west Marin and Sonoma counties, CA, USA, working in a series of restored and remnant strips of riparian tree cover. We focused on in-stream conditions, including the benthic macroinvertebrate community and habitat features likely to affect this community. Macroinvertebrates are useful indicators of in-stream conditions because they are ubiquitous, have well-studied food preferences, relatively small ranges as larvae, and taxa-specific responses to stressors in agricultural and grazing land uses (Rosenberg & Resh 1993; Matthaei et al. 2010; Larsen et al. 2011). As a result, they integrate site conditions over time.

We hypothesized that long corridors would result in lower water temperatures, less fine sediment, more pools (due to stabilization of dead wood and increased scour) and a shift in food resources (less algae, more leaf litter). In turn, we predicted that if greater tree corridor lengths improved in-stream conditions, this would transform the macroinvertebrate community to include more intolerant or sensitive taxa (higher % EPT, % sensitive, & richness, lower tolerance value) and drive a shift in consumer types towards detritivores and away from grazers (Sponseller & Benfield 2001). In addition, we hypothesized that if tree cover traps fine sediment, we would observe a shift away from burrowing invertebrates and towards clingers, which require larger substrate.

Methods

Study design

Marin and Sonoma counties have a Mediterranean climate with average rainfall of 68 cm (National Centers for Environmental Information 2018). We sampled in 2015 and 2016. 2015 was the final year of a multi-year drought and had below average rainfall (609 mm in 2015 water year), whereas 2016 represented slightly above-average rainfall (690 mm) (National Centers for Environmental Information 2018).

Over the past 20 years, riparian vegetation has been restored throughout the region to improve water quality and manage erosion, as well as to support listed subpopulations of Coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*O. mykiss*). Restoration techniques included fencing to permit passive recovery, the planting of willow stakes, and planting of other tree species in small grazing exclosures (see Figure 1.1b). The consistent land use and many small, low order streams provided good replication for our study. A 2011 study (Lennox et al. 2011) of riparian tree survival and recruitment following re-vegetation in the same region found high survival of woody species, as well as increases in in-stream wood and pool depth. We build upon this study by evaluating in-stream responses to tree cover.

To isolate the effect of tree corridor length, we controlled for the well-documented effect of watershed condition by sampling streams within one land use type in a small geographic area, using a nested design with multiple sampling points on each stream. We also controlled for three factors that capture potentially larger-scale influences on the effect of corridor length: stream power (Bizzi & Lerner 2015), length of upstream gap in tree cover, and soil type (which determines substrate size). These controls allow us to focus on the potential for improvement to in-stream conditions with greater riparian corridor length.

Stream were selected based on four criteria: 1) dominant land use/land cover of grazed grasslands; 2) a break in riparian cover coupled with a downstream intact or restored riparian tree corridor (if restored, established a minimum of 7 years prior); 3) flowing water in April (i.e., not ephemeral); and 4) site access permission. Within a stream, we used GoogleEarth aerial photography to select a site with no tree cover immediately upstream of the riparian corridor and a site at the downstream end of the corridor (Figure 1.1). Where additional sites were available between the unbuffered and far downstream site, we sampled up to two additional points with intermediate lengths of upstream corridor. Each comparison set along a given stream represents a "stream group." In three cases, we had two stream groups on a single stream, but these were a minimum of 2 km apart. Five of the stream groups we sampled had remnant tree corridors, while seven had recently planted tree corridors and one contained reaches of both restored and remnant cover (Table 1A.1).

Tree corridors in our study area were narrow, with a median width of 10 m on each bank (maximum 30 m). Dominant tree species included *Salix* spp. (willows), *Quercus* spp. (oak), *Umbellularia californica* (California bay laurel), *Alnus* spp. (alder), and *Aesculus californica* (California buckeye). Restored corridors were dominated by willows, while remnant corridors had a more diverse mix of trees. All streams were narrow enough that riparian trees achieved complete canopy closure (Table 1A.1). We calculated standardized corridor length as corridor length / bankfull stream width, because as stream volume increases we expect a longer corridor to be required to impact stream condition (Parkyn et al. 2003). Several of the larger streams had long corridors, so this also reduced the problem of one stream group having multiple outlier points. We re-ran the analyses on unstandardized corridor length, and all significant coefficients were unchanged.

Within each site, we identified a sampling riffle and defined a 50-m sampling reach working upstream from the selected riffle. We selected riffle habitats because they typically support the most productive and diverse macroinvertebrate communities in streams (Needham & Usinger 1956; Statzner et al. 1988; Ode 2007). We sampled at 25 sites within seven stream groups in 2015. In 2016, we resampled those sites as well as an additional 14 sites for a total of 39 sites within 13 stream groups. In four of our 13 stream groups, we could not access the unbuffered site, and instead sampled only within the area with tree cover (Table 1A.1). Only five sites (two stream groups) had perennial flow; the others are summer-dry intermittent streams. Most analysis includes both years, but in some cases (e.g., food resources quantification) we rely on only the 2016 dataset, which is more complete.

Habitat variables

We sampled in April 2015 and April-May 2016. At the downstream, mid, and upstream points of the reach (meter 0, 25, 50) we measured bankfull width and depth. We placed a HOBO Onset pendant continuous temperature logger at each site for 3 weeks. We calculated pool spacing as the number of channel widths per pool within the 50-m reach (50/ bankfull channel width / # pools) (Montgomery et al. 1995).

Across each sampling riffle we performed a pebble count of at least 100 pieces of substrate along perpendicular transects placed 0.25-1 m apart, depending on channel width and riffle length (Wolman 1954; Bunte & Abt 2001). Bed material smaller than 2 mm was recorded as "sand" and "fines." We calculated median diameter (d50) and % fines and sand.

In 2016, we took chlorophyll *a* samples from three cobbles within each sampling riffle. We scrubbed each cobble with a nylon brush until visibly clean, and then filtered a known proportion of the algal slurry through a Whatman glass fiber filter to the point of resistance (Arar & Collins 1997). We measured the three largest perpendicular axes for each cobble to estimate surface area using a spheroid approximation (Bergey & Getty 2006). In the lab, chlorophyll *a* (corrected for pheophytin *a*) was measured in μg/L concentrations using a TD-700 fluorometer (Arar & Collins 1997; Steinman et al. 2017). We then calculated the concentration of corrected chlorophyll *a* per rock area and took the median measurement from the three cobbles per site.

In 2016 we sampled riffles for coarse particulate organic matter (CPOM, ≥ 1 mm, e.g., leaf litter) and fine particulate organic matter (FPOM, <1 mm) using a 500-μm D-net. We cleaned cobbles from a 0.09-m^2 area into the net and disturbed the substrate for 30 seconds. We took three samples per site and combined them for a total sampling area of 0.27 m^2. We repeatedly elutriated the sample to remove gravel and sand, and then poured the sample through 1-mm mesh and removed all large invertebrates. CPOM was trapped on the net, towel-dried, and weighed in the field to the nearest gram. To estimate FPOM, the sample passing through the mesh was then filtered through a 500-μm net to capture FPOM ≥ 500 μm. We removed macroinvertebrates from the sample with forceps and preserved the remainder in formalin. In the lab, samples were dried, weighed, ashed, and reweighed to calculate the ash-free dry mass of this "large" fraction of fine particulate organic matter (Hutchens et al. 2017).

Invertebrates

We sampled the riffle invertebrate community using the sampling method outlined above. We elutriated samples in the field and preserved them in 95% ethanol. In the laboratory, samples with over 600 individuals were split into subsamples with a minimum count of 350 individuals. Invertebrates were sorted from the sample and identified to family for insects and class or order for non-insects using standard keys (Wiggins 1977; McCafferty 1981; Harrington & Born 2000; Cummins et al. 2008).

To assess community response to riparian tree cover, we used community metrics that have high discrimination and high stability for intermittent streams in this region (SFBRWQCB 2007) and that are less sensitive to our limited taxonomic resolution (e.g., not Diptera richness). We calculated **mean tolerance value** using California Tolerance Values (Ode 2003), based on the Hilsenhoff Biotic Index, assigning each taxon a value from 0 (intolerant or sensitive) to 10 (extremely tolerant) and using an abundance-weighted average to calculate a community tolerance value. We also calculated **percent intolerant** or sensitive taxa (scores 0-2). We calculated **% EPT** (Insecta orders Ephemeroptera, Plecoptera, and Trichoptera), a common measure of stream condition. EPT orders are typically larger-bodied, diverse, and sensitive to environmental stress (Resh & Jackson 1993). We compared % EPT and tolerance values in our sites to minimally disturbed intermittent streams in the San Francisco Bay Area (SFBRWQCB 2007).

To explore response to food resources, we assigned insects to **functional feeding groups** (shredder, grazer, other) using classifications based on mouthpart morphology (Ode 2003). Due to lack of taxonomic resolution, we did not assign a functional feeding group to *Chironomidae*, which made up a median of 36% of invertebrates at a site, and is most commonly comprised of predators, collector-filterers, and collector-gatherers, rather than shredder or grazer specialists (Ode 2003; Bogan et al. 2015). To assess responses to differences in habitat, we assigned taxa as clingers, burrowers, or other based on **functional form** (Poff et al. 2006). Median relative frequencies were extremely low for burrowers (0) so we only analyzed clingers. We also calculated **rarefied richness** based on our minimum count of 361 individuals.

Landscape characterization

Using GPS waypoints and ESRI ArcGIS, we located each site and measured corridor width and upstream corridor length from aerial imagery, as well as the length of the continuous unshaded distance upstream of the riparian corridor for each stream group, which we refer to as the upstream gap in cover We used a 3-m digital elevation model (USGS 2018) to map upstream drainage area and the channel slope. We estimated stream power using mean bankfull width and depth multiplied by slope.

Using the soil map for Marin County (Kashiwagi 1985) we assigned sites to one of two broad classifications of soil type: moderately drained fine coastal soils or well-drained coarser inland soils.

Data analysis

We completed all analyses and data manipulation in R (R Core Team 2017), and constructed figures using the package *ggplot2* (Wickham 2009). We compared characteristics of remnant and restored corridors using a Wilcoxon rank-sum test.

Models

We modeled six habitat outcomes (mean water temperature, pool spacing, % fines and sand, chlorophyll a concentration, FPOM, CPOM), and five invertebrate outcomes (% clingers, % EPT, % intolerant, tolerance value, rarefied richness) separately, constructing a total of eleven models. We define significance at $p \leq 0.05$ and report standard error.

To assess the effect of tree cover on habitat and invertebrate metrics, we performed a series of regressions using the R package *lme4* (Bates et al. 2015). We applied a logit transformation to proportion outcomes (Warton & Hui 2011), adding 0.001 where values included zero. We modeled concentrations and weights (chlorophyll *a*, CPOM, FPOM) and pool spacing using a generalized linear mixed effects model (GLMM) with a gamma distribution and a log link. Other responses were modeled using linear mixed effects models (SI Table A2). All predictors were centered and rescaled prior to analysis (Gelman 2008). We constructed regression models using both standardized corridor length and tree presence as predictors and stream group and year (where applicable) as random effects. Corridor length and tree presence are nested and collinear; tree presence functions as a covariate, allowing us to estimate the effect of corridor length independent of the potentially strong effect of presence. For models where corridor length did not have a significant relationship with the outcome variable, we performed a paired t-test comparing sites with no tree cover to the nearest downstream site with trees to test for an effect of tree presence. We did not perform this test for models with a significant effect of corridor length, as a significant effect of length indicates that trees do affect stream condition.

To assess whether the effect of corridor length varied with larger-scale conditions, we tested for significant interactions between corridor length and three covariates in turn: coarse vs. fine soil type, stream power (log-transformed), and length of the upstream gap in tree cover (log-transformed). We only tested those relationships for which we had hypotheses: we tested stream power for all terms, and soil type and upstream gap for all invertebrate outcomes but only a subset of the habitat outcomes (Table A2). We compared each of the resulting models to the corridor length and presence only model using a likelihood ratio test. We include the additional terms where they improved the model ($p \leq 0.05$). The interaction term for stream power did not improve any of the models, so it is not included in the table.

To assess model goodness of fit, we calculated pseudo-R^2 for the linear mixed effects models (Nakagawa & Schielzeth 2013) using the package *piecewiseSEM*

(Lefcheck 2016), which calculates the variance explained by the fixed effects alone (marginal pseudo R^2) and the total variance explained. Equivalent methods are not available for the generalized linear mixed effects models with log links that we used.

We assessed influential points and influential stream groups (random effects) using Cook's distance calculated with the package *influence.ME* (Nieuwenhuis et al. 2012): we visually inspected plots for high leverage points, excluded any streams or sites that were highly influential, re-ran the models on the reduced dataset, and compared to the models from the original full dataset. Results were qualitatively unchanged in all cases, so we retained the full dataset. We plotted residuals against predicted values, standardized corridor length, tree presence and any selected covariates; there were no patterns. We also plotted the random effects. The model output for pool spacing was counterintuitive, so we assessed the correlations between pool spacing and depth, as well as between pool spacing and wood pieces.

Ordination

We evaluated differences in whole community composition in response to watershed and stream condition using non-metric multidimensional scaling (NMDS) ordination and the R package *vegan* (Oksanen et al. 2017). To avoid autocorrelated repeated measures, we plotted only 2016 data. We performed a Hellinger transformation on the data prior to constructing the ordination to standardize total abundances and de-emphasize rare species (Legendre & Gallagher 2001). For visualization, we created four categories of standardized corridor lengths: no tree cover, short (0-50 m), medium (51-200 m), and long (>200 m), which represent roughly equal proportions of our data. To visualize the relationship between the invertebrate community and the environmental variables, we used the *envfit* function to create a series of vectors that maximize the correlation between environmental variables and the NMDS ordination and overlaid these on the ordination. We assessed significance by permuting the environmental variables 999 times and comparing estimates.

We performed a partial redundancy analysis (pRDA) on the transformed 2016 community data to test whether restored vs. remnant cover predicted community composition, after controlling for the effect of soil type (Legendre & Legendre 2012). We partialled out the effect of soil type, calculated the proportion of the remaining variance in the transformed species matrix explained by restored vs. remnant cover, and permuted the dataset 999 times to calculate a p-value (Borcard et al. 2011; Legendre & Legendre 2012; Oksanen et al. 2017).

Results

Comparison of restored and remnant corridors

Restored corridors were an average of 6 m narrower on the narrower side than remnant corridors ($w=165$, $p=0.02$), but had similar upstream gaps in cover ($w=108$, $p=0.9$). Shading ($w=131$, $p=0.4$) was also similar for the two groups. Partial redundancy analysis estimated that 5% of the variation in invertebrate communities was explained by whether the site was restored or remnant, after controlling for the effect of soil type.

Habitat response to tree corridor length

The habitat variables representing physical conditions (temperature, substrate, pools) responded to tree corridor length (Table 1, Table A2). Mean temperature (over 3 weeks) declined with tree corridor length more strongly where the upstream gap in cover was longer. The reductions in temperature over 1 km of tree cover brought the average site below the thermal threshold for salmonids, even in sites with long gaps in upstream cover (Figure 1.2a,e), and the median 3-week mean water temperature was 16˚C for unshaded sites and 14.7˚C for shaded sites. The proportion of fine substrate and sand on riffles declined with corridor length, although more dramatically for fine soil types (Figure 1.2b,f). Counter to expectations, pool spacing increased with tree corridor length, so that there were fewer pools at longer corridor lengths (Figure 1.2c,g). Pool depth and spacing had a weak negative correlation ($R=-0.3$, $p=0.04$), such that depth and frequency of pools decreased together.

Food resources did not respond consistently to corridor length. FPOM was higher at sites with long gaps in cover regardless of corridor length and increased (marginally significant) with corridor length at sites with long upstream gaps in cover, but did not change with corridor length at sites with short gaps in cover (Figure 1.2d,h). The other two food resources measurements (chlorophyll *a* and CPOM) did not change in response to corridor length (Table 1A.2).

When we compared sites with and without trees on the same stream using t-tests, CPOM significantly increased at sites with short corridors compared to sites with no tree cover (p=0.02), and chlorophyll *a* decreased (p=0.05). There was no difference in FPOM (Table 1A.2).

Invertebrate response to corridor length and tree presence

The relative abundance of EPT, clingers, and intolerant taxa all increased with corridor length (Figure 1.3, Table 1.1, Table 1A.2). Rarefied richness and tolerance value were not affected by tree corridor length. The inclusion of soil type as an interaction term improved predictions: increases in % EPT and % clingers with corridor length were greater in the fine soil type streams. Richness, % EPT, % clingers, and % intolerant were all higher in sites with the coarse soil type. In addition, richness was higher at sites with smaller gaps in upstream cover. Streams with large upstream gaps also experienced greater increases in % EPT with corridor length than streams with small upstream gaps (Table 1A.2).

Neither richness nor tolerance value (the invertebrate variables with a non-significant response to corridor length) differed significantly between sites with and without tree cover when we compared them using a t-test (Table 1.1, Table 1A.2).

Most sites failed to meet the regional reference intermittent stream values for % EPT and tolerance value (Figure 1.3b,d). Sixty-nine percent of samples (44 of 64) had a tolerance value greater than the “minimally disturbed” maximum for intermittent streams in the region, and 89% of samples (57 of 64) had % EPT values lower than the minimally disturbed intermittent stream minimum. No stream group had all sites within the minimally disturbed range for % EPT, and only one stream group was within the minimally disturbed range for mean tolerance value.

The most common functional feeding group across all sites was collector-gatherer (median 30%). Shredders and grazers were both extremely rare (median relative abundance below 2%), so we did not model their response to tree cover.

Differences in community composition

The whole community ordination (NMDS) was more strongly influenced by stream group than tree cover treatment (Figure 1.4). When we overlaid vectors of environmental variables, the strongest relationships were with % fines and sand (R^2=0.62, p=0.001) and median substrate size (R^2=0.52, p=0.001), followed by watershed area (R^2=0.49, p=.0001), and FPOM (R^2=0.43, p=0.001). NMDS plots showed a strong clustering of the five sites with perennial flow (Figure 1.4).

Discussion

Longer riparian corridors predicted a reduction of two highly-limiting stressors in aquatic ecosystems: water temperature and fine sediment. Corresponding increases in the relative abundances of sensitive invertebrate taxa with corridor length suggest that this reduction in stressors was ecologically meaningful. Although larger-scale influences limited conditions in our sites (e.g., ongoing land use, gaps in tree cover, soil type), and some metrics did not differ with corridor length, our results suggest that increasing the length of even small-scale riparian corridors can provide important benefits.

Improvements in habitat quality with corridor length

High water temperatures can severely stress endangered coldwater fishes as well as invertebrates, and temperatures are likely to increase in the study region under climate change (Isaak et al. 2012; Beer & Anderson 2013). Tree corridors compensated for upstream unshaded reaches within relatively short distances (<1 km), and temperatures continued to drop as corridor length increased. We were not able to map or account for groundwater influence in this study, but the consistent finding across many sites suggests that the temperature result was not groundwater dependent. Many of these streams dried towards the end of our temperature monitoring period, so these reductions occurred during one of the most stressful periods of the year in these streams, when temperatures are likely to be the highest.

Fine sediment is another critical stressor in most grazed systems (Suttle et al. 2004; Burdon et al. 2013), and we observed a substantial reduction of fine sediment by long narrow buffers. There are two potential mechanisms for local sediment reductions: either sediment is prevented from entering the channel, or once in the channel, sediment is removed from riffle habitats and stored elsewhere. Vegetation can slow erosion (Larsen et al. 2009) or filter sediment from overland flow (Sweeney & Newbold 2014), but in streams with fragmented tree corridors (as in this study), some amount of removal is likely required to meaningfully reduce fine sediment on riffle habitats. Once in the channel, fine sediment can be removed from riffle habitats and stored in pools, particularly if tree cover supports pool development through contributing and stabilizing dead wood in the channel (Opperman & Merenlender

2007). However, our model predicted fewer, shallower pools with increased corridor length (and a non-significant effect of tree presence), suggesting that pool development does not explain the reduction in fine sediment. Sediment can also be trapped by the vegetation during overbank flow (Pluntke & Kozerski 2003; Zong & Nepf 2011) and stored off-channel (Corenblit et al. 2007; Gurnell 2014), which seems likely given the density of growth we observed along these streams. To our knowledge, mechanisms for fine sediment reduction over short distances in small streams are not well understood, and further exploration of these mechanisms could be informative to managers.

Regardless of the mechanism, the higher relative abundance of sensitive invertebrate taxa with greater corridor length suggests that the reductions in fine sediment and temperature represent ecologically meaningful improvements. The response of clingers and EPT to both increased corridor length and less fine sediment supports the findings of others that fine sediment reduces richness and favors certain life history traits over others (Larsen et al. 2011; Burdon et al. 2013). Although we did not evaluate salmonid presence or habitat explicitly in this study, fine sediment and temperature can also strongly limit salmonid spawning and survival (Pusey & Arthington 2003), and these reductions would likely benefit these fishes as well.

In addition to physical habitat quality (fine sediment, temperature), we were interested in the response of invertebrate food resources (FPOM, CPOM, chlorophyll *a*), and whether shifts of these resources would affect the relative abundances of functional feeding guilds of invertebrates. Although CPOM and chlorophyll a density differed between exposed sites and sites with short corridor lengths, there was no effect of corridor length on any of the food resources metrics we included. Larger scale patterns that we did not include (such as changes in peak flow, nutrient loads, fish presence, etc.) are likely more important; for example, FPOM increased with upstream gap in tree cover. Correspondingly, the proportions of specialist consumers (many of which are sensitive to land use impacts) were low in all samples, and generalist collector-gatherers were dominant. Generalists can also be more tolerant of sediment (Larsen et al. 2011), which may help them thrive in these agricultural streams.

Influence of watershed-scale condition

While some improvements corresponded to increasing corridor lengths, overall outcomes were strongly limited by watershed-scale conditions. Site conditions became more similar to the reference condition with the restoration or protection of long riparian corridors, but most sites did not match the regional intermittent stream least-disturbed reference condition. To permit full recovery of a stream, the source of the stressor (land use, in this case) would need to be removed (Beechie et al. 2010). Regional references were available for two of our metrics: %EPT and mean tolerance value. Average values for %EPT in the coarse soil type almost doubled between short and long corridor lengths (for the median coarse soil stream %EPT increased from 16% to 30%), although the regional reference would require dramatic additional improvements (57% EPT).

The lack of measured response by the two whole-community metrics we modeled (invertebrate richness and mean community tolerance value) may reflect either our lack of taxonomic resolution, or the presence of other factors limiting recovery. Both richness and tolerance value are often used as a measure of community stress, and we expected them to show an improvement with corridor length. However, even with higher taxonomic resolution, others have failed to find strong responses of invertebrate richness to tree presence (Roy et al. 2003; Kail et al. 2015). This may be because invertebrate recovery is limited by watershed constraints, such as poor water quality, altered hydrology, or a lack of colonists to disperse to newly improved habitats (Sundermann et al. 2011; Hawley 2018).

We purposely selected sites within a small geographic area and consistent land use type to minimize the influence of watershed-scale conditions, which can exert a strong influence on stream condition and recovery (Allan et al. 1997; Sundermann et al. 2013; Villeneuve et al. 2018). Sites on the same stream are likely to be more similar due to variation in watershed area and management practices, and the rapid movement of water, carrying fine sediment, warm water, nutrients, and dispersing invertebrates from upstream (Wilson & McTammany 2014; Tonkin et al. 2018). However, despite our focus on a relatively homogenous study area, conditions varied widely between streams. The strong grouping by stream and weak grouping by corridor length in the ordination analysis highlighted the differences by watershed. The terms highlighted by the ordination as most important are set at large scales, not at the reach scale (watershed area, median substrate size). The strong grouping of sites by stream demonstrates the value of our nested design to detect patterns that might have otherwise been overwhelmed by differences between streams. These strong patterns also highlight the importance of considering site-specific constraints (such as soil type) and adjusting expectations accordingly (Stoddard et al. 2006).

Conclusion

The substantial literature that explores minimum riparian corridor widths for effective filtration of overland flow has largely ignored the potential benefits of increasing riparian corridor length to improve stream condition. Even very short corridors reduce water temperature and result in shifts to food resources, whereas long riparian corridors can support reductions in fine sediment and temperature as well as more sensitive invertebrate communities, with likely effects throughout the aquatic food web. However, we also find evidence that land use stress does continue to limit improvement, and we caution managers to carefully evaluate existing constraints and expectations. We find strong benefits of riparian corridors given the degraded starting condition, but we find no evidence of complete recovery with long corridor lengths. The similarities between sites with restored and remnant cover suggests that restoration of long corridors can provide many of the same benefits as protection of existing corridors. We also found that restored and remnant corridors supported similar invertebrate communities.

Given the constraints of land use, coordinating the placement of many small restoration projects may enable managers to maximize improvements to stream condition. In landscapes with multiple private landowners, the creation of longer riparian corridors could be accomplished by prioritizing restoration and protection of sites near existing riparian corridors. As others have suggested, regional efforts to support the creation and protection of long riparian corridors may help reduce land use stress and increase landscape connectivity with accompanying co-benefits for mammals, birds, and reptiles (Fremier et al. 2015).

Figures and tables

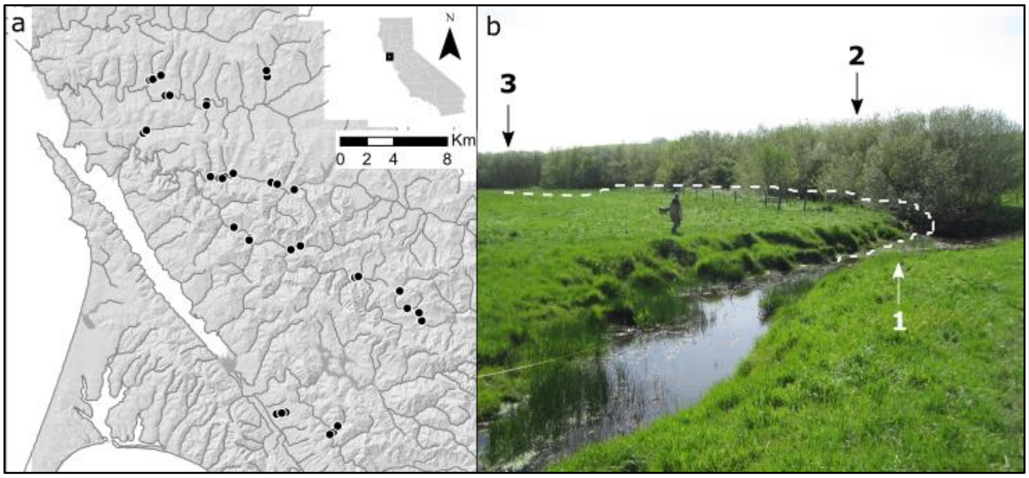


Figure 1.1. Study design

a) Study area in west Marin and Sonoma counties with sampling sites shown as black dots. b) The upstream unbuffered site (1) represents the unrestored state, and moving downstream along a restored (or remnant) tree corridor subsequent sampling points (2, 3) capture the effects of progressively longer tree corridors on stream condition.

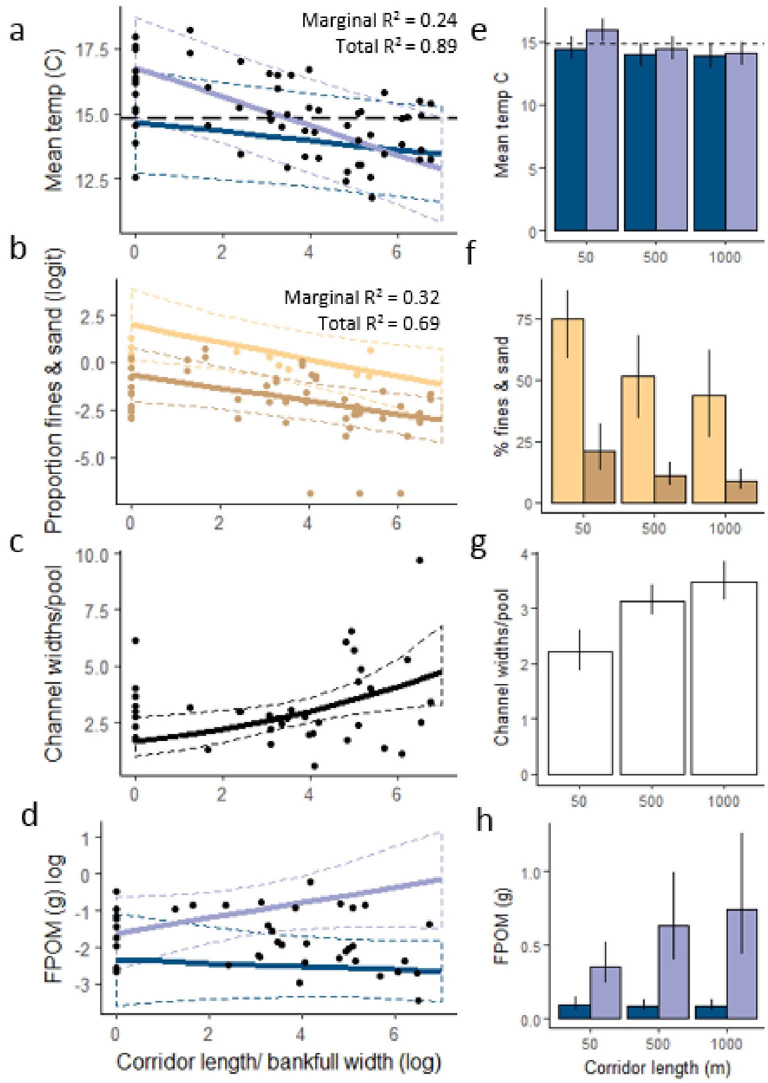


Figure 1.2. Selected model predictions for habitat metrics as a function of standardized corridor length.

A-D: Solid lines indicate coefficient estimates and dashed lines indicate 95% confidence intervals. Dots show observed data. Dark and light blue lines in A & D represent short and long gap in upstream riparian cover, respectively. Dark and light yellow on B represent coarse and fine soil type, respectively. E-H: Back-transformed model predictions for the average stream; lines show SE, and colors match A-D. Dashed line on A and E shows the maximum weekly mean water temperatures tolerable for salmonids (SFBRWQCB 2007). A and B include 2 years of data (A:N=59; C:N=64), C and D are from 2016 only (N=39). R^2 presented for linear models only (marginal R^2 is variance explained by fixed effects only, total R^2 is variance explained by model overall).

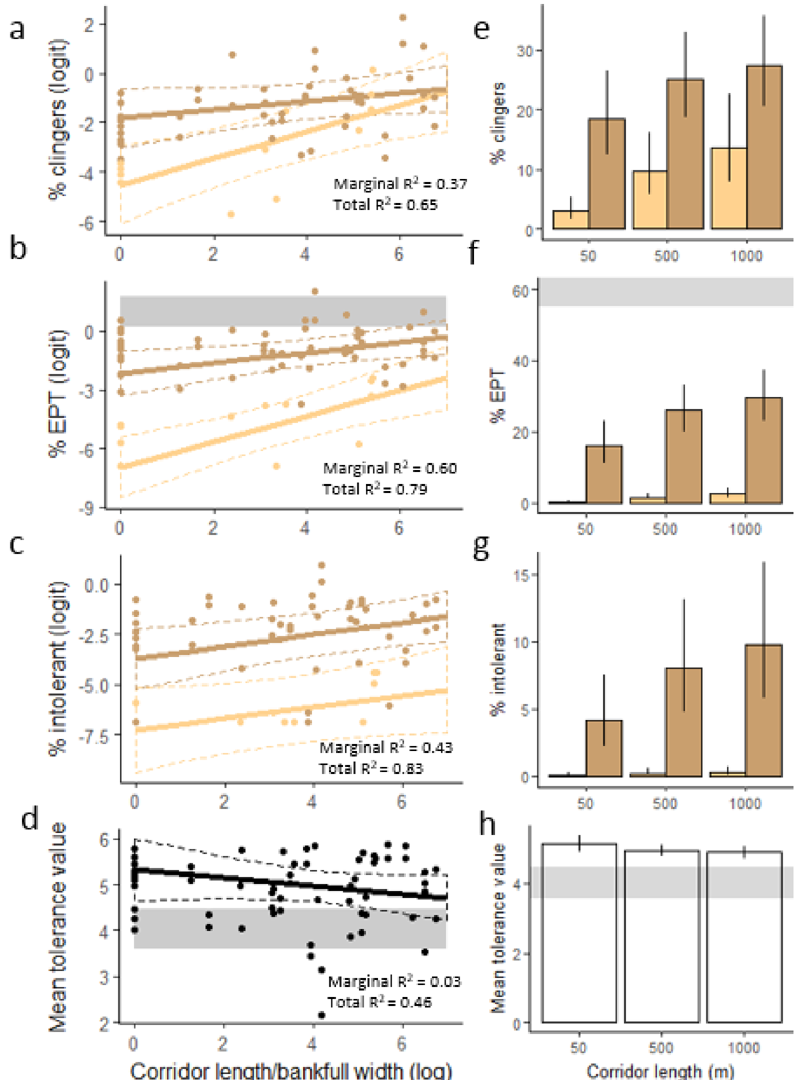


Figure 1.3. Selected model predictions for invertebrate metrics as a function of standardized corridor length. Where soil type was significant, light yellow indicates finer soil type, tan is coarser soil type. A-D: Solid lines indicate coefficient estimates and dashed lines indicate 95% confidence intervals. E-H: Back-transformed model predictions at three corridor lengths for the average stream; lines show standard error. Grey rectangles represent values for least-disturbed intermittent stream reference condition in the region (SFBRWQCB 2007). N=64. Marginal R^2 is variance explained by fixed effects only, total R^2 is variance explained by model overall.

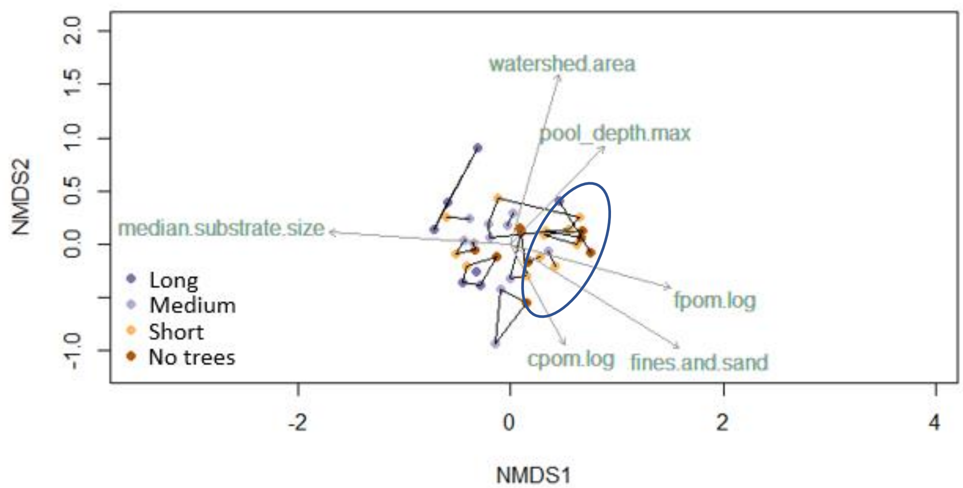


Figure 1.4. NMDS plot of sites overlaid with vectors representing environmental variables of interest.

Stress = 0.16, 2 dimensions, non-metric R^2=0.97. Colors represent coarse categories of corridor length. Connecting lines group sites within the same stream group. Vectors show subset of environmental variables with $p<0.05$ associated with the two NMDS axes. fpom.log is log of fine particulate organic matter, cpom.log is log of coarse particulate organic matter. The five sites in the blue oval have perennial flow; the remainder are intermittent.

Table 1.1. Summary of hypothesized and measured direction of response to riparian tree presence and standardized riparian corridor length for each of the 11 measured outcome variables. For statistics refer to Table A2. The results presented are all significant at $p \leq 0.05$ (or 95% CI not crossing zero).

			Findings		
	Outcome	***Hypothesis***	**Length**	**Covariate impact**	**Presence**[1]
Habitat	3-week mean temperature	↓	↓	Effect of corridor length weaker as watershed-scale riparian cover increases	
	Pool spacing (channel widths per pool)	↓	↑	None	
	% fines	↓	↓	Baseline shifts with soil type	
	CPOM (g)	↑	= & ↑	Increases with corridor length at high watershed-scale riparian cover	--
	FPOM (g)	↑	--		--
	Chlorophyll *a* (μg/L)	↓	--		↓
Invertebrates	% EPT	↑	↑	Effect of corridor length stronger in fine soil type, baseline shifts with soil type	
	% clingers	↑	↑	Effect of corridor length stronger in fine soil type, baseline shifts with soil type	
	% intolerant	↑	↑	Baseline shifts with soil type	
	Tolerance value	↓	--		--
	Rarefied richness	↑	--	Baseline shifts with soil type	--

[1] Assessed using a paired t-test contrasting sites with and without cover in the same stream group. T-tests were only performed for models with no effect of corridor length.

Supporting information

Table A1.1. Basic attributes and corridor lengths of the 13 stream groups.

Watershed Area (km2)	Flow	Min buffer sampled (m)	Max buffer sampled (m)	Number of sites	Mean buffer width (m)	Slope (m/m)	Bankfull width (m)	Soil type	Restoration year	Years sampled
1	intermittent	0	700	3	26	0.055	6	coarse	Remnant	2016
3	intermittent	0	901	4	41	0.03	5	coarse	Remnant	2015, 2016
7	intermittent	0	660	2	4	0.008	3	fine	~2005	2015, 2016
8	intermittent	0	170	2	9	0.009	4	coarse	Remnant	2015, 2016
10	intermittent	140	630	2	18	0.005	5	fine	~1995	2016
12	intermittent	0	4150	5	19	0.01	6	coarse	~1980	2015, 2016
13	perennial	160	1052	2	24	0.009	8	coarse	Remnant	2016
23	intermittent	350	650	2	31	0.006	5	coarse	~2005	2016
31	intermittent	0	640	3	10	0.007	11	coarse	?	2015, 2016
48	intermittent	0	1136	6	15	0.006	8	coarse	~2000 & remnant	2015, 2016
80	intermittent	0	280	2	5	0.003	8	fine	?	2016

Watershed Area (km2)	Flow	Min buffer sampled (m)	Max buffer sampled (m)	Number of sites	Mean buffer width (m)	Slope (m/m)	Bankfull width (m)	Soil type	Restoration year	Years sampled
82	perennial	4700	6740	3	27	0.004	10	coarse	Remnant	2015, 2016
97	intermittent	0	340	3	10	0.002	13	fine	~2005	2016

Table A1.2. Model estimates and standard error

for the relationship between corridor length and key habitat and invertebrate variables. All models included an intercept, tree corridor length/channel width, and tree presence. Where additional covariates improved the model, we indicate the model estimate for the interaction between the covariate and corridor length. All predictors were standardized. Stream group and year are included as random effects in all models as appropriate. Where we tested an interaction term and found that it did not improve the model, we indicate "n.s." Blanks indicate that we did not perform the test.

		Model					Covariates, included as interactions with corridor length		T-test[b]
Outcome	N	Model type	Intercept	Corridor length/channel width (log)	Interaction[c]	Tree presence[d]	Coarse soil type (binary)	Length of upstream gap (log)	Effect of tree presence (df=8)
Habitat - physical									
Mean temperature	59	linear	14.57 (0.94)	-1.55 (0.4)	-1.24 (0.41) upstream gap	-0.12 (0.42)		0.59 (0.32)	
% fines	64	linear, logit transformation	-1.68 (0.45)	-1.65 (0.56)	0.51 (0.74) coarse soil type	1.35 (0.6)	-2.27 (0.72)	n.s.	

[b] Paired t-test comparing site with no cover to nearest downstream site on the same stream, only performed where corridor length did not significantly predict the outcome variable

[c] Interaction between riparian corridor length and covariate (either soil type or upstream gap in riparian cover, as indicated)

[d] Tree presence is collinear with corridor length, so tree presence estimates are not meaningful and are included only for completeness.

		Model					Covariates, included as interactions with corridor length		T-test[b]
Outcome	N	Model type	Intercept	Corridor length/channel width (log)	Interaction[c]	Tree presence[d]	Coarse soil type (binary)	Length of upstream gap (log)	Effect of tree presence (df=8)
Pool spacing (channel widths per pool)	39	Gamma family, log link	1.16 (0.08)	0.69 (0.27)		-0.67 (0.31)	n.s.		
Habitat - food									
CPOM (g)	39	Gamma family, log link	-1.44 (0.18)	-0.66 (0.47)		-0.51 (0.51)		n.s.	Y (t=-3.0, p=0.02, mean log increase 0.8)
FPOM (g)	39	Gamma family, log link	-1.76 (0.27)	0.29 (0.43)	0.82 (0.45) upstream gap	-0.22 (0.44)		1.12 (0.39)	N (t=-0.005, p=1, mean log decrease 0.002)
Chlorophyll *a* (µg/L)	\39	Gamma family, log link	-1.44 (0.18)	-0.66 (0.47)		-0.51 (0.51)			Y (t=2.4, p=0.05, mean log decrease 0.4)
Invertebrates									
% EPT	64	linear, logit transformation	-1.72 (0.29)	1.53 (0.51)	-1.78 (0.67) coarse soil type	-0.97 (0.55)	3.49 (0.63)		

		Model					Covariates, included as interactions with corridor length		T-test[b]
Outcome	N	Model type	Intercept	Corridor length/channel width (log)	Interaction[c]	Tree presence[d]	Coarse soil type (binary)	Length of upstream gap (log)	Effect of tree presence (df=8)
			-2.15 (0.56)	1.47 (0.59)	1.25 (0.62) upstream gap	-0.42 (0.64)		-0.69 (0.5)	
% clingers	64	linear, logit transformation	-1.56 (0.35)	1.13 (0.51)	-1.73 (0.68) coarse soil type	0.01 (0.56)	1.53 (0.58)	n.s.	
% intolerant	64	linear, logit transformation	-3.36 (0.47)	1.37 (0.58)	0.09 (0.76) coarse soil type	-0.14 (0.62)	3.61 (1)	n.s.	
Tolerance value	64	linear	4.99 (0.17)	-0.4 (0.32)		0.18 (0.34)	n.s.	n.s.	N (t=0.6, p=0.5, mean decrease=0.1)
Rarefied richness	64	linear	12.14 (1.34)	0.98 (1.64)	-1.48 (2.2) coarse soil type	0.14 (1.8)	5.22 (1.72)		N (t=-1.4, p=0.2, mean increase=1.9)
			11.73 (1.46)	0.1 (1.76)	2.96 (1.87) upstream gap	2.02 (1.92)		-2.52 (1.37)	

Chapter 2: Meta-analysis of the effects of upstream land cover on stream recovery

Abstract

Unpredictable or variable ecosystem recovery from disturbance presents a challenge to conservation, particularly as the scale of human disturbance continues to increase. Theory suggests that land cover and disturbance characteristics should influence recovery, but individual studies of disturbance and recovery frequently struggle to uncover generalizable patterns due to high levels of site-specific variation. To understand how land cover, disturbance type, and disturbance duration influence ecosystem recovery, we performed a global meta-analysis of stream recovery from disturbances that affect water quality (e.g., oil spill, fire, waste water), using studies documenting recovery of 50 streams. We extracted upstream natural and urban cover percentages for each site and performed model selection and averaging to identify influences on recovery completeness. Most streams improved following the end of a disturbance but did not recover fully to baseline pre-disturbance condition within the studied period (median 60% of baseline). Scale of disturbance in time and space were not important in our models, but we found that higher percentages of upstream natural land cover corresponded to less complete recovery, possibly due to higher baseline conditions in these systems. Our findings suggest that impacts to systems with low anthropogenic stress may be more irreversible than impacts to already modified systems. We call for more long-term evaluations of ecosystem response to

disturbance and the inclusion of regional references and pre-disturbance reference conditions for comparison. A more thorough understanding of the role of the surrounding landscape in shaping stream response to disturbance can help managers calibrate expectations for recovery and prioritize protection.

Introduction

Human impacts to ecosystems continue to intensify, and managers often must choose where to allow impacts and where to prioritize ecosystem protection. Accurate expectations for recovery can help promote strategic choices. However, assessment of the relative risks and benefits is limited by incomplete knowledge of ecosystem recovery trajectories following a disturbance. Anthropogenic disturbances such as pollution, fire, and invasive species introductions can have lasting impacts on ecosystem condition, and accurate predictions for recovery following a disturbance remain elusive (Benayas et al. 2009; McCrackin et al. 2016; Meli et al. 2017; Moreno-Mateos et al. 2017; Jones et al. 2018). Restoration and recovery are highly site-specific, and there is a need for better understanding of which sites are likely to be particularly vulnerable and how recovery trajectories may vary.

Two potential influences on ecosystem recovery from a disturbance are the scale of the disturbance and the condition of site surroundings. The scale of a disturbance in time and space influences both the size of the initial impact and availability of dispersing organisms (Peterson 2002; Standish et al. 2014): for example, a landscape-scale disturbance could eliminate colonist sources much more effectively than a localized disturbance, and a disturbance of long duration might result in more permanent alterations to species assemblages and ecosystem functions (Lake 2000).

Many researchers have established the influence of surrounding land cover on ecosystem condition (Fahrig 2003; Leite et al. 2013), but the influence of land cover on ecosystem recovery has been difficult to quantify. More natural cover might speed recovery by creating more varied habitats that can serve as refugia (Sedell et al. 1990) or higher connectivity to other high quality habitats, supporting dispersal (Holl & Aide 2011; Leite et al. 2013). In addition, communities that are well-connected to high quality habitats may receive subsidies from their surroundings, promoting stability over time (Baxter et al. 2005). However, complete recovery could also take longer in sites with more natural cover because baseline conditions tend to be higher: higher quality habitats typically have higher species richness and support more sensitive species (Fahrig 2003), while the simplified communities in sites with high human impacts might be more tolerant or quick to recover. In this case, the different baselines against which recovery is assessed in highly impacted versus higher quality locations could create differences in the rate or completeness of recovery.

Streams are nested within catchments and are highly sensitive to surrounding land cover (Allan 2004), making them particularly good model systems to evaluate questions about the effect of the surrounding landscape on recovery progress. Streams in areas with high natural cover typically also contain more sensitive species, links to other patches of high quality habitat, and complex habitat that contains refugia for aquatic organisms such as fish and invertebrates (Urban et al. 2006). Human land uses affect stream ecological community composition, with increased land use intensity typically resulting in more tolerant, less diverse biological assemblages (Allan 2004; Lorenz & Feld 2013; Roy et al. 2016). Thus, human land use is also likely to affect stream recovery and reassembly patterns.

To assess the influence of disturbance scale and land cover on stream recovery, we conducted a meta-analysis examining the response of stream ecosystems to disturbance. We limited our assessment to disturbances affecting stream condition primarily through their effect on water quality (such as oil spills, waste water discharges, and experimental pollution) because water quality impacts occur at a range of spatial and temporal scales and are common across a broad range of landscape types, from highly natural to highly altered contexts. The influence of land cover on recovery can be difficult to determine from an individual study because of site-specific variation and the difficulty of conducting a single research project across a variety of land cover types. Meta-analysis is a powerful tool that allows the combination of results across studies and settings to detect patterns that may be contradictory, weak, or otherwise not captured within an individual study. Although a few meta-analyses have evaluated stream recovery (e.g. Miller et al. 2010; Smucker & Detenbeck 2014; Sievers et al. 2017), only one included information on land cover type. Miller et al. (2010) used coarse categories indicated by study authors to categorize land cover types. They found that streams in forested sites did improve, but they found no significant differences among land cover types. In this study, we create standardized assessments of surrounding land cover to explicitly evaluate whether scale of disturbance and type of surrounding land cover affects the ability of streams to reach the pre-disturbance or goal condition (complete recovery) following disturbances that affect water quality.

We asked the following questions: 1) How much recovery is achieved by streams following disturbances that affect water quality? 2) How do surrounding land cover and scale of disturbance in time and space affect stream recovery? Together these questions can help identify vulnerable sites and those that are likely to recover.

Methods

Study selection

We conducted a search for stream recovery studies using Web of Science on September 26, 2016. We performed a topic search for papers in English using terms for stream (stream or river or aquatic or creek) recovery (recover* or restor*) from a disturbance impacting water quality (pollut* or "water quality") worldwide. We reviewed papers covering a broad suite of disturbances that impact rivers primarily through impacts to water quality, as described by study authors (e.g. mining spill, oil spill, logging). Because our primary interest was the ecological response, we also included a term for an ecological indicator (abundance or diversity or ecolog* or richness or similarity or composition), which excluded those papers focusing solely on water quality recovery. Thus, the search included terms for stream + recovery + water quality + ecological and returned 1,643 papers.

We retained 29 papers for extraction using the following criteria (Fig. A2.1): the paper 1) presents quantitative data on in-stream biological condition of a stream or river, 2) documents recovery from a disturbance that affects water quality, including data from at least two time periods after the disturbance ended, 3) includes a reference condition (using either a nearby site or pre-disturbance data at the impact site), 4) is not a mesocosm (there must be a catchment affecting stream condition), and 5) includes information adequate to place the site in a GIS. We combined these papers with similar databases of published studies from Jones et al. (2018) and Meli et al. (2014), which were more general meta-analyses but included some studies of stream recovery from a water quality disturbance. Our final database contained 37 studies that document 50 streams recovering from disturbances affecting water quality.

Data extraction

For each recovered stream, we collected data describing three states: **impacted** (immediately after the cessation of the disturbance), **recovered** (the last time point recorded by the authors), and **reference** (a site not subject to the disturbance). The "reference" condition was either the same site before the disturbance or a nearby site. Some studies followed a before-after-control-impact design and included two un-impacted references. In those cases, we chose the same-site, pre-disturbance data as the reference. In cases where there were multiple reference sampling sites on the same stream, we chose the site closest to the water quality disturbance source. In cases with multiple data collection efforts over time, we chose the earliest post-disturbance data point as “impacted” and the last data point as “recovered”, while matching for season when possible. We extracted data from figures using DataThief v1.7 (Tummers 2006). We extracted data for all response variables that the study authors used to measure recovery. Where provided within a study also describing ecological condition, we collected data on water quality and physical habitat so that we could compare these to biotic condition.

Effect size calculation

In meta-analysis the log response ratio is a measure of effect size, which represents the difference between the control and treatment case for the variable of interest (Gurevitch & Hedges 2001). Response ratios are a standard unitless approach to scale the values of response variables so that effect sizes for many different types of response variables can be compared (Hedges et al. 1999; Gurevitch & Hedges 2001). Response ratios do not require estimates of standard error, an important consideration, as most data in our study lacked error estimates. We calculated **recovery completeness** using a response ratio comparing the recovered and initial baseline state (ln(recovered/reference)). For comparison, we also calculated **improvement** over the perturbed condition (ln (recovered/impacted)). Finally, we calculated the relative **impact magnitude** (ln (impacted/reference)). As other recent restoration meta-analyses have done (Meli et al. 2017), we reversed the sign for variables whose values increased rather than declined under disturbance. Recovery assessments are presented in the text in back-transformed values for easier interpretation. All plots and models use the original log ratios.

Eighteen percent of our dataset (105 responses) included zero values, which represent meaningful data. To avoid undefined response ratios we added 0.01 to the numerator and denominator of all values (Fig. A2.2).

Predictor variables

To characterize the surrounding landscape, we used information within each paper to spatially locate each study site, and used ArcGIS online to calculate the catchment area and ESRI ArcMap to calculate land cover upstream of each site, using existing stream and land cover datasets. We calculated two cover types: natural cover, which we define as all non-urban and non-agricultural land cover types (including forest, wetlands, grasslands, scrub, tree plantations), and urban cover, because urban cover is frequently particularly stressful to aquatic ecosystems (Walsh et al. 2005). Although tree plantations may function very differently than natural forests, most datasets did not distinguish the two, so we refer to all non-urban and non-agricultural lands as "natural cover." We extracted land cover percentages at a variety of buffer widths. However, both percent natural cover and percent urban cover were consistent across scales so we selected one representative scale for each land cover type (Fig. A2.3). The representative scale was chosen for its high correlation with other scales (natural cover: 600m wide and 1km upstream; urban cover: 120m wide and 5km upstream).

In addition to information on land cover, we collected information on study design and disturbance characteristics (Table 2.1, Table 2.2). To assess the importance of disturbance scale, we categorized the water quality disturbance in each study as either point source (from a single, identifiable point, such as a drainage pipe), or nonpoint source (a more diffuse landscape-scale impact, such as fire). We also included the log response ratio for impact magnitude (see above) as a covariate. Impact magnitude is the impacted condition scaled to the reference condition so that it can be compared across studies and metric types. We included this term so that we could account for differences in impact to more meaningfully compare recovery across studies, and expected higher impact magnitude to result in lower recovery completeness.

Prior to model construction, we assessed the correlation of our predictor variables using the R package *corrplot* (Wei & Simko 2016). We found high correlation (Pearson's r = -0.6) between catchment area and natural land cover – larger catchments had lower percentages of natural cover. To assess which variable was a better predictor, we compared the full model with natural cover as a predictor to the full model with catchment area as a predictor. Natural cover had a lower AICc score (delta AICc=4.5), so we used natural cover only in our models, acknowledging that some of the effect may be due to differences in catchment area.

Meta-regression

We constructed two types of models. First, we used meta-regression to model both overall recovery completeness and overall improvement. We constructed a simple model for each, with response metric type as fixed effect and a site-level random effect to account for non-independence of multiple observations from the same site (Zuur et al. 2009).

Second, we constructed a “full” model to assess which factors predicted recovery completeness, including response metric type, impact magnitude, % natural cover, % urban cover, point source vs. nonpoint source, study duration, reference type, and disturbance duration as fixed effects, and site as a random effect (Table 2.2). To enable comparison of the importance of different predictors, we standardized all continuous terms ((x-mean(x))/ 2sd(x)) (Gelman 2008). Our full model contained two categorical variables: metric type and reference type. In both cases, we set the largest category as the reference level: abundance is the reference level of metric type, and pre-disturbance is the reference level of reference type. Models were constructed using the R package *metafor* (Viechtbauer 2010). All analyses were completed using R 3.4.3 (R Core Team 2017).

We compared subsets of the full model using the bias-corrected Aikake Information Criterion (AICc) (Burnham & Anderson 2002), and the R package *glmulti* (Calcagno 2013). We forced the inclusion of metric type in every model but evaluated all combinations of the other variables. There was no clearly preferred model, so we selected models within 2 AICc values of the lowest AICc score, and used these to calculate model-averaged coefficients weighted by the relative AICc scores (Burnham & Anderson 2002; Grueber et al. 2011). We used these model-averaged coefficients to evaluate the relative importance of different predictors. Model-averaged coefficients can be misleading where terms are collinear (Cade 2015), so we also present the full and the top model to aid interpretation (Table 2.A2). We evaluated model residuals for fit, leverage, and skew; there were no influential outliers. Although funnel plots and other assessments of publication bias are typically recommended for meta-analysis, they are ineffective for random effects models and for models with missing variance and multiple effect sizes per study as we have here (Lajeunesse 2009).

Meta-analysis requires an estimate of within-study variance to weight studies (Gurevitch et al. 2018). However, only 96 of our 575 data points (17%) included estimates of error. Using *metafor*, we constructed models using the known variance where possible, and using the model to estimate a uniform variance for the points with unknown variance (Viechtbauer 2010). We then assigned this model-estimated variance to all points with missing variance prior to completing model selection. We performed two checks on our variance estimate. To assess model sensitivity to the estimated variance value, we re-ran the top-selected model with estimated sampling variance 1/5 and 2x the model-determined value. Estimates were qualitatively the same, so we used the model-assigned values (Fig. A2.4). Second, since the true variance for those points where variance is unknown is not likely to be uniform across studies, we applied a cluster-robust variance correction to data grouped at the level of the study (Angrist & Pischke 2009; Viechtbauer 2010): we assumed that errors are more likely to be correlated within a study, due to similar data collection methods. We present results with this corrector in the text; results both with and without are in Table A2.2. We were unable to apply the robust corrector during model selection; instead we applied this corrector to the top selected model (Table A2.2).

To explore whether impact magnitude also varied by land cover, we performed a secondary analysis modeling impact magnitude using the same set of terms (metric type, study duration, point source vs nonpoint source, reference type, disturbance duration, natural cover, urban cover) and performed model selection and model averaging using the same methods described above.

Results

Summary of dataset

The final dataset includes 37 studies, documenting the recovery of 50 streams, of which 26 were in the were in the US and 11 in Europe, with the remainder in Argentina (6), Malaysia (3), Australia (2), New Zealand (1), and Canada (1). Only 3 study sites were in the tropics, and only 3 study sites had non-perennial flow.

The studies reported a total of 575 responses that met our criteria. Abundance of one or more types of organisms was the most common response metric (60% of responses), and macroinvertebrates were the most common taxonomic group studied (56% of responses, and measured in some way in 31 of the streams) (Fig. A2.5). Nine streams were affected by natural disturbances (fire=2 streams, hurricane=1, and volcanic eruption=6) whereas the remainder were human-caused, including logging (4), experimental treatments of biocide (3), and nutrients (2), as well as opportunistic studies of recovery from spills (8), waste water discharges (7), logging (7), and mining (7). When we assessed natural cover by disturbance category, each category occurred either only in high natural cover sites or across a broad range of percent natural cover (Fig. A2.6).

Forty-two percent of sites (21 sites) were compared to pre-disturbance conditions; the remainder were compared to a nearby reference on another stream (32%) or an upstream reference on the same stream (26%). None of the sites were actively restored.

The median study duration (elapsed time between initial post-disturbance and final post-disturbance data collection) was 2 years, and study duration ranged from 20 days to 62 years post-disturbance (IQR= 1 - 3.8 years, mean ± sd = 5.3 ± 9.9). Seven streams were monitored for more than 10 years, and seven for less than one year.

Do streams recover following a disturbance affecting water quality?

The median site recovered to 60% of reference condition when averaged across all measured responses (mean site = 60%, CI=51, 71). Although most sites did not recover completely, 30% of measured responses achieved "over-recovery" - a final state above the reference condition. In addition, most sites did improve after the disturbance ended: the median site improved to 240% of disturbed condition (mean = 337%, CI=236, 480).

There were no significant differences between recovery of abundance and other metrics (Table A2.2), but biotic integrity, diversity, and water quality recovered to reference condition on average, while abundance and diversity did not (Fig. 2.1a). Abundance, diversity, and water quality improved significantly over the disturbed condition, while biotic integrity and physical habitat metrics did not (Fig. 2.1b).

When we evaluated only fish and invertebrate abundance, neither recovered completely (Fig. A2.7).

Influences on recovery

Recovery to the level of the pre-disturbance baseline (recovery completeness) was predicted by higher impact magnitude, less natural cover, and metric type (Fig. 2.2, Table A2.2, Table A2.3). As natural cover increased, recovery completeness decreased (Fig. 2.3). Recovery completeness also declined with longer study duration, suggesting that systems that fail to recover are studied for longer. Point source vs. nonpoint source was included in the top model set but had no clear effect on recovery: confidence intervals were much larger than the coefficient estimate.

The top selected model for recovery completeness (metric type + impact magnitude + natural cover + study duration) was better than the model with metric type and impact magnitude only (delta AICc=7). With the cluster-robust estimator applied to the top selected model, confidence intervals for study duration crossed zero, but natural cover and impact magnitude were still important (Table 2.A2).

When we modeled influences on impact magnitude, more urban cover and less natural cover predicted higher impact magnitude, although 95% confidence intervals on the model-averaged model coefficients crossed zero (Table 2.A4). Shorter study duration and use of a nearby or upstream (rather than pre-perturbation) reference also predicted higher impact magnitude.

Discussion

Overall, we found that streams improved but did not recover fully following disturbances affecting water quality. In addition, we found that streams with higher percentages of upstream natural cover were less likely to recover to the pre-disturbance baseline condition, but also less likely to experience severe impacts.

Measuring stream recovery

Overall, streams failed to recover to baseline. Other recent reviews and meta-analyses looking at other types of disturbances have found low recovery completeness in wetlands (Moreno-Mateos et al. 2012), forests (Meli et al. 2017), and multiple ecosystem types (Benayas et al. 2009; Jones et al. 2018). Meta-analyses and reviews of active stream restoration have similarly found either mixed results (Miller et al. 2010; Palmer et al. 2010; Smucker & Detenbeck 2014; Sievers et al. 2017) or lack of improvement (Stranko et al. 2012). We found improvement of most metrics and some cases of complete recovery: our focus on a discrete, reversible impact likely contributes to our more positive findings.

Measured recovery completeness might have increased if the study period had been longer. Ecosystems typically recover over decades to centuries or longer (Jones & Schmitz 2009), but studies in our database were overwhelmingly short-term, with almost 50% (24 studies) lasting less than two years, and 18% lasting less than one year. In many cases, the final condition captured in these studies is likely not true recovery. However, recovery completeness did not improve with longer study duration. Instead, we found a (weak) negative relationship between study duration and recovery completeness. To better evaluate recovery, longer studies are needed, a call echoed by much of the restoration ecology literature (Bernhardt et al. 2007).

Our results also point to three important components of study design. First, the choice of metric type affected measured stream recovery: abundance did not recover fully, while diversity and biotic indices did, and fish recovered slightly more than invertebrates. Most metrics in our analysis related to ecological structure rather than function, which may reflect the fact that our search terms related more strongly to structure than function. A more in-depth evaluation of the recovery of ecological function might reveal different patterns.

Second, our findings show that the choice of how we calculate recovery influences the patterns we observe. Although we restricted our sample to studies that included both a degraded reference (impacted site) and an unaffected reference (reference site), the degraded condition and unaffected reference condition are each frequently used individually as baselines in the restoration literature (Miller et al. 2010; Weber & Peter 2011; Morandi et al. 2014; Suganuma & Durigan 2015). These two different measures of the ecosystem recovery process – improvement (vs. degraded condition) and recovery completeness (vs. reference condition) – showed different patterns in our dataset: for example, abundance showed high improvement and low recovery completeness. We urge researchers and practitioners to continue to use BACI and other designs with multiple reference types whenever possible to ensure capture of both improvement and progress towards the target reference condition.

Finally, our analysis was limited by data availability. Better reporting of study statistics (including sampling error and sample size) would have strengthened our ability to draw conclusions from this analysis (Gerstner et al. 2017). In addition, our study sites were heavily biased towards temperate and perennial streams; patterns for tropical or temporary streams may differ. Due to the limited available data, our results are only suggestive, and we hope that in future, more detailed work will be possible to further explore these questions.

Influence of land cover and disturbance scale on recovery

More natural cover predicted less complete recovery. We explored whether impacts were also more severe in areas with natural cover, but found that more natural cover (or less urban cover) weakly predicted less severe impacts. Together, these findings suggest that streams with more upstream natural cover are likely to be more stable over time, but may fail to recover fully. Disturbances in already degraded ecosystems are more likely to have a temporary effect, while disturbances to natural ecosystems may be more irreversible. These findings emphasize the importance of avoiding impacts to high quality ecosystems.

The more complex and diverse communities typically found in streams with lower human impacts may explain the lower recovery and lower impacts in these systems. Stream condition varies predictably with land cover (Allan 2004; Norton et al. 2009) (but see Baker 2005), with increasing richness and abundances of sensitive taxa as anthropogenic cover in a catchment decreases (Paul & Meyer 2001; Roy et al. 2003). Streams with a high quality baseline condition are effectively held to a higher standard of recovery, with a larger complement of species (Stoddard et al. 2006); but a larger complement of species also means that the loss of a couple of species would result in a smaller measured impact magnitude. To allow direct comparison of recovered conditions across streams, studies need to include a regional reference. Researchers working in a variety of ecosystems have called for the use of a regional "quantitative optimum" reference representing best attainable regional condition (Stoddard et al. 2006; Morandi et al. 2014). The use of a regional reference condition would have allowed us to quantify the condition of each stream and separate the effects of initial condition and surrounding cover on recovery. Although a few of the studies in our database did include a regional reference (e.g., Arce et al. 2014), most did not, limiting the conclusions we could draw about ecosystem condition.

Less complete recovery in more natural areas could also occur because streams with more natural cover have more sensitive, specialist, and rare species (Roy et al. 2003), so that recolonization may take more time or might never reassemble the original species composition. Less sensitive and more homogenous taxa are likely to be widespread in streams with high anthropogenic land use (Urban et al. 2006), and these communities may be more resilient to additional disturbance because all of the intolerant species have already been lost (Stoddard et al. 2006). In this study, natural cover was correlated with small catchment area, so some of the observed effect may also indicate that headwaters are less able to recover to baseline, possibly because they lack upstream sources of colonists.

To further explore these results, we call for studies of stream recovery across a broader variety of stream types. The effect of natural cover on recovery completeness may interact with hydrological variability, particularly for intermittent streams

(Matono et al. 2012). Although our study included a range of upstream catchment area sizes, most were perennial, temperate streams, and patterns might differ by climate and flow regime.

The scale of disturbance in time and space did not predict recovery completeness. We expected long-lasting and large-scale disturbances (such as chronic nonpoint source water pollution) to result in less complete recovery. However, our results suggest that landscape condition and data collection methods have a larger influence on measured recovery than disturbance scale.

Our results echo the findings of others (e.g., Jones et al. 2018) that ecosystem recovery following disturbance is uncertain and often incomplete. Catchments with more natural cover may be less able to return to baseline conditions than catchments with extensive human land use and already simplified communities, so avoiding impacts in natural systems may be even more critical than in already modified streams. The limited number of studies available for this meta-analysis highlights the need for more rigorous studies of ecosystem improvement to support strategic conservation investments in the future.

Figures and tables

Table 2.1. Response metric categories with frequencies and examples.

The five super-categories were used in analysis; frequencies represent total data points within each category.

Category	Frequency	Example
Abundance	351	
Abundance	259	*Abundance of collector gatherers*
Density	49	*Mean density of trout*
Production	31	*Bedrock substrate annual production of scrapers (g AFDM/m^2/yr)*
Biomass	6	*Habitat weighted biomass of insects (g AFDM/m^2)*
Growth	5	*Arctic grayling cohort-specific growth rate (% /day)*
Percent cover	1	*Bryophyte cover (%)*
Diversity	98	
Richness	79	*Number of Mollusca taxa*
Diversity	15	*Shannon-Wiener diversity of diatom*
Community composition	4	*% Ephemeroptera richness*
Biotic integrity	47	
Health	14	*Fish stomach fullness*
Index	25	*Ecotoxicological rating (includes inverts, water quality)*
Survival	8	*Asian clam % survival*

Category	Frequency	Example
Physical habitat	43	
Organic matter	33	*Volume of large wood within bankfull width ($m^3 m^{-2}$)*
Sediment	7	*Unit channel sediment storage ($m^3 m^{-2}$)*
Channel	3	*Mean bankfull width (m)*
Water quality	36	
Pollution	20	*Copper concentration in water mg/L*
Nutrients	16	*Concentration of nitrate and nitrite (µg N/L)*

Table 2.2. Summary of terms included in the models,

including both predictor and response variables; all models also included site as a random effect.

Term	Type	Transformation[e]	Method of evaluation
Predictors			
Metric type[f]	Categorical, 5 levels	--	See Table 2.1. Categories are abundance, diversity, biotic integrity, physical habitat, water quality
Natural cover (%)	Continuous	none	Extracted from existing land cover datasets, evaluated cover 1 km upstream and 600 m wide. See Table A1.
Urban cover (%)	Continuous	none	Extracted from existing land cover datasets, evaluated cover 5 km upstream and 120 m wide. See Table A1.
Reference type	Categorical, 3 levels	--	Designates whether the undisturbed reference is nearby, upstream, or a pre-disturbance measurement
Point source vs. nonpoint source	Binary	--	Assigned based on the spatial scale and type of the disturbance to water quality
Disturbance duration (years)	Continuous	log	Number of years from beginning to end of disturbance to water quality
Study duration (years)	Continuous	log	Time between "impacted" and "recovered" measurements
Impact magnitude	Response ratio	log	ln (impacted[g] / reference[h])

[e] All continuous terms were standardized: (x-mean(x)) / 2*sd(x)

[f] Metric type was included in all models, other terms were included only in full model.

[g] "Impacted" refers to the earliest data taken following the end of the disturbance.

[h] "Reference" refers to data taken at the reference site (either nearby, upstream, or pre-disturbance), and represents the pre-disturbance baseline.

Term	Type	Transformation[e]	Method of evaluation
Responses			
Recovery completeness[i]	Response ratio	log	ln (recovered[j] / reference)
Improvement	Response ratio	log	ln (recovered / impacted)
Impact magnitude	Response ratio	log	ln (impacted / reference)

[i] Primary response variable used in this study.

[j] "Recovered" refers to latest data taken following the end of the disturbance.

Figure 2.1. Mean effect size by response metric type

for a) recovery completeness and b) improvement over degraded condition. Dashed line in (a) represents complete recovery; in (b) represents no improvement. Error bars represent cluster-robust 95% CI, and estimates are considered different from zero if CI do not overlap zero. Numbers indicate number of data points (number of sites).

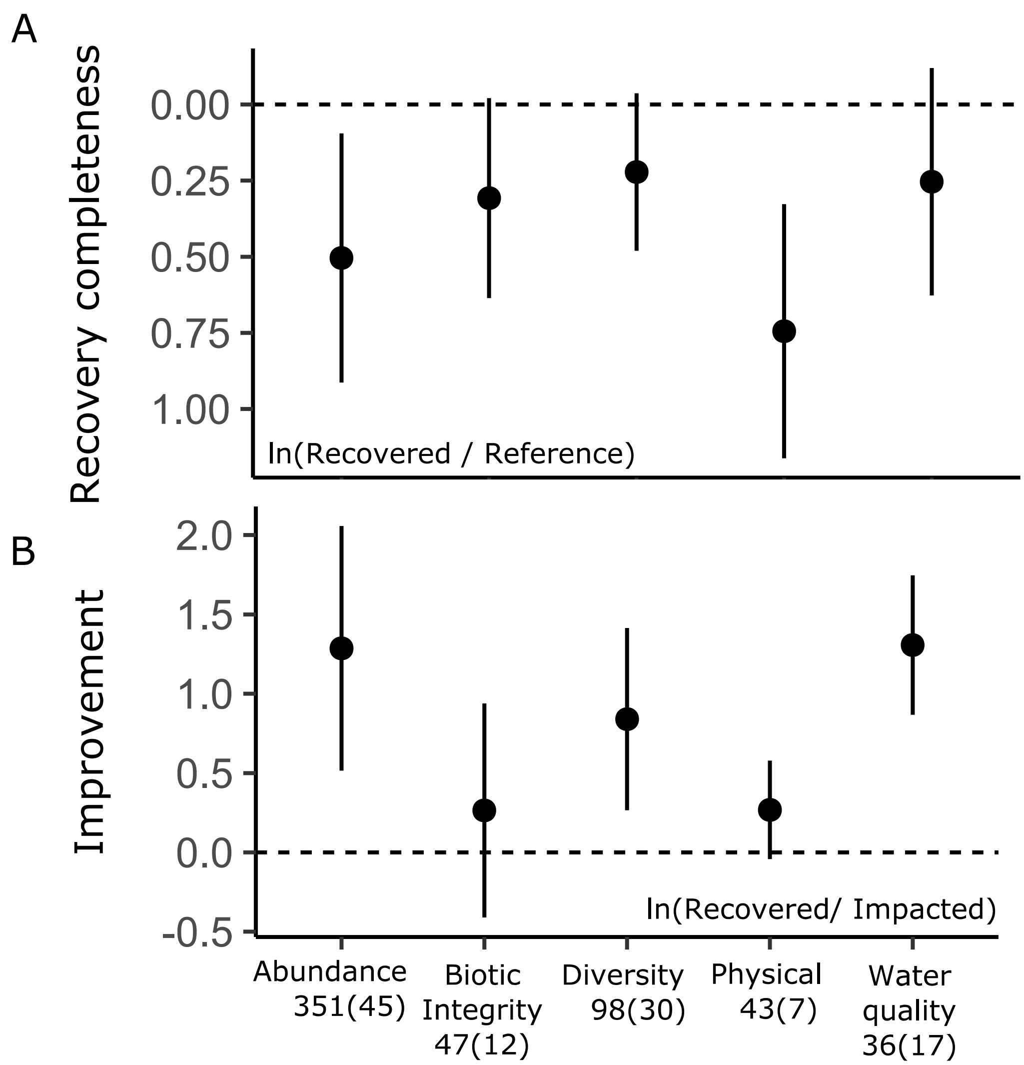


Figure 2.2. Model-averaged regression coefficients

(bars) +/- 95% CI (lines) for the top predictors of recovery completeness (delta AICc = 2). Predictors have been standardized so that coefficients can be directly compared, and estimates are considered different from zero if confidence intervals do not overlap zero. Models also include metric type; see Table A3.

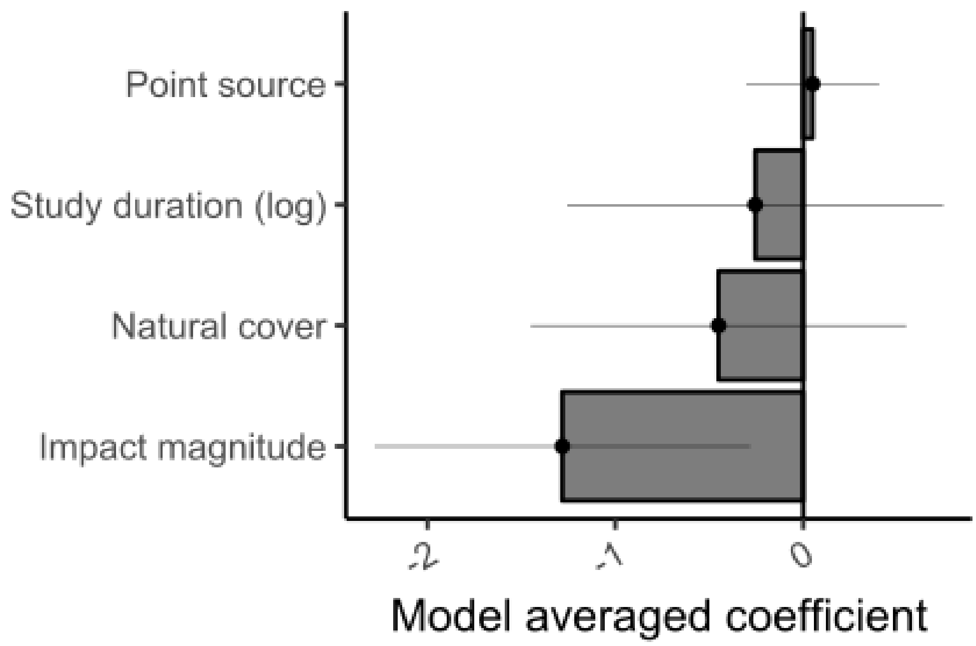
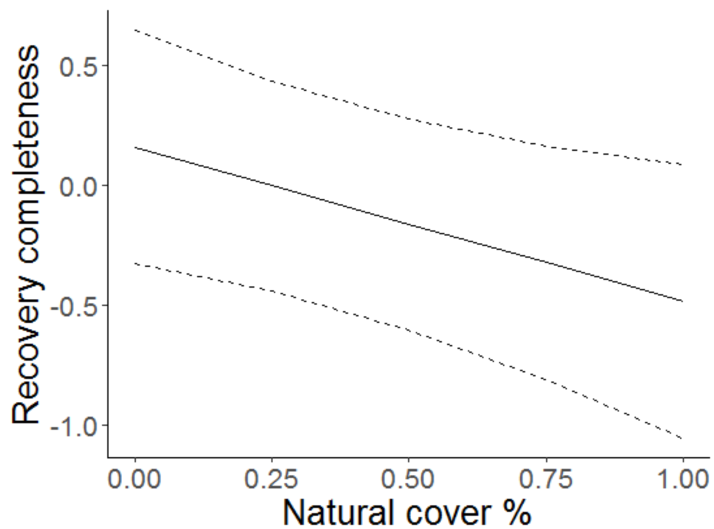


Figure 2.3. Estimated effect of land cover on recovery

to pre-disturbance baseline. Solid line is top-selected model estimate and dashed lines show robust 95% CI. Other terms in the top-selected model included metric type, study duration, and impact magnitude.



Supporting information

Figure A2.1. PRISMA diagram showing study selection process.

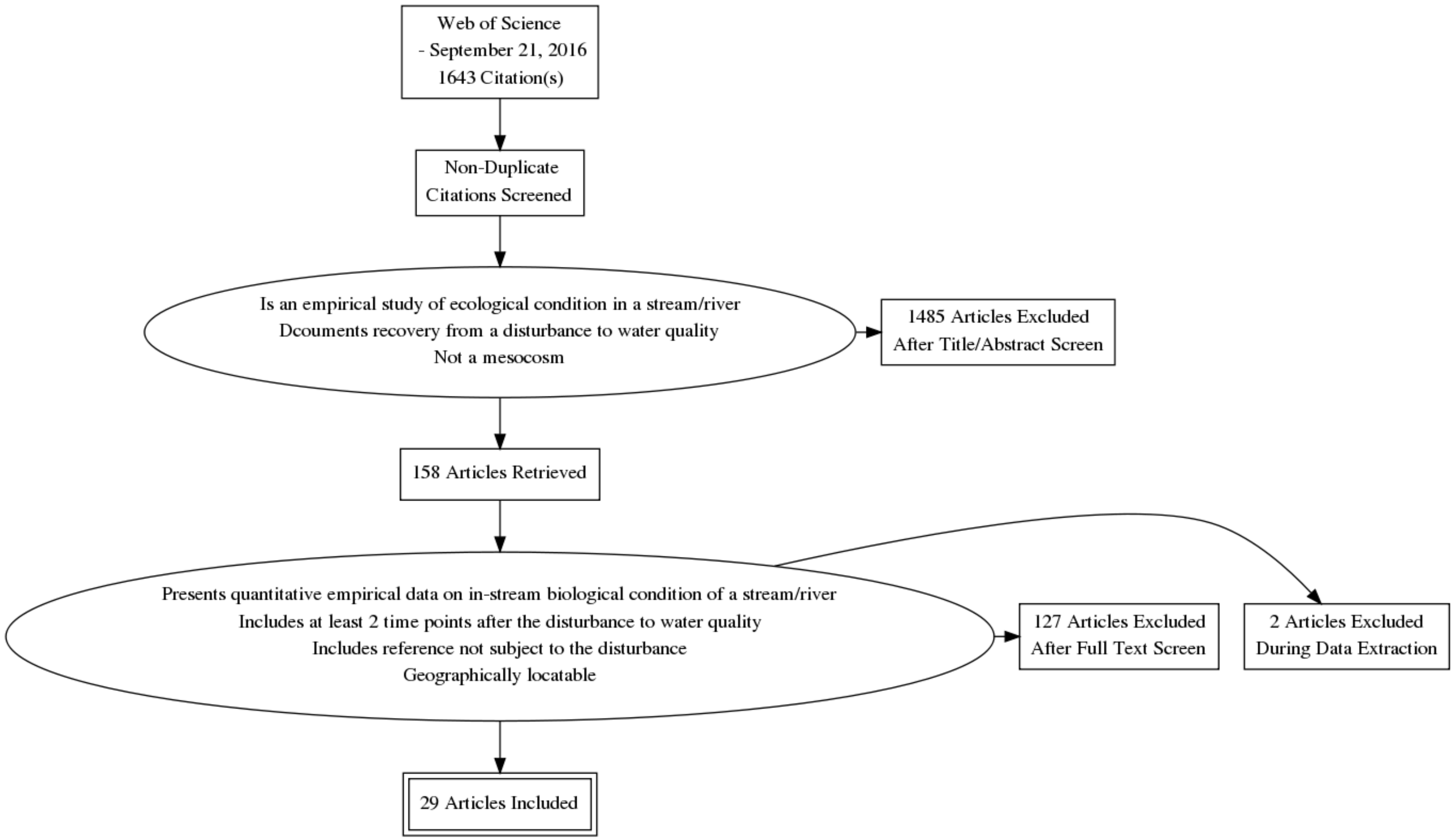
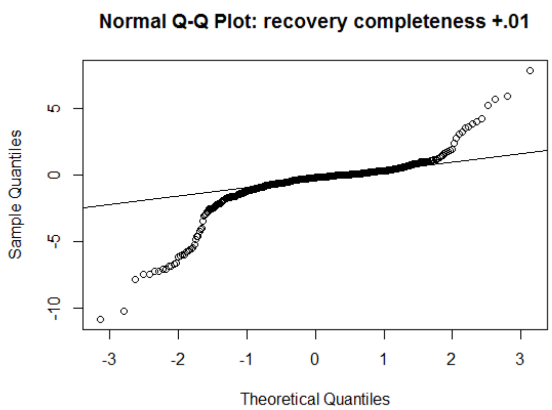


Figure A2.2. Comparison of raw and transformed values (+0.01) for recovery completeness.

Adding a small number to zero values is commonly done in recovery meta-analyses (Meli et al. 2017; Jones et al. 2018). To assess changes to the distribution, we plotted the raw and transformed response ratios against each other, and evaluated standardized recovery completeness effect sizes against the normal quantiles to compare the distribution of data with and without the transformation. We also compared the median and confidence intervals of the transformed distribution with zeros to the nonzero untransformed dataset using a Wilcoxon rank sum test, and found no difference at the p=0.05 level. Including the zero values and transformed data resulted in a shift of median recovery from -0.18 (no zeros) to -0.2 (zeros included).



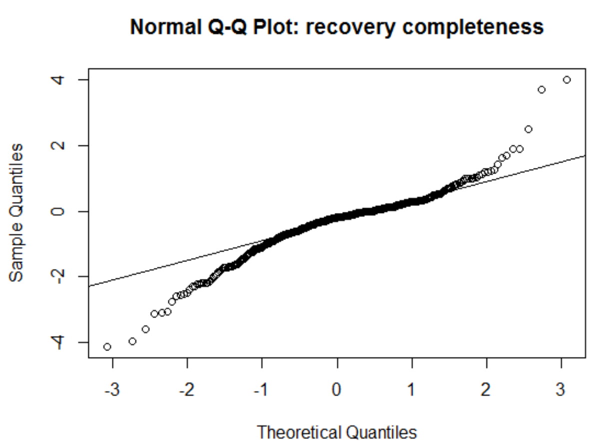


Figure A2.3. Land cover correlation matrix across all calculated scales, showing Pearson's correlation coefficients as percentages. Buff60_500m_Natural represents percent natural land cover within a buffer extending 60m either side of the stream and 500m upstream of the site. Each site is represented by one data point. Data are arranged such that more similar scales are clustered together. Boxes indicate the scales used in the analysis.

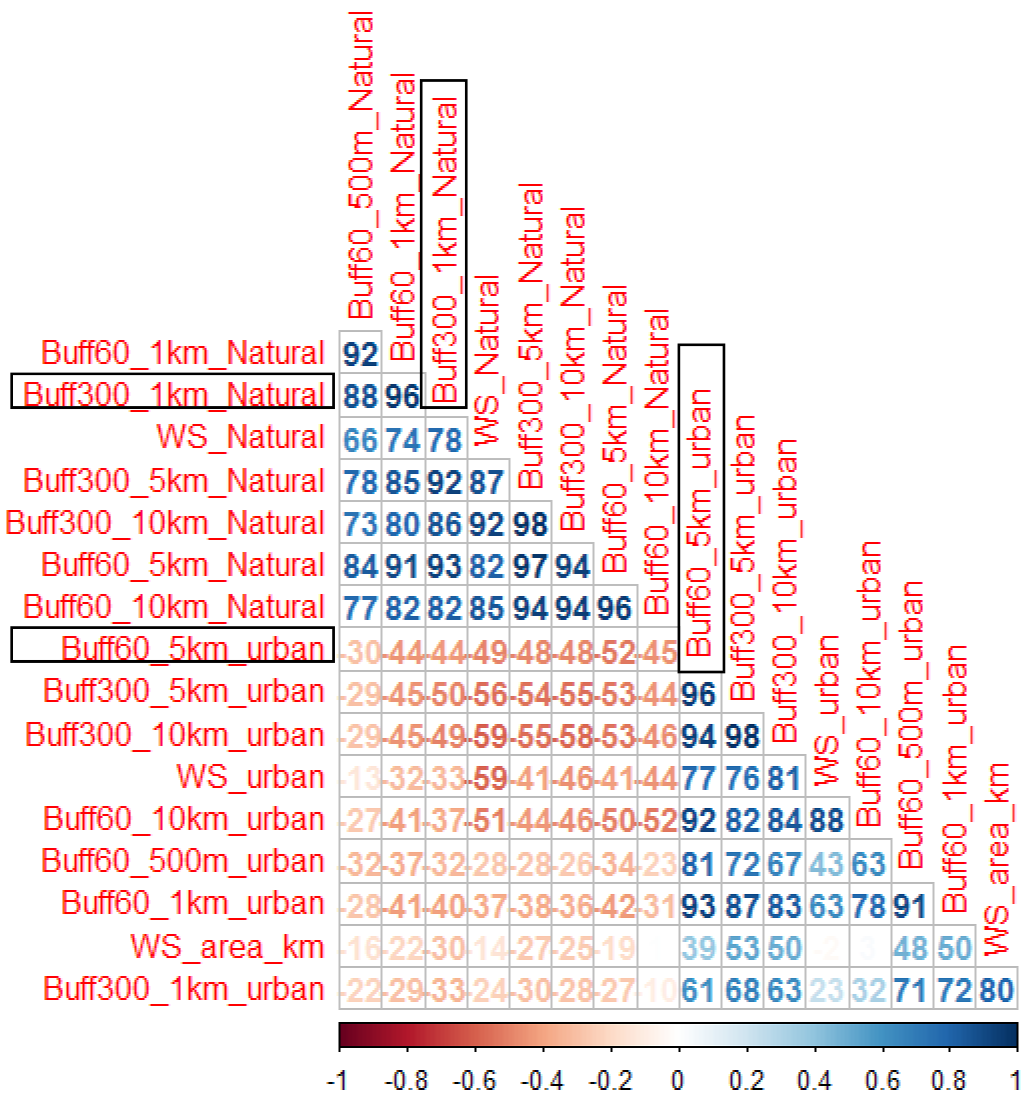


Figure A2.4. Model sensitivity to estimated variance

83% of our datapoints lack estimates of variance, which is required for meta-analysis. We used the *metafor* package to estimate a constant variance for these data in the process of fitting the full model (Viechtbauer 2010). In model selection we applied this estimate to all of those data lacking variance. Since this was only an estimate, we tested the sensitivity of the top selected model coefficients to different variance values. We fit the model using the estimated value (3.38), and then refit the model using twice the variance and 1/10th the variance. In all cases, we retained the original estimates for those 17% of data containing variance. The figure below shows the model estimates and 95% confidence intervals for each of the top selected predictors of recovery completeness using models fit with each of the three variance estimates. Despite some variation in values, our conclusions are unchanged.

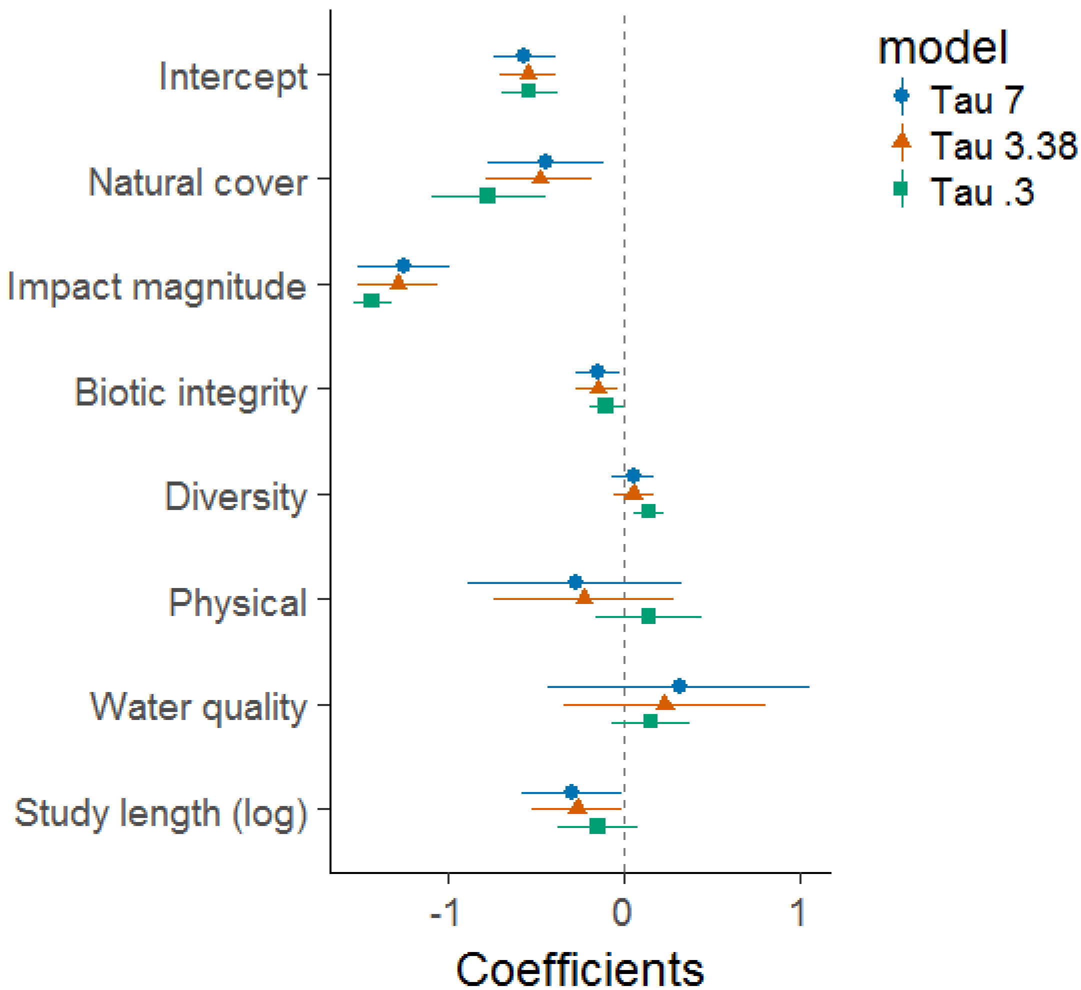


Figure A2.5. Frequency of each response metric type and taxonomic group within the dataset. Each study contains multiple response metrics.

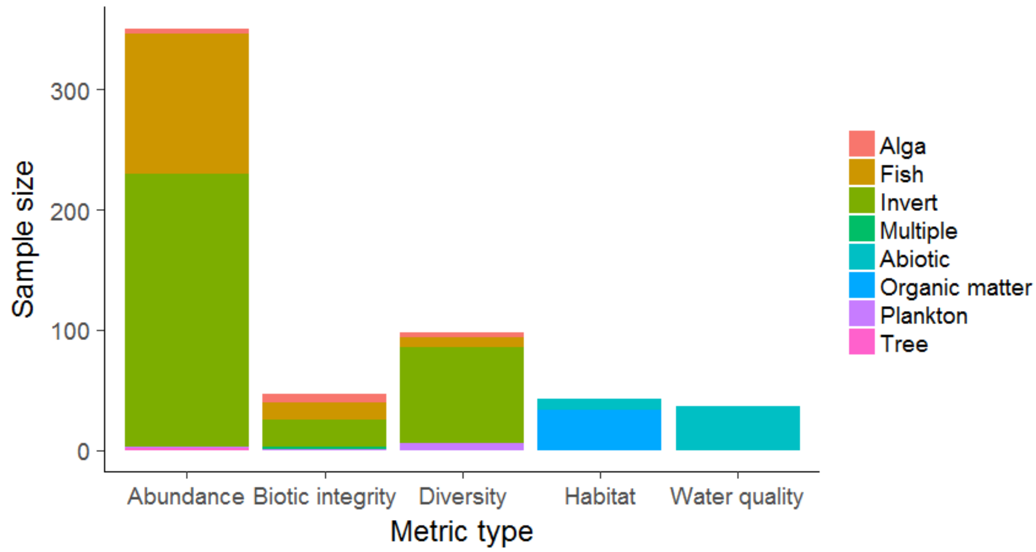


Figure A2.6. Type of impact to water quality plotted against % natural land cover.

Each site is represented by one dot.

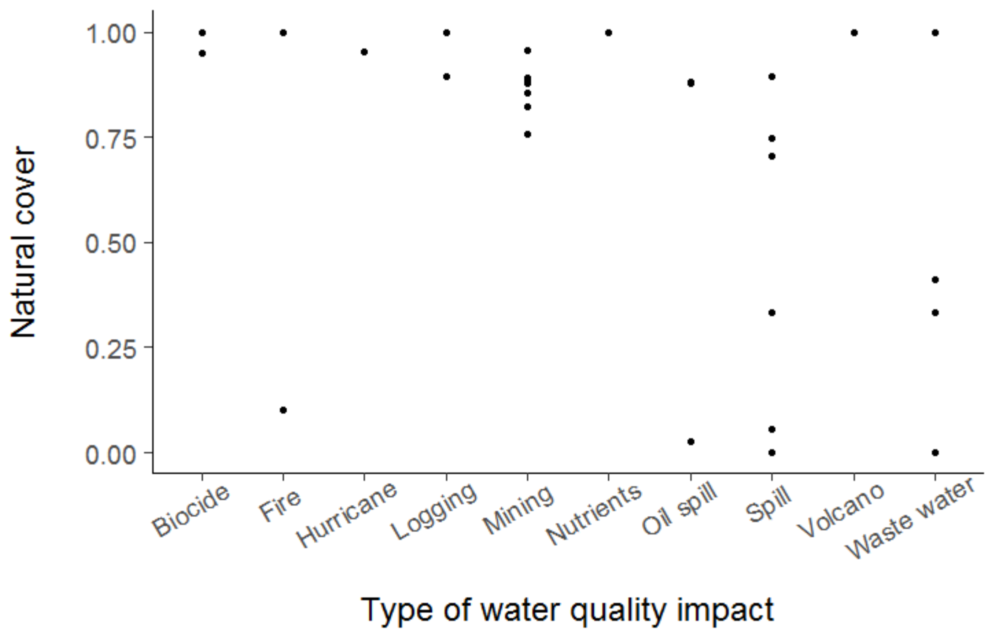


Figure A2.7. Recovery completeness of fish and invertebrate abundance
Model estimate and robust 95% confidence intervals for fish and invertebrate abundance.

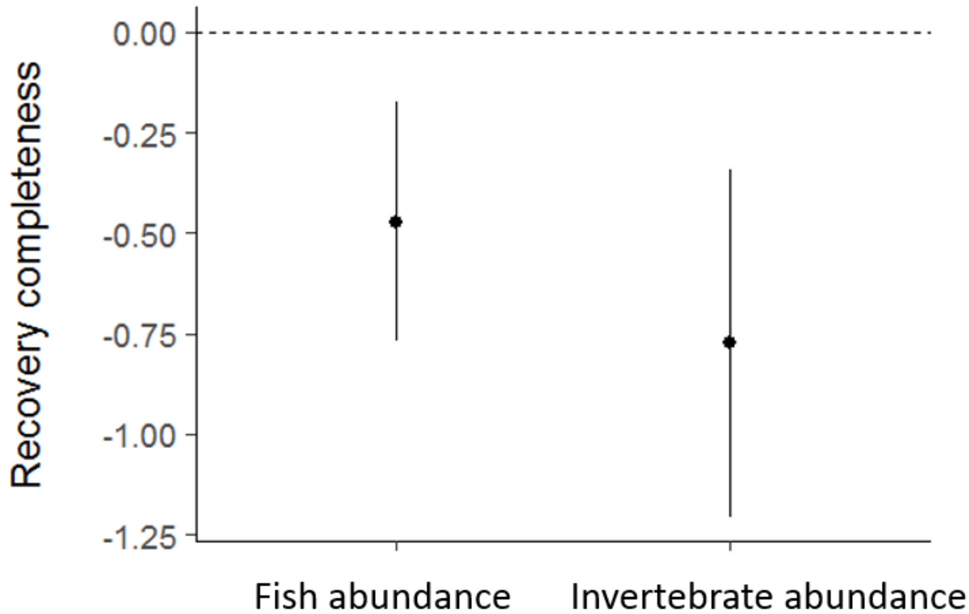


Table A2.1. Data sources and resolution for land cover and stream network by region.

Cover in the following countries or regions was estimated from aerial imagery due to poor quality or unobtainable land cover datasets: Alaska, Argentina, Brazil, and Malaysia.

Region	Data type	Data layer	Source	Resolution
USA	Land cover	National Land Cover Database (NLCD) 2006	Multi-Resolution Land Characteristics Consortium	30 x 30m
	Hydrology	National Hydrography Dataset (NHD)	U.S. Geological Survey	
Europe	Land cover	Corine Land Cover 2006	European Environment Agency	100 x 100m
	Hydrology	Catchment Characterization and Modeling (CCM) v2.1	Joint Research Center	
Canada	Land cover	Provincial Landcover 2000	Ontario Ministry of Natural Resources	25 x 25m
	Hydrology	Ontario Integrated Hydrology Data, 2012	Ministry of Natural Resources and Forestry	
Australia	Land cover	Victorian Land Use Information System 2014/ 2015	Department of Economic Development, Jobs, Transport, and Resources	
	Hydrology	National Surface Hydrology Database	Geoscience Australia	
New Zealand	Land cover	Land Cover Database v4.1	Land Resource Information Systems Portal	1 ha

Region	Data type	Data layer	Source	Resolution
	Hydrology	River Environment Classification v2.0 (REC2)	National Institute of Water and Atmospheric Research	30m DEM

Table A2.2. Recovery completeness models with robust correction.

All factors have been standardized (subtract the mean and divide by 2 sd), to enable direct comparison of coefficients. Columns present the three standard and robust-correction meta-regression models for recovery completeness: response metric type only, the full model, and the top model-selection model. Reference (intercept) levels are set at abundance for response type and pre-perturbation for reference type. P-values indicated as follows: ***(0-0.001), ** (0.0011-0.01), *(0.011-0.05), . (0.051-0.1).

Terms	Response only	Robust response only	Full model	Robust full model	Top selected model	Robust top selected model
Intercept	*-0.51 (0.08)****	*-0.51 (0.2)**	*-0.64 (0.15)****	*-0.64 (0.32).*	*-0.55 (0.08)****	*-0.55 (0.2)**
Broad metric						
Biotic integrity (vs Abundance)	*0.2 (0.05)****	*0.2 (0.18)*	*-0.15 (0.06)**	*-0.15 (0.29)*	*-0.15 (0.06)**	*-0.15 (0.26)*
Diversity (vs Abundance)	*0.28 (0.05)****	*0.28 (0.2)*	*0.06 (0.06)*	*0.06 (0.24)*	*0.05 (0.06)*	*0.05 (0.22)*
Physical (vs. Abundance)	*-0.24 (0.26)*	*-0.24 (0.27)*	*-0.28 (0.3)*	*-0.28 (0.42)*	*-0.23 (0.26)*	*-0.23 (0.31)*
Water quality (vs. Abundance)	*0.25 (0.3)*	*0.25 (0.22)*	*0.19 (0.29)*	*0.19 (0.28)*	*0.23 (0.29)*	*0.23 (0.28)*
Natural cover			-0.44 (0.2)*	-0.44 (0.25).	-0.48 (0.15)**	-0.48 (0.22)*
Urban cover			0.02 (0.26)	0.02 (0.28)		
Reference type						

Nearby (vs pre)	*0.16 (0.24)*	*0.16 (0.29)*		
Upstream (vs pre)	*0.21 (0.25)*	*0.21 (0.31)*		
Point source (vs. nonpoint source)	0.01 (0.23)	0.01 (0.35)		
Disturbance duration (log)	-0.04 (0.19)	-0.04 (0.29)		
Study duration (log)	-0.29 (0.14)*	-0.29 (0.31)	-0.27 (0.13)*	-0.27 (0.22)
Impact magnitude	-1.3 (0.12)***	-1.3 (0.69).	-1.28 (0.12)***	-1.28 (0.62)*

Table A2.3. Top selected models (a) and model averaged coefficients (b) for recovery completeness.

Models within 2 AICc scores of the top model were retained and are presented below. Akaike weights sum to one over the total model set, and represent the relative weight of each model. We forced the inclusion of metric type in each model. Each model contains an identical random effect term representing study site.

a

Top selected models for recovery completeness	AICc	delta AICc	weight
Metric type + natural cover + impact magnitude + study duration (log)	2725.18	0	0.23
Metric type + natural cover + impact magnitude + study duration (log) + point source	2726.39	1.21	0.12

b

	Estimate	Importance	CI
Intercept	-0.57	1	(-0.76, -0.38)
Broad metric			
Biotic integrity (vs. Abundance)	-0.15	1	(-0.27, -0.03)
Diversity (vs. Abundance)	0.05	1	(-0.06, 0.16)
Physical (vs. Abundance)	-0.22	1	(-0.73, 0.29)
Water quality (vs. Abundance)	0.22	1	(-0.35, 0.79)
Impact magnitude	-1.28	1	(-1.51, -1.05)
Natural cover	-0.45	1	(-0.75, -0.15)
Study duration (log)	-0.26	1	(-0.51, -0.01)
Point source (vs. nonpoint source)	0.05	0.35	(-0.14, 0.24)

Table A2.4. Top selected models (a) and model-averaged coefficients (b) for impact magnitude

Models within 2 AICc scores of the top model were retained and are presented below. Akaike weights sum to one over the total model set, and represent the relative weight of each model. We forced the inclusion of metric type in each model. Each model contains an identical random effect term representing study site.

a

Top selected models for impact magnitude	delta AICc	weight
Metric type + reference + urban cover + study duration	0	0.17
Metric type + reference + urban cover + study duration + natural cover	0.79	0.11
Metric type + reference + study duration + natural cover	1.41	0.08
Metric type + urban cover + study duration	1.62	0.07
Metric type + urban cover + study duration + natural cover	1.98	0.06
Metric type + reference + urban cover + study duration + point source (vs. nonpoint source)	1.99	0.06

b

	Estimate	Importance	CI
Intercept	1.69	1	(1.13, 2.24)
Broad metric			
Biotic integrity (vs. Abundance)	-1.26	1	(-1.36, -1.15)
Diversity (vs. Abundance)	-0.89	1	(-1.01, -0.78)
Physical (vs. Abundance)	-0.58	1	(-1.34, 0.18)
Water quality (vs. Abundance)	-0.69	1	(-1.31, -0.06)
Study duration (log)	-0.62	1	(-1.03, -0.2)
Urban cover	0.79	0.85	(-0.2, 1.79)
Ref - upstream	0.58	0.76	(-0.3, 1.47)
Ref - nearby	0.37	0.76	(-0.37, 1.11)
Natural cover	-0.28	0.46	(-1.06, 0.5)
Point source (vs. nonpoint source)	-0.02	0.11	(-0.15, 0.12)

Table A2.5. Papers included in the meta-analysis:

Adams SM, Ryon MG, Smith JG. 2005. Recovery in diversity of fish and invertebrate communities following remediation of a polluted stream: Investigating causal relationships. Hydrobiologia **542**:77–93.

Adams SM, Ham KD. 2011. Application of biochemical and physiological indicators for assessing recovery of fish populations in a disturbed stream. Environmental Management **47**:1047–1063.

Arce E, Archaimbault V, Mondy CP, Usseglio-Polatera P. 2014. Recovery dynamics in invertebrate communities following water-quality improvement: Taxonomy- vs trait-based assessment. Freshwater Science **33**:1060–1073.

Benstead JP, Green AC, Deegan LA, Peterson BJ, Slavik K, Bowden WB, Hershey AE. 2007. Recovery of three arctic stream reaches from experimental nutrient enrichment. Freshwater Biology **52**:1077–1089.

Carreira JA, Viñegla B, García-Ruiz R, Ochoa V, Hinojosa MB. 2008. Recovery of biochemical functionality in polluted flood-plain soils: The role of microhabitat differentiation through revegetation and rehabilitation of the river dynamics. Soil Biology and Biochemistry **40**:2088–2097.

Champeau THR, Stevens PW, Blewett DA. 2009. Comparison of fish community metrics to assess long-term changes and hurricane impacts at Peace River, Florida. Florida Scientist **72**:289–309.

Constantine JA, Pasternack GB, Johnson ML. 2005. Logging effects on sediment flux observed in a pollen-based record of overbank deposition in a northern California catchment. Earth Surface Processes and Landforms **30**:813–821.

Davies PE, Nelson M. 1993. The effect of steep slope logging on fine sediment infiltration into the beds of ephemeral and perennial streams of the Dazzler Range, Tasmania, Australia. Journal of Hydrology **150**:481–504.

de Miguel RJ, Gálvez-Bravo L, Oliva-Paterna FJ, Cayuela L, Fernández-Delgado C. 2016. Recolonization process and fish assemblage dynamics in the Guadiamar River (SW Spain) after the Aznalcóllar Mine toxic spill. River Research and Applications **32**:1196–1206.

Death RG, Baillie B, Fransen P. 2003. Effect of *Pinus radiata* logging on stream invertebrate communities in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research **373**:507–520.

Dinger EC, Marks JC. 2007. Effects of high levels of antimycin A on aquatic invertebrates in a warmwater Arizona stream. North American Journal of Fisheries Management **27**:1243–1256.

Duchrow RM. 1982. Effects of barite tailings on benthos and turbidity of two Ozark streams. Transactions of the Missouri Academy of Science **16**:55–65.

Ely DT, Wallace JB. 2010. Long-term functional group recovery of lotic macroinvertebrates from logging disturbance. Canadian Journal of Fisheries and Aquatic Sciences **67**:1126–1134.

Erős T, Takács P, Czeglédi I, Sály P, Specziár A. 2015. Taxonomic- and trait-based recolonization dynamics of a riverine fish assemblage following a large-scale human-mediated disturbance: the red mud disaster in Hungary. Hydrobiologia **758**:31–45.

Gomi T, Sidle RC, Noguchi S, Negishi JN, Nik AR, Sasaki S. 2006. Sediment and wood accumulations in humid tropical headwater streams: Effects of logging and riparian buffers. Forest Ecology and Management **224**:166–175.

Gunn J, Sarrazin-Delay C, Wesolek B, Stasko A, Szkokan-Emilson E. 2010. Delayed recovery of benthic macroinvertebrate communities in Junction Creek, Sudbury, Ontario, after the diversion of acid mine drainage. Human and Ecological Risk Assessment: An International Journal **16**:901–912.

Kennedy RJ, Rosell R, Hayes J. 2012. Recovery patterns of salmonid populations following a fish kill event on the River Blackwater, Northern Ireland. Fisheries Management and Ecology **19**:214–223.

King JJ. 2015. Ecology and economics of fish kills: Mortality and recovery of brown trout (*Salmo trutta L*.) and Atlantic salmon (*Salmo salar L*.) in an Irish river. Biology and Environment **115B**:157–170.

Klose K, Cooper SD, Bennett DM. 2015. Effects of wildfire on stream algal abundance, community structure, and nutrient limitation. Freshwater Science **34**:1494–1509.

Kruk A, Cieplucha M, Zieba G, Tybulczuk S, Tszydel M, Marszal L, Blońska D, Galicka W, Przybylski M. 2016. Recovery of fish fauna in the upper Warta

River, Poland: Long-term (1963-2012) monitoring study. Ecological Informatics **33**:109–118.

Kubach KM, Scott MC, Bulak JS. 2011. Recovery of a temperate riverine fish assemblage from a major diesel oil spill. Freshwater Biology **56**:503–518.

Lyon JP, O'Connor JP. 2008. Smoke on the water: Can riverine fish populations recover following a catastrophic fire-related sediment slug? Austral Ecology **33**:794–806.

Lytle DA, Peckarscky BL. 2001. Spatial and temporal impacts of a diesel fuel spill on stream invertebrates. Freshwater Biology **46**:693–704.

Mallin MA, Posey MH, Shank GC, McIver MR, Ensign SH, Alphin TD. 1999. Hurricane effects on water quality and benthos in the Cape Fear Watershed: Natural and anthropogenic impacts. Ecological Applications **9**:350–362.

Meade R. 2004. Fish and invertebrate recolonization in a Missouri prairie stream after an acute pollution event. North American Journal of Fisheries Management **24**:7–19.

Miserendino ML, Archangelsky M, Brand C, Epele LB. 2012. Environmental changes and macroinvertebrate responses in Patagonian streams (Argentina) to ashfall from the Chaitén Volcano (May 2008). Science of the Total Environment **424**:202–212.

Montz GR, Hirsch J, Rezanka R, Staples DF. 2010. Impacts of copper on a lotic benthic invertebrate community: response and recovery. Journal of Freshwater Ecology **25**.

Nelson SM, Roline RA. 1996. Recovery of a stream macroinvertebrate community from mine drainage disturbance. Hydrobiologia **339**:73–84.

Poulton BC, Callahan E V, Hurtubise RD, Mueller BG. 1998. Effects of an oil spill on leafpack-inhabiting macroinvertebrates in the Chariton River, Missouri. Environmental Pollution **99**:115–122.

Ryder L, de Eyto E, Gormally M, Sheehy Skeffington M, Dillane M, Poole R. 2011. Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications. Biology and Environment: Proceedings of the Royal Irish Academy **111B**:41–60.

Ryon MG. 2011. Recovery of fish communities in a warm water stream following pollution abatement. Environmental Management **47**:1096–1111.

Sabater S. 2000. Diatom communities as indicators of environmental stress in the Guadiamar River, S-W. Spain, following a major mine tailings spill. Journal of Applied Phycology **12**:113–124.

Schöll K, Szövényi G. 2011. Planktonic rotifer assemblages of the Danube River at Budapest after the red sludge pollution in Hungary. Bulletin of Environmental Contamination and Toxicology **87**:124–128.

Schultheis AS, Sanchez M, Hendricks AC. 1997. Structural and functional responses of stream insects to copper pollution. Hydrobiologia **346**:85–93.

Schulz UH, Costa PF. 2015. The effects of press and pulse disturbance by long and short-term pollution on the fish community in the Sinos River, RS, Brazil. Brazilian Journal of Biology **75**:S36–S44.

Simon ML, Cherry DS, Currie RJ, Zipper CE. 2006. The ecotoxicological recovery of Ely Creek and tributaries (Lee County, VA) after remediation of acid mine drainage. Environmental Monitoring and Assessment **184**:2559–2574.

Stone MK, Wallace JB. 1998. Long-term recovery of a mountain stream from clearcut logging: the effects of forest succession on benthic invertebrate community structure. Freshwater Biology **39**:151–169.

Walter CA, Nelson D, Earle JI. 2012. Assessment of stream restoration: Sources of variation in macroinvertebrate recovery throughout an 11-year study of coal mine drainage treatment. Restoration Ecology **20**:431–440.

Whiles MR, Wallace JB. 1995. Macroinvertebrate production in a headwater stream during recovery from anthropogenic disturbance and hydrologic extremes. Canadian Journal of Fisheries and Aquatic Sciences **52**:2402–2422.

Chapter 3: Where and why does restoration happen? Ecological and sociopolitical influences on stream restoration in coastal California

Abstract

The distribution of conservation effort on the landscape is affected by both ecological and social priorities and constraints. Together these influences can result in bias towards certain types of ecological or human communities. We evaluate the distribution of restoration projects on the California Central Coast, USA, to evaluate sociopolitical and biophysical influences on the type and distribution of one type of conservation effort. We compiled data on 700 publicly funded stream restoration and management projects completed in the past 30 years and the biophysical and sociopolitical characteristics of the 310 sub-catchments in our study area. Our database contains three categories of stream projects: ecological restoration to benefit natural ecosystems, human-oriented projects to enhance ecosystem services, and data collection projects for planning and monitoring. Both ecological and human-oriented restoration efforts were clustered near the coastline. Stream activities of all kinds were highest in sub-catchments with water quality impairment, high population density, high pro-environmental voting, and a highly educated, wealthy, non-Hispanic white population. Ecological restoration and data collection were also greater in catchments with higher native fish richness. Our findings indicate that restoration activity is aligned with, and perhaps responding to, ecological need, and that restoration efforts are concentrated near human population centers and restoration

organizations. Disparities in conservation effort by income, race, and education are concerning and should be evaluated in more depth and in other regions.

Introduction

As humans degrade natural ecosystems, conservation has become a large and growing need. The distribution of ecological management effort across a landscape affects both ecological and human communities. Ecosystems receiving more protection or restoration may regain lost resilience and ecosystem function, experience enhanced connectivity, and support a broader host of species (Ruiz-Jaen & Aide 2005; Benayas et al. 2009). Human communities situated near high-quality natural areas or restoration projects may experience enhanced health and recreational opportunities (Brancalion et al. 2013; Wolch et al. 2014), protection from erosion and floods (Clewell & Aronson 2006; Nilsson et al. 2018), employment opportunities (BenDor et al. 2015), and connection to nature and community (Light 2006; Moran 2010; Egan et al. 2011, although see e.g., DeFries et al. 2007).

In recent years, tools and frameworks have proliferated to assist managers and funders in selecting appropriate sites and projects to maximize (largely ecological) benefits given limited conservation budgets (Norton et al. 2009; Jellinek 2017). Such tools often prioritize conservation in sites with high ecological value or condition, such as regional reference sites or refuges for endemic or endangered species, in part because agencies often have regulatory mandates to protect these areas. In contrast, restoration projects seeking to maximize benefit per dollar spent might prioritize sites unlikely to recover unassisted (Fullerton et al. 2006), or highly impaired sites that negatively impact the surroundings (e.g. through spreading invasive species or changing the disturbance regime) (Leite et al. 2013).

The literature on prioritization tools has a largely normative focus; it provides guidance for where conservation projects *should* be located, usually from a purely ecological perspective (e.g., Moilanen et al. 2009), and pays little attention to the empirical outcomes, i.e. where projects *are* located in practice. The actual locations, however, may differ from the normative guidance. First, decision-makers may be motivated by the potential for social benefit, for example through ecosystem service enhancement or protection (Chan et al. 2006; Standish et al. 2012). Researchers traditionally have not emphasized socioeconomic benefits of restoration (Aronson et al. 2010; Wortley et al. 2013), although these can be substantial (Hillman 2004; Millennium Ecosystem Assessment 2005; BenDor et al. 2015). Urban parks and urban stream restoration are both examples of ecological management strategies often undertaken for social benefit (Cockerill & Anderson 2014; Flies et al. 2017). In other cases, systems with high ecological value are prioritized for conservation due to co-benefits such as visitation, recreational uses such as hunting or fishing, or improved municipal water quality (e.g. Turner & Daily 2007).

Second, the spatial distribution of conservation efforts and benefits may be influenced by socioeconomic and political factors. Conservation may be more feasible in locations with high public interest in environmental issues if this results in the presence of more environmental organizations, funding opportunities, and access to private lands (Christian-Smith & Merenlender 2010; Langridge 2016). Similarly, wealthy pro-environmental communities may mobilize to advocate for their communities, resulting in capture of a greater share of regional or statewide conservation budgets (Mohai et al. 2009). Indeed, studies of the distribution of both urban greening and wetland mitigation projects have found uneven distribution across socioeconomic groups (Ruhl & Salzman 2006; BenDor et al. 2008; Stewart et al. 2014). More broadly, policy making, at least in the United States, often reflects the preference of economic elites rather than the population more broadly (Bartels 2008; Gilens & Page 2014).

Finally, availability of funding may constrain the types of projects that are accomplished in an area or the choice of project location. Funding may promote a particular goal (e.g., invasive species management, human access, endangered species protection), and may also be restricted to certain locations.

To explore influences on the allocation of conservation effort, we focused on the regional-scale distribution of restoration projects in relation to ecological values and human communities. Restoration is an intensive and frequently costly type of ecological management: it can involve a transformation of the ecological community through the addition and removal of species or barriers to connectivity, and can have substantial impacts on human communities (Suding et al. 2015). A regional analysis allows us to compare priorities across many different land use types and human communities while maintaining the ability to perform a fine-grain analysis of factors potentially influencing decision-making. Analyzing the types of locations that currently receive restoration effort can reveal whether certain types of ecosystems or human communities receive disproportionate attention.

Streams present an excellent case study for questions about restoration. Streams are a particularly important ecosystem type for considering distribution of effort because their sensitivity to human activities (e.g., water diversions, land use change) results in widespread need for active stream management and restoration. In addition, due to their special protection under the US Clean Water Act, stream impairment is unusually well documented and regulated, and streams are well-mapped compared to other ecosystem types. Despite these features, patterns and drivers of stream restoration effort are poorly quantified (e.g. Bernhardt et al. 2005). Previous efforts have quantified ecological and management aspects of stream restoration, focusing on types of restoration activity (Bernhardt et al. 2005; Kondolf et al. 2007; Christian-Smith & Merenlender 2010), motivations and land use context (Bernhardt et al. 2007; Moran 2010), or match between restoration and actions called for by endangered species recovery plans (Barnas et al. 2015). However, to our knowledge the relationship between the spatial distribution of effort and both ecological and sociopolitical patterns has not been evaluated.

Whether restoration projects are effective in achieving their conservation goals is a question beyond the scope of this paper (see Suding 2011; Maron et al. 2012; Jones et al. 2018). Indeed, research to date has found high uncertainty among outcomes for stream-based projects (Bernhardt & Palmer 2011; Wohl et al. 2015b).

However, regardless of ecological outcomes, the spatial distribution of restoration projects is intrinsically important as a reflection of intent and resource allocation.

We selected the California Central Coast region, USA for our study for its high biodiversity and unusual variety of biophysical conditions, land use types, and human communities. Within the California Central Coast, we compared the distributions of stream restoration projects focused on ecological goals, such as fish habitat, water quality, or riparian condition; projects focused on human well-being, such as flood control and access; and projects collecting pre- or post-project data. We analyze how restoration effort varied spatially using biophysical and sociopolitical factors as indicators of both intended priorities and unintended biases affecting restoration effort. We ask: which natural and human communities benefit from restoration efforts? What implicit priorities can we detect in the distribution of projects?

Methods

Study design

We mapped stream restoration and management projects within the five counties of the California Central Coast (Santa Cruz, Monterey, San Luis Obispo, Santa Barbara, and San Benito). The study area extends 500km along the coast and 60km inland. California is among the top three regions in the United States for density of stream restoration projects (Bernhardt et al. 2005), and the Central Coast has active local agencies and a robust monitoring program documenting the condition of streams. The Central Coast ranges in elevation from sea level to 1,700m, and includes both the foggy, redwood-covered Santa Cruz Mountains and the dry Carrizo Plain. It is largely rural and agricultural (including the highly productive Salinas Valley) but contains several large urban centers, including Monterey, Santa Cruz, and Santa Barbara.

We measured restoration effort as 1) number of restoration project sites and 2) the amount of public restoration spending occurring within each catchment unit (defined here as the 12-digit Hydrologic Unit Code or HUC, the smallest nationally defined hydrologic unit). The study area contains 310 catchment units; dividing the region in this way provided a natural unit of analysis. Catchment units within the study area have mean area of 85 km^2 (sd = 35) and 144 stream km (sd = 71). We clipped the study area to match California Regional Water Quality Control Board Region 3, which administers much of the funding, monitoring, and regulation for the region (Central Coast Regional Water Quality Control Board et al. 2016); this resulted in the exclusion of eight catchment units in eastern San Benito County. Our study also excludes the five easternmost catchment units of San Luis Obispo County and the Channel Islands because key datasets lacked information for these areas.

Data collection

We compiled databases documenting publicly funded restoration over the past 30 years (Table A1). We focus on publicly funded projects because public funds support over 80% of all stream restoration in the US (Bernhardt et al. 2007) and were more consistently tracked across jurisdictions. We identified potential databases using personal contacts and internet searches.

We used a modified version of the National River Restoration Science Synthesis classification system (Bernhardt et al. 2005) to classify each project by restoration type. In our analysis, we combined several NRRSS categories and added categories describing stream management for human benefit (Table 2.1). If multiple activities were described, we assigned the project to the activity that appeared to be the primary motivation based on the project title and brief description (e.g., a project to control bank erosion using riprap and the restoration of native vegetation would be coded as bank stabilization). Throughout, we refer to all entries in our final database as “restoration sites.” On-the-ground projects with ecological goals are classified as “ecological restoration,” projects undertaken for human benefit are “human-oriented”, and projects focusing on planning, research, or monitoring are grouped together as “research/monitoring.” Each of the individual databases we combined included multiple project types (e.g., both ecological and human-oriented) (Table A1).

We geolocated projects in ESRI’s ArcMap software using coordinates, place names, or catchment unit numbers, as available. We were interested in the number of unique restoration project sites per catchment unit, so where multiple projects had the same project focus (e.g., fish, infrastructure) and location (latitude and longitude values, rounded to two decimal points), we counted each group as one unique project site. We focused on sites rather than projects because multiple projects with the same goal in the same location are not likely to be independent.

For each catchment unit, we compiled measures of (i) ecological characteristics that may affect the need for or effectiveness of restoration efforts; and (ii) social characteristics that may influence the prioritization of restoration projects (Table 2). Ecological characteristics included stream type, ecological need, and ecological value. We characterized stream type using catchment steepness, stream order, and percent natural cover. We used native fish richness as a proxy for ecological value, as well as potential and current habitat (defined as "critical habitat") for steelhead (*Oncorhynchus mykiss*), which is threatened throughout the Central Coast (National Marine Fisheries Service 2013). We characterized ecological need using habitat condition and water quality impairment.

Certain categories are both ecological and social. Human impact represents both the number of people able to benefit from an improvement and the likely ecological impairment. We included percent impervious cover and human population density as proxies for human impact. Land ownership is also ecological and social: we included the percent of the catchment in protected areas owned by non-profits or public agencies and protected for "open space values" (GreenInfo Network 2017), because access for ecological restoration may be easier within these areas than in private lands.

Social characteristics included demographic indicators and expressed public interest in environmental management. Commonly used demographic indicators include income, education, and race; disparities by any of these indicators can suggest inequitable resource capture by more powerful or advantaged social groups (e.g. Ash

& Fetter 2004; Pearce et al. 2006). We characterized local human communities using median income, percent of population with a college education, and percent of population that is non-Hispanic white, because these indicators differentiate communities in the California Central Coast. Public interest can be measured using voting records (Kahn 2002; Wu & Cutter 2011): we used yes” votes on a recent pro-environmental state ballot measure. To select the ballot measure we used in our models, we compared several recent statewide measures, and chose the measure that elicited the most variation in voting. As a robustness check, we repeated the analysis with other ballot measures and found both a high correlation between voting on different measures, and consistent results using different ballot measures (see Appendix B).

To map the locations of Central Coast organizations doing restoration, we performed internet searches on the lead agency listed for each project. We identified the type and location for each lead agency and used the R package *sp* (Pebesma & Bivand 2005; Bivand et al. 2013) to calculate the distance between each project and the office of the responsible lead agency. Government agencies included city, county, state, and federal agencies (e.g., Parks districts, Conservation Corps, Agricultural Commissioner, Resource Conservation District). Non-profits included a range of land trust, stewardship, and watershed groups, as well as some national and international nonprofits. We grouped together individuals and consultants as “private.” Finally, “partnerships” included academic institutions and public-private partnerships that did not fit neatly into the other categories.

Statistical analysis

All analyses were conducted in R version 3.4.2 (R Core Team 2017). We used two response variables for each of the three restoration types: the number of sites per catchment unit, and spending per catchment unit.

To analyze the number of restoration sites per catchment unit, we estimated negative-binomial models using the *glm.nb* function in the R package *MASS* and a log link (Venables & Ripley 2002). Negative-binomial models are commonly used for over-dispersed count data (Zuur et al. 2009), and fit our data better than Poisson models. The full models contain all of the variables from Table 2. We standardized each variable by subtracting the mean and dividing by two standard deviations, which places continuous variables on the same scale as binary variables to allow for direct comparison of coefficients (Gelman 2008). Not all catchment units were the same size, so we included number of stream kilometers within the catchment unit as a covariate in all models. We defined significance as $p<0.05$, such that 95% confidence intervals did not overlap zero.

Several of the variables in the full models were highly collinear, limiting our ability to determine the influence of a variable or group of related variables (Figure A3.1). We evaluated cases where two or more terms a) represented the same category (e.g., human community characteristics), and b) were correlated with a Pearson's r of >0.5. In these cases, we evaluated the coefficient estimates in the ecological model and removed the term with the larger p-value (Figure A3.1a; Table A3.2). We also dropped mean slope and natural cover because they were correlated with percent protected area and population density (Figure A3.1a). The resulting models are our final "reduced" models, which we present graphically in Figure 3.3. We used the same terms for each of the models to enable comparison between models. Median income, percent of population completing four-year college degree, and percent of population that is non-Hispanic white were correlated, so in our models percent non-Hispanic white represents this suite of demographic characteristics.

To model public restoration spending by catchment, we fit linear models to log-transformed spending data. We used the same "reduced" terms for these models. We also fit linear models to number of organizations performing restoration within a catchment unit to test whether the distribution of these organizations was biased.

To evaluate distribution of restoration across different stream types, we compared the stream order and flow regime of sites with ecological restoration to all potential reaches in the Central Coast. Using a 30m digital elevation model we generated stream network and flow direction layers which we used to determine stream order throughout the study area. We used a one-sample t-test to compare the mean stream order of all 1km reaches to the mean stream order of restoration sites. To compare flow regimes, we used the National Hydrologic Dataset (NHD+) to extract the flow regime (perennial, intermittent, ephemeral) for each restoration project site, and calculated the number of one-km reaches by flow regime across the Central Coast. We tested whether the two distributions of stream types differed using a

Pearson's Chi-squared test with p-values calculated using 2,000 Monte-Carlo simulations. We repeated this process for human-oriented stream management sites.

We evaluated the residuals of all full and reduced models for normality, fit, and leverage, and plotted predicted against observed values: residuals were evenly distributed. We calculated variance inflation factors for each model to test for collinearity (max VIF<1.87). We assessed fit using explained deviance (pseudo R^2) (Zuur et al. 2009) or adjusted R^2, as appropriate.

We performed additional analyses to assess the robustness of our results. To more explicitly test the role of habitat quality, we re-ran the models on the 30% of catchments with standardized physical habitat monitoring data. To test whether the models were simply describing steelhead-bearing streams, we re-ran the reduced ecological model on the subset of catchments that contained critical steelhead habitat (National Marine Fisheries Service (NOAA Fisheries) 2005). By constraining our analysis to these steelhead-bearing streams, we were able to remove the influence of steelhead presence on the results. Finally, we compared projects completed before and after 2002 (the median year in our dataset), and found consistent results across both time periods. Additional robustness checks are presented in Appendix B.

Results

Characteristics of restoration efforts

Our final database contained 699 restoration sites, documenting restoration efforts between 1983 and 2017. Half of all sites (54%) were on-the-ground restoration for an ecological goal (Figure 3.1a). The most common type of ecological project sought to benefit fish; 94% of all ecological projects took place within catchments designated as steelhead critical habitat (Figure A3.2). Restoration is clustered in three areas along the coast: southern Santa Barbara County, Morro Bay, and Santa Cruz (Figure 3.2).

Over the 34-year period, we recorded US$341M total public funding for ecological restoration and US$404M for human-oriented projects. In many cases, this funding required matching funds, so actual spending may be twice as high. Eighty-two percent of projects included information on spending. Total spending was highest for human-oriented projects, despite a lower total number of projects (Figure 3.1b). Median per-project spending for infrastructure and ecosystem services was over three times higher than median per-project spending for ecological projects (~US$100,000 vs. ~US$30,000). Only 77% of ecological restoration projects reported spending information, compared to 90% and 92% for human-oriented and research-monitoring, respectively (Table A1), so if more expensive projects are more likely to include spending information, the difference in median spending may be even greater.

Stream sites selected for restoration efforts were biased towards larger and perennial streams. The mean stream order of restoration sites (2.83) was higher than the mean stream order for the study area (1.84) ($t=18.86$, $df=575$, $p<0.0001$). The flow regime in reaches with ecological and human-oriented projects differed significantly from the distribution of reaches in the region overall (Ecological: $\chi^2=4841$, $p=0.0005$; Human: $\chi^2=18151$, $p=0.0005$). Although perennial reaches make up only 8% of stream kilometers on the Central Coast, they were the site of 54% of ecological restoration projects and 26% of human-oriented projects. By contrast, ephemeral streams, which constitute 70% of stream kilometers received only 9% of ecological and 27% of human-oriented restoration projects: ephemeral streams receive 1/7 of the number of restoration project sites that would be predicted by chance. Intermittent streams were selected for restoration projects in proportion to their occurrence on the Central Coast (~15%).

Seventy-six percent of restoration and management projects in the database listed an organization or agency, with numerous projects completed by each of government (32%), non-profit (40%), and other organizations (private or partnership; 25%). We were unable to place all of these organizations: for example, 20% of projects were completed by federal or international organizations where no single, regional office was associated with the project. For the 56% of projects with lead organizations that we could identify and place on a map, we found that the organizations performing restoration and management were clustered near restoration sites (Figure A3.3). Seventy-five percent of locatable organizations only led projects at sites less than 60km distant, and 50% completed work exclusively within 16 km. Government organizations performed the most local work (median distance to office 10km; cities median distance 1.7km), followed by non-profits (median distance 12km). Other organizations, largely partnerships based in public universities, showed a strikingly different pattern, and were much more likely to work far away (median 65km), likely due to the existence of university holdings far from the main campus.

Ecological and sociopolitical factors influencing stream restoration effort

The number of ecological restoration sites per catchment unit was positively related to ecological variables (native fish richness, water pollution), human population density, and sociopolitical variables (pro-environmental voting, percent non-Hispanic white population) (Figure 3.3a; Table A3.2). The explained deviance was 58%. The model for the subset of the 153 catchment units designated as steelhead critical habitat had almost identical coefficient patterns, with one exception: native fish richness was no longer significant (Table A3.2; explained deviance 45%). For the third of catchments with physical habitat information, poor habitat condition was highly correlated with population density and predicted increased restoration effort (Table A3.2; explained deviance 53%).

For human-oriented stream management, population density was the most important predictor of number of restoration projects, along with the presence of a water quality impairment (Figure 3.3a, Table A3.2; explained deviance 57%). A wealthier, whiter, and/or more highly educated population, and higher rates of pro-environmental voting also predicted more project sites.

Predictors of research and monitoring projects were similar to ecological restoration project with one important exception: population density did not significantly predict research and monitoring projects (Figure 3.3a, Table A3.2; explained deviance 45%). Native fish richness, water pollution, voting patterns, and race/education/income were important predictors.

The predictors for the distribution of public spending matched the predictors for number of sites for both ecological restoration (model adjusted R^2=0.25) and research and monitoring (model adjusted R^2 = 0.22) (Figure 3.3b, Table A3.2). Public spending for human-oriented stream management was predicted by population density and native fish richness (model adjusted R^2=0.32); no other predictors were significant.

Organizations performing restoration were more likely to be in areas with high population density, pro-environmental voting, and a more affluent, educated, non-Hispanic white population (Table A3.2; explained deviance 71%).

Discussion

Stream restoration effort on California's Central Coast is aligned with many measures of potential ecological and social benefit, including native fish richness, poor water quality and habitat condition, and high population density. While we might expect that ecological projects would be guided by ecological need and value, and human-oriented projects might be guided by potential for human benefit, we found similar patterns in the distribution of project sites for all types of restoration and management. The distribution of sites was skewed towards perennial streams, population centers, spatial locations near the performing organizations, and areas with higher percentages of wealthy, college-educated, non-Hispanic white populations.

Alignment of ecological restoration with ecological need

More ecological restoration occurred in catchments with higher ecological value (represented by native fish richness) and ecological need (represented by water quality impairment and habitat condition), suggesting that these goals may guide restoration effort. However, restoration was also strongly concentrated in a specific type of value and need: in catchments that provide potential steelhead habitat and catchments near population centers.

The siting of 95% of ecological restoration efforts in steelhead-bearing catchments, the focus on perennial streams, and the many projects to support fish habitat are predictable due to the status of steelhead. Coastal California steelhead populations are listed as threatened under the federal Endangered Species Act, so steelhead-oriented projects in California have a legal mandate (National Marine Fisheries Service 2013). In addition, the high restoration effort in Santa Cruz County may be due to the presence of an additional listed salmonid in that county, *Oncorhynchus kisutch*, and mandated funding for the protection of these two species. Although intensive efforts may be necessary to support these salmonids, the focus on one or two cold-water fish species may result in a lack of diverse stream habitat priorities, a challenge with most focal species approaches (Lindenmayer et al. 2002). Intermittent and even ephemeral streams can support distinct and diverse ecological communities that are frequently undervalued (Bogan et al. 2013; Acuña et al. 2017). Appropriate management of these other stream types appears to require a different prioritization process or new funding sources.

Even within steelhead streams, more ecological restoration took place in catchments with water quality impairment and high population densities (Table A3.2). These impacted sites are likely degraded and in need of restoration, and may benefit more people (Moran 2010), but they are also likely to have more limited restoration potential (Roth et al. 1996; Wahl et al. 2013). Many studies have documented the so-called "urban stream syndrome," a suite of impairments and reduced restoration potential in streams with impervious cover in the upstream catchment (Walsh et al. 2005; Urban et al. 2006; Roy et al. 2009). A restoration focus on highly impacted sites with low recovery potential makes monitoring and post-project assessment even more crucial to determine whether conditions at these sites are improving.

Unfortunately, our study also highlights a lack of monitoring and data collection. Projects focused specifically on monitoring were extremely rare in our dataset. Even assuming that many projects included some monitoring efforts that were not listed as a project priority (Kondolf et al. 2007), the lack of evidence of designated funding for monitoring and project upkeep in our database supports other findings that monitoring is underfunded and underemphasized (Jähnig et al. 2011; Gilvear et al. 2012; Hagger et al. 2017). Data quality was also a problem in this study: we had to exclude 8% of projects from our database due to lack of information on location or activity. Of the projects in our database, 24% did not list a lead agency, and 18% had no information on spending. Given the uncertainty about restoration outcomes, continued effort to track and make public these types of information is critical to understanding the outcomes of public investment. We join others in calling for more support for monitoring and assessment of projects to enhance our ability to conserve and manage ecosystems and guide effective investment of funds (Suding 2011; Gilvear et al. 2012).

Social bias

Stream management and restoration of all types (ecological, human-oriented, and research/monitoring) occurred most frequently in a particular type of human community – a community with higher percentages of residents who are wealthy, non-Hispanic white, and have college degrees. This was true even when we re-ran the models on only steelhead-bearing catchments. These three demographic characteristics are strongly correlated in this region, making the causal relationships difficult to parse.

At least three related mechanisms could explain this pattern. First, communities with higher incomes might be better able to provide the matching funds required for most of these grants. Our measure of community interest in restoration (pro-environmental voting) also has a strong positive relationship with restoration effort, so restoration may be more likely in communities with both high interest and ability to pay for restoration (see also Babin et al. 2016; Langridge 2016).

Second and relatedly, communities with more income and education might be more adept at securing a disproportionate share of restoration funding. This would be consistent with a large literature documenting that poor and minority communities receive disproportionately less access to high quality green spaces (Sanchez et al. 2013; Stewart et al. 2014), investment in ecological improvements (Watkins et al.

2017) and fewer public services (Hastings 2009). The grant process may also favor established organizations with higher organizational capacity (Moran 2010).

Thirdly, local demographics may influence the location of restoration organizations, which in turn may influence restoration site selection. We showed that restoration organizations are more likely to occur in catchments with wealthy, white, and educated populations, and that restoration is often dependent on the existence of a local organization. Restoration organizations nationwide are also staffed overwhelmingly by non-Hispanic white employees, despite interest in environmental issues across groups (Taylor 2014). If these white staffers are more likely to initiate work (whether launching conservation nonprofits or creating a stream restoration program in their position at a public agency) where they live, rather than only in those locations with the greatest need, the unrepresentative conservation workforce could help explain the uneven distribution of restoration effort. The hyper-local nature of restoration can be positive. However, combined with the current diversity gap in the conservation workforce, an extremely local focus could result in the disproportionate concentration of restoration near populations with more wealthy non-Hispanic white residents. This is particularly true for restoration projects initiated by non-profit organizations, the locations of which are likely to be driven by community demographics.

Conclusion

In this mixed-use coastal region, we found that both ecological and social factors predict the distribution of stream restoration effort. We found that, cumulatively, restoration practitioners and funders prioritize impaired sites, sites with high human populations and impervious cover, and, for ecological restoration, sites with more native fishes (particularly salmonids). We encourage managers to consider whether a focus on the most highly impaired sites is necessarily improving condition overall. In addition, although restoration effort appears to be addressing many of the places of greatest ecological need, we identified some concerning disparities in social equitability of restoration effort. A regional or larger catchment-scale approach to restoration may help to improve the distribution of both social and ecological benefits from restoration, and we join others in calling for regional coordination to improve restoration planning and outcomes (Bernhardt & Palmer 2011; Gilvear et al. 2012; Lorenz & Feld 2013; Vietz et al. 2016). We also encourage attention to a broader array of stream types and human communities. Evaluating social equity and social justice concerns is crucial to ensuring that all communities benefit from publicly funded stream restoration projects.

Figures and tables

Figure 3.1. Number of sites by project type (a), and spending by project type (b) on stream restoration and management for the California Central Coast.

"Ecological" projects refer to on-the-ground restoration that seeks to achieve an ecological goal. "Human-oriented" projects refer to projects for stream management that seek to benefit people. "Research & monitoring" are projects performing pre- and post-project information collection or planning, but do not directly restore or alter habitats. See Table 3.1 for more information on categories. All sites with project type "unknown" were excluded from the models.

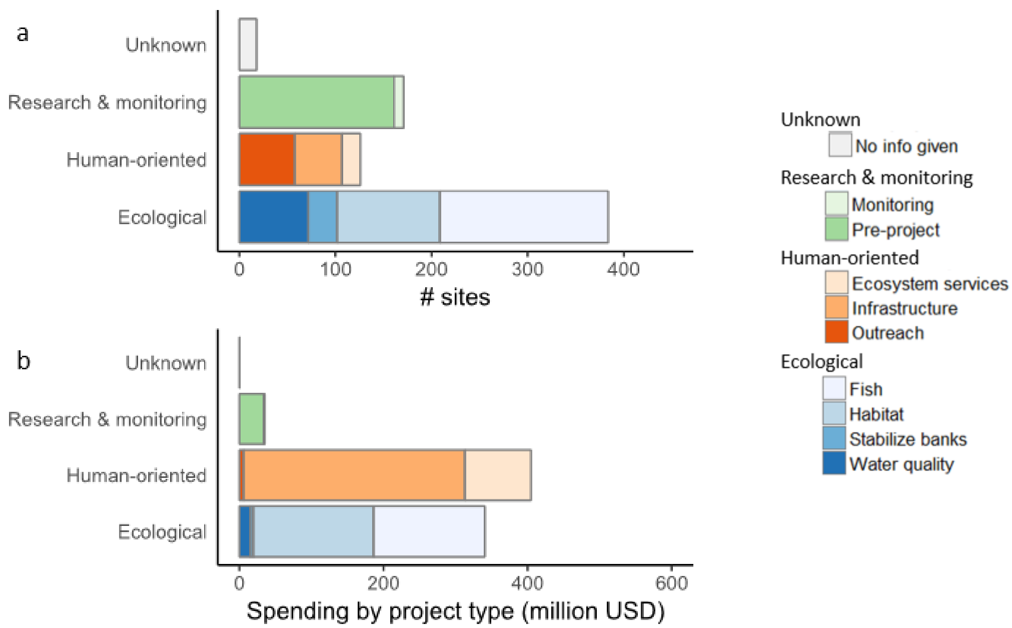


Figure 3.2. Distribution of stream restoration and management sites on the California Central Coast.

Note that most catchment units contain no restoration projects. The catchment units depicted delineate the study area.

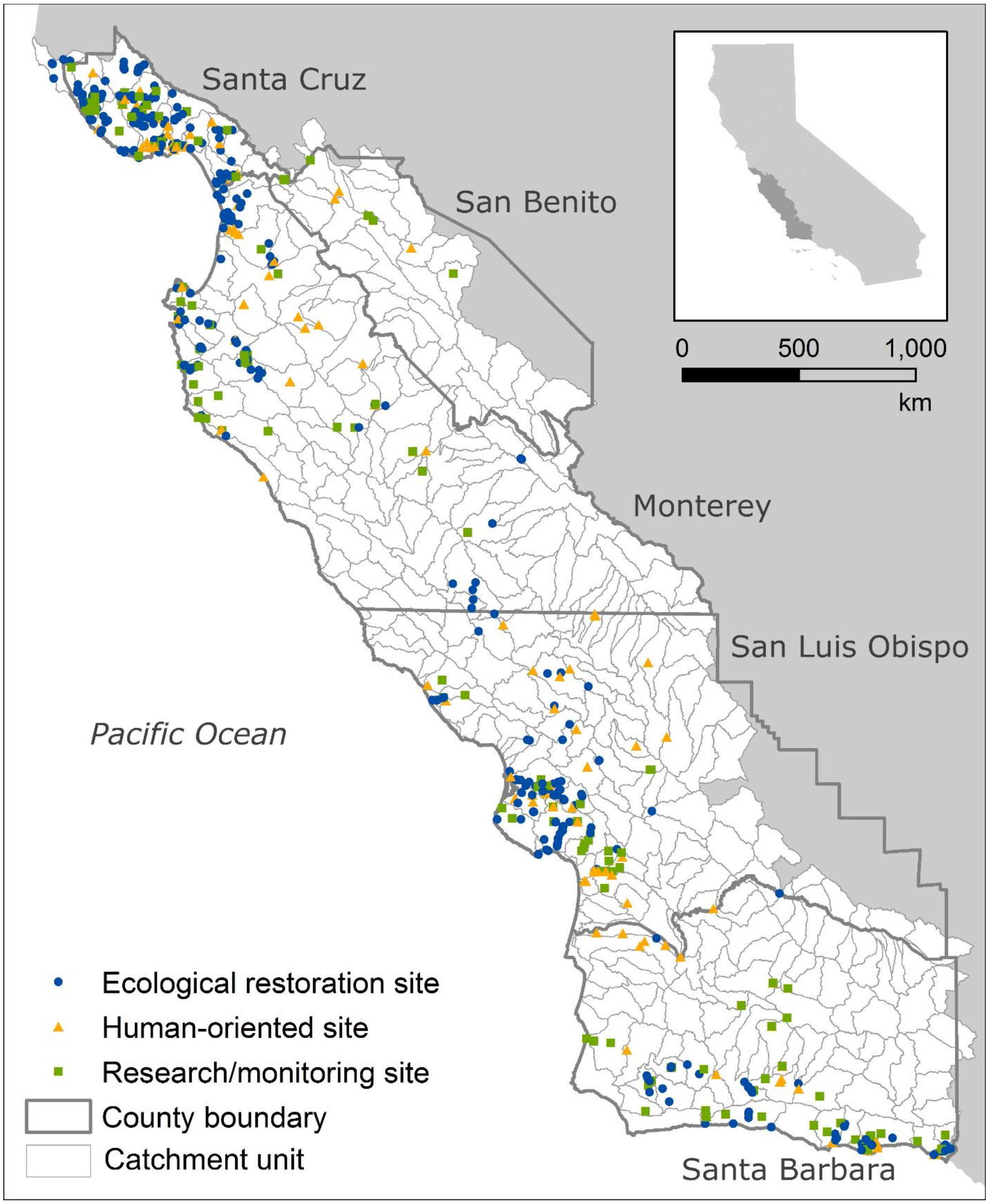


Figure 3.3. Variables predicting (a) number of restoration sites per catchment unit and (b) spending per catchment unit.

Each symbol and color represents the output of a different model. Shapes represent the standardized coefficient estimates and lines represent 95% confidence intervals. Estimates are significant when 95% confidence intervals do not overlap zero. N=310. Results also presented in Table A2.

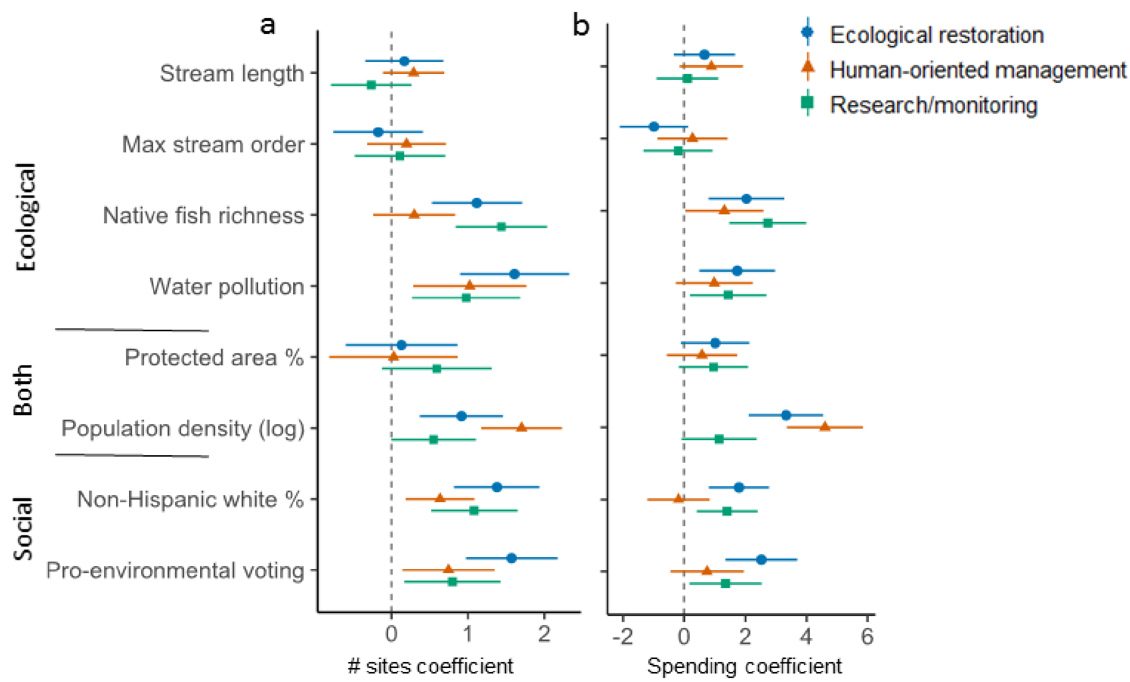


Table 3.1. Restoration project types

Description of the categories used in the analysis, including a crosswalk to the National River Restoration Science Synthesis (NRRSS) categories (Bernhardt et al. 2005).

Type	Focus	Examples	NRRSS categories
Ecological	Fish	Habitat improvement for fish, including for spawning; barrier removal; fish rearing	Fish passage, dam removal/retrofit, in-stream species management
Ecological	Habitat	In-stream physical habitat restoration; floodplain reconnection; habitat protection; riparian restoration	In-stream habitat improvement; flow modification; floodplain reconnection; land acquisition; riparian management
Ecological	Stabilize banks	Bank stabilization/erosion control	Bank stabilization
Ecological	Water quality	Sediment reduction; upland sediment retention; TMDL projects	Water quality management; storm water management
Human	Ecosystem services	Flood protection; aesthetics and public access	Aesthetics/recreation/ education
Human	Outreach	Planning for human benefit; education and outreach; training	Aesthetics/recreation/ education
Human	Infrastructure	Building roads, bridges, etc. that cross streams	
Research & monitoring	Monitoring	Maintenance; monitor project outcomes	
Research & monitoring	Pre-project	Planning; research; modeling	

Table 3.2. Variables characterizing catchments on the Central Coast.

In each case, we listed the broad categories that we wanted to capture, identified one or more indicator datasets that could provide a measurement of that category, and then calculated a metric from each dataset for use in the models. Final model terms (in bold) were selected using a correlation matrix (Appendix A) to identify independent terms representing each category of interest. All data were processed in ArcGIS and R. Data were available for entire study area unless noted otherwise.

Category	Indicator	Year	Selected metric	Source	Scale
Stream type	Steepness		Mean slope	1 arc-second Digital Elevation Model, National Elevation Dataset, USGS	30m
	Stream order		**Maximum in catchment unit**	**National Hydrography Dataset (NHD+), USGS**	
	Natural cover	2011	%	National Land Cover Database, Multi-Resolution Land Characteristics Consortium (DOI & USGS)	30m
Ecological value	**Native fish richness**	**2014**	**Count**	**123 PISCES, California Department of Fish and Wildlife**	**Catchment unit**
	Steelhead critical habitat[11]	2005	Presence/absence	National Marine Fisheries Service, NOAA	Reach

[11] Critical habitat is designated under the Endangered Species Act and is defined as “areas…[within or outside the geographical area occupied by the species]…essential to the conservation of the species” (ESA Sec 3(5)(A); 50 CFR Sec 424.02). This represents the actual or potential range of steelhead in this area.

Ecological need/ impairment	**Impaired water bodies**	**2012**	**Presence/absence of 303(d) listed waters**[12]	**California State Water Resources Control Board**	**Reach**
	Habitat condition		Minimum CRAM score[13]	EcoAtlas	Point
	Aquatic invertebrate communities	1999-2015	Minimum CSCI score[14]	California State Water Resources Control Board	Point
Access	**Protected lands**	**2016**	**% of catchment unit in protected areas**[15]	**California Protected Areas Data Portal - GreenInfo Network**	
Human impact / Population density	Impervious cover	2011	Mean percent developed imperviousness	National Land Cover Database, Multi-Resolution Land Characteristics Consortium (DOI & USGS)	30m
	Population density	**2010**	**Area-weighted mean #/km2**	**US Census Bureau, Decennial Census**	**Census block**
Human communities	Income	2015	Area-weighted mean median	US Census Bureau, American Community	Census block group

[12] Impaired water quality is defined as listing on the US Clean Water Act 303(d) list of impaired waters. Lists are generated by states and submitted to the federal Environmental Protection Agency for approval. States must evaluate "all existing and readily available information" in developing their 303(d) lists (40 C.F.R. §130.7(b) (5)).

[13] California Rapid Assessment Method (CRAM). Data present for only 1/3 of catchment units.

[14] California Stream Condition Index (CSCI). Data present for only 1/3 of catchment units.

[15] Including fee-owned areas and areas explicitly set aside as open space (excluding e.g. DoD lands). Excludes easements on private lands.

			annual income	Survey 5-year estimates	
	Education	2015	% over 25 completing 4-year college, area-weighted mean	US Census Bureau, American Community Survey 5-year estimates	Census block group
	% non-Hispanic white	**2010**	**Area-weighted mean %**	**US Census Bureau, Decennial Census**	**Census block group**
Public interest	**Pro-environmental voting**	**2006**	**% of votes cast voting yes on Proposition 84[16], area-weighted mean %**	**The Statewide Database**	**Precinct**

[16] Proposition 84 was approved in 2006 and authorized California to sell $5.4 billion in bonds to fund "water quality, safety and supply. Flood control. Natural resource protection. Park Improvements". The bond allocated 35% of the funding to conservation, and 44% to drinking water, water quality, and flood control (bondaccountability.resources.gov).

Supporting information

Figure A3.1. Correlation plot of the factors included in the full models.

All terms have been centered and standardized, and correlations are presented as percentages. Numbers represent Pearson's r. Terms representing the same category (see Table 2) were removed if correlated above 0.5.

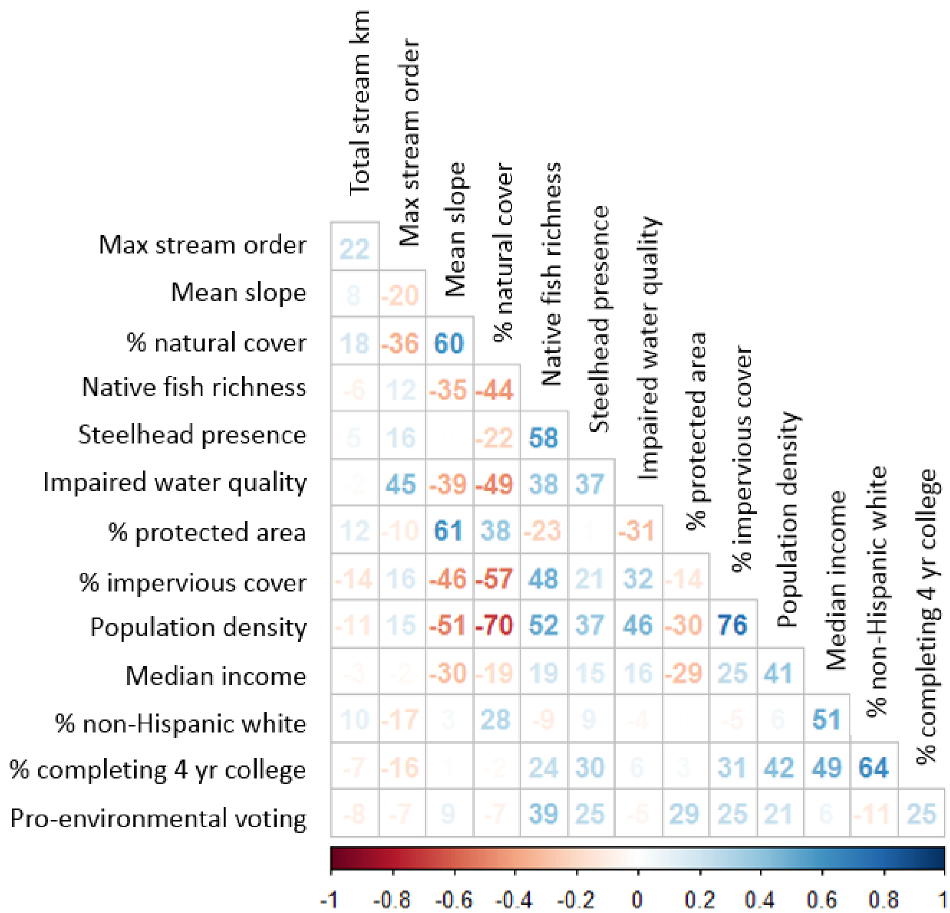


Figure A3.2. Variation of sociopolitical and biophysical predictors on the Central Coast

Each of the final model predictors is shown for the study area. One value per predictor was calculated for each HUC12, as shown here (see Table 2 for data sources). We also show steelhead-bearing streams.

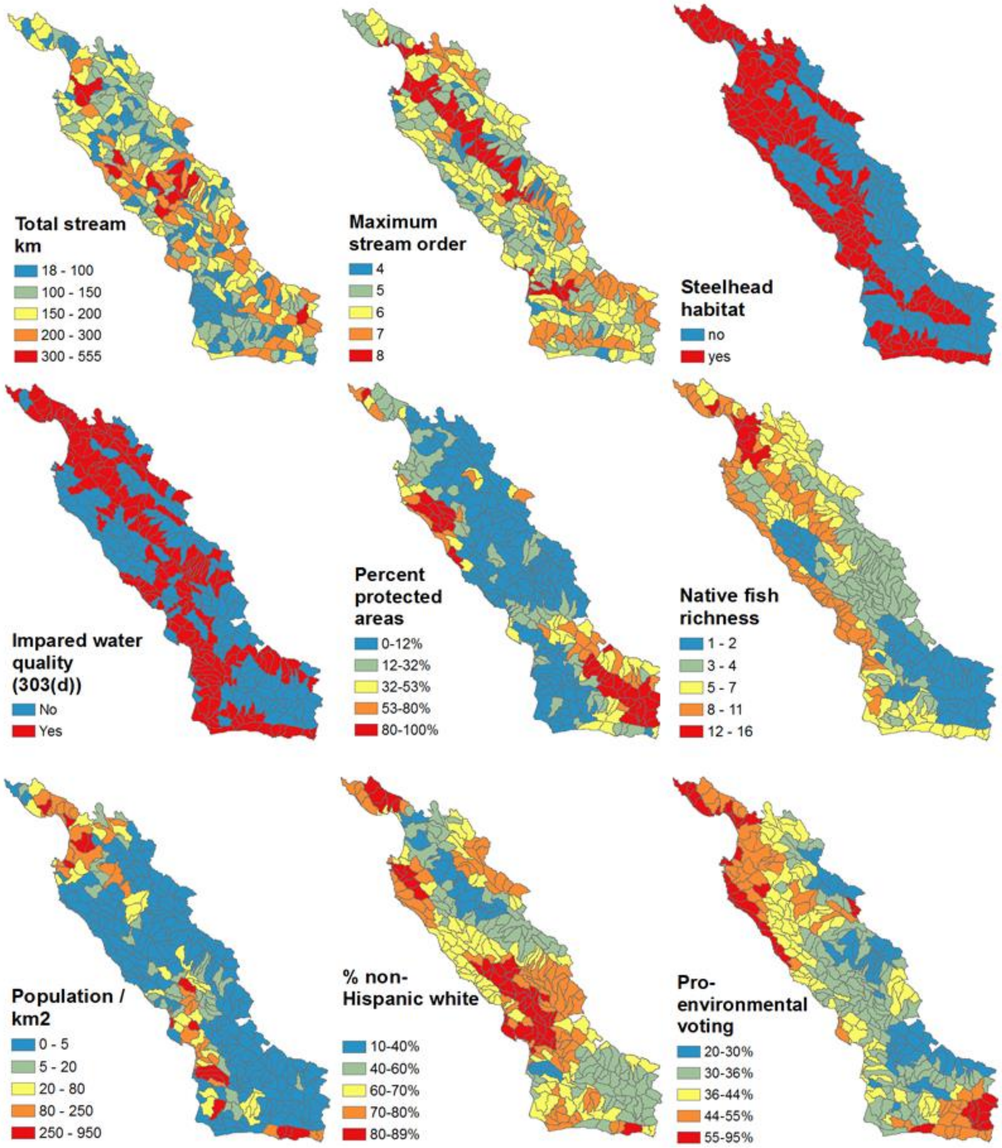


Figure A3.3. Location of restoration organizations on the Central Coast.

Distribution of organizations involved in restoration on the Central Coast. Circle size is scaled to the number of restoration projects performed by that organization.

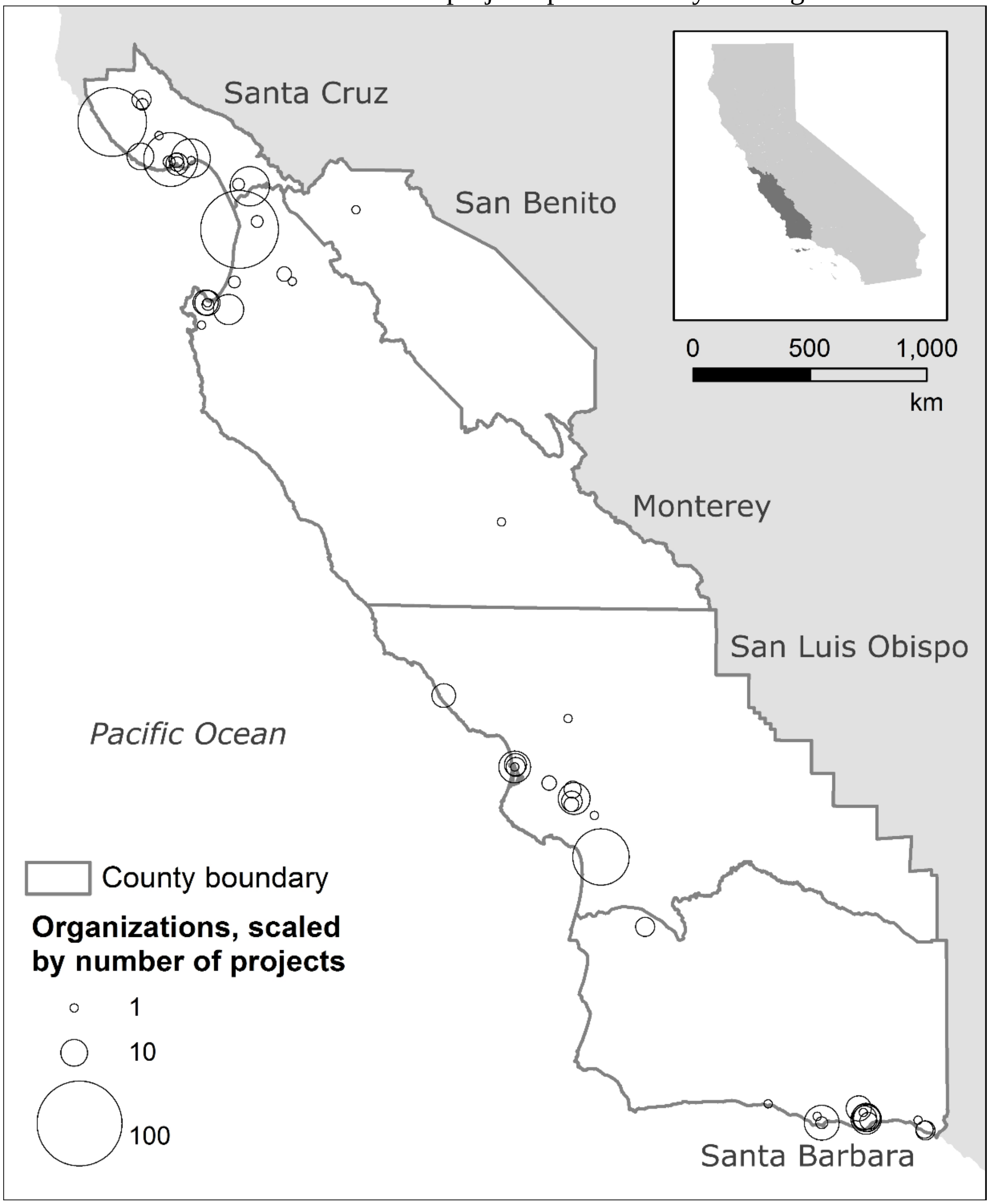


Table A3.1. Restoration database sources.

The restoration sites described in this paper were collected from the sources listed here. Most funding sources required a local funding match. The numbers presented here represent the number of unique restoration and management sites included in the final database used for modeling. Projects in category "unknown" did not provide any information about actions or goals.

Source Database	Managing/Granting agency	Description	Projects in final database	Ecological projects	Human projects	Info projects	Un-known	Spending info?
Fisheries Restoration Grant Program	California Department of Fish and Wildlife	Grant program to support salmonids and improve deteriorating fish habitat through projects from sediment reduction to watershed education throughout coastal California. Funded projects types include a broad range of in-stream, riparian, upslope, monitoring, planning, and fish barrier projects.	354	190	46	116	2	Yes
California Proposition 84	State Water Resources Control Board	Safe drinking water, water quality and supply, flood control river and coastal protection bond act. Supports grant program for water quality and storm water management.	129	44	56	29		Yes
National River Restoration Science Synthesis	Multiple	Compilation of existing project databases, see Bernhardt et al. 2005	45	20	6	16	3	Partial
EcoAtlas	Multiple	Projects completed by Moss Landing Marine Labs and entered into EcoAtlas	66	53	7	1	5	No

Source Database	Managing/Granting agency	Description	Projects in final database	Ecological projects	Human projects	Info projects	Un-known	Spending info?
Federal Clean Water Act 319H funding	State Water Resources Control Board and USEPA	Federal nonpoint source pollution control program providing grant funds for projects to control impacts to beneficial uses and limit pollutant effects.	37	30		6	1	Partial
California Natural Resources Project Inventory	Multiple	Database of over 8000 natural resource projects in California. Not currently maintained.	29	21	6	1	1	Partial
California Proposition 12	California State Parks	Safe Neighborhood Parks, Clean Water, Clean Air, and Coastal Protection Bond Act of 2000. Supports grant program for state investment in public open spaces and clean water protection.	7	5	1		1	Yes
Habitat Conservation Fund	California Department of Parks and Recreation	Grant program to support public outdoor recreation projects, including nature interpretation programs, protection of plant and animal species, and acquisition and development of wildlife corridors and trails	6	5	1			Partial
California Proposition 1e	California Department of Water Resources	Disaster preparedness and flood protection bond act of 2006.	13	10	1	2		Yes

Source Database	Managing/Granting agency	Description	Projects in final database	Ecological projects	Human projects	Info projects	Un-known	Spending info?
Urban Streams Restoration Program	California Department of Water Resources	Grant program dispersing funds from California Propositions 40 and 13 for reducing flooding and erosion, restoring ecological value of streams, and promoting community stewardship	5	3	2			Yes
Other/ unknown		Small local agencies or source unknown	18	11			2	No
Total			**709**	**392**	**126**	**171**	**15**	

Table A3.2. Model coefficients for restoration effort by catchment unit.

All factors have been standardized (subtract the mean and divide by 2 sd), to enable direct comparison of coefficients. Each column represents a different model; the reduced models are also presented in Figure 3. The first seven models and the steelhead and organizations models are negative binomial models (log link). The three spending models are linear models (log transformed). The final model (Steelhead catchments) models ecological restoration for the 153 catchment units designated as (potential) steelhead habitat, and "with habitat" models the 101 watersheds with habitat information. The other models all have N=310. P-values indicated as follows: ***(0-0.001), ** (0.0011-0.01), *(0.011-0.05), . (0.051-0.1)

Category	Indicator	Ecological			Human-oriented		Research/monitoring		Spending			Organization	Steelhead catchments
		Full	Reduced	With habitat	Full	Reduced	Full	Reduced	Ecological	Human	Research		
	(Intercept)	-2.68 (0.37)***	-2.14 (0.29)***	-0.29 (0.34)	-2.78 (0.36)***	-2.57 (0.31)***	-2.58 (0.37)***	-2.03 (0.28)***	1.6 (0.42)***	1.98 (0.4)***	1.71 (0.4)***	-4.61 (0.63)***	-0.82 (0.34)*
Covariate	Total stream km	0.13 (0.27)	0.16 (0.26)	-0.13 (0.29)	0.32 (0.22)	0.29 (0.21)	-0.32 (0.28)	-0.26 (0.27)	0.57 (0.54)	0.89 (0.52).	0.11 (0.52)	0.67 (0.3)*	0.16 (0.28)
Water-shed type	Stream order (max)	-0.05 (0.31)	-0.18 (0.3)	0.02 (0.33)	0.18 (0.28)	0.2 (0.26)	0.13 (0.31)	0.11 (0.3)	-0.15 (0.61)	0.27 (0.58)	-0.2 (0.58)	0 (0.43)	-0.2 (0.35)
	Steepness (mean)	0.24 (0.36)			0.42 (0.36)		0.17 (0.35)						
	Natural cover (%)	0.33 (0.43)			-0.32 (0.36)		0.09 (0.43)						
Ecological value	Native fish richness	0.91 (0.34)**	1.12 (0.3)***	0.7 (0.38).	0.19 (0.29)	0.3 (0.27)	1.14 (0.34)***	1.44 (0.31)***	1.9 (0.67)**	1.31 (0.65)*	2.73 (0.64)***	0.59 (0.43)	0.66 (0.34).
	Steelhead presence (Y/N)	1.08 (0.39)**			0.4 (0.35)		1.08 (0.4)**						
Ecological condition	Water pollution (Y/N)	1.35 (0.37)***	1.66 (0.37)***	1.01 (0.43)*	0.99 (0.4)*	1.02 (0.38)**	0.67 (0.37).	0.98 (0.36)**	1.4 (0.67)*	0.98 (0.65)	1.44 (0.64)*	0.78 (0.65)	1.2 (0.42)**
	Physical habitat condition (min)			-1.02 (0.43)*									
Access	Protected area (%)	-0.03 (0.42)	0.12 (0.37)	-0.32 (0.52)	-0.14 (0.46)	0.03 (0.43)	0.35 (0.42)	0.59 (0.37)	0.75 (0.61)	0.58 (0.59)	0.96 (0.58).	0.51 (0.83)	-0.03 (0.47)
Human impact	Impervious cover (mean)	-0.14 (0.32)			0.2 (0.24)		-0.15 (0.31)						
	Population/ km2 (log)	1.13 (0.48)*	0.89 (0.28)**	0.53 (0.4)	1.3 (0.45)**	1.7 (0.27)***	0.46 (0.48)	0.55 (0.28).	2.22 (0.66)***	4.61 (0.64)***	1.15 (0.63).	2.96 (0.56)***	0.96 (0.31)**
Social	Household income (median)	0.32 (0.32)			-0.07 (0.29)		0.47 (0.32)						
	Non-Hispanic white (%)	0.54 (0.48)	1.39 (0.29)***	1.36 (0.34)***	0.43 (0.43)	0.64 (0.23)**	0.17 (0.49)	1.08 (0.29)***	1.65 (0.54)**	-0.19 (0.52)	1.4 (0.51)**	0.89 (0.38)*	1.37 (0.33)***
	Education (% w/ 4yr degree)	0.24 (0.46)			0.36 (0.43)		0.46 (0.46)						
Interest	Pro-environmental voting (%)	1.34 (0.33)***	1.6 (0.31)***	1.44 (0.41)***	0.64 (0.35).	0.75 (0.31)*	0.55 (0.35)	0.8 (0.32)*	2.08 (0.63)**	0.74 (0.61)	1.35 (0.6)*	1.27 (0.52)*	1.53 (0.39)***

Appendix 3B. Additional methods: Robustness checks

Most catchment units have zero restoration sites, so to compare the performance of our model to a model simply describing the presence or absence of restoration within the catchment, we modeled presence/absence of ecological restoration using a logistic model and the reduced model terms. We also re-ran the reduced ecological restoration model on the full dataset using a log-linear model instead of negative binomial to test for sensitivity to distribution choice. In both cases, the sign and significance of coefficients were unchanged.

We were concerned that the inclusion of two university towns (Santa Cruz and Santa Barbara) skewed our results, but the observed patterns were consistent even after these two towns and their immediate surroundings were excluded from our dataset. We also re-ran the ecological restoration and research & monitoring models after excluding all Coho habitat from the model (in 12 Santa Cruz County catchments), and found consistent results.

We compared patterns in voting for our chosen proposition (CA Proposition 84) and other recent statewide environmental bond measures (Proposition 1E: passed in 2006; Proposition 1: 2014; Proposition 12: 2000). At the county level, the correlation between percent of votes cast in support of each proposition was >0.8. We substituted voting records for Proposition 1E at the HUC12 level into the model, and found the same relationship with restoration (coefficient estimate=0.96, SE=0.36, model AIC=605; as compared to values for Proposition 84: coefficient=1.43, SE=0.29, model AIC=594). Although voting for each proposition varies, the general patterns appear to be maintained, as others have found (Press 2003). We chose Proposition 84 for our models because it elicited the most variation in votes.

To assess whether our results were driven by the fish focus of the largest restoration database in our study (the Fisheries Restoration Grant Program), we excluded all projects from that database and reran our models. The results were robust - although there were slight changes in coefficient estimates and slightly larger confidence intervals following the exclusion of the many projects in this database, there were no changes in significance or direction of effect for any of the model terms.

We tested the residuals from our full ecological model for spatial autocorrelation using plots of semivariance (Zuur et al. 2009). We used the *gstat* package in R to calculate and plot variograms of the deviance residuals, and found no evidence of spatial autocorrelation (Pebesma 2004; Gräler & Pebesma 2016).

The linear models for spending had poor fit due to the many zero values. We refit the models using a tobit regression (for censored data; package *censReg* (Henningsen 2017)). The coefficient estimates were larger, but patterns of significance and direction were unchanged, so we retained the simpler linear models in the paper.

References

Acuña V, Hunter M, Ruhí A. 2017. Managing temporary streams and rivers as unique rather than second-class ecosystems. Biological Conservation **211**:12–19.

Allan JD. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics **35**:257–284.

Allan JD, Erickson DL, Fay J. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology **37**:149–161.

Angrist JD, Pischke J-S. 2009. Mostly harmless econometrics: an empiricist's companion. Princeton University Press, Princeton, NJ.

Arar EJ, Collins GB. 1997. In vitro determination of Chlorophyll a and Pheophytin a in marine and freshwater algae by fluorescence.

Arce E, Archaimbault V, Mondy CP, Usseglio-Polatera P. 2014. Recovery dynamics in invertebrate communities following water-quality improvement: taxonomy- vs trait-based assessment. Freshwater Science **33**:1060–1073.

Aronson J et al. 2010. Are socioeconomic benefits of restoration adequately quantified? A meta-analysis of recent papers (2000-2008) in Restoration Ecology and 12 other scientific journals. Restoration Ecology **18**:143–154.

Ash M, Fetter TR. 2004. Who lives on the wrong side of the tracks? Evidence from the EPA's risk screening environmental indicators model. Social Science Quarterly **85**.

Babin N, Mullendore ND, Prokopy LS. 2016. Using social criteria to select watersheds for non-point source agricultural pollution abatement projects. Land Use Policy **55**:327–333. Elsevier Ltd.

Baker ME, Weller DE, Jordan TE. 2006. Improved methods for quantifying potential nutrient interception by riparian buffers. Landscape Ecology **21**:1327–1345.

Barnas KA, Katz SL, Hamm DE, Diaz MC. 2015. Is habitat restoration targeting relevant ecological needs for endangered species? Using Pacific Salmon as a case study. Ecosphere **6**:1–42.

Bartels LM. 2008. Unequal Democracy: The Political Economy of the New Gilded Age. Princeton University Press, Princeton.

Bates D, Maechler M, Bolker B, Walker S. 2015. Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software **67**:10–48.

Baxter C V, Fausch KD, Carl Saunders W. 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. Freshwater Biology **50**:201–

220.

Beechie T, Pess G, Roni P, Giannico G. 2008. Setting River Restoration Priorities: A Review of Approaches and a General Protocol for Identifying and Prioritizing Actions. North American Journal of Fisheries Management **28**:891–905.

Beechie TJ, Sear D a., Olden JD, Pess GR, Buffington JM, Moir H, Roni P, Pollock MM. 2010. Process-based Principles for Restoring River Ecosystems. BioScience **60**:209–222.

Beer WN, Anderson JJ. 2013. Sensitivity of salmonid freshwater life history in western US streams to future climate conditions. Global Change Biology **19**:2547–2556.

Benayas JMR, Newton AC, Diaz A, Bullock JM. 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. Science **325**:1121–4.

BenDor T, Brozovic N, Pallathucheril VG. 2008. The social impacts of wetland mitigation policies in the United States. Journal of Planning Literature **22**:341–357.

BenDor TK, Livengood A, Lester TW, Davis A, Yonavjak L. 2015. Defining and evaluating the ecological restoration economy. Restoration Ecology **23**:209–219.

Bergey EA, Getty GM. 2006. A review of methods for measuring the surface area of stream substrates. Hydrobiologia **556**:7–16.

Bernhardt ES et al. 2005. Synthesizing US river restoration efforts. Science **308**:636–637.

Bernhardt ES et al. 2007. Restoring rivers one reach at a time: results from a survey of U.S. river restoration practitioners. Restoration Ecology **15**:482–493.

Bernhardt ES, Palmer MA. 2011. River Restoration : the fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecological Applications **21**:1926–1931.

Bivand RS, Pebesma E, Gomez-Rubio V. 2013. Applied spatial data analysis with R. Springer, New York, NY.

Bizzi S, Lerner N. 2015. The use of stream power as an indicator of channel sensitivity to erosion and deposition processes. River Research and Applications **30**:16–27.

Bogan MT, Boersma KS, Lytle D a. 2013. Flow intermittency alters longitudinal patterns of invertebrate diversity and assemblage composition in an arid-land stream network. Freshwater Biology **58**:1016–1028.

Bogan MT, Hwan JL, Carlson SM. 2015. High aquatic biodiversity in an intermittent coastal headwater stream at Golden Gate National Recreation Area, California. Northwest Science **89**:186–195.

Borcard D, Gillet F, Legendre P. 2011. Numerical Ecology with R. Page (Gentleman R, Hornik K, Parmigiani GG, editors). Springer.

Brancalion PHS, Cardozo IV, Camatta A, Aronson J, Rodrigues RR. 2013. Cultural ecosystem services and popular perceptions of the benefits of an ecological restoration project in the Brazilian Atlantic forest. Restoration Ecology **22**:65–71.

Bunte K, Abt SR. 2001. Sampling surface and subsurface particle-size distributions in wadable gravel- and cobble-Bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. USDA Forest Service, Rocky Mountain Research Station.

Burdon FJ, McIntosh AR, Harding JS. 2013. Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. Ecological Applications **23**:1036–47.

Burnham KP, Anderson DR. 2002. Model selection and multimodel inference: a practical information-theoretic approach, 2nd edition. Springer-Verlag, New York.

Cade B. 2015. Model averaging and muddled multimodal inferences. Ecology **96**:2370–2382.

Calcagno V. 2013. glmulti: Model selection and multimodel inference made easy. R package version 1.0.7.

Central Coast Regional Water Quality Control Board, State Water Resources Control Board, California Environmental Protection Agency. 2016. Water quality control plan for the Central Coast Basin.

Chan KMA, Shaw MR, Cameron DR, Underwood EC, Daily GC. 2006. Conservation planning for ecosystem services. PLoS Biology **4**:2138–2152.

Christian-Smith J, Merenlender AM. 2010. The disconnect between restoration goals and practices: A case study of watershed restoration in the Russian River Basin, California. Restoration Ecology **18**:95–102.

Clewell AF, Aronson J. 2006. Motivations for the restoration of ecosystems. Conservation Biology **20**:420–428.

Cockerill K, Anderson WP. 2014. Creating false images: Stream restoration in an urban setting. Journal of the American Water Resources Association **50**:468–482.

Corenblit D, Tabacchi E, Steiger J, Gurnell AM. 2007. Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: A review of complementary approaches. Earth-Science Reviews **84**:56–86.

Cummins KW, Merritt RW, Berg MB. 2008. Ecology and distribution of aquatic insects. Pages 105–122in R. W. Merritt, K.W.Cummins, and M. B. Berg, editors.An introduction to the aquatic insects of North AmericaFourth. Endall/Hunt Publishing Company, Dubuque, IO.

DeFries R, Hansen A, Turner BL, Reid R, Liu J. 2007. Land use change around protected areas: management to balance human needs and ecological function. Ecological Applications **17**:1031–8.

Dixon SJ, Sear DA, Odoni NA, Sykes T, Lane SN. 2016. The effects of river restoration on catchment scale flood risk and flood hydrology. Earth Surface Processes and Landforms **41**:997–1008.

Egan D, Hjerpe EE, Abrams J, editors. 2011. Human dimensions of ecological restoration: Integrating science, nature, and culture. Society for Ecolgoical Restoration. Island Press.

Fahrig L. 2003. Effects of Habitat Fragmentation on Biodiversity. Annual Review of Ecology, Evolution, and Systematics **34**:487–515.

Flies EJ, Skelly C, Negi SS, Prabhakaran P, Liu Q, Liu K, Goldizen FC, Lease C, Weinstein P. 2017. Biodiverse green spaces: A prescription for global urban health. Frontiers in Ecology and the Environment **15**:510–516.

Fremier AK, Kiparsky M, Gmur S, Aycrigg J, Craig RK, Svancara LK, Goble DD, Cosens B, Davis FW, Scott JM. 2015. A riparian conservation network for ecological resilience. Biological Conservation **191**:29–37.

Fullerton AH, Beechie TJ, Baker SE, Hall JE, Barnas KA. 2006. Regional patterns of riparian characteristics in the interior Columbia River Basin, Northwestern USA: Applications for restoration planning. Landscape Ecology **21**:1347–1360.

Gelman A. 2008. Scaling regression inputs by dividing by two standard deviations. Statistics in Medicine **27**:2865–2873.

Gerstner K, Moreno-Mateos D, Gurevitch J, Beckmann M, Kambach S, Jones HP, Seppelt R. 2017. Will your paper be used in a meta-analysis? Make the reach of your research broader and longer lasting. Methods in Ecology and Evolution **8**:777–784.

Gilens M, Page BI. 2014. Testing theories of American politics: elites, interest groups, and average citizens. Perspectives on Politics **12**:564–581.

Gilvear DJ, Casas-Mulet R, Spray CJ. 2012. Trends and issues in delivery of integrated catchment scale river restoration: lessons learned from a national river restoration survey within Scotland. River Research and Applications **28**:234–246.

GreenInfo Network. 2017. California Protected Areas Database. Available from http://www.calands.org/data (accessed July 20, 2017).

Grueber CE, Nakagawa S, Laws RJ, Jamieson IG. 2011. Multimodel inference in ecology and evolution: Challenges and solutions. Journal of Evolutionary Biology **24**:699–711.

Gurevitch J, Hedges L V. 2001. Meta-analysis: combining the results of independent experiments. Page in S. M. Scheiner and J. Gurevitch, editors. Design and Analysis of Ecological Experiments. Oxford University Press, Incorporated, Cary, NC.

Gurevitch J, Koricheva J, Nakagawa S, Stewart G. 2018. Meta-analysis and the science of research synthesis. Nature **555**:175–182. Nature Publishing Group.

Gurnell A. 2014. Plants as river system engineers. Earth Surface Processes and Landforms **39**:4–25.

Gurnell AM, Corenblit D, García de Jalón D, González del Tánago M, Grabowski RC, O'Hare MT, Szewczyk M. 2016. A conceptual model of vegetation-hydrogeomorphology interactions within river corridors. River Research and Applications **32**:142–163.

Hagger V, Dwyer J, Wilson K. 2017. What motivates ecological restoration? Restoration Ecology **25**:832–843.

Harrington J, Born M. 2000. Measuring the Health of California Streams and Rivers: a methods manual for water resource professionals, citizen monitors, and natural resources studentssecond. Sustainable Land Stewardship International Institute, Sacramento, CA.

Hastings A. 2009. Poor neighbourhoods and poor services: Evidence on the "rationing" of environmental service provision to deprived neighbourhoods. Urban Studies **46**:2907–2927.

Hawley RJ. 2018. Making Stream Restoration More Sustainable: A Geomorphically, Ecologically, and Socioeconomically Principled Approach to Bridge the Practice with the Science. BioScience **68**:517–528.

Hedges L V, Gurevitch J, Curtis PS. 1999. The meta-analysis of response ratios in experimental ecology. Ecology **80**:1150–1156.

Hillman M. 2004. The importance of environmental justice in stream rehabilitation. Ethics, Place and Environment **7**:19–43.

Holl KD, Aide TM. 2011. When and where to actively restore ecosystems? Forest

Ecology and Management **261**:1558–1563. Elsevier B.V.

Holl KD, Crone EE, Schultz CB. 2003. Landscape Restoration: Moving from Generalities to Methodologies. BioScience **53**:491.

Hutchens JJ, Wallace JB, Grubaugh JW. 2017. Transport and Storage of Fine Particulate Organic Matter. Page Methods in Stream Ecology. Elsevier Inc.

Isaak DJ, Wollrab S, Horan D, Chandler G. 2012. Climate change effects on stream and river temperatures across the northwest U.S. from 1980 – 2009 and implications for salmonid fishes:499–524.

Jähnig SC, Lorenz AW, Hering D, Antons C, Sundermann A, Jedicke E, Haase P. 2011. River restoration success: a question of perception. Ecological Applications **21**:2007–15.

Jellinek S. 2017. Using prioritisation tools to strategically restore vegetation communities in fragmented agricultural landscapes. Ecological Management and Restoration **18**:45–53.

Jones HP, Jones PC, Barbier EB, Blackburn RC, Benayas JMR, Holl KD, McCrackin M, Meli P, Montoya D, Moreno-Mateos D. 2018. Restoration and repair of Earth's damaged ecosystems. Proceedings of the Royal Society B: Biological Sciences **285**.

Jones HP, Schmitz OJ. 2009. Rapid recovery of damaged ecosystems. PloS ONE **4**:e5653.

Kahn ME. 2002. Demographic change and the demand for environmental regulation. Journal of Policy Analysis and Management **21**:45–62.

Kail J, Brabec K, Poppe M, Januschke K. 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. Ecological Indicators **58**:311–321.

Kashiwagi JH. 1985. Soil survey of Marin County, California. United States Department of Agriculture, Soil Conservation Service.

Kondolf GM, Anderson S, Lave R, Pagano L, Merenlender A, Bernhardt ES. 2007. Two decades of river restoration in California: What can we learn? Restoration Ecology **15**:516–523.

Lajeunesse MJ. 2009. Meta-Analysis and the Comparative Phylogenetic Method. The American Naturalist **174**:369–381.

Lake PS. 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society **19**:573–592.

Langridge S. 2016. Social and biophysical context influences county-level support for collaborative watershed restoration: Case study of the Sacramento River, CA, USA. Ecological Restoration **34**:285–296.

Larsen S, Pace G, Ormerod SJ. 2011. Experimental effects of sediment deposition on the structure and function of macroinvertebrate assemblages in temperate streams. River Research and Applications **27**:257–267.

Larsen S, Vaughan IP, Ormerod SJ. 2009. Scale-dependent effects of fine sediments on temperate headwater invertebrates. Freshwater Biology **54**:203–219.

Lefcheck JS. 2016. piecewiseSEM: Piecewise structural equation modeling in R for ecology, evolution, and systematics. Methods in Ecology and Evolution **7**:573–579.

Legendre P, Gallagher ED. 2001. Ecologically meaningful transformations for ordination of species data. Oecologia **129**:271–280.

Legendre P, Legendre L. 2012. Numerical EcologyThird. Elsevier.

Leite MDS, Tambosi LR, Romitelli I, Metzger JP. 2013. Landscape ecology perspective in restoration projects for biodiversity conservation : a review. Natureza & Conservacao **11**:108–118.

Lennox MS, Lewis DJ, Jackson RD, Harper J, Larson S, Tate KW. 2011. Development of vegetation and aquatic habitat in restored riparian sites of California's North Coast rangelands. Restoration Ecology **19**:225–233.

Light A. 2006. Restorative relationships: From artifacts to natural systems. Page in R. France, editor. Healing Nature, Repairing Relationships: Landscape Architecture and the Restoration of Ecological Spaces. The MIT Press, Cambridge, MA.

Lindenmayer DB, Manning AD, Smith PL, Possingham HP, Fischer J, Oliver I, McCarthy MA. 2002. Approach and Focal-Species a Critique Landscape Restoration : Conservation Biology **16**:338–345.

Lorenz AW, Feld CK. 2013. Upstream river morphology and riparian land use overrule local restoration effects on ecological status assessment. Hydrobiologia **704**:489–501.

Louhi P, Mykrä H, Paavola R, Huusko A, Vehanen T, Mäki-Petäys A, Muotka T. 2011. Twenty years of stream restoration in Finland: little response by benthic macroinvertebrate communities. Ecological Applications **21**:1950–61.

Maron M, Hobbs RJ, Moilanen A, Matthews JW, Christie K, Gardner T a., Keith D a., Lindenmayer DB, McAlpine C a. 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biological Conservation **155**:141–148. Elsevier Ltd.

Matono P, Bernardo JM, Oberdorff T, Ilhéu M. 2012. Effects of natural hydrological variability on fish assemblages in small Mediterranean streams: Implications for ecological assessment. Ecological Indicators **23**:467–481. Elsevier Ltd.

Matthaei CD, Piggott JJ, Townsend CR. 2010. Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. Journal of Applied Ecology **47**:639–649.

Mayer PM, Reynolds SK, McCutchen MD, Canfield TJ. 2007. Meta-analysis of nitrogen removal in riparian buffers. Journal of environmental quality **36**:1172–80.

McCafferty WP. 1981. Aquatic Entomology: the fishermen's and ecologists' illustrated guide to insects and their relatives. Science Books International, Boston, MA.

McCrackin ML, Jones HP, Jones PC, Moreno-Mateos D. 2016. Recovery of lakes and coastal marine ecosystems from eutrophication: A global meta-analysis. Limnology and Oceanography **62**:507–518.

Meli P, Benayas JMR, Balvanera P, Ramos MM. 2014. Restoration enhances wetland biodiversity and ecosystem service supply, but results are context-dependent: A meta-analysis. PLoS ONE **9**:e93507.

Meli P, Holl KD, Rey Benayas JM, Jones HP, Jones PC, Montoya D, Moreno Mateos D. 2017. A global review of past land use, climate, and active vs. passive restoration effects on forest recovery. Plos One **12**:e0171368.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: Multiscale assessments, Vol. 4. Synthesis report series. Island Press, Washington, D.C.

Miller SW, Budy P, Schmidt JC. 2010. Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. Restoration Ecology **18**:8–19.

Mohai P, Pellow D, Roberts JT. 2009. Environmental Justice. Annual Review of Environment and Resources **34**:405–430.

Moilanen A, Wilson KA, Possingham H, editors. 2009. Spatial conservation prioritization: Quantitative methods and computational tools. Oxford University Press, Oxford, UK.

Montgomery DR, Buffington JM, Smith RD, Schmidt KM, Pess G. 1995. Pool Spacing in Forest Channels. Water Resources Research **31**:1097–1105.

Moore AA, Palmer MA. 2005. Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. Ecological Applications **15**:1169–1177.

Moran S. 2010. Cities, creeks, and erasure: Stream restoration and environmental justice. Environmental Justice **3**:61–69.

Morandi B, Piégay H, Lamouroux N, Vaudor L. 2014. How is success or failure in river restoration projects evaluated? Feedback from French restoration projects. Journal of Environmental Management **137**:178–188.

Moreno-Mateos D, Barbier EB, Jones PC, Jones HP, Aronson J, López-López JA, McCrackin ML, Meli P, Montoya D, Rey Benayas JM. 2017. Anthropogenic ecosystem disturbance and the recovery debt. Nature Communications **8**:14163.

Moreno-Mateos D, Power ME, Comín FA, Yockteng R. 2012. Structural and functional loss in restored wetland ecosystems. PLoS biology **10**:e1001247.

Muller I, Delisle M, Ollitrault M, Bernez I. 2016. Responses of riparian plant communities and water quality after 8 years of passive ecological restoration using a BACI design. Hydrobiologia **781**:67–79.

Naiman J, Decamps H. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics **28**:621–658.

Nakagawa S, Schielzeth H. 2013. A general and simple method for obtaining R2 from generalized linear mixed-effects models. Methods in Ecology and Evolution **4**:133–142.

National Centers for Environmental Information. 2018. Climate Data Online. Available from https://www.ncdc.noaa.gov/cdo-web/datasets (accessed May 14, 2018).

National Marine Fisheries Service. 2013. South-Central California Coast Steelhead Recovery Plan. West Coast Region, California Coastal Area Office, Long Beach, California.

National Marine Fisheries Service (NOAA Fisheries). 2005. Endangered Species Act Critical Habitat: Steelhead. Available from https://map.dfg.ca.gov/metadata/ds0122.html (accessed October 8, 2017).

Needham PR, Usinger RL. 1956. Variability in the macrofauna of a single riffle in Prosser Creek, California, as indicated by the Surber sampler. Hilgardia **24**.

Nieuwenhuis R, Grotenhuis M te, Pelzer B. 2012. influence.ME: tools for detecting influential data in mixed effects models. R Journal **4**:38–47.

Nilsson C, Riis T, Sarneel JM, Svavarsdóttir K. 2018. Ecological Restoration as a Means of Managing Inland Flood Hazards. BioScience **68**:89–99.

Norton DJ, Wickham JD, Wade TG, Kunert K, Thomas J V, Zeph P. 2009. A method for comparative analysis of recovery potential in impaired waters restoration planning. Environmental Management **44**:356–68.

Ode P. 2003. List of Californian Macroinvertebrate Taxa and Standard Taxonomic

Effort. CAMLnet Aquatic Bioassessment Laboratory, CA Department of Fish and Game, Rancho Cordova, CA.

Ode PR. 2007. Standard operating procedures for collecting macroinvertebrate samples and associated physical and chemical data for ambient bioassessments in California. California State Water Resources Control Board Surface Water Ambient Monitoring Program (SWAMP) Bioassessment SOP 001.

Oksanen J et al. 2017. vegan: Community Ecology Package. R package version 2.4-5. https://CRAN.R-project.org/package=vegan.

Opperman JJ, Merenlender AM. 2007. Living trees provide stable large wood in streams. Earth Surface Processes and Landforms **32**:1229–1238.

Palmer MA, Menninger HL, Bernhardt E. 2010. River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? Freshwater Biology **55**:205–222.

Parkyn SM, Davies-Colley RJ, Halliday NJ, Costley KJ, Croker GF. 2003. Planted riparian buffer zones in New Zealand: Do they live up to expectations? Restoration Ecology **11**:436–447.

Paul MJ., Meyer JL. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics **32**:333–365.

Pearce J, Kingham S, Zawar-Reza P. 2006. Every breath you take? Environmental justice and air pollution in Christchurch, New Zealand. Environment and Planning A **38**:919–938.

Pebesma EJ, Bivand RS. 2005. Classes and methods for spatial data in R. R News **5**.

Peterson GD. 2002. Estimating Resilience Across Landscapes. Ecology and Society **6**:17.

Pluntke T, Kozerski HP. 2003. Particle trapping on leaves and on the bottom in simulated submerged plant stands. Hydrobiologia **506**–**509**:575–581.

Poff NL, Olden JD, Vieira NKM, Finn DS, Simmons MP, Kondratieff BC. 2006. Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. Journal of the North American Benthological Society **25**:730–755.

Pusey BJ, Arthington AH. 2003. Importance of the riparian zone to the conservation and management of freshwater fish : a review. Marine and Freshwater Research **54**:1–16.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Resh VH, Jackson JK. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. Pages 195–133in D. M. Rosenberg and V. H. Resh, editors.Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, NY.

Roni P, Hanson K, Beechie T. 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North American Journal of Fisheries Management **28**:856–890.

Rosenberg DM, Resh VH. 1993. Introduction to freswhater biomonitoring and benthic macroinvertebrates. Pages 1–10in D. M. Rosenberg and V. H. Resh, editors.Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, NY.

Roth NE, Allan JD, Erickson DL. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. Landscape Ecology **11**:141–156.

Roy AH, Capps KA, El-Sabaawi RW, Jones KL, Parr TB, Ramirez A, Smith RF, Walsh CJ, Wenger SJ. 2016. Urbanization and stream ecology: diverse mechanisms of change. Freshwater Science **36**:1–7.

Roy AH, Purcell AH, Walsh CJ, Wenger SJ. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society **28**:908–910.

Roy AH, Rosemond AD, Paul MJ, Leigh DS, Wallace JB. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). Freshwater Biology **48**:329–346.

Ruhl JB, Salzman J. 2006. The effects of wetland mitigation banking on people. National Wetlands Newsletter **28**.

Ruiz-Jaen MC, Aide TM. 2005. Restoration Success: How is it being measured? Restoration Ecology **13**:569–577.

Sanchez GM, Nejadhashemi AP, Zhang Z, Woznicki SA, Habron G, Marquart-Pyatt S, Shortridge A. 2013. Development of a socio-ecological environmental justice model for watershed-based management. Journal of Hydrology **518**:162–177.

Sedell JR, Reeves GH, Hauer FR, Stanford JA, Hawkins CP. 1990. Role of refugis in recovery from disturbances:modern fragmented and disconnected river systems. Environmental Management **14**:711–724.

SFBRWQCB. 2007. Water quality monitoring and bioassessment in nine San Francisco Bay Region watersheds: Walker Creek, Lagunitas Creek, San Leandro Creek, Wildcat Creek/San Pablo Creek, Suisun Creek, Arroyo Las Positas, Pescadero Creek/Butano Creek, San Gregorio Creek, an. Oakland, CA: Surface Water Ambient Monitoring Program, San Francisco Bay Regional Water Quality Control Board.

Sievers M, Hale R, Morrongiello JR. 2017. Do trout respond to riparian change? A meta-analysis with implications for restoration and management. Freshwater Biology **62**:445–457.

Smucker NJ, Detenbeck NE. 2014. Meta-analysis of lost ecosystem attributes in urban streams and the effectiveness of out-of-channel management practices. Restoration Ecology **22**:741–748.

Solari L, Oorschot M Van, Belletti B, Hendriks D, Rinaldi M, Vargas-Luna A. 2016. Advances on modelling riparian vegetation - hydromorphology interactions. River Research and Applications **32**:164–178.

Sponseller RA, Benfield EF. 2001. Influences of land use on leaf breakdown in southern Appalachian headwater streams: a multiple-scale analysis. Journal of North American Benthological Society **20**:44–59.

Standish RJ et al. 2014. Resilience in ecology: Abstraction, distraction, or where the action is? Biological Conservation **177**:43–51.

Standish RJ, Hobbs RJ, Miller JR. 2012. Improving city life: Options for ecological restoration in urban landscapes and how these might influence interactions between people and nature. Landscape Ecology **28**:1213–1221.

Statzner B, Gore JA, Resh VH, Journal S, American N, Society B. 1988. Hydraulic stream ecology: observed patterns and potential applications. Journal of the North American Benthological Society **7**:307–360.

Steinman AD, Lamberti GA, Leavitt PR, Uzarski DG. 2017. Biomass and pigments of benthic algae. Pages 223–241in G. A. Lamberti and F. R. Hauer, editors.Methods in Stream Ecology, Volume 1Third. Els.

Stewart IT, Bacon CM, Burke WD. 2014. The uneven distribution of environmental burdens and benefits in Silicon Valley's backyard. Applied Geography **55**:266–277.

Stoddard JL, Larsen DP, Hawkins CP, Johnson RK, Norris RH. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications **16**:1267–76.

Stranko SA, Hilderbrand RH, Palmer MA. 2012. Comparing the fish and benthic macroinvertebrate diversity of restored urban streams to reference streams. Restoration Ecology **20**:747–755.

Suding BK, Higgs E, Palmer M, Callicott JB, Anderson CB, Gutrich JJ, Hondula KL, Lafevor MC, Larson BMH, Randall A. 2015. Committing to ecological restoration. Science **348**:638–640.

Suding KN. 2011. Toward an era of restoration in ecology: Successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics

42:465–487.

Suganuma MS, Durigan G. 2015. Indicators of restoration success in riparian tropical forests using multiple reference ecosystems. Restoration Ecology **23**:238–251.

Sundermann A, Gerhardt M, Kappes H, Haase P. 2013. Stressor prioritisation in riverine ecosystems: Which environmental factors shape benthic invertebrate assemblage metrics? Ecological Indicators **27**:83–96. Elsevier Ltd.

Sundermann A, Stoll S, Haase P. 2011. River restoration success depends on the species pool of the immediate surroundings. Ecological Applications **21**:1962–71.

Suttle KB, Power ME, Levine JM, McNeely C. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications **14**:969–974.

Sweeney BW, Newbold JD. 2014. Streamside Forest Buffer Width Needed to Protect Stream Water Quality, Habitat, and Organisms: A Literature Review. Journal of the American Water Resources Association **50**:560–584.

Taylor DE. 2014. The state of diversity in environmental organizations. Green 2.0.

Tonkin JD, Altermatt F, Finn DS, Heino J, Olden JD, Pauls SU, Lytle DA. 2018. The role of dispersal in river network metacommunities: Patterns, processes, and pathways. Freshwater Biology **63**:141–163.

Tummers B. 2006. DataThief III. <http://datathief.org/>.

Turner RK, Daily GC. 2007. The ecosystem services framework and natural capital conservation. Environmental and Resource Economics **39**:25–35.

Urban MC, Skelly DK, Burchsted D, Price W, Lowry S. 2006. Stream communities across a rural-urban landscape gradient. Diversity and Distributions **12**:337–350.

USGS. 2018. 3DEP products and services: The National Map.

Venables WN, Ripley BD. 2002. Modern Applied Statistics with S. Springer, New York, NY.

Viechtbauer W. 2010. Conducting meta-analyses in R with the metafor package. Journal of Statistical Software **36**:1–48.

Vietz GJ, Rutherfurd ID, Fletcher TD, Walsh CJ. 2016. Thinking outside the channel: Challenges and opportunities for protection and restoration of stream morphology in urbanizing catchments. Landscape and Urban Planning **145**:34–44. Elsevier B.V.

Villeneuve B, Piffady J, Valette L, Souchon Y, Usseglio-Polatera P. 2018. Direct and indirect effects of multiple stressors on stream invertebrates across watershed, reach and site scales: A structural equation modelling better informing on hydromorphological impacts. Science of the Total Environment **612**:660–671. Elsevier B.V.

Violin CR, Cada P, Sudduth EB, Hassett BA, Penrose DL, Bernhardt ES. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications **21**:1932–49.

Wahl CM, Neils A, Hooper D. 2013. Impacts of land use at the catchment scale constrain the habitat benefits of stream riparian buffers. Freshwater Biology **58**:2310–2324.

Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan RP. 2005. The urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society **24**:706–723.

Warton DI, Hui FKC. 2011. The arcsine is asinine: The analysis of proportions in ecology. Ecology **92**:3–10.

Watkins SL, Mincey SK, Vogt J, Sweeney SP. 2017. Is planting equitable? An examination of the spatial distribution of nonprofit urban tree-planting programs by canopy cover, income, race, and ethnicity. Environment and Behavior **49**:452–482.

Weber C, Peter A. 2011. Success or failure? Do indicator selection and reference setting influence river rehabilitation outcome? North American Journal of Fisheries Management **31**:535–547.

Wei T, Simko V. 2016. corrplot: Visualization of a correlation matrix. R package version 0.77. https://CRAN.R-project.org/package=corrplot.

Wickham H. 2009. ggplot2: Elegant graphics for data analysis. Springer-Verlag, New York.

Wiggins GB. 1977. Larvae of the North American Caddisfly Genera (Trichoptera). University of Toronto Press, Toronto and Buffalo.

Wilson MJ, McTammany ME. 2014. Tributary and mainstem benthic macroinvertebrate communities linked by direct dispersal and indirect habitat alteration. Hydrobiologia **738**:75–85.

Wohl E, Bledsoe BP, Jacobson RB, Poff NL, Rathburn SL, Walters DM, Wilcox AC. 2015a. The natural sediment regime in rivers: broadening the foundation for ecosystem management. BioScience **65**:358–371.

Wohl E, Lane SN, Wilcox AC. 2015b. The science and practice of river restoration. Water Resources Research **51**:5974–5997.

Wolch JR, Byrne J, Newell JP. 2014. Landscape and urban planning urban green space, public health, and environmental justice: The challenge of making cities "just green enough." Landscape and Urban Planning **125**:234–244. Elsevier B.V.

Wolman MG. 1954. A methods of sampling coarse river-bed material. Transactions of the American Geophysical Union **35**.

Wortley L, Hero J-M, Howes M. 2013. Evaluating ecological restoration success: A review of the literature. Restoration Ecology **21**:537–543.

Wu X, Cutter B. 2011. Who votes for public environmental goods in California? Evidence from a spatial analysis of voting for environmental ballot measures. Ecological Economics **70**:554–563. Elsevier B.V.

Zong L, Nepf H. 2011. Spatial distribution of deposition within a patch of vegetation. Water Resources Research **47**:1–12.

Zuur A, Ieno EN, Walker N, Saveliev AA, Smith GM. 2009. Mixed effects models and extensions in ecology with R. Springer Sciences.