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COMPENSATORY MITIGATION FOR STREAMS UNDER THE CLEAN WATER ACT: REASSESSING SCIENCE AND REDIRECTING POLICY[1]

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ABSTRACT: Current stream restoration science is not adequate to assume high rates of success in recovering ecosystem functional integrity. The physical scale of most stream restoration projects is insufficient because watershed land use controls ambient water quality and hydrology, and land use surrounding many restoration projects at the time of their construction, or in the future, do not provide sufficient conditions for functional integrity recovery. Reach scale channel restoration or modification has limited benefits within the broader landscape context. Physical habitat variables are often the basis for indicating success, but are now increasingly seen as poor surrogates for actual biological function; the assumption "if you build it they will come" lacks support of empirical studies. If stream restoration is to play a continued role in compensatory mitigation under the United States Clean Water Act, then significant policy changes are needed to adapt to the limitations of restoration science and the social environment under which most projects are constructed. When used for compensatory mitigation, stream restoration should be held to effectiveness standards for actual and measurable physical, chemical, or biological functional improvement. To achieve improved mitigation results, greater flexibility may be required for the location and funding of restoration projects, the size of projects, and the restoration process itself.

(KEY TERMS: environmental regulations; stream restoration; aquatic ecology; rivers/streams; water policy.)

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INTRODUCTION

The overarching goal of the United States (U.S.) Clean Water Act is to sustain and restore the physical, chemical, and biological integrity of the nation's waters. Under Section 404 of the Clean Water Act, permitted impacts to "aquatic resources" must be mitigated. The term "aquatic resources" includes both streams and wetlands, but although the science, policy, and economics of wetland mitigation have received considerable attention (NRC, 2001), stream mitigation has not (Lave *et al.*, 2008).

Mitigation under the Clean Water Act was regulated for years by a series of guidance documents, but in 2008, the "Rule for Mitigation of Impacts to Aquatic Resources" was finalized jointly by the U.S. Army Corps of Engineers (Corps) and the Environmental Protection Agency (EPA). This rule now governs mitigation (33 CFR Parts 325 and 332, "Compensatory Mitigation for Losses of Aquatic Resources," April 10, 2008; §332.5 and §332.6) and is hereafter referred to as the "2008 Rule." Mitigation prioritizes avoiding and minimizing impacts, and allows "compensatory mitigation," or the preservation, enhancement, or restoration of a site in order to

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compensate for unavoidable stream impacts elsewhere. The goal of such compensatory mitigation is "no net loss" of stream function. However, to make the program practicable, regulatory agencies use the metric of stream length rather than metrics of stream function to assess no net loss: impacts to X meters of stream require at least X meters of stream be restored elsewhere (the key regulatory agencies are the Corps and the EPA, which co-administer CWA Section 404 with the Corps being the permitting agency). Critically, the underlying and necessary assumption of compensatory stream mitigation is that streams can be intentionally manipulated to increase specific functions of interest, and thus restored or enhanced, thereby providing the compensation to offset losses elsewhere.

Following the practice of compensatory wetlands mitigation, many states and districts of the Corps have been moving toward two broad paths for compensatory stream mitigation. First is the practice of entrepreneurial "mitigation banks," which restore selected stream reaches and sell the resulting mitigation credits to entities that are responsible for stream impacts elsewhere (Figure 1). In addition are in lieu fee programs, which accept fees from developers who impact streams and then accumulate those fees to fund restoration. There are other variants on compensatory mitigation (e.g., permittee responsible), and these are reviewed in detail elsewhere (BenDor *et al.*, 2009; Womble and Doyle, 2012). Data on the growth of compensatory wetland mitigation and banking are more readily available than for streams. Gillespie (2005) reported that the first stream mitigation bank was set up in 2000, with "about two dozen"

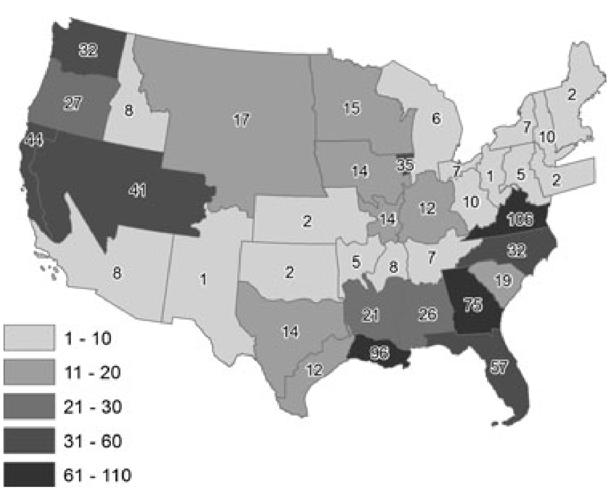

FIGURE 1. Distribution of Mitigation Banks in the U.S. as of February 2011 (courtesy of Todd BenDor). Data are spatially organized by Corps of Engineer District, the federal agency regulatory lead for compensatory mitigation under Clean Water Act Section 404.

in operation by 2005. The National Mitigation Banking Association (http://www.mitigationbanking.org) lists stream or wetland banks in 14 states and provides specific examples of stream mitigation banks in North Carolina, Montana, Virginia, and Louisiana, although the number of stream mitigation banks is growing rapidly. As of October 2005, there were 28 state-run mitigation programs (in lieu fee programs), which had conducted compensatory mitigation activities on 1,789,245 linear feet of stream (Environmental Law Institute, 2006). As of May, 2011, the Corps' RIBITS (Regulatory In-lieu Fee and Bank Information Tracking System) recorded 240 compensatory stream mitigation banks nationwide (ribits.usace.army.mil).

Regardless of the state or mechanism (e.g., banking *vs*. in lieu fee), stream mitigation in general follows the same logic: stream impacts are compensated by stream restoration. Because of this logic, stream restoration provides the cornerstone and justification of the burgeoning compensatory stream mitigation practice and industry.

Following almost a decade of implementing compensatory mitigation of wetlands, the National Research Council conducted a systematic review and showed a series of systemic failures (NRC, 2001). At the time of the review, compensatory stream mitigation was relatively rare, as stream impacts could be compensated by wetland restoration. However, many states and Corps' districts have begun requiring greater adherence to "in-kind" mitigation – that stream impacts be mitigated with stream restoration – and the 2008 Rule increases the requirement for in-kind compensatory mitigation. The impact of this policy evolution has been not only to bolster the demand for stream restoration but also to place stream restoration increasingly under the auspices of compensatory mitigation.

As compensatory stream mitigation becomes more widely practiced throughout the U.S., it is important to develop assessments of the practice, similar to that done by the NRC (2001) for wetlands. Because stream restoration plays an important if not central role in compensatory stream mitigation, we briefly synthesize some of the available literature on the efficacy of existing stream restoration projects to understand this component of mitigation. The working assumption by regulators, practitioners, and many academics appears to be that stream restoration, as typically practiced (see Bernhardt *et al.*, 2005), produces increased physical, chemical, and biological integrity. This assumption is necessary for the current implementation of compensatory mitigation to be an option in the CWA 404 permitting program. Our review shows that this assumption is questionable, and that many traditional stream restoration projects are largely ineffective at restoring chemical and biological functions.

Based on our assertion that traditional stream restoration is largely ineffective for meeting stated goals, we propose a set of policy adaptations to the current state of restoration science. We suggest that these adaptations would facilitate more effective, long-term compensatory stream mitigation programs and provide economic incentives for those projects that are more functionally effective. Findings may be applied to refine compensatory stream mitigation policies or to develop policies for Endangered Species Habitat Conservation and Mitigation, or Federal Power Act mitigation associated with hydropower relicense for Federal Energy Regulatory Commission (FERC) projects, as well as mitigation programs that may emerge for newer alternative energy development programs and the expanded federal infrastructure construction (Wilkinson *et al.*, 2009).

DOES RESTORATION WORK?

Stream restoration, as considered here, is defined as the intentional manipulation of stream channel conditions in order to re-establish or enhance specific stream functions. Restoration often involves physical manipulations such as remeandering, bioengineered bank stabilization, among others. Functions of interest are normally linked to the stated goal of the CWA: to maintain and restore the chemical, physical, and biological integrity of the nation's waters. Restoration often involves return to a predegradation trajectory (Society for Ecological Restoration, 2004), and thus involves lofty goals. Many scientists and regulators have moved away from the expression "restoration" in favor of "rehabilitation" or "re-establishment" as a more appropriate term (e.g., 2008 Rule at §332.2).

Few projects are ever examined in light of any original goals: only ~10% of the stream restoration projects identified by a national survey (not just mitigation projects) involved data collection on habitat, biological populations, or ecological processes (Palmer *et al.*, 2007). In lieu of such data that would allow systematic evaluation of success or failure, we use available published studies to assess the practice of stream restoration in terms of its track record of restoring the physical, chemical, and biological integrity. Our review is not intended to be comprehensive, but rather to be representative of stream responses to restored conditions. We drew conclusions from the available and robust peer-reviewed literature rather than the vast gray literature (e.g., agency reports). We contend that the findings of the peer-reviewed published literature provide a more defensible foundation for assessing the state of the science, and do not represent a biased sampling of possible restoration outcomes; that is, we see no reason why published literature would systematically prefer the selection of successful or unsuccessful projects for study, and that such bias is not possible when studies involve pre- and postrestoration evaluations. However, we acknowledge that our review does not, and could not, distinguish mitigation-driven restoration from other mechanisms or funding of restoration. Whether mitigation-driven restoration is systematically different is a critical research question.

Physical Integrity

For physical integrity, we draw from published studies of flood attenuation and sediment stabilization. Flood peak attenuation and reduction in sediment loads are often used to justify stream restoration (e.g., NCDENR, 2008). Flood attenuation refers to the reduction of the flood peak discharge by reversing the effects of channelization and other channel changes associated with the development of watersheds. Channel straightening, enlargement and removal of large wood reduce hydraulic resistance, and the loss of in-channel and floodplain storage all combine to reduce the duration but increase peak discharge of flow hydrographs (Campbell *et al.*, 1972; Doyle and Shields, 1998; Acreman *et al.*, 2003). These changes may be exacerbated when combined with watershed deforestation or increasing imperviousness (Dunne and Leopold, 1978).

As stream restoration involves a return to more natural channel form, many restoration workers have simply assumed that their projects (e.g., introducing large wood, channel remeandering, reconnecting the channel with the floodplain) would attenuate floods. Few studies have tested these assumptions, and these rely almost exclusively on modeling (Wolff and Burgess, 1994; Anderson *et al.*, 2006; Sholtes and Doyle, 2011). Importantly, these studies have found that restoration of long reaches (on the order of 10 km) could produce flood attenuation. In contrast, most stream restoration projects treat reaches <1 km long (Bernhardt *et al.*, 2005), and flood attenuation by projects of this scale is too small to be documented empirically or relied upon (Figure 2) (Sholtes and Doyle, 2011). Many short reach projects in headwater streams might combine to reduce downstream flood peaks (Liu *et al.*, 2004), but such an approach would require coordinated planning at the watershed scale, and even then, complex channel network routing dynamics can negate these potential effects on downstream

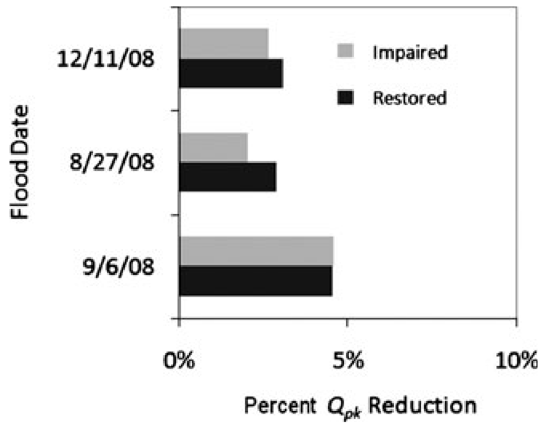

FIGURE 2. Effect of Stream Restoration on Flood Peaks and Flood Attenuation (adapted from Sholtes and Doyle, 2011). Flood attenuation effects of typical short-reach stream restoration projects are too small to detect.

waters of interest (Emerson *et al.*, 2005). Many aspects of stream restoration (addition of large wood or instream structures, remeandering, riparian zone revegetation, reconnection with floodplains) have the potential to increase local flow resistance and thus local flood peaks (Shields and Gippel, 1995; Hydraulics Research, 1988). Thus, the available research indicates that changes in flood characteristics due to short-reach (~1 km) stream restoration projects are too small to measure using available technology. Whether mitigation-driven restoration projects are systematically longer or larger than nonmitigation-driven projects is currently unknown, but modeling suggests that significant reach lengths (~10 km) are needed to have measurable benefits.

The second physical function for streams to be changed by restoration is sediment loads. For instance, about 20% of the projects in the National River Restoration Science Synthesis database (http://nrrss.nbii.gov/) lists "bank stabilization" as a project goal, and many projects constructed for other intents include bank re-vegetation or erosion control features. Restoration often involves reconfiguring channels as well as stabilizing stream beds and banks using plant materials (e.g., Gray and Sotir, 1996) or other inert structures. Rigorous evaluations of the performance of sediment controls associated with channel restoration are largely lacking. Those evaluations that do exist are often based on a link between stability of channel dimensions rather than decreased sediment concentrations or loads (e.g., Shields *et al.*, 1995b; Buchanan *et al.*, 2010). Bank stabilization with plant materials or combinations of plant materials and structure may succeed or fail over the short term (Shields *et al.*, 1995a; Pezeshki *et al.*, 2007), and over the longer term plants and large wood may fail (Shields *et al.*, 2008b) or perform inadequately (Price and Birge, 2005). That is, the efficacy of stream restoration for stabilizing channel dimensions (i.e., not increasing downstream sediment loads via channel adjustments) remains an area of rich research but equivocal long-term success.

But the central assumption of stream restoration for sediment loads is that it results in a net decrease. Few studies have examined channel erosion control effects on watershed sediment yield. Shields (2009) examined long-term (>10 years) sediment yield records from a channel stabilization effort, which involved an expenditure of more than US\$309 million, of over US\$40,500/km^2 in 16 Mississippi watersheds. Only one watershed experienced a statistically significant decline in sediment loads, and this watershed was treated by building several small reservoirs as well as in-channel structures. Only one subwatershed within the 16 watershed region had a documented sediment load reduction (Kuhnle *et al.*, 1996). Thus, local reduction in sediment loads associated with restoration activities may be possible, but whether this "signal" is observable at the scale of an entire watershed remains an open question.

An additional aspect that has received little attention is the role that restoration projects themselves play in sediment mobilization due to construction activities, channel adjustments, or project failure (Thompson, 2005; Miller and Kochel, 2010; Radspinner *et al.*, 2010; Buchanan *et al.*, 2010). If restoration is to result in net decrease of sediment loads, then these effects must be considered as part of the net changes. To our knowledge, no studies have documented the sediment load changes associated with pre-, during, and postrestoration activities. We revisit this below.

Chemical Integrity

Stream restoration projects are also sometimes credited with mitigating stream water quality degradation. Documentation of such benefits is scarce, although the concept is increasing in popularity (Craig *et al.*, 2008). Some workers have shown that restoring hydrologic connectivity between streams and floodplains boosts the potential for denitrification of riparian waters (Kaushal *et al.*, 2008; Richardson *et al.*, 2011), and denitrification has been measured in *groundwater* passing though vegetated riparian zones by many workers (Jacobs and Gilliam, 1985; Lowrance, 1992; Newbold *et al.*, 2010). But few studies link restoration directly to improvements in *stream water* quality and some that do suffer from confounding influences.

Studies of restoration on stream water quality have produced mixed results; Roberts *et al.* (2007) showed that large wood addition to very small streams (watershed size = 0.33-3.7 km^2) increased NH_4 uptake, but noted that the literature yields conflicting results regarding links between transient storage and nutrient uptake. More recently published studies of the effects of wooded riparian buffers (Sutton *et al.*, 2010; Newbold *et al.*, 2010) and instream restoration on small stream water quality also contain equivocal results regarding the overall efficacy of restoration measures for stream quality remediation. For instance, Kasahara and Hill (2006) found that riffle-step restoration in nitrogen (N)-rich lowland streams intensified hyporheic exchange but accounted for the removal of only 0.003-0.06% of stream NO_3 loads. Sudduth *et al.* (2011) found that stream metabolism did not vary between restored and degraded streams; although nitrate uptake was higher in restored streams, they found that this was explained by the removal of riparian vegetation for channel modifications, not the direct effect of enhanced nitrate uptake kinetics. Similarly, Bukaveckas (2007) found not only higher nitrogen retention in restored streams but also significantly higher temperature associated with riparian vegetation removal. Similarly problematic results come from Richardson *et al.* (2011), who found restoration of 600 m of a small stream was effective in reducing nitrate and total phosphorus (P) loads by estimated amounts 64 and 28%, respectively. But their project included restoration of 2.1 ha of connected wetlands and was located in the flood backwater region of a constructed impoundment, both of which certainly confound interpretation of their findings.

Restoration has also been advocated as a mechanism to reduce stream temperatures by increasing shade, reducing width/depth ratios or increasing surface-groundwater exchange (Poole and Berman, 2001; Arrigoni *et al.*, 2008; Kaushal *et al.*, 2010). Shading and geomorphic effects may be simulated by models (e.g., Bartholow, 2000), and watershed-scale treatments can be effective. However, temperature shifts due to increased hyporheic exchange are more complex. Streams with a substantial fraction of bed sediments finer than gravel experience very low levels of transient storage in the hyporheic zone (Stofleth *et al.*, 2008). Other studies have found that stream temperature changes due to induced hyporheic exchange are so slight ($\sim$0.01 C) even at the local, patch scale that they could be difficult to measure (Hester and Doyle, 2009), and the cumulative impact of local changes has not been established. Like changes in flood attenuation, water temperature changes due to typical mitigation projects (<1 km reach length) are likely too small to be detectable with available monitoring technology or without considerable efforts; it is also questionable whether such changes are meaningful or important in an ecological context.

In cases where restoration can be linked to a water quality change, there may be critical issues of whether those changes are important in comparison with what happens during more ecologically effective flows (Doyle *et al.*, 2005b). Many of the types of actions typical of stream mitigation projects may produce benefits at base flow but have diminishing benefits at higher flows (e.g., Shields and Gippel, 1995; Sholtes and Doyle, 2011). If water quality goals are primarily focused on downstream load reduction, then amelioration of low-flow conditions is of marginal relevance when the bulk of nutrient loads are transported during high discharge events. A study of Midwest agricultural watersheds showed that most N and P is exported during spring runoff events; the authors suggested that little potential existed for restoration efforts that targeted low-flow regimes to affect downstream nutrient loads (Royer *et al.*, 2006). A similar study of mixed land use watersheds in the Baltimore, Maryland area found that N export was increasingly associated with high flows as watersheds urbanized but was greater for base flows in forested watersheds (Shields *et al.*, 2008a). Thus, restoration benefits may help in forested watersheds but not in urban watersheds. A study of large wood addition to a deeply incised Mississippi stream showed that the rehabilitation work had no impact on water quality because flashy hydrology that was related to channelization and resultant incision dominated water quality loading and processes (Shields *et al.*, 2010).

In all, the utility of stream restoration for generating measurable and meaningful water quality benefits, as restoration is currently practiced and for the common scale of practice, is doubtful. This is true for influencing local water quality but is even more so the case for influencing the quality of downstream receiving water bodies such as reservoirs or estuaries.

Biological Integrity

Biological integrity is, or should be, the ultimate goal of most restoration efforts (Karr, 2006). Due to the cost and difficulty of monitoring biological populations, many studies of stream restoration effectiveness have focused on physical habitat rather than actual organisms (Skinner *et al.*, 2008). Stream mitigation projects usually feature physical habitat manipulation at the reach scale that does not address hydrologic or water quality factors imposed by upstream or watershed conditions (Shields *et al.*, 2007). For instance, restoration of pool-riffle morphology in an

Illinois stream resulted in improved habitat and fish abundance over a two-year postconstruction monitoring period (Schwartz and Herricks, 2007). However, the resulting biological (fish) metrics were lower when compared with less degraded streams in the same ecoregion, and the restored fish community was dominated by tolerant, pool species rather than intolerant, riffle-dwelling fishes, which were the target species. Similarly, a study of stream remeandering in Indiana showed that habitat metrics improved at one site and declined at another following restoration, and that even with habitat improvement, most metrics of fish communities did not surpass levels observed in a channelized, disturbed reference stream (Moerke and Lamberti, 2003). The authors concluded that, "Streams that have been degraded by poor watershed land use cannot be restored by focusing solely on instream conditions. Factors degrading stream communities need to be identified, and management activities should be targeted to the highest level at which negative influence occurs (e.g., the watershed)." In a study of restored streams in North Carolina, Violin *et al.* (2011) found that macroinvertebrate communities in restored streams were indistinguishable from urban, degraded streams, and they concluded that reach-scale restoration was generally unsuccessful.

Regional studies and review papers have produced more general but equally mixed results. Brookes *et al.* (1996) reviewed 17 reports of instream habitat enhancement and found that a majority reported positive results on habitat provision. An 11-year study of incised streams in northwestern Mississippi showed that watershed treatment alone was inadequate to produce habitat and fish population recovery (Shields *et al.*, 2007). Smiley *et al.* (in press) found that herbaceous riparian buffers had little effect on habitat quality or biota of midwestern channelized streams, but that sites that experienced both watershed erosion control and instream habitat work, responded well. Studies of 13 lowland UK rivers treated with instream structures (flow deflectors or artificial riffles) found no significant impact on macroinvertebrate taxon richness (Harrison *et al.*, 2004) or fish communities (Pretty *et al.*, 2003) despite positive effects on local physical habitat. The authors concluded that restoration effectiveness was constrained by larger-scale factors, such as watershed condition. Stewart *et al.* (2006) conducted a quantitative meta-analysis of, "Any studies examining the impact of in-stream structures on the abundance of salmonids." They found 137 studies that met review criteria, a subset of which was usable for quantitative analysis. Their synthesis found "a weakly significant positive impact" of instream habitat structures on salmonid populations but concluded that "No ecologically significant impact on salmonid population size or habitat preference was present. There [were] no significant relationships between the effectiveness of engineering in-channel structures and hydrological or ecological variables at a population level," although they did find some positive relationships when woody debris was used for restoration. Roni *et al.* (2008) conducted a global review of 345 published evaluations of stream restoration techniques, including studies on road improvement, riparian rehabilitation, floodplain connectivity and rehabilitation, instream habitat improvement, nutrient addition, and other, less-common techniques. The review by Roni *et al.* (2008) was self-admittedly biased toward western North America and successful projects. Particular restoration techniques were found to provide noticeable benefits (see particularly their tables 3 to 5). Some techniques, such as reconnection of isolated habitats, rehabilitation of floodplains, and placement of instream structures, were shown to be effective for improving habitat and increasing local fish abundance under many circumstances. Regarding instream structures, Roni *et al.* (2008) found that, "When implemented properly these techniques can produce dramatic improvements in physical habitat and biota, particularly for salmonid fishes." However, they also noted that, "While placement of instream structures appears to be successful at increasing local fish abundance, particularly that of salmonids, results are highly variable among species, life stages, and structure types, and little positive benefit has been documented for nonsalmonids." Whiteway *et al.* (2010) presented a meta-analysis of data from 211 stream restoration projects designed to improve salmonid habitats. Significant increases in salmonid density and biomass were reported, although large differences in response were noted among various species. Physical habitat effects were also statistically significant. An important caveat in any of these studies is whether the treatment of restoration resulted in production or attraction: were more organisms able to grow and reproduce, or did the restoration work simply attract organisms that were nearby anyway?

Investigators, who report marginally successful stream restorations, and many practitioners facing less-than-ideal results often suggest that more time will allow fuller ecosystem recovery. Although this may indeed be true, there are very few long-term projects that provide any support for this idea. Rather, the limited evidence may be pointing in the opposite direction. First, watershed context and upstream land use are particularly important on project success; in many, if not most, restoration scenarios, it is more likely that upstream land use will become *more* developed through time rather than *less* developed through time, which will most likely reduce project success (e.g., Moerke and Lamberti, 2003). Indeed, based on their review of successful projects, Roni *et al.* (2008) suggested that, "The potential benefits of most instream structures will be short lived (<10 years) unless coupled with riparian planting or other process-based restoration activities that can lead to long-term recovery of deficient processes." Perhaps most importantly is a single known study that conducted a retrospective, quantitative meta-analysis of historic restoration projects: Thompson (2006) reviewed 79 reports published prior to 1980 regarding impacts of instream habitat enhancement structures on fish populations and found only seven provided evidence of beneficial effects.

Conclusions from the Literature

The balance of published evidence suggests that current practices of stream restoration – in terms of scale and technique – cannot be assumed to provide demonstrable physical, chemical, or biological functional improvements. Although there are some, perhaps many, stream restoration projects that have been successful for specific and modest goals, there is not compelling evidence from the peer-reviewed literature that successful projects are the majority or that success can and should be presumed. The rate of success for restoration of habitat and some types of biological populations (particularly for salmonid fishes) appears to be attainable in specific types of cases such as forested or less disturbed catchments, although success varies widely from project to project and cannot be presumed. Restoration in urban watersheds may be prohibitively difficult. Measurable and meaningful levels of attenuation of floods, reduction of sediment loads, and improvement of water quality appear to be unrealistic and ungrounded goals for the predominant practice and scale of stream restoration.

Within the context of compensatory stream mitigation, systemic shortfalls of restoration have important implications. Stream impacts are assumed to lead to loss of function at impacts sites. When permitted stream impacts are mitigated via traditional stream restoration, given the lack of demonstrated functional improvement at restoration sites, we are led to conclude that that there has been, and continues to be, a national net loss of physical, chemical, and/or biological stream function.

POLICY ADJUSTMENTS AND OPPORTUNITIES

Given this assessment of restoration efficacy, it is necessary to look for opportunities to adjust specific or general policies to increase the likelihood of successful projects. This is in part based on our assumption that eliminating compensatory mitigation is not an option for regulators.

One of the important emphases of the 2008 Rule is the increased emphasis on quantifying the changes in stream ecosystem functions at restoration sites [see 2008 Rule at §332.5(b)]. The above review suggests three factors that should be considered in implementation of this rule. First, stream restoration may not improve functions; that is, it may not work in the first place. Second, even when it does work, the changes in functions associated with stream restoration may be so subtle as to be difficult to quantify using available techniques and technology. Third, surrogates are very often poor indicators of actual functions of interest (e.g., habitat can be restored while actual species of interest are largely absent).

In light of these consideration factors, we suggest several possible types of adjustments in current compensatory mitigation policy application and particularly ways in which the 2008 Rule could be applied so as to maximize the potential utility of stream restoration as compensatory mitigation.

Emphasize Change Rather Than Endpoints

In applying requirements of success criteria and monitoring [2008 Rule at §332.5 and §332.6(c)(1)], some specific functions should be recast as "changes" rather than specific, targeted endpoints. Monitored restoration projects may be compared with an existing, natural or less disturbed site [2008 Rule at §332.5(b)]. For example, the BACI (Before-After-Control-Impact) (Green, 1979) study design is often used to compare restored sites before and after restoration with an unaltered control site. The selection of the control site is pivotal; we suggest that when this evaluation approach is applied, multiple control sites be chosen for comparison (Shields *et al.*, 1998; Moerke and Lamberti, 2003; Richardson *et al.*, 2011). First, the prerestoration condition at the treated site should be compared with a similarly degraded disturbed site. Second, the treated or restored reach should be compared with a lightly degraded "target" site. For the initial years following restoration, it is imperative that success criteria be developed and compared with the disturbed site, and that initial success be defined as a "change from existing conditions." This takes the approach of first ensuring that restoration is doing no harm. This also allows the mitigators to show a trajectory of change in a positive direction (if such changes in fact occur), that is, from disturbed conditions toward less degraded, "natural" conditions.

Ensuring at least benign effects of restoration may be important success criteria to articulate in mitigation policies as part of required success criteria and performance standards. Indeed, an important aspect of restoration that to date has been ignored is the impact of restoration activities themselves on stream ecosystems. For no-net-loss of ecosystem function at a site, the long-term restored condition must not just be an improvement of prerestoration, but it must make up for the impacts of the restoration itself. For instance, consider the function of sediment load reduction at a site with a five-year monitoring period. There can be large quantities of sediment mobilized during stream restoration and bank stabilization (e.g., bioengineering) projects due to the heavy earth-moving that is often involved. Although some of this can be avoided by working "in the dry" (under drained, pumped conditions), there can still be considerable sediment mobilization during the first few high flows passing through the channel. Thus, for there to be a net reduction in sediment loads attributable to restoration, the postrestoration sediment load must be less than the prerestoration sediment loads plus the sediment mobilized during restoration. Yet, most restoration projects do not collect data during the restoration activities themselves. Achieving no-net-loss in terms of sediment loads for the entire project could be challenging given how much sediment is mobilized during typical restoration projects (Figure 3), creating a substantial quantity of sediment retention that must be made up over the lifetime (or monitoring period) of the project. If the project is not able to account for these restoration impacts, then the restoration can be seen as a net source of stream ecosystem degradation. We argue that, at a minimum, restoration must show a change in the right direction – a trajectory toward recovery that can exceed the impacts of restoration itself.

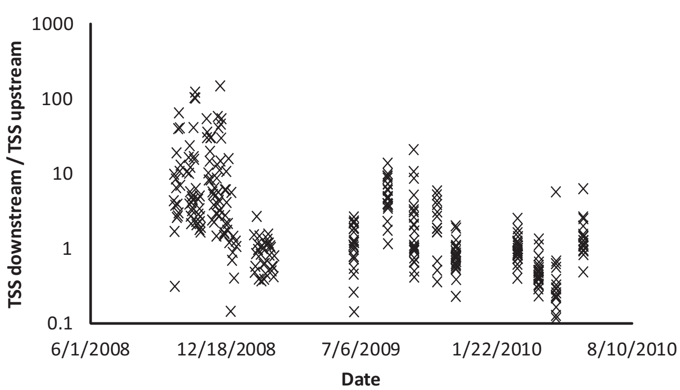

FIGURE 3. Effect of Stream Restoration on Total Suspended Solids (TSS) at Ellerbe Creek, North Carolina. Values >1 indicate export of TSS from the restored site. Data are shown for base flow only, that is, flood discharge data are removed. Restoration activities occurred in late summer and winter of 2008. Note that while ratio of TSS export gradually declines, during the time of restoration TSS export concentrations were substantially greater downstream than upstream.

It is important to emphasize that this is a rather modest goal; over time, as more is learned, more refined and specific goals must be adopted, but at present, it appears that this minimalist approach is a needed starting point, as it is unclear whether many restoration projects have avoided being a cause of degradation themselves.

Different Metrics for Different Projects, Released Over Time for Metrics Met

A greater number of metrics are needed to quantify stream ecosystem changes as part of compensatory mitigation requirements [2008 Rule at §332.5 and §332.6; also §332.8(o)(2)]. Ecosystems are complex, but at present there are few metrics employed with most focus being on channel form (geomorphology), physical aquatic habitat, and possibly some limited biotic metrics (e.g., indicators of biotic integrity). In a 2011 (and ongoing) review of the success criteria of 12 states and Corps' guidance documents for stream mitigation, all states and districts required quantitative data on geomorphology of restored streams but none required quantitative biological or chemical data; that is, no states required all projects that restoration empirically demonstrate positive biological or chemical changes as part of determining whether a restoration project was deemed successful (M. Doyle, unpublished manuscript). As noted above, physical habitat restoration is a necessary but not sufficient condition for biological or chemical functional recovery; some states did require qualitative estimates of biological conditions, but these were generally limited to visual assessments of hydraulic habitat, not the actual organisms or water quality changes.

As an alternative approach to relying on geomorphic metrics alone, mitigators could be given a choice of multiple metrics of water quality, sediment loads, biota, and hydrology, among others. Some projects may have little impact on sediment loads but may exhibit strong aquatic macroinvertebrate recovery, or vice versa. The goal should be to have classes of metrics (chemical, physical, and biological), each containing several members. A collection of metrics should be used at each site to establish project success, and credits should be released proportional to the metrics met.

A notable example of this approach is the criteria used in North Carolina (developed by the Corps, Wilmington District, and state water quality regulatory agency) for evaluating dam removals for compensatory stream mitigation that assigns credits based on restoring water quality (with specific metrics), establishing a lotic aquatic community, and restoring threatened or endangered species re-colonization (U.S. Army Corps of Engineers, 2008). A portion of the total possible number of credits is given for each specific function restored; if only habitat and channel morphology is quantified, then a large portion of the credits associated with biotic metrics will not be released. Most relevant here as an example is how the dam removal guidance assigns credits for endangered species; credits for this category are divided into three parts ("layers") and released upon demonstration (i.e., quantification) of particular metrics. If habitat is quantified, then one-third of the endangered species portion of credits is released. If indicator species are documented, then another third can be released. If actual target species are demonstrably restored as a sustainable population, then the remainder of the credits is released. The critical aspect of this approach is that the simple claim of restoring habitat for species is insufficient for receiving all the credits for restoration; actual documentation of species is required. Furthermore, as credits are released in proportion to functions demonstrably restored rather than the "all or nothing" approach that is more often taken, mitigators will have greater incentive to carefully select sites and strategies. Restoration firms may optimize their channel designs or site locations for certain ecological functions while de-emphasizing others – a more realistic expectation of restoration. Different stream mitigation projects require different success criteria, and adaptation of success metrics will be needed because of the range of conditions encountered by different projects. As Moerke and Lamberti (2003) stated, "a single set of measurements may not be appropriate for evaluating all restorations; rather, metrics must be carefully chosen to fit the specific restoration." Although there is enormous latitude in the 2008 Rule for which criteria or monitoring are needed for projects (2008 Rule at §332.5 and §332.6), many states and Corps' districts in their interpretation of this rule for local guidance set the success criteria for all projects to a narrow group of measurements, most often specific geomorphic metrics that are assumed to be associated with high-quality streams. The lack of flexibility in success metrics may have unintended consequences on restoration practices, as mandating uniform success metrics across a wide geographic and ecological range will cause mitigators to adapt their design to create the metrics rather than to restore functions appropriate for the site.

Finally, requiring a range of success criteria metrics makes the mitigators propose specific functions that they will monitor, as it is essential that the features/processes measured be commensurate with the scale and scope of the compensatory mitigation project [§332.6(a)(1)]. This enables designers to maximize some functions while losing others; this is a much more honest and realistic tradeoff approach to restoration, meaning that mitigators will have to optimize which functions they are most capable of achieving. This policy will place the onus on regulators to establish realistic weightings for different functions. For instance, streambank erosion control and associated sediment production can be achieved with generous application of riprap, but this may not move toward the Clean Water Act biological integrity goals. Developing sustainable populations of sensitive aquatic biota, therefore, should be given greater weight than reducing bank erosion. Alternatively, regulators could weight specific functions for the specific watershed of interest. In a watershed with known water quality problems that is upstream of a drinking water reservoir, water quality improvement would be weighted more heavily. Alternatively, flood attenuation might be weighted heavily for watersheds where flash floods are a problem.

Allow Flexibility in Location: Emphasize Relative Location Rather Than Absolute Location

The 2008 Rule emphasizes the "watershed approach" to mitigation [2008 Rule, §332.3(c), particularly §332.3(c)(4)]. The "watershed approach" has been left undefined by both the Corps and the EPA, although District Engineers of the Corps must balance the ecological needs associated with compensatory mitigation as well as economic realities. The primary way that the idea of watershed approach has been implemented has been through the use of geographic service areas (GSAs), the area within which mitigation sites can compensate for impacts. In addition, watershed approach has been applied by requiring comparable stream orders or types; for example, impacts to a third-order stream could not be mitigated by restoration of a headwater, first-order stream. There is substantial variability in GSA regulation between states and Corps' districts, although many districts simply constrain GSAs to eight-digit Hydrologic Unit Catalog areas, and there is considerable discretion for locating mitigation projects or waiving the GSA requirement for specific projects (for national review, see Womble and Doyle, 2012).

Ensuring spatial proximity of impact to mitigation sites is important to ensure the proper gains from restoration for some stream functions but not necessarily all [2008 Rule at §332.8(d)(6)(ii)(A)]. For example, water quality degradation by impacts should be mitigated within the same watershed of the targeted receiving water body (e.g., estuary or drinking water supply). Similarly, flood attenuation functions could be important to keep within the same watershed and perhaps as far upstream as possible. Other functions, however, may be more appropriately mitigated out of watershed. For endangered species habitat, for instance, the goal is sustaining a viable population of a particular species, or sustaining appropriate genetic diversity, in which case spatial proximity to impact is probably a detriment rather than an asset. For these types of functions, it might be more appropriate to seek out distant, large, isolated banks that are proven locations of such species and that will be less likely to suffer degradation due to encroaching watershed development over longer periods of time. Thus, it cannot be simply assumed that mitigating all functions should be done as closely to the impacts as possible; different functions may require different GSAs.

Another aspect of location is the landscape context of the restoration site because upstream watershed conditions can exert as much influence on stream ecosystems as those within or immediately adjacent to the restored reach (Moerke and Lamberti, 2003; Walsh *et al.*, 2007; Roni *et al.*, 2008; Shields *et al.*, 2010), although this is not always the case (Moore and Palmer, 2005; Richardson *et al.*, 2011). The most notable aspect of specifics of site location is the reality that water pollution can overwhelm channel restoration effects. Nutrient concentrations in streams draining urban or agricultural watersheds can reach surprisingly high levels (Stanley and Maxted, 2008), and chronic toxicity associated with elevated nutrient concentrations can affect the growth, respiration, reproduction, and survival of many sensitive aquatic biota (Lewis and Morris, 1986; Camargo *et al.*, 2005; Camargo and Alonso, 2006). Also, high nutrient concentrations can reduce or eliminate the ability of streams to retain nutrients (Earl *et al.*, 2006). In addition, incised, enlarged channels, impervious surfaces and stormwater systems that bypass riparian zones cause high peak flows, lower base flows, lower groundwater tables, and degraded water quality. All of these effects can negate any potential channel restoration benefits (Groffman *et al.*, 2003; Shields *et al.*, 2008b and 2010).

Although local conditions can be important, broader, watershed-scale conditions can limit the "restorability" of many streams. Urban watersheds can have extremely limited restoration potential, as can those in intensively cultivated watersheds with water quality problems. It is not that these streams should be ignored, but rather that these limitations must be recognized explicitly in goals and channel designs. There is a great need to place emphasis on the limitations and other opportunities, as well as the need to adapt restoration design in these settings. In some urban watersheds with poor water quality and flashy hydrology, restoration should focus on improving hydrology (flood attenuation) and water quality while de-emphasizing biotic recovery, as the latter may simply not be feasible. Moreover, a diversity of channel, wetland and impoundment measures might optimize outcomes while fully acknowledging that all stream functions cannot be achieved along the entire reach of stream (Richardson *et al.*, 2011).

The take-home message is that "place matters." The same restoration design that can be successful in a suburban or forested setting may have little chance of success in a highly urban setting. As noted by wetland restorationists (Bedford, 1999; Zedler, 2000), the goals of restoration should not be the same for all locations; different locations need different goals to produce desirable effects at the watershed scale. *The most important aspect of preproject planning is to more carefully identify the specific need of that particular stream reach within that particular watershed*, that is, more precisely identifying what watershed processes and factors are limiting ecosystem integrity. To date, there has been too little consideration of project site location in evaluating proposed stream mitigation projects. Regulators in charge of evaluating proposed projects should consider whether the proposed functions may realistically be achieved given the constraints set by the geographic context of the proposed site.

Allow Flexibility for Large or Unique Projects

Requiring quantifiable changes in success metrics we suspect will inherently incentivize larger projects. Larger projects are needed in order to produce some types of measurable functional gains, such as flood attenuation, habitat, and water quality improvements. For instance, Ardon *et al.* (2010) found that a substantial length and area of stream and wetland restoration was needed in order to attenuate floods sufficiently to allow biogeochemical retention of nutrients: approximately 3 km of stream with immediate connection to over 440 ha of riparian wetlands. Further, fluvial systems frequently respond to external drivers in nonlinear or threshold fashions such that functional gains are not obtained until a minimum project length or area is exceeded. Gains beyond that point may be nonlinear up to a limit that is approached asymptotically.

As small gains are often not measurable, increased application of appropriate, quantitative metrics will devalue smaller projects. If functional gains are measured relative to simultaneous changes at degraded reference sites, even larger effects may be required for detection. Existing policies obstruct the ecologically attractive larger restoration projects. Most notably, small GSAs limit the spatial scale over which credits can be sold. Large restoration projects tend to be located in less developed areas often distant from stream reaches impacted by ongoing development. Cash flows for larger projects may be unattractive to mitigators as they require larger upfront investments with considerable economic risk, but their remote locations reduce demand for the credits they generate.

Flexibility in Process: Emphasize Results Not Process

To date, there has been heavy reliance on restoration via active manipulation of channel form, such as remeandering, base-level raising, or building instream habitat structures. Such approaches have mixed records of success and tend to be costly. Other approaches are available that rely on natural processes and are thus less invasive, less costly, and potentially more effective over the long term. Beechie *et al.* (2010) propose that restoration planners focus on process-based actions such as forest road removal, environmental flow restoration, levee removal, riparian forest recovery, beaver introduction, and dam removal rather than creating certain habitat conditions by actively modifying channels. As another example, riparian zone fencing for livestock exclusion can rapidly improve riparian vegetation, bank stability, and instream habitat (Roni *et al.*, 2002, 2008; but see Sarr, 2002), without sediment remobilization associated with the construction of instream structures or remeandering.

Strategic removal or abandonment of infrastructure such as dams, levees, or road crossings may prove particularly attractive as a restoration option. Over the past few decades, more than 300 dams have been removed in the U.S. Most of these structures were small, privately owned and operated dams that no longer served their original purpose (e.g., grist mills, textile mills). Many also posed safety hazards or economic liability to the owners and local communities. Removing such structures thus reduces liability and furnishes ecosystem restoration benefits. In the few cases where dam removal was studied ecologically, stream functions were restored in <10 years, and often in <5 years (Beasley and Hightower, 2000; Burdick and Hightower, 2006; Catalano *et al.*, 2007; Kanehl *et al.*, 1997; Doyle *et al.*, 2005a). Dam removals have also been used on at least two occasions under the auspices of compensatory stream mitigation; in these cases, the dam removals were associated with recovery of state and federally listed endangered species and accomplished large-scale restoration at costs substantially less than that required for more orthodox stream restorations (J. Preyer, Restoration Systems, LLC, April 23, 2010, personal communication).

Dam removals are not without problems, such as the release of sediments, nutrients, and other contaminants stored in former impoundments, and in some cases watershed conditions can continue to dominate restoration potential (Kareiva *et al.*, 2000). Abandonment or decommissioning of levees is another potentially effective restoration strategy but is less frequently practiced than dam removal. In a few cases, levees have been intentionally abandoned following damage by floods or as part of larger watershed restoration plans (Haase and Blodgett, 2009). Following the 1993 Midwest floods, levee districts used federal funds to put formerly leveed floodplains under conservation easement or purchased them outright for wildlife preserves (e.g., Big Muddy Fish and Wildlife Reserve, Lisbon Bottoms, Missouri River, Missouri). In these cases, the gains in ecosystem services are substantial, and minimal disturbance in the form of fill, dredging, or excavation was required. And like dam removal, there are substantial gains to public safety and reduced public spending that can be gained through these types of decommissioning aging infrastructure (Doyle *et al.*, 2008).

Unfortunately, existing policies do not encourage the use of a full suite of approaches to generating mitigation credits. Rather than requiring a specific approach to restoration, such as natural channel design or instream structures, there is great need for flexibility. In order to foster innovation, policies should be implemented that regulate the end result rather than the process.

ECONOMIC INSTRUMENTS FOR REDIRECTING RESTORATION EFFORTS

One of the key shortcomings in current policy is the lack of incentives to ensure long-term effectiveness. Although the 2008 Rule includes language to increase long-term management and sustainability [§332.7, particularly §332.7(b)], the details for how this is to be accomplished are vague beyond the requirements for real estate instruments. However, the 2008 Rule does allow flexibility for creating ways to increase financing of long-term management [§332.7(d)(3)], and we suggest that there may be some policy adaptations that could provide greater incentive for long-term monitoring and demonstrated gains in ecological function.

Incentivizing Monitoring via Economic Discounting

Economic discounts in the form of trading ratios might be used to provide incentives for monitoring and success. Trading ratios are often used to require that mitigation sites be larger or more numerous than impacted sites, as mitigation will never fully regain functions provided by an unimpacted stream (see Womble and Doyle, 2012). For example, a 5:1 trading ratio requires 5 m of stream be restored for every 1 m impacted for compensatory mitigation credit. By accepting restoration projects that are suboptimal but more numerous than impacted sites, liability for mitigation failure is widely distributed. However, uniform trading ratios can result in better projects being penalized for the bad. Trading ratios may be adjusted downward to provide incentives for (1) setting success criteria and monitoring projects and (2) projects with quantitative data showing some real restoration of certain ecosystem services. Trading ratios may be refined downward as the scientific basis for restoration becomes stronger (Olander, 2008). Such an approach would not penalize mitigators willing to pay for monitoring but would instead provide an incentive for credit purchasers to find projects that have actual data and success criteria, as this would lower their mitigation costs.

Restoration Portfolios

At the broadest level of revamping, the mitigation program would be to develop a "restoration portfolio" type program for both impactors and restorers. Restoration portfolios would be based on the assumption that ecosystems are bundles of ecosystem functions, and impacted sites will lose a certain portion of those functions. Restored sites will enhance a certain portion of those functions, but it is highly unlikely that those functions lost are perfectly matched by those functions gained at a mitigation site. Rather, the functions lost would need to be mitigated by two or, possibly, three different mitigation sites. For instance, a particular restoration site in an urban setting might provide sufficient flood attenuation and water quality enhancements, but the impacted site also provided known habitat to threatened species. In this case, additional credits, specifically toward those species, would be sought so that the combination of two sites would mitigate the three functions of interest. The mitigator would thus be in the business of building a diverse range of restoration sites that could service a broad need of many different types of impactors. The mitigator would be able to take advantage of a portfolio of restoration sites across a wide geographic area, the furthest of which might provide substantial endangered species habitat, whereas the closer might provide sufficient water quality benefits.

Although appealing in the abstract, this approach could likely suffer from severe accounting problems; each site would have to be evaluated for an extensive number of potential credits (and debits) that are gained from restoration. Numerous ecosystem functions from a small number of restoration sites could be listed. The other potential problem is that this approach would require some degree of discernment and flexibility on the part of the regulators for evaluating what functions may be lost and gained. Nevertheless, such an approach would provide greater opportunities for restorers to develop a portfolio of restoration sites, which together could be used to mitigate specific impacted sites of high value. The key advantage to mitigators is that it provides some economic insurance if one particular project is not able to provide a particular function.

Restoration Trust Funds for Long-Term Adaptive Management Approaches

Currently, there is heavy incentive to actively modify damaged streams but little incentive to collect monitoring data or proceed with passive or minimal interventions (e.g., beaver introductions; fencing riparian areas). As our earlier review has argued, actively modifying streams does not necessarily provide real restoration, and so adjustment to incentivize the shortest, least cost method of getting to ecosystem recovery is needed.

The "trust fund" approach would begin with identification of an acceptable active restoration plan, rigorous success metrics, and setting aside adequate funds to complete the planned project. But rather than proceeding with construction, the mitigators would perform incremental, minimal work and highly active monitoring. As soon as the project reach met or exceeded the proposed success metrics, the active intervention would cease. Monitoring would continue until regulators were satisfied. The remaining set-aside funds could be used to provide incentives using the approach that is currently used in some "Rigs to Reefs" programs for managing decommissioned offshore oil platforms (Frumkes, 2002; Kaiser, 2006): savings are divided between mitigators and an environmental or state trust fund.

For example, consider a 2-km reach of suburban stream identified by a mitigator as a project site. The mitigator designs a restoration project and, in collaboration with regulators, estimates cost to be US$1 million. The mitigator then proposes to use the "trust fund" approach to restore the site. The mitigator sets up a trust fund that would mature into US$1.5 million over 10 years, builds off-stream watering facilities and riparian fencing and easements for two local ranchers, and purchases an easement to eliminate cattle to access the stream corridor. He excludes cattle via fencing and does minimal riparian vegetation work but no channel earth moving. A rigorous monitoring program is established through a link with a local university. After five years, monitoring data show that the project reach has met 70% of its goals as gaged by water quality and habitat metrics, but no target fish are present. Regulators then release a portion of the credits but withhold that portion associated with fish population restoration; the mitigator decides to pursue the additional credits and installs a limited number of instream structures but finds these are not effective. He then installs woody debris jams in Year 7, continues to monitor, and detects the presence of several fish species that were targeted for recovery. This was accomplished without the need for full channel re-alignment. After 10 years, costs for administration, design, easements, construction, and monitoring total US$900,000, which is US$600,000 below the yield from the maturing trust fund. Half of those savings then go to the local state environmental trust fund for future easements and preservation work, and the mitigator "pockets" the remaining half.

The goal of such an approach, regardless of policy details or financial reality of the above example, is to create a program that incentivizes and rewards success rather than actions with no success. Current policies reward the activity of restoration but do not necessarily adequately require or reward the restoration of function. A significant drawback of this approach, however, is the potential for it to reduce costs of impacting streams; by reducing costs for mitigation credits, costs are also reduced for impacts. Such tradeoffs present great difficulties for the agencies that regulate compensatory mitigation.

CONCLUSIONS

Available evidence suggests low rates of success for stream restoration projects. Because of the use of stream restoration in compensatory mitigation, we are led to conclude that there has been, and continues to be, a national net loss of physical, chemical, and/or biological integrity of streams. Although some innovative approaches appear to be working, and more successful cases may emerge, success of restoration cannot be assumed, and the capability of designers to produce restoration projects that compensate for impacts on stream ecosystem processes remains uncertain. Accordingly, there should be greater emphasis on avoidance and minimization. Success criteria for restoration projects should become substantially more rigorous. There are many benefits of such increased rigor, but most importantly, more stringent success criteria would increase mitigation costs and therefore decrease impacts in the first place. Discounting the use of restoration portfolios and trust funds, more flexibility in project size and location, and emphasis on long-term documented effects would be appropriate policy approaches to the high level of uncertainty in current restoration practice.

Future stream restoration research should examine available restoration meta-analyses to compare and contrast effects of enhancement and restoration. A critically limiting but necessary assumption underlying this present work is that restoration of the past 10 years is a little different from that driven by mitigation. We have no data to suggest that this is not the case, but this remains an important arena of research.

Clear guidelines are needed to define economically feasible stream ecosystem monitoring approaches. What is measurable? What functions are of interest, and what functions can be measured with available technology? If metric components are not measurable, then the metric should not be applied. Accordingly, a science of project size is needed to define the role of project size on providing measurable changes in ecosystem functions. Similarly, tools are needed to quantify relative benefits of restoration locations within a given watershed to watershed-scale ecological services.

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