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Consideration of spatial and temporal scales in stream restorations and biotic monitoring to assess restoration outcomes: A literature review, Part 1

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Abstract

Stream and river restoration practices have become common in many parts of the world. We ask the question whether such restorations improve freshwater biotic assemblages or functions over time, and if not, can general reasons be identified for such outcomes. We conducted a literature survey and review of studies in which different types of stream restorations were conducted and outcomes reported. These restoration types included culvert restoration; acid mine restoration or industrial pollutant restoration; urban stream restoration; dam removal, changes in dam operation, or fish passage structures; instream habitat modification; riparian restoration or woody material addition; channel restoration and multiple restorations. The streams ranged from headwater streams to large rivers, and the regions included North America, Europe, Australia and New Zealand, and a small number of sites in Asia and Africa. In this part of the review, we describe the methods used for the review and present reviews for the first three types of stream restorations. For culvert restorations, the small sample size and variable study design and biotic responses limited generalizing about temporal and spatial scale effects for that restoration type. The complex and often lengthy time to restore streams from acid mine drainage and industrial pollutants often resulted in positive biotic responses, but restored sites had reduced responses compared to reference sites. Most urban stream restorations had minimal or mixed improvements in biotic responses, with one mismatch in spatial scale evidenced by hydraulic structures used in a restoration unable to withstand peak discharge.

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DISCLAIMER

This review was prepared at the USEPA, ORD, National Center for Environmental Assessment, Cincinnati Division. It has been subjected to the agency's peer and administrative review and approved for publication. However, the views expressed are those of the authors and do not necessarily represent the views or policies of the USEPA. The authors declare that there are no conflicts of interest.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

Keywords

stream and river restoration; biotic assemblages; biotic functions; outcome assessment; spatial and temporal scales

1 | INTRODUCTION

Human activities have altered either directly or indirectly stream ecosystems at various scales throughout the world. Direct alterations include building of dams; associated headraces, tailraces and other structures that divert flows from the stream channel; the reconstruction of stream channels to straighten, simplify, deepen, move or otherwise modify channels; or the release of effluents that affect aquatic biota and ecosystem functions. Indirect alterations include changes in riparian zone or catchment land cover and use that alter energy and material inputs along with flow regimes and sediment supplies. To counter these human alterations and their effects on biotic assemblages and function, various types of stream or river restorations have been conducted at substantial expense (Alexander & Allan, 2006; Bernhardt et al., 2005). Early reviews of the literature reporting the effects of stream restoration criticized the studies for not conducting follow-up monitoring to assess the outcomes. Recent studies have begun to assess the results of these restorations, and some reviews have begun to assess determinants of success (Jähnig et al., 2011; Nilsson et al., 2015; Wortley, Hero, & Howes, 2013). We conducted a literature survey of studies in which stream restorations were conducted for different sources of impairment and in which an assessment was conducted to determine whether the effects on the local aquatic biotic assemblage or functions were positive or not. Based on our prior reading of other papers on stream restoration (e.g., Smucker & Detenbeck, 2014; Smucker et al., 2016), we wanted to assess whether the improvement in biotic assemblages or function associated with different types of restorations are related to how well the spatial scales of the impairment and restoration match and to the quality and timing of biotic monitoring after restoration.

Stream ecosystems are organized into a nested hierarchy of habitats with differing linear spatial and temporal scales of potential persistence (Benda et al., 2004; Fausch, Torgersen, Baxter, & Li, 2002; Frissell, Liss, Warren, & Hurley, 1986; Gurnell et al., 2016). However, geomorphic and biotic functions, such as water flow, sediment loading, organic matter (OM) input and transport or nutrient spiraling (Gregory, Swanson, McKee, & Cummins, 1991), as well as metacommunity dynamics of the biotic assemblages (Altermatt, 2013; Heino, 2013; Leibold et al., 2004), can occur at scales of much longer stream segments—up to the entire catchment and to the landscape scale (Freeman, Pringle, & Jackson, 2007; Newson & Newson, 2000). Geomorphic functions and biotic assemblages may also respond to alterations of the landscape at much larger spatial and temporal scales (Minshall, 1988). Conversely, for practical reasons, biotic monitoring is often limited to a reach scale, such as a reach that is 20 times the mean wetted width or the length within a certain number of riffle and pool units or meanders in a stream.

2 | METHODS

We conducted a search of the peer-reviewed literature published in English since January 1, 2000 using Web of Science to identify studies of stream restorations conducted in catchments where the sources of stressors were urbanization; agriculture, including livestock grazing; forestry management; industrial or mine effluents; channel alterations; or dams. The search was carried out in August 2016, and an iterative strategy was used to develop the search string and narrow the search. The final search string, “TOPIC: (Stream* OR river* OR creek*) AND TOPIC: (Restor* OR “dam removal” OR dechannel*) AND TOPIC: (Stressor* OR disturbance OR urban* OR dam OR dams OR channel* OR degrade* OR erosion OR [sediment AND load*] OR suburbanization OR agriculture OR eutrophication OR [nutrient* AND load*] OR livestock OR grazing)” refined by: “LANGUAGES: (English) AND RESEARCH AREAS: (environmental sciences ecology OR ecology OR water resources OR marine freshwater biology OR fisheries OR biodiversity conservation),” resulted in 4225 studies.

To further narrow the results to those studies most likely to be relevant, the results of this search strategy were downloaded and analyzed with the approach described by Varghese, Cawley, and Hong (2018) that uses semi-supervised machine learning algorithms for topic extraction and supervised clustering to classify the studies as considered and not considered. This approach identified 1710 studies, which were tagged as “Considered” and added to the Stream Restoration database in the US Environmental Protection Agency’s (USEPA’s) Health & Environmental Research Online (HERO) database (USEPA, 2018a).

A study was chosen for annotation and further review if it included descriptions of (1) the stressors being remediated and their sources; (2) the type of restoration conducted; (3) monitoring to assess any physiochemical changes and changes to one or more biotic assemblages (i.e., fish, macroinvertebrates, macrophytes, and periphyton) or their functioning (i.e., production, respiration, and nutrient retention) resulting from the restoration; and (4) a monitoring design that allowed an assessment of whether or not the restoration improved the biotic assemblage or its functioning. Implicit in these criteria is that an intended objective of a reviewed restoration was the improvement of a biotic assemblage or its functioning and not just the improvement of some physical or chemical characteristic of the stream.

Initial title and abstract screens were conducted by two independent contractor analysts followed by a contractor technical expert (ICF, Fairfax, VA) to further divide the “Considered” studies between those that were potentially relevant or not relevant. The relevant category included 643 papers. The full text of each potentially relevant study was then evaluated and annotated by a single contractor analyst. The resulting number of annotated studies included 381 papers.

These 381 annotated papers were reviewed by the coauthors of this review to check whether the studies met the above criteria and assess whether the biotic assemblage or function was improved or not. We also grouped the studies into restoration types: (1) culvert restoration; (2) acid mine restoration or industrial pollutant restoration; (3) urban stream restoration;

(4) dam removal, changes in dam operation, or fish passage structures; (5) instream habitat modification; (6) riparian restoration or woody material addition; (7) channel restoration and (8) multiple restorations. These groups of restoration types were based on a recognition that scales of restoration and the stressors appear to vary among types, and separate consideration of this variation within and among restoration types would simplify our synthesis of the results of these diverse studies. However, we recognize that these types are not mutually exclusive. For our purposes, we considered a restoration to have improved the biotic assemblage or functioning if there was a statistically significant (generally $p < 0.05$ or as indicated by the authors) improvement in the reported measurements of the biotic assemblages or their function compared to an unrestored control, which was usually the same site prior to the restoration. Also, when available, we considered whether the restored stream became more similar to an undegraded reference site. However, we recognize that this may be a minimal definition of improvement. In a number of cases, statistically significant results were small increases in taxa richness (i.e., 1 or 2 species) or abundance of 1 taxon. These cases were difficult to judge as to their biological significance (Johnson, 1999; Nakagawa & Cuthill, 2007), particularly when no data was presented for relatively undegraded reference sites that could represent a restoration target.

To characterize the streams where the restorations occurred, we extracted any data supplied by the authors that described stream size, such as stream order, width, mean annual discharge, or catchment area, along with the location, including the country, major administrative subdivision (i.e., state or province), and freshwater ecoregion according to Omernik (1987) for the USA and Canada (Wilken, Nava, & Griffith, 2011) and Abell et al. (2008) for other countries. For dam removals, we also recorded data on the dam size, and for urban restorations, we recorded the city or town. To assess the temporal scale of the restoration and its monitoring, we also recorded the available information of the dates of any biological monitoring prior to the restoration, during the restoration and after the restoration. To characterize the stream restorations, we summarized the restoration activity and the observed results for each study.

3 | RESULTS

Of the 381 annotated papers, 198 papers were found to lack at least one of the criteria as described in Supplementary Table 1. In many cases, studies were excluded because the papers lacked a quantitative measurement of a biotic assemblage or function. In 24 cases, 2 to 4 papers that assessed the same restoration were reviewed as a single restoration. This left 127 distinct restorations that were evaluated in this review. Geographically, most of the studies were conducted in either North America (69) or Europe (38) with some studies conducted in Australia (9), New Zealand (6), Asia (4) and Africa (1). Following are reviews of the studies by each restoration type for the first three types: culverts, acid mine drainage and industrial pollutants, and urban stream restoration. Details on the individual restoration studies are compiled in Supplementary Table 2 and Tables 1–3. For these three restoration types, which had fewer examples, we describe the analysis of individual papers that lead to our synthesis regarding the restoration type. The remaining five restoration types are reviewed in Part 2 (Griffith & McManus, 2020).

Although the literature search was limited to peer-reviewed papers published from January 1, 2000 to August 2016, some papers that met all stated criteria reported restorations that often began before 2000. This time lag was due, in part, to the time, often a number of years, required to follow up restorations with biological monitoring.

3.1 | Culverts

Culvert restorations had the fewest papers, four, returned from the literature search among the eight topics (see Supplementary Table 2 and Table 1 for details of the reviewed studies). The replacement or restoration of culverts at the intersections of streams and roads is meant to improve fish passage along a stream network. Such restoration projects range from replacing the culvert with an arched bridge to retrofitting a culvert with baffles to increase surface roughness and reduce water velocities (Favaro, Moore, Reynolds, & Beakes, 2014; Franklin & Bartels, 2012; Ogren & Huckins, 2015). In two culvert restoration studies in the Manistee River, Ogren and Huckins (2015) observed that neither macroinvertebrate nor fish IBI scores at restored sites differed from control stream sites, while Evans, Riley, and Lamberti (2015) found the proportional similarity of fish assemblages increased upstream and downstream of a culvert replacement by a bridge but acknowledged that other drivers could have produced such a response. Another culvert restoration combined a fish ramp with the installation of baffles in the culvert (Franklin & Bartels, 2012). Species richness and fish density both increased in the reach upstream of the restoration. In a comparative study of culvert restoration in tributaries of the Lower Fraser River in urban Vancouver, Favaro et al. (2014) analyzed fish response variables in relation to stream type: reference (no culvert), culvert with baffles and culvert without baffles. The effect of stream type on fish densities was species specific. Also, fish communities differed among stream types, and species richness increased with the time (years) since baffles were installed in the culverts, suggesting that the effect of restoration may take decades.

Culverts are installed where roads intersect small streams and affect the hydrology and physical habitat of streams (Favaro et al., 2014). While culvert restoration is undertaken at a specific location on the stream network, evaluating the effectiveness of that restoration is difficult for two reasons (Table 3). First, the time lag between the restoration completion and the expected biotic response is unknown. That is, over what duration should the monitoring be done? Second, it is unclear at what spatial extent monitoring should be conducted to evaluate whether the dispersal barriers caused by the culvert have been eliminated. Three of the four culvert studies evaluated specific replacement or modifications of culverts, so monitoring data were collected over small geographic extents and over time spans of one to 8 years (Supplementary Table 2). The Vancouver study relied on a single, summer field season to collect fish monitoring data at more sites, presumably over a larger geographic area. The combination of only four studies with quite variable study designs and responses precludes assessing if the scale of the culvert restorations corresponded to the scale of the stressor impacting fishes and benthic invertebrates.

3.2 | Acid mine drainage and industrial pollutants

Current and historical hard rock and coal mining operations can produce multiple stressors on stream biota that include acidification of stream water, increased toxic

metal concentrations and iron hydroxide coating of stream substrate (Walter, Nelson, & Earle, 2012). These multiple stressors often require multiple amelioration methods (see Supplementary Table 2 and Table 2 for details of reviewed studies). Stream restoration efforts are focused on low -ordered streams in catchments because mining operations frequently occur there, although the effects of mining can persist tens to hundreds of stream kilometers downstream (Hornberger, Luoma, Johnson, & Holyoak, 2009).

An assessment of restoration efforts in a metal-contaminated stream in the upper Arkansas River spanned 17 years (Clements, Vieira, & Church, 2010). Downstream of the Arkansas River headwaters, the catchment around California Gulch, a tributary, was designated a USEPA Superfund site. Restoration efforts included construction of two water treatment facilities, removal of mine tailings and revegetation of riparian areas. Biotic responses showed the greatest improvement at upstream sites, nearest the restoration sites, with slower responses at downstream sites, and the poorest response at the site furthest downstream was attributed to residual mine tailings on the floodplain (Clements et al., 2010). The duration of the follow-up monitoring was long enough to detect both seasonal and episodic climatic effects.

Although not continuously monitored, Williams and Turner (2015) evaluated restoration of acid mine drainage by comparing a 1967 biological and chemical stream survey as pre-restoration data to post-restoration- data collected in 2011. The multiple restoration activities included surface mine revegetation and sealing deep mines along with an anoxic limestone drain and settling ponds. Seven of the eight biotic response variables were all greater in 2011 than in 1967. The same time period saw increases in pH and alkalinity and decreases in free acidity, SO_4^{2-} and Fe concentrations. Another restoration project spanning 11 years evaluated if wetlands constructed to treat iron hydroxides from abandoned coal mine drainage led to responses in the stream macroinvertebrate community (Walter et al., 2012). Downstream of the wetland treatments, taxa density increased over time, and benthic communities were more similar between pairs of upstream and downstream sites after the wetland treatment.

Gunn, Sarrazin-Delay, Wesolek, Stasko, and Szkokan-Emilson (2010) showed that benthic macroinvertebrates differed significantly at a site where acid mine drainage was diverted to a water treatment plant compared with control sites. While some improvements in water chemistry were observed, other stressors potentially affecting the invertebrate response included urbanization and a lack of coarse particulate organic matter (CPOM; i.e., tree leaves) to the stream.

Industrial discharges into stream networks can introduce a range of chemicals that necessitate stream remediation. Some of those chemicals may include polycyclic aromatic hydrocarbons, chlorinated organics and heavy metals. Restoration efforts for such stream contaminants have been the focus of research at Oak Ridge National Laboratory. Much of the research has been done on the East Fork Popular Creek (EFPC), whose headwaters have received various contaminants from point-source- discharges for about 45 years (see Supplementary Table 2 and Table 3 for details of reviewed studies)(Adams & Ham, 2011; Adams, Ryon, & Smith, 2005). A series of restoration activities were implemented to reduce

the contaminants from these point-source discharges. Monitoring was conducted before, during and after restoration at 3 – 6 contaminated sites on the EFPC stream network and at one or two reference sites in nearby uncontaminated streams (Adams et al., 2005; Ryon, 2011; Smith, Brandt, & Christensen, 2011). Fish assemblage monitoring from 1985 to 2004 showed sensitive species richness increased for EFPC sites furthest from the source of contaminants but was still lower than the reference sites indicating a partial recovery of the fish community. For benthic invertebrates, the uppermost two sites closest to the restoration work had a positive trend over 18 years in total and Ephemeroptera-Plecoptera-Trichoptera (EPT) richness, but both richness measurements were below that of the reference sites. A Before-After-Control-Impact (BACI) comparison for effects of specific pollution abatement periods on total richness, EPT richness and EPT density showed improvement in most of those metrics at the two uppermost sites, but at a site further downstream, only three of the nine comparisons were statistically significant. Smith et al. (2011) concluded that recovery was ongoing at the two headwater sites even 7 years after the last restoration activity and attributing responses to those activities at sites 9 km or further downstream was difficult. Even with long-term monitoring on a small catchment, the EFPC studies illustrate the complexities of evaluating restoration activities. Some of those complexities include multiple stressors that can act in concert to impact a stream and a lag in the response of ecological indicators to remediation actions. Furthermore, chemical contamination was even detected at what was once considered a reference site.

For acid mine and industrial pollutants restoration, the source of stressors typically lies in headwaters, and multiple and expensive restorative methods have been implemented over several years or decades. Three of the four acid mine restoration activities saw improvements in fish and macroinvertebrate responses (Clements et al., 2010; Walter et al., 2012; Williams & Turner, 2015), and such responses were also observed in the EFPC studies at Oak Ridge (Adams et al., 2005; Ryon, 2011; Smith et al., 2011). However, in both types of restoration, the impacted sites often still differed from reference sites. The time span of monitoring data covered 8 to 44 years, with fish and benthic invertebrate data collected at 1 to 12 sites. Gunn et al. (2010) observed improvements in water chemistry, but other stressors, namely urbanization and lack of CPOM, were suggested as causing the impacted site to differ in the benthic invertebrate assemblage from the reference sites even after 8 years following restoration. This result suggests that the scale of the restoration was not sufficient to fully mitigate the stressors, but at the other four study locations, the scale of restoration efforts did correspond to that of the stressors.

3.3 | Urban stream restoration

Urban stream restoration is complex because multiple stressors are associated with increased impervious surface area (Paul & Meyer, 2001). Such stressors include high peak flows and low base flows, erosion of stream banks, thermal stress and chemical pollutants (Palmer & Bernhardt, 2006). It is difficult to define reference streams for comparison to restored streams, and there are many constraints on feasible restoration activities, because of the presence of private property and infrastructure that typically require protection from flooding.

Of the reviewed studies, 2, 8 and 7 showed overall positive, minimal and mixed improvements in biotic effects, respectively, to urban stream restoration (see Supplementary Table 2 and Table 3 for details of reviewed studies). The study of Nine Mile Run illustrated best the mismatch between the scale of the stressor and stream restoration (Bain et al., 2014). Even the 2 studies with positive improvements acknowledged that the small length of stream restored would be unlikely to lead to full biotic recovery given catchment-scale drivers, such as piped streams and urban hydrology (Neale & Moffett, 2016; Rios-Touma, Prescott, Axtell, & Kondolf, 2015).

A multidisciplinary approach was used to evaluate an urban stream restoration in Nine Mile Run in Pittsburgh (PA, USA) (Bain et al., 2014). While pre- and post- restoration NO_3^- concentrations of water samples showed limited changes, fish and macroinvertebrate measurements improved only modestly. Hydraulic structures installed for the restoration had to be repaired several times, because their design flow was initially underestimated. This restoration project highlights the mismatch between one of the stressors, peak discharge, and the scale of the restoration implemented.

A stream restoration project that produced a wide, deep, low-velocity channel also had mixed results (Arango, James, & Hatch, 2015). The restoration had no effect on NH_4 and P concentrations, but their uptake velocities and the algal biomass accrual rate were greater in the restored reach versus the upstream reference. Aquatic insect density also increased after restoration. However, the initial post-restoration fish assemblage did not persist, and fish densities did not differ before and after the restoration (Arango et al., 2015). The lack of improved fish species richness or composition after restoration in association with increased habitat complexity may occur because the reach scale restoration did not address the other multiple catchment-scale stressors (Palmer, Menninger, & Bernhardt, 2010).

Altered hydrologic regimes, modified channel and riparian areas and inputs of pollutants and sediment affect urban streams (Blakely & Harding, 2005). Taxa richness and Trichoptera taxa rich were not different among a restored and two unrestored streams, and Blakely and Harding (2005) attributed the lack of effects to the presence of high heavy metals, intermittent flows and barrier to colonization. In Christchurch, New Zealand, invertebrate responses were quite variable after 5 years and had no strong association with the restoration, which was riparian planting with or without channel modification (Suren & McMurtrie, 2005). In three Melbourne, Australia, streams, changes in macroinvertebrate biomass were inconsistent between reaches with riparian planting of at least 15-year-old *Eucalyptus* and *Acacia* compared to an upstream open reach that lacked native riparian vegetation (Thompson and Parkinson, 2011). However, macroinvertebrate assemblage composition improved and reflected greater inputs of allochthonous detritus in planted reaches. This is consistent with riparian restoration in non-urban catchments (Giling, Grace, Mac Nally, & Thompson, 2013; McTammany, Benfield, & Webster, 2007; Quinn, Croker, Smith, & Bellingham, 2009). However, all the reaches were still dominated by pollution tolerant macroinvertebrate, and Thompson and Parkinson (2011) suggest these urban streams were also affected by previously observed flashy hydrology and high nutrient loads (Walsh et al., 2005). These effects may have been ameliorated by an ongoing drought,

but drainage systems carrying stormwater and nutrients bypass the riparian restoration of planting vegetation (Thompson & Parkinson, 2011).

Streambank stabilization in urban streams ranges from bioengineering using nonstructural methods (i.e., live cuttings and geotextile fabrics), to hard structural methods (i.e., riprap and concrete). Sudduth and Meyer (2006) examined reaches with bioengineered, stabilized streambanks and found no differences in total macroinvertebrate abundance, diversity or taxa richness on those banks between unrestored and reference reaches. Macroinvertebrate biomass was greater in reference sites and the 7- and 9-year old restoration sites compared to the 3-year old restoration site, and they concluded that planting and live cuttings inserted between riprap should not be used, because it provides little OM for macroinvertebrates. Also, bank stabilizations at the reach-scale were ecologically ineffective in an urbanized catchment (Sudduth & Meyer, 2006). Similarly, no or slight improvements in invertebrate indices were recorded in a stream restoration study that used bioengineered material, as well rock armoring and plantings of native vegetation (Selvakumar, O'Connor, & Struck, 2010). However, NH_4 uptake length was shorter in restored stream sites compared with the unrestored sites (Hines & Hershey, 2011), which was attributed to the greater surface area of the rock structures where biofilms developed and NH_4 uptake occurred.

The difficulty of defining reference sites for urban stream restoration activities is illustrated by a study of four stream restoration projects undertaken in the Tryon Creek watershed that used three reference reaches outside of the catchment (Rios-Touma, Prescott, Axtell, & Kondolf, 2015). Unimpacted reference sites within the watershed could not be found. The restoration activities enhanced stream habitat complexity, stabilized banks, reconnected floodplains and planted more vegetation in riparian and upland areas. Taxa and EPT richness increased from before to after the restoration, but both richness metrics were less than in the reference sites. The differences were attributed to lower CPOM in the restored reaches and to differences in catchment land use (Rios-Touma et al., 2015).

Restoration of much larger rivers flowing through large urban areas often entail structural and connectivity modifications. Along with urban development, the Danube River in Vienna, Austria, was impacted by channel straightening and construction of a hydroelectric power plant. Loss of riverine inshore habitat occurred because riprap was the dominant shoreline structure of the spoil island prior to the rehabilitation (Chovanec, Schiemer, Waidbacher, & Spolwind, 2002). The restoration produced side channels, shallow water areas, coves, gravel banks, small permanent backwaters and temporary waters. Fish from nine restored sites in 1999 were compared to a single preresoration sample collected from a cove in 1993. One rehabilitated site clustered tightly with the reference site based on fish species presence/absence, and fish species richness increased with connectivity. An urban floodplain restoration project was also undertaken on the Danube River to evaluate the effects of increasing surface water connectivity to several backwaters, and the most downstream backwater functioned as a sink for dissolved P and chlorophyll *a*, with the function attributed to dense stands of submerged macrophytes (Bondar-Kunze, Preiner, Schiemer, Weigelhofer & Hein, 2009). The restoration in the Spree river in Berlin created a bay and artificial structures for spawning and nursery habitat (Wolter, 2010). Species diversity did not increase, but fish density did increase around the artificial structures.

A restoration unique to urban streams, daylighting, which opens a stream that had previously been buried or piped, as was undertaken on two reaches on adjacent streams in Auckland, New Zealand (Neale & Moffett, 2016). A pre- to post treatment comparison showed a statistically significant improvement in the stream invertebrate communities with the greater difference in the south reach attributed to forested headwaters having a source population for colonists. Another novel urban stream restoration method constructed artificial ponds to serve as settling basins for the gradual deposition and suspension of pollution in the Sokołówka River in Łódź, Poland (Zelazna-Wieczorek & Nowicka-Krawczyk, 2015). Benthic samples of diatoms were collected and used to construct a Specific Pollution Sensitivity Index, Eutrophication/Pollution Index and Trophic Diatom Index. Based on those indices, a longitudinal profile showed that 13 of the 14 sites had an ecological status of poor or bad, with only one site being moderate. Zelazna-Wieczorek and Nowicka-Krawczyk (2015) concluded that the construction of a cascade of artificial ponds was not an effective means of restoring urban streams.

In the Baltimore (MD, USA) urban Long-Term Ecological Research project, portions of Minebank Run were restored to stabilize the stream geomorphometry and reduce channel erosion and incision (Harrison, Groffman, Mayer, & Kaushal, 2012). This increased the number of pools in the study reach, but the number of riffles and runs did not differ before and after restoration. However, reach nitrification, denitrification and methane production did not differ between pre- and post-restoration (Harrison et al., 2012). While Minebank Run and Spring Branch Run, another restored stream, each had lower surface velocities than two unrestored streams, the surface velocities and nutrient uptake metrics in the restored streams were not significantly different from the unrestored streams (Klockner, Kaushal, Groffman, Mayer, & Morgan, 2009). Among forested, restored urban and degraded urban streams, molar C:N ratios were statistically significantly greater in forested streams compared with the other two stream types (Newcomer et al., 2012). Denitrification rates were not different between restored and degraded urban streams but were significantly greater than those in the forested streams. Using natural organic carbon sources, denitrification rates were reported to be greater for degraded urban streams than for forested and restored sites, which were not significantly different (Newcomer et al., 2012). In North Carolina, denitrification rates were significantly greater in restored sites that used Natural Channel Design compared with unrestored sites, and restored sites also had significantly greater channel complexity (Tuttle, McMillan, Gardner, & Jennings, 2014).

The complexity of evaluating urban stream restorations is shown by the decadal synthesis of hydrologic, geomorphic, water quality and biotic data of the large (19.4 km²) catchment of Nine Mile Run (Bain et al. 2014). The altered hydrology of urban streams is illustrated by the prevalence of piped or buried drainages in the maps of Bain et al. (2014) and Neale and Moffett (2016). Those studies, along with other studies, often attributed the small extent of the urban stream restoration relative to the catchment's water quality, CPOM, stormflow intensity and bedload intensity as rendering the restoration ineffective (Bain et al. 2014, Neale and Moffett 2016, Rios-Touma 2015, Selvakumar et al. 2010, Sudduth and Meyers 2006, Tuttle et al. 2014). Other studies were less precise in both the spatial characterization of the catchments and extent of the restorations. Thompson and Parkinson (2011) called

for research at multiple scales of impacts on urban streams to evaluate effects of local restoration measures.

4 | SUMMARY

Two of the restoration types reviewed here, acid mine and industrial pollutants and urban streams, reported studies with either improved biotic measures or no statistically significant improvements associated with the temporal and spatial extent of the restoration in their respective catchments. Acid mine and industrial pollutants and urban stream studies were implemented over small areas. While these studies were smaller in scale, they often involved complex restoration activities phased in over several years or decades. Unlike other restorations that address mainly hydrological alterations to streams or rivers, these restoration activities involve treating water contamination, such as through the use of wastewater treatment plants.

For the acid mine and industrial pollutants restoration, the likely source of the stressor was known and located on low-order streams. Restoration of acid mine drainage and industrial pollutants tended to focus on treating stream chemistry at locations along the stream network. Although, altered stream chemistry was treated, improvements in biota did not always occur throughout the stream network. For example, in one mining restoration, a floodplain source of residual contaminants was thought to affect recovery of biota furthest downstream (Clements et al., 2010), and in another, possible new stressors in the form of urbanization and reduced CPOM were thought to prevent recovery of the macroinvertebrate community (Gunn et al., 2010).

In contrast, in urban stream restoration where the primary stressor, impervious surfaces are distributed throughout catchment, the consequent effects from high peak flows and low base flows, stream bank erosion, thermal stress and chemical pollutants are spatially distributed over the area and stream network (Palmer & Bernhardt, 2006). The two retrospective urban stream restoration studies, Bain et al. (2014) and Rios-Touma et al. (2015), provided interpretations as to why those restorations did not succeed. The multidisciplinary approach used by Bain et al. (2014) examined hydrological, water chemistry and biotic responses to a variety of restoration projects considered in aggregate in the catchment, and observed hydrological restoration failures due to under designed flows used in the planning. Stream restoration projects either in urban or nonurban areas require multidisciplinary expertise in engineering, hydrology, geomorphology and ecology as well as sociology (Palmer & Bernhardt, 2006).

In the Tryon Creek watershed, several different types of restoration were implemented, and the differences between restored and reference sites were attributed to land use differences between catchments and lower CPOM in Tryon Creek (Rios-Touma et al., 2015). This was one of the few studies that posed a conceptual model of restoration at instream and catchment levels affecting system changes and then ecological responses. Conceptual models may better help match an appropriately-scaled restoration to the stressors affecting particular streams along with biotic monitoring to assess the outcomes of the restoration activities.

In Part 2 (Griffith & McManus, 2020), we will review the five addition stream restoration types and discuss other considerations when planning stream restorations and biomonitoring of the outcomes.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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TABLE 1.

Details and results of studies in which the restoration was culvert restoration

Citation	Details	Results
Ogren and Huckins (2015)	Put in full spanning structures to replace culverts. At three sites used BACI-paired series and measured cold water fishes and macroinvertebrate biotic integrity indices.	Not able to detect improvements in IBI scores for fish or macroinvertebrates. Sample sizes were small because there were only 2–4 years of pre-monitoring data and 3–5 years of post-monitoring data.
Evans et al. (2015)	Replaced three culverts with bridges.	Prior to culvert removal, South Pine Creek upstream to downstream similarity was 68.7%, and 3 years post-culvert removal it increased to 93.7%. North Pine Creek went from 77.9% similarity prior to culvert removal to 86.8 % 3 years after culvert removal. Hinton Creek had 86.8% similarity prior to culvert removal, and it increased to 95.9% 3 years after culvert removal.
Franklin and Bartels (2012)	Initially installed pool and ramp from Waikato River to connect to Bankwood Stream where a culvert was located to solve perched culvert problem. Fish ramp alone failed to provide passage for some fishes so plastic spoiler baffle sheets were installed within culvert to lower water velocities and provide surface roughness.	Mean species increased from 3.0 in Jan. 2006/2007 before ramp to 5.5 in Jan. 2010/2011 surveys after spoiler baffles installed, and fish density increased.
Favaro et al. (2014)	Streams with nonbaffled culverts ($n = 7$), streams with baffled culverts ($n = 13$), reference (unculverted) streams ($n = 6$)	Culverts had a species-specific effect. Densities of two sculpin species were significantly lower in culvert and culvert-baffled streams compared to reference streams. Rainbow and cutthroat trout densities were lower in reference streams. Fish assemblage differed among the stream types. Species richness had positive relationship with the age, in years, of the baffles.

BACI, Before-After-Control-Impact; IBI, Index of Biotic Integrity

TABLE 2.
Details and results of studies in which the restoration was acid mine drainage or industrial pollutant restoration

Citation	Details	Results
Clements et al. (2010)	Installed water treatment facility in 1992 to treat upper portion of watershed, and in 1994 another such facility was installed for lower portion of watershed. In 1999, removal of mine tailings from California Gulch and revegetation of riparian areas.	The monitoring design for macroinvertebrates and fish compared two upstream sites to two downstream, highly contaminated, sites, over the 17 years, and comparisons were also made to reference streams ($n = 62$) for the macroinvertebrates. The macroinvertebrate and fish responses at the downstream sites furthest from restoration treatments lagged responses at the upstream sites, with that lagged response at the most downstream site attributed to residual mine tailings in the floodplain.
Williams and Turner (2015)	Between 1970 and 2000, recontouring and revegetating surface mines in the headwaters, injecting and sealing deep mines with a limestone slurry, plugging abandoned gas and oil wells and using an anoxic limestone drain and two settling ponds to treat acid mine drainage discharge.	Seven of the eight biotic response variables, five for macroinvertebrates and three for fish, were all greater in 2011 than in 1967, with only % EPT not being statistically different between the years. Also, observed increases in pH and alkalinity and decreases in free acidity, SO_4^{2-} , and Fe concentrations in 2011 compared with 1967.
Walter et al. (2012)	A BACI monitoring design was used to compare sites upstream of the wetlands to downstream sites in macroinvertebrate responses. Constructed wetlands used to treat iron hydroxides from abandoned coal mine drainage, and macroinvertebrates were sampled from 1994–2004.	Prior to the wetland treatment, community similarity indices between pairs of upstream and downstream sites were low, but after the wetland treatment, those indices became higher, indicating more similarity.
Gunn et al. (2010)	Restoration done in the headwaters of the Frood Branch of Junction Creek entailed damming and diverting AMD runoff and pumping that water to a water treatment facility, which discharged the treated water to Junction Creek at a location outside of the Frood Branch.	Monitoring design for the benthic macroinvertebrate community compared 1 year before the diversion to 8 years after the diversion and compared the sampled macroinvertebrates from that site to nine reference headwater stream sites. Benthic macroinvertebrate metrics and community at the restored site were significantly different from the reference sites. The diversion reduced metal concentrations, but those concentrations still exceeded OPWQO.
Adams and Ham et al. (2011); Adams et al. (2005), Ryon (2011), Smith et al. (2011)	A series of restoration efforts were implemented beginning in 1986 through 2004, which included a new wastewater treatment plant, construction of a lined retention basin, dechlorination of major outfalls, flow augmentation, and permanent flow bypass of lined basin.	Monitoring was done before, during, and after those restoration efforts at contaminated sites ($n = 3$ to 6) on the EFPC stream network and at one or two reference sites on other nearby streams outside of the EFPC watershed. Fish community monitoring from 1985 through 2004 showed sensitive species richness increased for EFPC sites furthest from the source of contaminants but were still lower than the reference sites. For benthic invertebrates, the uppermost two sites, closest to the restoration work, had a positive trend over 18 years in total and EPT richness, but both richness measurements were below that of reference sites. A BACI comparison for effects of specific pollution abatement periods on total richness, EPT richness, and EPT density showed improvement in most of those metrics at the two uppermost sites, but at a site further downstream, only three of the nine comparisons were significant.

AMD, acid mine drainage; BACI, before-after-control-impact; EFPC, East Fork Poplar Creek; EPT, Ephemeroptera-Plecoptera-Trichoptera; Fe, iron; OPWQO, Ontario Provincial Water Quality Objectives; SO_4^{2-} , sulfate

TABLE 3.

Details and results of studies in which the restoration was of urban streams

Citation	Details	Results
Arango, James, and Hatch (2015)	A 118-m reach that had been straightened and confined was excavated to create a new meandering channel with two backwater channels and two deep pools separated by three short riffles. Fixed LWM with root wads and large boulders were added and native riparian vegetation was planted. The new meandering channel was increased to 134 m. Monitoring design was an upstream-reference site compared to downstream, restored site, except for fish, which were compared pre- vs. post-restoration.	Restoration had no effect on ammonium and phosphorus concentrations, but uptake velocities of those two nutrients were greater in the restored reach versus the upstream reference reach, and the rate of algal biomass accrual was greater in the restored reach compared to the upstream reference reach. Fish community composition differed from pre-restoration at 1 and 6 months after restoration, but by 11 months after restoration there was no difference in community composition and nor were there before versus after restoration differences in mean fish density for the four most abundant species.
Bain et al. (2014)	Multiple restoration objectives included making physical and chemical changes to the stream, such as improving stream stability and WQ, diminishing peak discharges and meeting societal needs for improving aesthetic quality and enhancing recreational opportunities in the watershed. Components of the restoration project included rerouting the stream channel, removing impervious surfaces, removing chemical barrier to fish passage, installing hydraulic structures, removing sediment, establishing wetlands and repairing hydraulic structures.	Pre- and post-restoration nitrate concentrations, as a measure of WQ, showed limited changes, and modest improvements occurred in fish and benthic invertebrate measurements. Hydraulic structures installed in the stream channel are threatened by extreme storm discharges because the design flow used for restoration planning was too low. Those structures had to be repaired three times, with the clast size finally being increased during the third repair.
Blakely and Harding (2005)	Restoration spanned 5 years and included riparian plantings, in-stream modification by addition of boulders, wetland plantings, sediment trap construction and channel meandering.	No differences were detected in mean taxonomic richness per stream or mean number of caddisfly taxa among the three streams. Water chemistry was very similar among the three streams. Sediment copper concentration was significantly higher in the restored Okeover stream versus two control streams. While sediment lead concentration was not significantly different among the streams, it was above agency trigger values for the Okeover stream and one of the control streams.
Chovanec, Schiemer, Waidbacher, and Spolwind (2002)	Restoration produced artificial side channels, shallow water areas, coves, gravel banks, small permanent backwaters and temporary waters on a spoil island, which previously had riprap as dominant shoreline structure. Fish were sampled from nine sites in 1999, post-restoration, and compared to single sample collected from a cove in 1993, pre-restoration.	One of the restored sites clustered tightly with the reference site based on fish species presence/absence, and fish species richness at the sites increased with connectivity.
Bondar-Kunze, Preiner, Schiemer, Weigelhofer, and Hein (2009)	Evaluated the effects of increasing surface water connectivity to several backwaters through an artificial side channel of the Danube River.	The backwater section having the surface water connection to the artificial side channel exported more total phosphorus per day than the backwater section further downstream, with that downstream backwater section serving as a sink for dissolved phosphorus and chlorophyll <i>a</i> .
Hines and Hershey (2011)	Restoration included planting riparian vegetation, stabilizing banks and installing three types of structures, riffles, step pools and rock vanes. Monitoring compared restored versus unrestored streams in macroinvertebrates and nutrient uptake.	No significant difference in NCB1 between restored and unrestored streams. A nutrient-processing measurement, ammonium uptake length, was significantly greater in unrestored stream sites compared to the restored sites. The shorter length in restored sites was attributed to the in-stream rock structures providing more surface area for biofilm development, where ammonium uptake could occur.
Klockner, Kaushal, Groffman, Mayer, and Morgan (2009)	Spring Branch restoration had upstream stormwater ponds installed below the storm drain outfall, plunge pools constructed below storm pipe outfalls, catch basins used to attenuate flow and hydrologically "connected" floodplain access for bankfull discharges, with the restoration goal being to manage stream flow for erosion and flood control. Minebank Run restoration included reshaping of the stream banks to create hydrologically "connected" floodplains, re-establishing riparian vegetation and step-pool structures to improve stormwater management.	Nitrate retention metric were not significantly different among sites, and denitrification rates in surface sediments were not consistently different among restored versus degraded sites.

Citation	Details	Results
Harrison, Groffman, Mayer, and Kaushal (2012)	See Minebank Run restoration above.	Restoration increased the number of pools in the study, compared to before the restoration, the number of riffles and runs did not differ between those two periods. Reach nitrification, denitrification and methane production did not differ between pre- and post-restoration.
Newcomer et al. (2012)	See Spring Branch and Minebank Run restorations above.	The effects of three stream types, forested, restored urban and unrestored urban on denitrification potential rate were quite variable. Denitrification rates for restored, urban and unrestored, urban streams were not significantly different, but those rates were significantly greater than rates in the forested streams. Denitrification measurements taken with natural organic carbon sources were significantly greater for degraded, urban streams compared to forested and restored sites, while the rates for forested and restored sites were not significantly different.
Neale and Moffett (2016)	Two reaches on adjacent streams in the Avondale watershed each had 90 m of concrete pipe removed, along with soft-engineering of the stream channels and banks and a replacing of the uniform concrete substrate with one of boulders, cobble, wood and silt.	A pre- to post-treatment comparison showed a significant difference in the stream invertebrate communities based on a Bray-Curtis similarity matrix of macroinvertebrate abundance with the greater difference in the South reach attributed to forested headwaters having a source population for colonists. Of six univariate metrics, such as EPT richness, percentage EPT abundance, etc., tested at the two sites, only one showed a significant trend over the pre- and post-daylighting sample dates.
Rios-Touma et al. (2015)	Restoration actions entailed enhancing habitat complexity in the stream, stabilizing banks, reconnecting floodplains and planting more vegetation in riparian and upland areas. Had pre- and post-restoration monitoring along with reference sites.	Taxonomic richness and EPT richness increased significantly from before restoration to after, but both richness metrics were significantly less than the reference sites.
Selvakumar, O'Connor, and Struck (2010)	Bioengineered material, as well as rock armoring and planting of native vegetation, was used to restore ~500 m of degraded stream channel. Evaluation of the restoration included benthic macroinvertebrate samples collected above, within and below the restored area and before and after the restoration.	Biotic condition showed only slight improvements based on the VSCI and HBI, and sites were still scored as impaired based on VSCI. No significant differences were detected in water samples for chemical constituents and bacterial indicator taken pre- and post-restoration in addition to upstream and downstream of the restoration.
Sudduth and Meyer (2006)	Examined four reaches that had bioengineered, stabilized streambanks of 200 to 330 m in length and compared habitat and invertebrates from those banks to an unrestored reach and reference reach, with restoration having been completed 1, 3, 7, or 9 years before the study.	Site differences were not detected in total abundance, diversity and taxon richness of the bank macroinvertebrates, but macroinvertebrate biomass was significantly greater in the reference, 9- and 7-year-old restoration sites versus the 3-year-old restoration site, which had the smallest biomass
Suren and Memurtie (2005)	Restoration included either riparian planting only or a combination of riparian planting and channel modifications.	Among the five sites, very few of the macroinvertebrate community metrics, such as taxonomic richness, species evenness, species diversity, EPT richness and percentage EPT showed much response to the restoration. Invertebrate responses were quite variable, with no strong association to the stream enhancements.
Thompson and Parkinson (2011)	Monitoring design evaluated whether replantings, which were at least 15 years old, at three urban stream sites could restore invertebrate communities based on comparisons to upstream sites that were open, that is, without native riparian vegetation.	Invertebrate biomass was greater in open reaches than forested reaches, with significant differences also occurring between the streams and a reach type by stream interaction, and community composition biomass differed between open versus forested reaches and among the three streams.
Tuttle, Mcmillan, Gardner, and Jennings (2014)	Natural channel design restoration was used on six of the streams and included floodplain regrading, replanting riparian vegetation, installing grade control structures and stabilizing banks. The restored streams were compared to two other streams: one was a preurbanization stream and the other was an urban, unrestored stream.	Denitrification rates were significantly greater for restored sites compared to unrestored sites, and restored sites had significantly greater channel complexity than unrestored sites.
Wolter (2010)	Restoration included creation of an artificial, shallow bay and a littoral zone protected from waves by a trapezoidal wall with openings. Fish sampled at 12 sites.	Species diversity did not increase, but fish density did increase around the artificial structures in the Spree River.
Zelazna-Wieczorek and Nowicka-Krawczyk (2015)	Construction of seven artificial ponds served as settling basins for the gradual deposition of suspension and pollution in the river, and benthic samples of	Based on those indices, a longitudinal profile showed that 13 of the 14 sites had an ecological status of poor or bad, with only one site being moderate. Concluded that cascade construction of artificial ponds is not a good means for urban stream restoration.

Citation	Details	Results
	diatoms were collected at pond and stream sites, with the diatom data used to construct: SPSI, E/PI and TDI.	

E/PI, Eutrophication/Pollution Index; EPT, Ephemeroptera-Plecoptera-Trichoptera; HBI, Hilsenhoff's Biotic Index; LWM, large woody material; NCBI, North Carolina Biotic Index; SPSI, Specific Pollution Sensitivity Index; TDI, Trophic Diatom Index; VSCI, Virginia Stream Condition Index; WQ, water quality