Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals

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Abstract

Ecological restoration has grown rapidly and now encompasses not only classic ecological theory but also utilitarian concerns, such as preparedness for climate change and provisioning of ecosystem services. Three dominant perspectives compete to influence the science and practice of river restoration. A strong focus on channel morphology has led to approaches that involve major Earth-moving activities, such as channel reconfiguration with the unmet assumption that ecological recovery will follow. Functional perspectives of river restoration aim to regain the full suite of biogeochemical, ecological, and hydrogeomorphic processes that make up a healthy river, and though there is well-accepted theory to support this, research on methods to implement and assess functional restoration projects is in its infancy. A plethora of new studies worldwide provide data on why and how rivers are being restored as well as the project outcomes. Measurable improvements postrestoration vary by restoration method and measure of outcome.

> Could this old world, which man has overthrown, be rebuilded, could human cunning rescue its wasted hillsides and its deserted plains from solitude or mere nomade occupation, from barrenness, from nakedness, and from insalubrity, and restore the ancient fertility and healthfulness. . . . (Marsh 1864, p. 47)

1. INTRODUCTION

The first well-known attempt to restore land began in 1934, when Aldo Leopold launched the Curtis Prairie project in Wisconsin, though the idea was certainly not new. As the quotation above indicates, as early as 1864, when the first edition of *Man and Nature* by George Marsh was published, the concept of restoration was already being considered. Yet despite those early efforts, ecological restoration did not enter the mainstream of scientific thought until the 1980s. Today, restoration ecology as a science is extremely young, but it is growing rapidly. The number of research publications has risen exponentially over the past decade, and they are not limited to the journal *Restoration Ecology* but appear in a broad array of scientific journals. Although it has largely been an applied science and remains so, the theoretical basis of restoration ecology is firmly rooted in classic ecology; restoration projects provide unique opportunities to test much of that theory. There is a strong emphasis on understanding what factors enhance restoration of biodiversity (Rodrigues et al. 2011), on the role of physical habitat heterogeneity in the rate and degree of recovery (Holl et al. 2013), and on the use of resilience theory—thresholds, state changes, and feedbacks (Suding et al. 2004)—in understanding the potential for an ecosystem to be restored. Spatial ecology also plays a prominent role in restoration research, particularly including a focus on dispersal dynamics, metapopulation theory, and the landscape context of restoration sites (Reynolds et al. 2013).

Restoration ecologists today consider many of those ecological theories together with an environmental forecasting perspective to ask questions, such as: What species combinations can be expected to coexist postrestoration given future climate regimes? What is the relationship between those future assemblages, ecosystem function, and ecosystem services? Should ecologists embrace the concept of novel ecosystems and replace scientific work on restoration with a focus on intervention ecology that seeks to manipulate systems to meet future needs (Hobbs et al. 2011)? Or should they merely seek to restore desired ecological processes or products (Bullock et al. 2011)? These topics are not unlike those that philosophers and environmental ethicists have been struggling with for many years: Is restoration of nature even possible? What is a natural system and how does that relate to human intentionality (Gobster & Hull 2000)? However, for restoration ecologists these issues are not merely matters of philosophy. As policy makers, managers, and funders of restoration projects increasingly embrace the concept of ecosystem services, focus is shifting from restoration to achieving a previous or least-disturbed ecological condition to what ecosystems can provide for humans (Benayas et al. 2009, Kline et al. 2013). This shift in turn is changing the type of science demanded from restoration ecologists. For example, the scientific focus on testing ecological theory that underlies restoration (e.g., metapopulation dynamics) may instead be replaced by science focused on quantifying the role a particular species plays in supporting ecosystem processes or products useful to humans (Montoya et al. 2012).

A shift toward the more utilitarian view of restoration for ecosystem services is particularly pronounced for running-water ecosystems and, as we describe below, this view is having a major impact on the types of studies and findings that are being reported (Gilvear et al. 2013, Palmer et al. 2014). Perhaps one of the reasons a utilitarian view of restoration is so pronounced for running waters is because the wide dependence of people on riverine ecosystems—for water, transportation, food, and more—has rendered rivers objects of human use for centuries. Yet, because rivers occupy low-lying points on the landscape, all human activities in a watershed influence the recovery potential of rivers; thus, even in regions distant from the channel, meeting human needs and desires can be at odds with what is needed to restore a river (Kondolf & Yang 2008).

The dependence of humans on water has also resulted in major political and legal battles that have influenced river restoration (e.g., Gerlak et al. 2013). Because freshwaters are protected by law in many countries while property rights on the land that influences rivers are also protected, the outcomes of political, regulatory, and legal battles can lead to changes in how restoration is defined and what science is brought to bear (Doremus & Tarlock 2013, Iovanna & Griffiths 2006). One of the most important pieces of environmental legislation passed in the United States is the Clean Water Act, which makes explicit reference to restoration—the goal is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters" [33 U.S.C. § 1251(a)]. Because urban development and natural resource extraction often result in degradation of running waters, restoration of a degraded waterway elsewhere is legally required as compensatory mitigation. Less comprehensive but related requirements exist in Canada, Germany, the Netherlands, Sweden, and the United Kingdom (Tischew et al. 2010).

We begin this review by situating riverine restoration within the body of theory and science that underpins three dominant perspectives among restoration scientists on how to approach restoring a river. We discuss when and how these perspectives emerged as well as the role of personalities, human needs, and other social forces that influenced their adoption. We then turn to the body of science on the ecological effectiveness of restoration and how that relates to these three perspectives. Throughout when we use the phrase "river restoration," we are referring to the full range of running-water systems, from intermittent headwaters to mid-order streams to large rivers.

2. DOMINANT PERSPECTIVES INFLUENCING THE SCIENCE AND PRACTICE OF RESTORATION

The emergence of restoration ecology as applied to rivers and streams is much more recent than that for terrestrial ecosystems, and its theoretical basis has been more grounded in the physical sciences, especially hydrology and geomorphology, than in the ecological sciences. Dominant perspectives on the science and practice of river restoration are driven by social dynamics both external to and internal to the community of scientists and restoration practitioners. We highlight three perspectives that shape the field today that largely represent divergent views on which biophysical factors and approaches are most important in restoring river ecosystems. Our discussion also notes the complex social factors, changing policies, and legal decisions that have influenced their emergence and adoption by different groups of scholars, practitioners, and managers.

2.1. Restoration as Channel Design

Flow has long been considered a master variable in riverine ecosystems because it, along with sediment dynamics, directly affects channel form and consequently the biota and ecological processes within the channel. Researcher and practitioner communities agree that full ecological restoration requires consideration of flow and sediment regimes, but controversies arise over how similar the regimes must be to the historical or reference site "range of variability" (Poff et al. 2010). There is further disagreement over whether manipulating various aspects of the flow regime (e.g., restoring or reducing peak flows) is sufficient for full ecosystem recovery. A great deal of focus has been on restoring natural flow regimes by managing releases from dams to better mimic historical flood levels downstream (Arthington & Pusey 2003, Hart et al. 2002). However, reconfiguring the channel to accommodate degraded hydrogeomorphic regimes is more common than restoring the hydrologic and sediment regimes themselves. As a result, such projects become channel design activities and fail to address the issues necessary for true restoration, like water allocation or increasing infiltration capacity of watershed soils. Channel reconfiguration is especially common in urban and agricultural regions in which river channels may be reshaped and boulders, wood, and rock deflectors added to slow water flow. In addition, banks may be armored with material to minimize erosion from excessive runoff (Levell & Chang 2008, Walsh et al. 2005).

The design perspective has been championed largely by engineers but also by some hydrologists and geomorphologists. The unfounded assumption is that once the channel can handle the prevailing flow and sediment fluxes, then species assemblages, primary production, decomposition, nutrient processing, and other ecological processes will be restored (i.e., the "field of dreams hypothesis"; Palmer et al. 1997). The design perspective has been influenced more by a single individual—Dave Rosgen—than any other scientific researcher or research tradition (Lave 2013). Rosgen developed the Natural Channel Design (NCD) method, which uses a channel classification and form-based template approach to determine what morphologic configuration the design should seek to ensure stability (Rosgen 1998). He claims this approach will restore the chemical, physical, and biological functions of a river that is self-regulating and exhibits a stable channel (Rosgen 2011), yet the method does not address chemical or biological processes. Despite this shortcoming, NCD and other channel design approaches are the most common stream restoration approaches (González del Tánago et al. 2012, Nagle 2007).

The majority of channel design projects are implemented by consultants in the business of restoration, and social science scholars have argued that this has resulted in "privately produced science" dominating the field of stream restoration (Lave et al. 2010). Rosgen himself runs a major consulting business, produced a guidebook on restoration, and teaches classes focused on his stepwise method for channel design. Although failures are common (see Section 4), the NCD methods can be easily adopted by others, and thus Rosgen filled a training void for practitioners that academics have not. Methods like NCD that require heavy equipment, engineering designs, and construction personnel are expensive to implement and therefore very lucrative for businesses. The combination of training materials, the profit factor, and Rosgen's charismatic personality has contributed significantly to the heavy reliance on the NCD approach by practitioners in the private sector.

A fairly well-defined group of academic scientists have pushed for a process-based approach to channel design instead of the form-based classification approach of Rosgen (Simon et al. 2007). Nonetheless, the focus in this approach still centers on channel morphology—how to design a channel given the local discharge and sediment regime in the context of a particular watershed and landscape [see the stream restoration toolbox of the National Center for Earth-Surface Dynamics (**http://www.nced.umn.edu**); Smith et al. 2011].

Most stream restoration projects today are implemented with a primary focus on channel form or physical structures rather than on ecological processes (Lake et al. 2007, Wortley et al. 2013). Channel width, depth, and slope are manipulated such that, in theory, the channel will not aggrade or degrade under the local hydrogeomorphic conditions. A number of failures, as well as strong critiques of channel design for stability and the Rosgen NCD approach (Buchanan et al. 2012, Lave et al. 2010, Niezgoda & Johnson 2005, Simon et al. 2007, Smith & Prestegaard 2005), have prompted some hydrologists and geomorphologists to broaden the focus from fixed channel form to including the concept of a dynamic equilibrium in which the channel is free to change over time (Kline & Cahoon 2010, Wheaton et al. 2008). Some have emphasized that working to achieve a stable channel may in some cases help protect infrastructure but is not a form of restoration (Shields et al. 2003).

Although the dynamic approach focused on restoring processes is not yet broadly applied in practice, it calls for a shift in the design process from reliance only on a template or channel classification to a quantitative approach based on hydrogeomorphic processes, theory, empirical field methods, and limited modeling (Kline & Cahoon 2010, Wohl et al. 2005). Yet, ecologists have pointed out that though restoration of hydrogeomorphology is a critical consideration in restoring many streams, it is typically not sufficient for degraded channels, and it can even lead to worsening the ecological condition of the stream; i.e., it may be viewed as a disturbance itself (Louhi et al. 2011, Tullos et al. 2009). For example, in restoring floodplain overflow potential, if riparian trees are removed from a previously closed-canopy stream, the underlying energy regime may change from allochthonous resources to one driven by primary production, which may shift the stream further away from the desired ecological state and often toward algae-dominated streambeds and higher temperatures (Sudduth et al. 2011). Similarly, if the hydrologic regime is restored but there is no nearby source of invertebrate colonists, then the in-stream communities will remain unrestored (Sundermann et al. 2011). Finally, an over-reliance on channel design may obfuscate efforts to identify the factor that most limits recovery of a stream; quite often this factor is water quality, and thus ecological recovery will not occur until the source of pollutants is removed (Kail et al. 2012, Selvakumar et al. 2010).

2.2. Restoration of Ecological Function

An emerging emphasis in river restoration research is to include the restoration of ecological functions. This is in part related to a push by ecologists for a more comprehensive process-based restoration (Beechie et al. 2010), i.e., one that goes beyond hydrogeomorphic processes to include restoration of ecological processes. But this push also represents a backlash to the form-based approach and its high failure rate (Section 4.2), not only from a geomorphic perspective but from an ecological perspective (Bernhardt et al. 2005, Kondolf et al. 2006). Thus, functional ecological restoration includes efforts specifically targeted at restoring critical structural ecosystem features (e.g., riparian vegetation) and critical ecological processes, such as nutrient dynamics (e.g., flux or uptake of nutrients), the input of organic matter, and productivity (Beechie et al. 2010, Bernhardt & Palmer 2011). Which processes and structures are most critical to restore vary depending on what the stressors are for a particular channel and which of those stressors must be reduced or removed for the project to be successful over time.

Reducing stormwater or agricultural runoff to streams and restoring riparian vegetation are essential implementation measures for recovering natural processes in many degraded streams (see Section 2.3). Similarly, restoring longitudinal connectivity of river segments is needed for some restoration projects to ensure that dispersal of riparian plant propagules and fish is not deterred (Hart et al. 2002). Dispersal of stream insects to restored headwaters requires the near proximity of healthy headwaters with a colonist pool; restoration potential may be very limited when entire watersheds are impaired (Parkyn & Smith 2011). Biogeochemical processes and infiltration properties in riparian soils (González del Tánago & García de Jalón 2006), hyporheic exchange rates with the surface water (Hester & Gooseff 2010), and hydraulic connectivity with the floodplain (Opperman et al. 2010, Schneider et al. 2011) must be similar to those in reference sites to ensure the recovery of hydrologic and biogeochemical processes (Roley et al. 2012).

Interest in functional restoration has also emerged for legal and social reasons. The increasing emphasis on ecosystem services by researchers, agencies, and nonprofit groups is leading to interest in how best to restore them; this in turn requires understanding what ecological processes support each service and how to recover them (Palmer & Filoso 2009). Indeed, the scientific community is pushing research forward to develop ecological production functions, or quantitative relationships between ecosystem services and ecosystem processes (Febria et al. 2014). Restoration scientists collaborating with practitioners can use such relationships to help identify what types of actions or combination of actions might best restore the desired processes (Bullock et al. 2011, Palmer & Filoso 2009).

Recent US court cases and scholarly challenges asserting that restoration practices used for compensatory mitigation do not comply with the legal requirements to replace all lost ecological functions have also contributed to the growing emphasis on functional restoration (Doyle & Shields 2012, Forman 2011). Ecologists are actively researching stream functions and their responses to restoration actions as well as how best to assess functional restoration (Bunn et al. 2010, Gabriele et al. 2013, Hoellein et al. 2012). Most of this research has focused on using direct measures of processes, such as whole-stream metabolism, nitrogen (N) uptake, or rates of decomposition, to assess levels of degradation and responses postrestoration (Palmer & Febria 2012). However, natural resource managers, agencies, and practitioners are largely following a separate path, seeking to identify assessment methods on the basis of a hierarchical classification of functions (Fischenich 2006) called a functional pyramid (Harman et al. 2012). The hierarchy is not meant to imply that the (hydrologic) functions at the base of the pyramid should be the primary focus of every restoration project but instead that hydrologic processes influence almost every other process in streams. Although the functional pyramid concept was designed to guide practitioners in functional assessment methods, many of the measurement parameters suggested (Harman et al. 2012, Appendix A, part c) are not true functional measures (i.e., rates of processes) but surrogates assumed to represent functions, such as measures of meander-width ratio assumed to represent channel migration/stability. For the pyramid assessment method to be scientifically sound, empirical evidence that each surrogate significantly predicts function needs to be empirically validated.

Although functional restoration and functional measures are widely viewed by ecologists, and increasingly by government agencies, as required for effective restoration and assessment, restoration in regulatory, resource management, and practitioner contexts remains largely a design process that is evaluated structurally (see Section 3.2). Process-based approaches and assessment are most often not used because they take longer and may require more engagement with stakeholders and because the science of functional restoration is still evolving.

2.3. Restoration Beyond the Channel and Beyond Disciplinary Silos

Most restoration projects are completed at reach scales—typically 1 km or less of stream length—and involve manipulating the channel itself. Yet the sources of impacts to streams are largely generated outside of the channel in the watershed. As with restoration of any ecosystem, the most successful and sustainable approaches should target the source of degradation and focus on the appropriate scale. Removal of nonnative species from streams could be considered restoration targeting the degradation source; however, the introduction or establishment of nonnative species is often related to factors outside of the channel. In any case, once stressors, such as nonnatives, uncontrolled runoff, or pollutant inputs, are removed, restoration theory suggests that a stream should recover on its own (Falk et al. 2006). This form of restoration is the ultimate type of functional restoration because the stressors exert their impact by influencing the processes, both ecological and physical, that define healthy rivers (Gilvear et al. 2013). Though they are rare compared with projects focused on channel form (Sections 3.1 and 3.2), projects involving dam removal, implementation of best management practices in the watershed (e.g., stormwater infrastructure in urban settings), and reforestation are excellent examples of functional restoration targeting problems at their source (Arthington & Pusey 2003, Walsh et al. 2005, Wang et al. 2002).

Restoring river channels through actions beyond the channel vastly expands the scholarship that is required. Researchers working at the interface of hydrology, geomorphology, and ecology are making important contributions to functional restoration by advancing our understanding of the complex interactions between vegetation, groundwater, river flows, channel morphology, and water quality (Booth & Loheide 2010, Hall et al. 2014). But much of the needed scholarship is from the social sciences. Despite the appeal of achieving a self-sustaining ecosystem through an out-of-stream approach directed at the underlying causes of degradation, immense social pressures work against this tactic (Christian-Smith & Merenlender 2010, Gobster & Hull 2000, Gross 2003). Actions that are implemented up in the watershed may butt up against individual property-owner rights and thus require extensive engagement with many landowners and stakeholders in the watershed (Fletcher et al. 2011). Nevertheless, recent failures to restore river ecosystems through in-stream interventions have led to a push to seek out-of-stream solutions (Kline & Cahoon 2010). This approach is particularly pronounced in urban settings, in which uncontrolled stormwater runoff is a major source of river degradation, yet little progress has been made in the ecological recovery of urban rivers. As a result, scientists are pressing managers and regulators to look to the watershed for opportunities to implement highly distributed projects that hopefully will have the additive effect of increasing infiltration in the uplands (Walsh et al. 2005). The number of projects monitoring streams before and following watershed-scale restoration is increasing, and there is now evidence of many successes (Smucker & Detenbeck 2014). For example, afforestation of an agricultural watershed has been shown to successfully reduce runoff, improve summer base flows, and decrease channel erosion, resulting in a macroinvertebrate community similar to forested regions (Quinn et al. 2009). Restoration of wetlands or implementations of other watershed best management practices have also led to enhanced stream biodiversity (Ramchunder et al. 2012, Wang et al. 2002).

3. QUANTITATIVE STUDIES OF RESTORATION PROJECTS

Beginning in 2005, a series of articles was published summarizing the status of river restoration in the United States that documented the major goals and motivating factors for restoration (Bernhardt et al. 2005, Palmer et al. 2005). These articles arose from a coordinated effort by more than 20 researchers (mostly ecologists) to document what was being done to restore rivers in the United States, who was doing it, how much it cost, and how effective it was. The National River Restoration Science Synthesis (NRRSS) project documented for the first time the complex reasons for which rivers are restored and the extremely low rate of monitoring relied upon to determine their ecological outcomes. Any monitoring that was completed was largely focused on whether a project was implemented as planned or whether the structural integrity of the newly built channel persisted for some fixed period of time. When the NRRSS project was completed there were insufficient objective data to determine the effectiveness of projects—particularly with respect to ecological status (Palmer et al. 2007). Even in the Pacific Northwest of the United States—a region that made up some 60% of the more than 37,000 projects in the NRRSS database, which were mostly focused on recovery of endangered salmon—almost half of the projects lacked success criteria, yet the majority of project managers interviewed believed their projects were successful (Katz et al. 2007, Rumps et al. 2007). Fortunately, lack of sufficient effectiveness data is no longer the situation.

3.1. Studies Synthesizing General Trends

The NRRSS project was the beginning of a dramatic increase in scientific interest in stream restoration on the part of ecologists as evidenced by a steady increase in journal articles focused on the topic. There has not yet been another synthesis of comparable size; however, there is now a wealth of published studies providing data on the outcome of restoration projects (Section 4.2). There are regional assessments of the state of river restoration (Gilvear et al. 2012, Haase et al. 2012, Robson et al. 2009, Sundermann et al. 2011) and major syntheses efforts for the Nordic countries (e.g., Hagen et al. 2013) and the European Union (EU; **www.reformrivers.eu**). In fact, the EU Water Framework Directive (WFD) was a major impetus for new work, because along with legislation that expanded the scope of water protection to all surface waters and groundwater, it set targets for achieving "good ecological status" for all rivers (Haase et al. 2012). Restoration became a major focus for achieving those targets. Significant funding by the EU and individual countries was made available to academic researchers to accelerate understanding of how best to restore freshwater ecosystems (Morandi et al. 2014).

Recent studies that summarize the status of river restoration for particular regions outside the United States suggest many similarities with what was found in the NRRSS studies (Bernhardt et al. 2007, Palmer et al. 2007). This is particularly the case for the types of actions undertaken for restoration (e.g., channel reconfiguration, addition of in-stream structures, etc.), even though ecological goals may differ among geographic regions (see REFORM's (REstoring rivers FOR effective catchment Management's) "FORECASTER" website at **http://wiki.reformrivers.eu/index.php/Forecaster_Description**). For example, as in the United States, reconfiguring channels with earth-moving equipment remains common throughout Australia, Canada, China, Europe, Japan, and beyond (Amirault 2012, Brooks & Lake 2007, González del Tánago et al. 2012, Morandi et al. 2014, Yu et al. 2012). In Japan, still waterways and aquatic biota are highly valued, and so restoration efforts have often focused on manipulating dam flow releases, widening channels, and augmenting channels with gravel (Nakamura et al. 2006). Less is known about river restoration in China, but what has been published suggests a strong focus on engineering and channel design similar to the NCD approach (Yong-Xing et al. 2008, Yu et al. 2010). River restoration in Korea is more recent and has a strong focus on engineered designs and aesthetics, although there has been more recent attention to ecological values (Jeong et al. 2011, Woo 2010).

3.2. Restoration Goals and Implementation Methods

Supplemental Material

We compiled information on 644 restoration projects from 149 published studies that provide quantitative information on the effectiveness of restoration projects implemented using different actions (**Supplemental Table 1**; follow the **Supplemental Material link** from the Annual Reviews home page at **http://www.annualreviews.org**). It is important to note that to ensure accuracy of the conclusions, we review only studies using direct field measurements and exclude those that modeled outcomes or estimated them using surrogate measures.

We found that the most common goals were related to increasing biodiversity, stabilizing channels, improving riparian and in-stream habitat, and improving water quality (**Figure 1*a***). There were also a small number of projects ($N = 11$ out of the 644) with the main goal of mitigation. The methods used to restore the streams were dominated by physical manipulations of the channels (**Figure 1*b***); this is a similar finding to the observation made 10 years ago by the NRRSS project. Channel hydromorphic projects using, for example, the NCD approach, involved reconfiguring the channel, such as moving it laterally, adding sinuosity, or raising/lowering the bed or floodplain for reconnection; these accounted for just 32% of the 644 projects (**Figure 1*b***). They often included addition of in-stream structures, such as boulders, logs, and gravel. Less

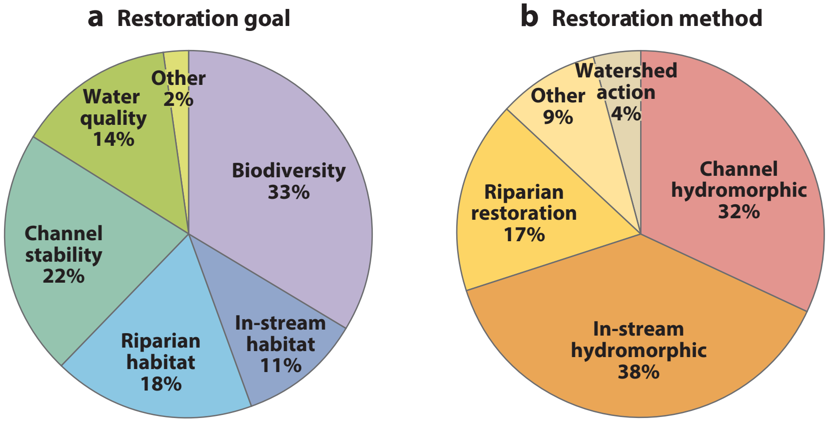

Figure 1

Summary of the most common restoration goals and implementation methods for 644 river or stream restoration projects having quantitative data reported in the published literature; values are percentages of projects using a goal or method. (*a*) The primary goal of each restoration project was identified and associated with one of six attributes to restore or improve: water quality, channel stability, riparian habitat, in-stream habitat, biodiversity, or other. Each project was also placed into one of several broad categories of (*b*) restoration methods depending on how the project was implemented. Channel hydromorphic projects involved reconfiguring the channel, such as moving it laterally, adding sinuosity, or raising/lowering the bed or floodplain for reconnection, and they often included addition of in-stream structures, such as boulders, logs, and gravel; in-stream hydromorphic projects were less intensive projects that involved only manipulating in-stream structures, adding large woody debris, armoring the bank, or creating artificial riffles without major channel excavation or reconfiguration; riparian restoration projects were those projects implemented by planting of riparian vegetation or removal of nonnative vegetation as the primary or sole restoration method; watershed action projects were those in which the project was implemented up in the watershed without manipulation of the channel, and they included, for example, addition of stormwater management, creation of wetlands, or use of cover crops; and "other" projects were varied, including, for example, treatment of acid mine drainage, dam removal, changes in reservoir releases to restore natural flow regime, or creation of an in-stream or riparian wetland.

invasive projects that involved only manipulating in-stream structures, adding large woody debris, armoring the bank, or creating artificial riffles without major channel excavation or reconfiguration (**Figure 1*b***, In-stream hydromorphic) were even more common (38% of the projects).

Most projects involved some riparian planting; however, only 17% of the projects implemented riparian planting or removal of nonnative riparian vegetation as the primary or sole restoration method (**Figure 1*b***, Riparian restoration); though some of these projects were associated with the goal of providing more riparian habitat, many aimed to improve water quality, stabilize the channel, or enhance some ecological function. Projects implementing actions in the watershed instead of the channel (**Figure 1*b***, Watershed action) were few in number (4%; **Figure 1*b***), as were studies quantifying the effects of treatment of acid mine drainage, dam removal, or changes in reservoir releases to restore more natural flow (**Figure 1*b***, Other; 9%).

4. ECOLOGICAL EFFECTIVENESS OF RESTORATION

Ecologists have long wrestled with the thorny problem of defining success in a restoration context. The Society for Ecological Restoration identified key attributes of successful restoration (SER 2004) that fall into four main categories: (*a*) species composition, (*b*) ecosystem function, (*c*) ecosystem stability, and (*d*) landscape context. It further defined potential indicators for each category that could be used to assess the outcome of projects in the field. Revisions and updates have been suggested (Shackelford et al. 2013), but measures of biological diversity, abundance, and ecosystem processes remain the indicators most commonly recommended for field assessments of restoration outcomes (Wortley et al. 2013).

Stream ecologists have recommended a number of additional informative indicators that can be linked to these categories and together provide a guide to appropriate identification of metrics for measuring project effectiveness (**Table 1**). Many of these were used in the 644 projects we reviewed, but a plethora of other metrics was also reported (**Table 2**). Across the 644 projects, biological metrics were most commonly used to evaluate outcome—some 71% of the projects were assessed biologically (e.g., invertebrate species diversity) at least once postrestoration and many projects several years after project completion. The next most common assessment approach used was the measurement of a physical structural attribute (more than 50% of the projects), with habitat and substrate size being the most frequently used. Often metrics only indirectly related to the project goal were used to assess outcome, e.g., measuring channel form when improving water quality was the goal.

4.1. Assessment Metrics

To facilitate our ability to draw general conclusions about project outcome as a function of goal and restoration method, we extracted from each of the 644 projects information on metrics that were used to quantitatively evaluate project outcome and sorted these according to one of four assessment metric categories (water quality, physical features, biophysical processes, biological characteristics; **Table 2**). For each assessment category, we used the metric from a given project that was most informative relevant to the stated project goals. For example, if one of the goals was to restore in-stream biodiversity, the project may have reported changes relative to prerestoration or a reference site using a regional index of biotic integrity (IBI); a species diversity index, such as the Shannon index; the number of EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa; the species richness or abundance; or the percentage of taxa belonging to a functional feeding group. When reported, we used the regional IBI or diversity index as the biological metric for evaluating biological outcome because it provides more information about community composition than number of EPT or species richness and because it takes into account relative abundances of all species regardless of their level of tolerance to some stressor. If the project had an additional goal that was linked to a different assessment metric category, such as water quality, we also extracted information on this goal's outcome, again using only the most informative metric within that assessment category; e.g., if a project reported changes in a toxicant or chemical of concern for a stream but also reported pH, conductivity, and/or oxygen concentrations, we used the toxicant or chemical metric because that factor was typically driving the water quality goal and was thus presumably the primary stressor.

When metrics were difficult to distinguish according to their informativeness, we used functional, process-based measures over structural ones if they were provided. For example, if the goal was to stabilize channels and the project provided data from direct measurements of bank erosion rates or channel migration over time, we used these over other measures, such as particle size on the bed. Similarly, if the goal was to reduce the downstream flux of N or total suspended sediments (TSS) and the project reported net annual export of N or TSS as well as average concentration of N and TSS pre- versus postrestoration, we used the export data. However, because functional (process rate) measures were far less common than point-in-time measures, such as N concentration, we used the latter data as needed.

Table 1 Criteria used for evaluating restoration effectiveness of ecological structures and functions

I. RESTORATION SUCCESS ATTRIBUTES[a]	II. RIVERINE ATTRIBUTES	III. INDICATOR METRICS FOR USE IN THE FIELD[b]
Structure		
Biotic		
Species composition as in reference system	Biotic community composition: Riparian vegetation; Macroinvertebrates; Fish assemblage	Diversity index, species composition,% sensitive macroinvertebrate species, presence or absence of native/nonnative species
All functional groups present or likely to colonize	Riparian vegetation providing: Habitat diversity, litter inputs, shade, evapotranspiration (ET)	Riparian zone width, vertical structure, plant density,% deciduous,% evergreens,% grasses,% herbs,% shrubs, direct measurements of ET or of water content, plant functional group composition
	Plant species richness critical to ecosystem functions	Nitrogen fixers, plant groups important to channel form (e.g., species that produce stabilizing roots, baffle flows, contribute to soil organic matter stocks, etc.)
	Animals: Macroinvertebrate functional groups	Shredders, collector/gatherers, filterers, scrapers, predators
	Ecosystem engineers	Burrowers, stone-rollers (fish), detritivorous fish, salmon, beaver, etc.
Diminished pollutant loads (biotic)	Pathogenic microorganisms	Count or concentration of bacteria, viruses, protozoans, etc.
Physical		
Landscape context and size of restoration site suitable for supporting reproducing populations and maintaining ecosystem functions	Position in catchment, river network characteristics (connectivity longitudinally, laterally, and vertically), project size	% of catchment forested or in natural vegetation, stream order, status of upstream water quality, length of stream restored, longitudinal connectivity of riparian vegetation, river network connectivity (i.e., barriers to flow), floodplain connectivity, hyporheic connectivity
Sufficient suitable habitat to support species	Channel form and in-channel structure	Pool: riffle sequence, sinuosity, discharge, spatial heterogeneity, streambed particle size distribution (D84:D50), large woody debris, macrophyte cover, bank refugia
Diminished pollutant loads (chemical)	Abiotic measures of water quality	Upstream and on-site: conductivity, pH, diel variability in temperature and oxygen, nutrients, sediment, chemical pollutants (point and nonpoint sources)
Function		
Physical		
Primary physical drivers of ecosystem structure within the natural range of variability of reference systems, no signs of dysfunction	Discharge and sediment regimes	Discharge over time: magnitude and timing of peak and low flows, sedimentation rate, bed-load transport, bank erosion rate, channel aggradation/degradation
Energy regime that supports food web as in reference system	Flux of light	% open canopy, direct measures of photosynthetically active radiation (PAR)
	Inputs of organic matter from terrestrial landscape and from upstream	Timing and amount of leaf litter and wood inputs, changes in benthic organic carbon stocks, dissolved organic carbon (DOC), subsidies of terrestrial insects

(*Continued*)

Table 1 (*Continued*)

I. RESTORATION SUCCESS ATTRIBUTES[a]	II. RIVERINE ATTRIBUTES	III. INDICATOR METRICS FOR USE IN THE FIELD[b]
Biological		
Primary and secondary production at levels comparable to reference systems	Ecosystem metabolism, riparian plants, macroinvertebrate assemblages, fish populations	Gross primary production, respiration, growth, survival, and age structure of plantings, macroinvertebrates, and fish species of interest
Rates of biogeochemical processes appropriate to support biota and maintain material fluxes supportive or protective of nearby ecosystems	Nutrient processes	Nutrient fluxes combined with standing stocks: nutrient storage, turnover, export, assimilatory uptake, denitrification, nitrogen fixation, phosphorus release from sediments, etc.
Stability		
Ecosystem resilient enough to endure "normal" stress events in local environment	Geomorphic and ecological recovery rate following flood flows	Channel form over time (in dynamic equilibrium), substrate composition over time, channel aggradation or degradation, invertebrate community composition,% survival, refugia
Restored ecosystem is as self-sustaining as its reference ecosystem and can evolve with changing environment	Little maintenance required following restoration actions, channel has room to adjust, invertebrate and riparian functional redundancy	Floodplain connectivity, riparian width, connectivity to source of colonists (plants and invertebrates), number of species per functional group

[a]Data taken from SER (2004), Shackelford et al. (2013).
[b]Data taken from Bady et al. (2005), Bunn et al. (2010), Casanova (2011), Clapcott et al. (2010), Davies et al. (2010), Diamond et al. (2012), González del Tánago & García de Jalón (2006), Hauer & Lamberti (2007), Kondolf & Piégay (2003), Marks et al. (2010), Naiman et al. (2005), Newcomer et al. (2012), Shackelford et al. (2013), Sudduth et al. (2011), Woolsey et al. (2005, 2007).

4.2. Reported Outcomes

To synthesize general restoration effectiveness across the 644 projects, we tabulated restoration outcome as a function of assessment metric category (water quality, physical features, biophysical processes, biological characteristics) and implementation method (channel hydromorphic, in-stream hydromorphic, riparian restoration, in-stream or riparian wetland creation, watershed action, other) (**Table 2**). If the project data provided for a metric indicated any improvement associated with the restoration, then the project was scored as making progress toward the goal for that metric and was included in the calculation of the percentage of projects reporting progress toward a goal. It is important to note that this provides a liberal estimate of restoration success because any improvement at all was scored positively even if authors indicated the stream was still impaired biologically.

The improvement rates for projects evaluating physical characteristics, such as habitat, substrate, channel form, and velocity, were among the highest compared with other outcome metrics (**Table 2**). For habitat, substrate, and channel form, this result is not particularly meaningful because the improvements are actually a reflection of the fact that the assessment metric is the same as what was manipulated for the restoration, i.e., adding boulders, different types of substrate, or meanders to a stream and then using their presence or arrangement as the measure of improvement. Velocity was measured at the local scale (within the restoration reach), and when improvements were found they were typically linked to reconfiguration of channel form or the addition of in-stream structures (e.g., Gardeström et al. 2013). Dynamic measures of hydrology that could indicate if underlying hydrological processes, not just local flows, were changed by the restoration were not common but included efforts to detect changes in the frequency of flood events (Schiff et al. 2011), floodplain inundation (Klein et al. 2007), or annual water yield (Quinn et al. 2009) after large-scale hydromorphic restoration projects. The apparently higher success rate of in-stream versus channel hydromorphic projects is confounded by differences in the processes measured because the in-stream projects assessed small-scale groundwater–surface water exchange.

Stability was also assessed at the reach scale ($N = 38$; **Table 2**) primarily for projects that involved large-scale hydromorphic restoration actions that included channel reconfiguration. Less than half of these projects showed improvements in channel stability compared with prerestoration regardless of how stability was measured and even though many of the projects involved the use of large boulders or other materials to hold the banks in place. Other studies have previously emphasized poor outcomes for such projects (e.g., Miller & Kochel 2010) that do not restore processes (Section 2.2) and rarely allow for lateral channel migration (Section 2.1).

Improvements in the five metrics within the water quality category (**Table 2**) were found for only 7% of the channel reconfiguration projects and for none of the in-stream channel projects (**Table 2**). However, for the biophysical process category, 5 of the 8 in-stream projects reported improvements in the processing of N or phosphorus (**Table 2**, Nutrient dynamics) and these were projects that measured nutrient removal at the reach scale after adding wood or other structures to increase hydraulic retention (Hines & Hershey 2011, Roberts et al. 2007). Measurement of nutrient loads (concentration $\times$ discharge) during both low flows and storm events were rarely made yet they are both required to determine whether a project actually results in less downstream export of a nutrient (Filoso & Palmer 2011, Richardson et al. 2011).

Studies using indicators of hydrologic or biogeochemical processes to assess project outcome exhibited a higher success rate. Projects seeking to restore organic matter dynamics as measured by in-stream retention of organic matter or rate of decomposition reported success rates of 100% for riparian restoration (**Table 2**). These were not, however, associated with increased aquatic biodiversity or recovery of sensitive species.

Unfortunately, recovery of biodiversity was rare for the vast majority of stream restoration projects, and this appears to be closely related to the restoration method. For the most common type of projects—those implemented using channel or in-stream hydrogeomorphic adjustments—only 16% showed any improvement in biodiversity (as a Shannon type index or an IBI), even though many showed substantial postrestoration improvements in habitat, channel form, substrate, or local velocity, and a number of them assessed invertebrate diversity many years postrestoration (e.g., Louhi et al. 2011). The lack of substantial improvements in biodiversity despite improvements in habitat and channel form (e.g., Ernst et al. 2012) is not surprising given the many studies that have emphasized the distinction between form and function (Section 2.2). Structural variables, like channel form and physical habitat, are not riverine processes, and even when they are rehabilitated, biological recovery is not common (Bond & Lake 2003, Niezgoda & Johnson 2005, Palmer et al. 2010, Ryder et al. 2011, Selvakumar et al. 2010). Put simply, habitat may be important ecologically, but it is not sufficient for assessing ecological outcomes (Doyle & Shields 2012), and in the vast majority of cases restoration of habitat does not lead to restoration biologically (Jähnig et al. 2010). In many cases, water quality is the most limiting factor biologically, and until that is addressed habitat improvements offer few benefits (Kail et al. 2012).

Improvements in taxa richness (number of taxa) but not biodiversity were found for a number of in-stream hydrogeomorphic projects (44 out of 47 projects) because new taxa were found postrestoration, but they were not characteristic of the reference site or the "desired" state of

Table 2 A synthesis of findings from published studies that quantitatively evaluated river or stream restoration project outcomes[a]

ASSESSMENT CATEGORY AND METRICS	RESTORATION METHOD											
	Channel hydromorphic		In-stream hydromorphic		Riparian restoration		In-stream or riparian wetland creation		Watershed action		Other[b]	
	% projects showing improvement	Number of projects	% projects showing improvement	Number of projects	% projects showing improvement	Number of projects	% projects showing improvement	Number of projects	% projects showing improvement	Number of projects	% projects showing improvement	Number of projects
Water quality												
E. coli, metals, ANC	–	2			30%	10			40%	5	80%	10
Turbidity	7%	14			80%	5			0%	3	0%	5
Temperature	–	1	–	1	–	1			–	1	–	1
Dissolved oxygen			–	1							60%	5
Nutrients or other desired chemical change					100%	32	–	1	60%	5		
Morphology/physical characteristics												
Flow (velocity)	44%	32	46%	26	14%	7			100%	2		
Hydrologic dynamics[c]	29%	14	60%	5	25%	4						
Substrate	75%	52	58%	12	25%	12			100%	4		
Stability	47%	36	–	1	–	1					–	1
Channel form	92%	12	29%	7	0%	9						
Habitat	80%	60	60%	5	100%	31					100%	17
Biophysical processes												
Primary production[d]	50%	12	17%	6	100%	6					–	2
Secondary production[e]	100%	5	12%	17							100%	4
Nutrient dynamics[f]	14%	7	63%	8	88%	17	25%	4				
Organic matter dynamics[g]			42%	12	100%	3			–	1	0%	5
Biological characteristics												
Index of biologic integrity	0%	41	12%	17	37%	19	–	1	0%	3	–	2
Diversity indices	35%	54	6%	50	–	2			–	1	0%	14
% EPT	33%	12			100%	31					0%	5

(*Continued*)

Presence/absence of desired species	64%	11	59%	27	56%	9			75%	8	78%	9
Chlorophyll a, algae, macrophytes	0%	11			0%	10						
Richness	14%	36	94%	47					–	1	61%	18
Other biological characteristics	14%	7	88%	8	0%	1					50%	2

[a]Project outcomes are shown by restoration method and by metric used to assess project outcome; they are listed as a percent of projects showing any improvement toward meeting their primary goals (examples include: reduce concentrations of *E. coli* or nutrients; increase biodiversity, organic matter or nutrient flux; shift rates of primary production or hydrology to be more similar to reference site) and as the number of projects (*N*). For categories with two or fewer projects, outcome frequency is not estimated and instead a dash is shown. Empty cells indicate there were no data in that category. Outcomes are relative to a reference site, the site prerestoration, or some target. Metric categories reflect how projects were assessed and do not necessarily reflect the stated goal for a project. (Assessment metrics for a project were often indirectly related to the stated project goal; i.e., a project with the goal of improved water quality may have used measurements of flow as the primary metric, not the concentration or flux of the constituent of water quality concern.) Study citations for each project are available in the **Supplemental Literature Cited**.
[b]Other includes treating acid mine drainage and manipulating flows or reservoir releases.
[c]Refers to functional hydrologic measures, such as hydraulic residence time or storage (Kasahara & Hill 2006, Quinn et al. 2009).
[d]Typically measured as community respiration in g O_2 per m^2 per day (Hoellein et al. 2012).
[e]A loose category for projects in which abundance or biomass was followed multiple years postrestoration (Sweka & Hartman 2006) or hatching success rate or fish growth was monitored (Sternecker et al. 2013, Vehanen et al. 2010).
[f]Rates of nutrient processes such as nutrient uptake and nitrogen fluxes/export (Bukaveckas 2007, Gabriele et al. 2013, Richardson et al. 2011).
[g]Rate of litter input to stream (Thompson & Parkinson 2011) or rate of leaf retention (Muotka & Laasonen 2002).
Abbreviations: ANC, acid neutralizing capacity; EPT, Ephemeroptera, Plecoptera, and Trichoptera.

Supplemental Material

the stream. Unlike diversity, taxa richness is not a particularly informative indicator of project outcome because it does not distinguish between tolerant and intolerant taxa. One of the most comprehensive studies of restoration outcomes (24 channel reconfiguration projects assessed) reported no significant change in diversity for two-thirds of the projects and only a slight increase in taxa richness in the other third that was associated with the addition of a few tolerant taxa characteristic of urban streams (Tullos et al. 2009).

A recent study has shown that watershed-scale, out-of-channel management practices to restore urban streams can be quite successful: "measures of biodiversity in restored streams were 132% of those in unrestored urban streams, and indices of biotic condition, community structure, and nutrient cycling significantly improved" (Smucker & Detenbeck 2014). These authors combined published study data and data from reports. Our study only relied on the former and in some cases older studies, hence our sample size for watershed projects. We found that the highest success rates biologically were for those projects that involved a primary focus on enhancing the riparian zone as the restoration action. Typically, these involved either planting native vegetation or removing nonnative vegetation. Across all riparian restoration projects that used one of the biological or biophysical process metrics to evaluate outcome, 69% of them showed improvement. In fact, 88–100% of the projects showed improvements in productivity, organic matter, nutrient dynamics, or percentage of EPT. The first three metrics are dynamic or functional metrics (e.g., rates of processes), which are believed to be better assessment indicators of early progress in restoration because they reflect how the system is functioning, not just its point-in-time or snapshot status (Palmer & Febria 2012). The EPT metric is a snapshot metric that reflects improvements in the number of mayflies, stoneflies, and caddisflies, but it is important to note that most of the projects using EPT to assess effectiveness of riparian restoration (**Table 2**) were from a single study (Orzetti et al. 2010) that focused on sites in which riparian cover had been restored for some time; the authors emphasize that the biological community typically takes 5–10 years to recover following restoration, and that is after water quality has improved.

5. BEYOND RESTORATION

> But changes like these must await great political and moral revolutions in the governments and peoples by whom those regions are now possessed. . . . (Marsh 1864, p. 47)

Although we find that outcomes of river restoration so far may be disappointing, it is important to remember that stream restoration science is very young compared with, say, forest or prairie restoration. Researchers and practitioners are still developing methods, and the problematic ecological outcomes of many or most structurally based restoration projects are only now becoming more widely acknowledged. A unified perspective on how to succeed in restoring rivers has yet to take hold. We show that a major emphasis remains on the use of dramatic structural interventions, such as completely reshaping a channel, despite growing scientific evidence that such approaches do not enhance ecological recovery, and the data we assembled (**Table 2**) suggest they are often ineffective in stabilizing channels when stability is the primary goal. Efforts at the watershed and riparian scales that target restoration of hydrological processes and prevention of pollutants from entering the stream appear to offer the most promise.

Restoring the ecological integrity of degraded waterways is tough, complicated work, and it is tempting to turn to familiar critiques of ecological restoration. These typically fall into two viewpoints: (*a*) restoration must meet current human priorities and should therefore involve engineering an ecosystem to maximize some natural process to meet those priorities, and (*b*) restoration of an ecosystem can never achieve some former or least-disturbed ecological state but instead must be forward-looking.

Such viewpoints are easily reconciled with the prevailing social and economic forces that drive the practice and regulation of river restoration. When compared with comprehensive recovery of ecological structure and function, maximizing/minimizing a select set of biophysical processes consistent with the human priorities in a particular setting, time, and geographic context may be easier to get approved and funded. Thus, redefining ecological restoration to be consistent with regional priorities that target some ecological processes at the expense of others (e.g., converting a stream to a novel ecosystem to meet regulatory requirements; Palmer et al. 2014) may make the work easier, but it is unlikely to truly mend the world's degraded rivers.

Arguments that restoration is not adequately focused on human benefits, that a watershed approach is impractical given land ownership and infrastructure constraints, and that it is no longer feasible given the pace of global change are commonly used for moving beyond ecological restoration. What would this mean in practice? Hobbs and others (Hobbs et al. 2011) have promoted the concept of "intervention ecology" for novel ecosystems, but the details of how intervention ecology would work are vague. Hallett et al. (2013) suggest that it would shift management from a focus on historical conditions to a focus on ecosystem functions and services. However, unlike ecological restoration, which focuses on recovering the full suite of processes and structures found in some reference ecosystem, enhancing ecosystem services and intervening to maximize specific ecosystem processes are targeted responses to particular social or economic demands. Often the priorities associated with those demands are not ecological (Smith et al. 2014). Thus, calls for intervention ecology and the anticipation of future ecological conditions represent new, socially driven aims for restoration that may or may not result in recovery of the full suite of ecological processes and structures.

Most ecologists do not claim that such shifts in definition are wrong from a normative perspective; there is wide recognition that social decisions are involved in selecting reference conditions for restoration targets. Instead, many would argue these changes represent something very different from ecological restoration. Further, it may open the door for the potential loss or degradation of other ecosystem attributes that are unrelated to the target ecosystem service(s) or that are incompatible with stakeholder goals (Palmer & Filoso 2009). Indeed, in some cases the intervention itself can transform a stream into a novel ecosystem (Filoso & Palmer 2011, Sanon et al. 2012). The most extreme example involves attempts to create streams where they did not previously exist. Newly created streams are now a legal form of restoration for compensatory mitigation in the United States (Palmer & Hondula 2014). This has enabled the destruction of healthy ecosystems (Moore & Moore 2013) and prompted new research on stream creation (Bronner et al. 2013, Hossler et al. 2011, Scrimgeour et al. 2013). Ongoing research indicates that attempts to create streams de novo have not resulted in streams that support the biodiversity or ecological processes characteristic of nearby intact streams. Such activities are all symptoms of the intense social and economic pressures that influence the practice and science of what is called "ecological" restoration. Restoration is hard, and forestalling the socio-economic incentives to invent new ecosystems rather than restore existing ones or to manipulate channels rather than rehabilitate watersheds will require great revolutions indeed.

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The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

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