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Consideration of spatial and temporal scales in stream restorations and biotic monitoring to assess restoration outcomes: A literature review, Part 2.

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Abstract

Stream and river restoration practices have become common in many parts of the world. To answer the question whether such restoration measures 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 assessed. In the first paper, we reviewed studies of culvert restorations, acid mine drainage or industrial pollution restoration; and urban stream restoration projects. Here, we review studies of restoration via dam removal, changes in dam operation or fish passage structures; instream habitat modification; riparian restoration or woody material addition; channel restoration and multiple restoration measures and develop some general conclusions from these reviews. Biomonitoring in different studies detected improvements for some restoration measures; other studies found minimal or no statistically significant increases in biotic assemblage richness, abundances or functions. In some cases, untreated stressors may have influenced the outcomes of the restoration, but in many cases, there were mismatches in the temporal or spatial scale of the restoration measure undertaken and associated monitoring. For example, either biomonitoring to measure restoration effects was conducted over a too short a time period after restoration for effects to be observed, or the sources and stressors needing remediation occurred at a larger catchment scale than the restoration. Also, many restoration measures lack observations from unimpaired reference sites for use in predicting how much of a beneficial effect might be expected.

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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.

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1 | INTRODUCTION

In this study, we conducted a literature survey of studies in which stream restoration measures 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. We wanted to assess whether the improvement in biotic assemblages or function associated with different types of restorations were related to how well the spatial scales of the impairment and restoration match and to the quality and timing of biotic monitoring after restoration.

In the first part of this study (Griffith & McManus, 2020)(Griffith & McManus, 2020), we reviewed studies of three restoration types: (1) culvert restoration; (2) acid mine drainage or industrial pollutant restoration and (3) urban stream restoration. The acid mine and industrial pollutants and urban stream studies were generally undertaken over small areas but had complex phased restoration measures that occurred over several years. Often, these involved treating water contamination. Culverts usually represent a limited barrier to animal movements, and the effect of their presence and removal is difficult to assess. In contrast, the stressors associated with urban streams are generally distributed throughout their catchments, and the relatively localized restoration measures are unlikely address the multiple stressors in these watershed. In this part, we review studies on five additional restoration types.

2 | METHODS

As described in more detail in Part 1 (Griffith & McManus, 2020), we conducted a search of the peer-reviewed literature published in English to identify studies of stream restoration measures conducted in catchments where the sources of stressors were urbanization; agriculture, including livestock grazing; forestry management; industrial or mine effluents; channel alterations; or dams. To narrow the results to those studies most likely to be relevant, the results of the search 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. 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 or periphyton) or their functions (i.e., production, respiration or 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 functions. Implicit in these criteria is that an intended objective of a reviewed restoration was the improvement of a biotic assemblage or its functions and not just the improvement of some physical or chemical characteristic of the stream.

Having grouped the studies into eight restoration types, we review five of those types in this paper: (1) dam removal, changes in dam operation, or fish passage structures; (2) instream habitat modification; (3) riparian restoration or woody material addition; (4) channel restoration and (5) multiple restoration measures.

3 | RESULTS

The following section presents the reviews of the studies by each restoration type for these five restoration types. Details on the individual restoration studies are compiled in Supplementary Tables 1 (i.e., stream size and location, restoration date, premonitoring dates, postmonitoring dates) and 2–6 (i.e., restoration measure details and biotic results). For these restoration types, which had more studies, we focus on the synthesis of the results of the individual papers. The tables in the text summarize the restoration and results for selected papers, but the supplementary tables provide details for all the papers reviewed for a restoration type.

3.1 | Dam removal, altered dam operation, or fish passage structures

Individually, dams are a source of alterations to streams and rivers, both upstream and downstream of the dam. Upstream, a dam generally creates a lentic habitat whose size varies with dam height and stream gradient that may accumulate fine sediment (Sethi, Selle, Doyle, Stanley, & Kitchel, 2004). The operation of dams can affect natural flow regimes downstream by reducing discharges or by altering the frequency and magnitude of elevated discharges (Doledec et al., 2015; Propst & Gido, 2004). Dams also alter the physical and chemical characteristics of downstream waters and stream channels. Hypolimnetic release dams typically decrease water temperatures during the summer and increase water temperatures during the winter relative to the normal stream temperature regime (Olden & Naiman, 2010), while surface release dams may warm summer water temperatures (Kornis et al., 2015). Hypolimnetic waters may also be low in dissolved oxygen (DO) (Bednarek & Hart, 2005). A lack of suspended sediment in discharged water from dams can result in erosion of sediments downstream resulting in channel downcutting and armoring (McManamay, Orth, & Dolloff, 2013), while flushing of plankton from the reservoir may change trophic resources for downstream fauna (Oswood, 1979). Dams are also physical barriers to the movement of organisms (Hatten et al., 2016; McManamay, Orth, Dolloff, & Mathews, 2013).

Individual dams are a relatively discrete stressor source to rivers and streams, both upstream and downstream of the dam site. Therefore, dam removal appears to generally match the spatial and temporal scales of the stressors associated with individual dams. Of the reviewed studies, 14, 4 and 3 showed overall positive, no and mixed improvements in biotic effects, respectively, of dam removal or changes in dam operation (see examples in Table 1 and full details of the reviewed studies in Supplementary Tables 1 and 2). One study used space for time substitution to show longer term recovery following dam removals (Hansen & Hayes, 2012).

Some stressors, like direct channel disturbance at the dam site by its removal or erosion of fine sediments from the former pool and readjustment of the channel, may persist or at least require some time for recolonization of the reach by the biotic assemblages (Orr, Kroiss, Rogers, & Stanley, 2008; Renofalt, Lejon, Jonsson, & Nilsson, 2013). While Hansen and Hayes (2012) estimated biotic recovery times greater than 15 years, other studies suggest only a few years (Pollard & Reed, 2004; Tuckerman & Zawiski, 2007), although the authors of the review believe this is likely an insufficient time to fully monitor ecological recovery. Insufficient data, such as the quantity of fine sediments being retained by the dam and the use of various measurement endpoints limits discerning why recovery time varies in many of these studies.

Dam removals eliminate barriers to movement of both anadromous and other native fish assemblages (Gardner, Coghlan, Zydlewski, & Saunders, 2013; Hogg, Coghlan, Zydlewski, & Gardner, 2015; Kiernan, Moyle, & Crain, 2012; Marks, Haden, O'Neill, & Pace, 2010; Muehlbauer et al., 2009; Propst & Gido, 2004). In two cases, dam removal not only removed a barrier but also either altered the downstream water temperature regime changing the fish assemblage from cool-water to cold-water (Kornis et al., 2015) or enlarged existing spawning habitat downstream for Coho salmon (Hatten et al., 2016).

Some of these studies found possible deleterious effects from dam removal, particularly with reference to elimination of the impoundment. Adverse effects include the potential for stranding and declines of less mobile animals (e.g., some unionid mussels) (Sethi et al., 2004). There is a question surrounding whether accumulated fine sediments should be removed prior to dam removal (Bushaw-Newton et al., 2002). Beatty et al. (2017) has argued against some dam removals suggesting these lentic habitats could become refuges from climate change, particularly in arid regions.

Many of these studies use a modified before-after-control-impact (BACI) design, which includes positive (i.e., samples from a site unaffected by the dam or its removal that represents target or reference conditions in the stream) and negative (i.e., samples from a site representing the conditions within the dam-affected reach before dam removal) controls. However, even with pseudoreplication- of individual biotic samples, the power of the analyses is low because there is often just one site, which can occur with other restoration types. Moreover, the results for various dam sites are likely idiosyncratic, being dependent on the width and depth of the stream, height of the dam, size of the pond or reservoir, location of the water release, stream gradient, amount of fine sediment accumulated in the reservoir, the presence of other dams, the effects of other stressors, the biotic assemblage and other factors (Brooks, Russell, Bevitt, & Dasey, 2011; Bushaw-Newton et al., 2002; Chiu, Yeh, Sun, & Kuo, 2013; McManamay, Orth, Dolloff, et al., 2013; Poulos et al., 2014; Thomson, Hart, Charles, Nightengale, & Winter, 2005).

While dam removal has become an increasingly viable ecological restoration option, particularly for dams that have outlived their historical function, many dams have important economic functions, such as producing electricity or storing water for human use (Propst & Gido, 2004). Keeping a dam in place and altering its operation may be preferable in some cases to ameliorate specific stressors, such as low DO (Bednarek & Hart, 2005) or alteration of natural flows, based on the studies we reviewed (Brooks et al., 2011; Doledec et al., 2015;

Kiernan et al., 2012; Lamouroux & Olivier, 2015; McManamay, Orth, Dolloff, et al., 2013; Mérigoux et al., 2015; Propst & Gido, 2004).

However, alterations of dam operation do not remove it as a barrier to animal movement. This effect might be ameliorated by constructing fish passage structures. A study of two tributaries of the Danube River (Zitek, Schmutz, & Jungwirth, 2008) found evidence of the beneficial use of some passage structures in a fish assemblage that included migratory riverine species. However, such structures have been used primarily for salmonids (Kiffney et al., 2009), have had limited success with Atlantic salmon and nonsalmonids like *Alosa sapidissima* (American shad) in Atlantic drainages (Brown et al., 2013) and have been used infrequently with dams on smaller inland streams (Bunt, Katopodis, & McKinley, 1999; Schmetterling, Pierce, & Liermann, 2002). Also, fish ladders generally assist only in upstream movement of fish, and injury and mortality can occur during downstream movement past dams, when fish may pass through the turbines (Brown et al., 2013; Eyler, Welsh, Smith, & Rockey, 2016). Therefore, there have been calls for and current projects to remove larger dams (rather than just create fish ladders), such as on the Elwha River, WA, USA (Brown et al., 2013; Elofson, 2008). However, no assessment of the biological effects of dam removals on the Elwha River have yet been published (East et al., 2015; Ritchie et al., 2018).

3.2 | Instream habitat modification

Modification of instream habitats are generally small-scale manipulations of habitat characteristics considered important for stream biota. Some of these manipulations are based on structures originally designed to supply critical habitat for valued species, particularly gamefish (Rosi-Marshall, Moerke, & Lamberti, 2006). Of the studies reviewed, 2, 6 and 9 showed positive, minimal or no significant and mixed improvements in measured biotic effects, respectively (see examples in Table 2 and Supplementary Tables 1 and 3 for details of the reviewed studies).

In several studies, the spatial extent of the restoration appeared to be insufficient to affect the biotic assemblage. The attempted restoration of a constructed stream channel in the Canadian Arctic only altered stream depths and substrates slightly, did not change instream resources and did not consistently alter the macroinvertebrate assemblage (Scrimgeour, Jones, & Tonn, 2013). Rock weirs in the upper Cache River did not consistently alter macroinvertebrate assemblages on woody snags or clay streambed (Walther & Whiles, 2008), but indicator taxa differed among the habitats, such as chironomids on the clay streambed and Ephemeroptera, Plecoptera and Trichoptera (EPT) on the rock weirs (Heinrich, Whiles, & Roy, 2014). The patches of added gravel downstream from two dams were relatively small and also potentially unstable (McManamay, Orth, & Dolloff, 2013; Merz & Chan, 2005). However, submerged and emergent macrophytes, a less commonly monitored assemblage, responded to such reach-scale restorations, as a study of 40 stream reaches ranging from 100 to 8000 m in length increased macrophyte cover and richness (Lorenz, Korte, Sundermann, Januschke, & Haase, 2012).

Manipulations of instream habitat can increase local abundance of fish or macroinvertebrates, if other nearby instream habitats can act as a source of new migrants

(Negishi & Richardson, 2003; Schwartz & Herricks, 2007). This is particularly clear for relatively mobile species like salmonids, for which such manipulations are commonly used, although the effects may be variable among species (Whiteway, Biron, Zimmermann, Venter, & Grant, 2010). Such enhancements may be attractive to these fish by supplying cover or other critical habitat (Rosi-Marshall et al., 2006), but may not affect other biotic assemblages.

Other fish and macroinvertebrates may slowly, if at all, colonize such habitats, particularly if the habitats are disconnected or are separated by enough distance to hinder dispersal (Heino et al., 2017; Parkyn & Smith, 2011; Raborn & Schramm, 2003; Tonkin, Stoll, Sundermann, & Haase, 2014). Consequently, habitat connectedness and species dispersal (i.e., metacommunity dynamics) need to be considered in setting expectations (Heino, 2013).

In some cases, important characteristics of the habitat may be damaged in the manipulations or may continue to adjust to the manipulations. In Finnish streams, use of heavy equipment to move rocks back into the channel removed aquatic mosses, an important macroinvertebrate habitat, that had not recovered after 3 years (Haapala, Muotka, & Laasonen, 2003; Louhi et al., 2011). When riprap was removed from a reach of the Danube River, fine sediments continued to erode from the restored bend and the river widened for some time following the restoration affecting the fish assemblage (Keckeis, 2014).

Other stressors, not affected by an instream habitat restoration, may limit the effects of a restoration. In the studies reviewed, such stressors include poor water quality (Pretty et al., 2003; Sarriquet, Bordenave, & Marmonier, 2007; Schwartz & Herricks, 2007) or riparian disturbance (Lepori, Palm, & Malmqvist, 2005). These results underscore the fallacy that just restoring stream habitats will result in recolonization by the biotic community (Bernhardt & Palmer, 2011; Bond & Lake, 2003). Often the cause is not from direct anthropogenic alterations to the stream habitats. Then, these instream habitat restorations may be attempting to treat smaller-scale geomorphological effects rather than the larger-scale causes (Filoso & Palmer, 2011; Harrison et al., 2004), such as alterations of stream flows or catchment land use (Gordon & Meentemeyer, 2006; Poff, Bledsoe, & Cuhaciyan, 2006).

3.3 | Riparian restoration and instream addition of woody material

Riparian zones are normally vegetated strips of land adjacent to streams. They are periodically inundated by flood flows, have shallower water tables than in more upland areas and usually support plant communities distinct from the adjacent uplands. The riparian zone acts as a source of materials, such as organic matter, moving from the land to the stream; a sink for materials, such as fine sediments, moving from the stream to the land; and as a filter for materials, such as nutrients, moving from more upland areas toward the stream. Streams are disturbed by land use changes in their catchments. Impacts may include removal or alteration of vegetation by forestry practices, row crop agriculture, conversion to pasture, or by commercial or residential development. Common restoration practices for riparian zones include replanting of native vegetation, fencing to exclude livestock and bank regrading.

An important material input from riparian zones to streams is woody organic matter. Studies of streams in temperate forested regions, particularly where some older growth forests remain (e.g., Pacific Northwest), have documented the role of woody material as structural and habitat elements of stream channels (Abbe & Montgomery, 1996; Benke & Wallace, 2010; Bilby & Ward, 1989; Wohl & Goode, 2008). This has led to the concept that woody material may be important to the restoration of streams, even though extant riparian forests along many streams are unable to produce large amounts of wood, particularly large pieces that are resistant to transport by storm discharges (Acuna, Diez, Flores, Meleason, & Elosegi, 2013). Moreover, the timeline for recovery of older growth forests can be very long, in the range of 100 years at least, and allowing such a recovery would be incompatible with the production of lumber and other wood products for human use and with other valued land uses. Therefore, rather than waiting for more long-term recovery of riparian zones or in the absence of restoration of the riparian zone, adding woody material or surrogates for woody material to streams may be an option. Of the reviewed studies, 14, 6 and 6 showed positive, minimal or no significant and mixed improvements in measured biotic effects, respectively (see Table 3 for examples and Supplementary Tables 1 and 4 for details of reviewed studies).

As a longitudinal feature of stream ecosystems, riparian zones can affect stream biotic structure and function at a relatively large scale. This is described by the river continuum concept and flood-pulse concept (Junk, Bayley, & Sparks, 1989; Vannote, Minshall, Cummins, Sedell, & Cushing, 1980), stream biotic structure and function and its relationship to the riparian zone varies longitudinally in river systems. Many riparian restoration measures have been undertaken at relatively limited scales, with restoration at the catchment scale being limited to relatively small, headwater catchments (Orzetti, Jones, & Murphy, 2010). More extensive riparian restoration measures may be limited by economic and other factors, and often, the dominant land uses, such as row crop agriculture or livestock grazing, can limit riparian zones to largely herbaceous vegetation. Moreover, riparian shading increases with the height and maturation of the riparian trees and requires a time scale of years (Quinn, Croker, Smith, & Bellingham, 2009; Ranganath, Hession, & Wynn, 2009; Teels, Rewa, & Myers, 2006).

Greater shading by riparian vegetation usually causes a shift from autochthonous production by algae or macrophytes to allochthonous production based organic detritus (Giling, Grace, Mac Nally, & Thompson, 2013). The decrease in food quality may reduce macroinvertebrate densities or biomass (Parkyn, Davies-Colley, Halliday, Costley, & Croker, 2003), but other measures may not follow this decrease (McTammany, Benfield, & Webster, 2007). Some management approaches, such as coppicing, may counter this change by leaving the canopy over the stream open (Clews & Ormerod, 2010; Clews, Vaughan, & Ormerod, 2010). However, less riparian shading can increase stream water temperatures, which may affect particularly cold water fauna, such as salmonids and many Ephemeroptera, Plecoptera or Trichoptera (EPT), (Broadmeadow, Jones, Langford, Shaw, & Nisbet, 2011; Sweeney & Newbold, 2014). Also, as observed by Minshall (1978) and others, some stream ecosystems, such as the alpine meadow stream studied by Herbst, Bogan, Roll, & Safford (2012), are naturally characterized by greater autochthonous production and fewer inputs of woody material. Other types of management can short-circuit interception functions of the riparian zone. Tile drainage bypasses the interception function by creating an alternate subsurface pathway for water flow (Smiley, King, & Fausey, 2011). Also, low organic accumulation on the soil surface may affect the interception of nutrients or fine sediments.

Other forms of riparian management, such as establishment of herbaceous buffer strips, fencing to exclude livestock and grazing management on pastures depend on maintenance of vegetative cover and can affect stressors, such as turbidity and fecal coliforms, but not necessarily nutrients and DO (Carline & Walsh, 2007; Smiley et al., 2011; Sovell, Vondracek, Frost, & Mumford, 2000; Weigelhofer, Fuchsberger, Teufl, Welti, & Hein, 2012). Fish and macroinvertebrate assemblages are less likely to change consistently with these types of restoration. However in high altitude (2400–2950 m), meadow streams of the Sierra Nevada, where herbaceous vegetation is more important in the riparian zone, removal of livestock grazing alone increased riparian vegetative cover and resulted in improvements in instream habitat quality and macroinvertebrates (Herbst et al., 2012).

Maintenance of native vegetation in riparian zones has been shown to have effects on streams in diverse ecosystems. These include removal of invasive *Tamarix* in the desert southwestern United States, removal of introduced *Acacia* and *Eucalyptus* from native shrub riparian habitats in South Africa and removal of invasive *Lonicera maackii* (Amur honeysuckle) along headwater streams in the midwestern United States (Keller, Laub, Birdsey, & Dean, 2014; McNeish, Moore, Benbow, & McEwan, 2015; Samways, Sharratt, & Simaika, 2011).

Recruitment of wood material capable of playing a significant role in stream geomorphology depends on it being of sufficient size and mass to resist being moved by floods or even by periods of elevated discharge less than bank full (Acuna et al., 2013). This generally requires older trees that produce woody material of greater mass and length and requires time far beyond that of the more recent restoration efforts we review here.

Large woody material (LWM) can affect very local habitats and the fish or macroinvertebrates using those habitats. LWM that creates cover for fish, stable substrates for macroinvertebrates, or pools can increase the local abundances of organisms, even in the absence of larger scale restorations (Bond & Lake, 2005; Coe, Kiffney, Pess, Kloehn, & McHenry, 2009; Howell et al., 2012; Hrodey & Sutton, 2008; Lester, Wright, & Jones-Lennon, 2007; Nicol, Lieschke, Lyon, & Koehn, 2004). In Australia, placement of woody structures in stream affected by sand slugs created cover for fish, but did not have the more extensive expected geomorphological effects (i.e., creation of pools), because no high-flows occurred as a result of an ongoing drought (Howson, Robson, Matthews, & Mitchell, 2012; Howson, Robson, & Mitchell, 2009, 2010). Also, functional changes may not occur at this localized scale (Entrekin, Tank, Rosi-Marshall, Hoellein, & Lamberti, 2008, 2009).

However, added LWM or surrogates that are not replaced by natural recruitment from the riparian zone will be unstable on a longer temporal scale because wood decomposes and the structures will break or be moved, particularly during periods of high flows (Acuna et al., 2013; Testa, Shields, & Cooper, 2011). Therefore, in the absence of reestablishment of a riparian zone with woody vegetation and natural recruitment of woody material, continued function would depend on active maintenance and replacement of the woody structures (Moore & Rutherfurd, 2017).

One problem with determining the effect of riparian restoration is identifying a target for the restored biotic condition. While most studies have a negative control that defines the biotic conditions in absence of the restoration, only some of the riparian restoration measures, and almost none of the wood addition studies, have a positive control that defines the biotic conditions in the absence of the cause of impairment. There are real difficulties finding undisturbed sites, particularly unlogged sites, in many catchments.

3.4 | Channel restoration

In many regions of the world, human activities have either directly or indirectly altered stream channels. These alterations have included conversion of meandering or multiple channels into single, often straight channels or even, rerouting the stream channels, dewatering side channels and destabilizing the streambed and banks. Various hydrogeomorphic approaches have been used to restore these alterations. Of the studies reviewed, 11, 11 and 4 showed positive, minor or no significant and mixed improvements in measured biotic effects, respectively. (see Table 4 for examples and see Supplementary Tables 1 and 5 for details of reviewed studies).

Alterations of stream channels often occur at relatively large scales, either because meandering, braided, or anastomosing channels were channelized and reduced to single, straightened channels (Colangelo, 2007; Jähnig, Brunzel, Gacek, Lorenz, & Hering, 2009a; Nakano & Nakamura, 2006, 2008; Obolewski & Glinska-Lewczuk, 2011; M. L. Pedersen, Friberg, Skriver, Baattrup-Pedersen, & Larsen, 2007) or in some cases, single channels were moved to an edge of their floodplain in attempts to reduce flooding or possibly increase arable land (Bukaveckas, 2007; Gregory, 2006). Other alterations of stream channels can occur because of geomorphologic changes, such as alteration to sediment supply due to changes in riparian or catchment land use or vegetation (Simon & Rinaldi, 2006). Similarly, channel restoration measures resulting in significant biotic improvements of those reviewed appear to be those undertaken at larger scales (Jordan & Arrington, 2014; Koebel, Bousquin, & Colee, 2014; Lüderitz, Speierl, Langheinrich, Voelkl, & Gersberg, 2011; Obolewski, Glinska-Lewczuk, Ozgo, & Astel, 2016; M. L. Pedersen et al., 2007), whereas those with the least effect were those where a small restored reach was embedded within unrestored reaches (Akasaka, Nakano, & Nakamura, 2009; Jähnig et al., 2009a; Jähnig & Lorenz, 2008; Nakano, Nagayama, Kawaguchi, & Nakamura, 2008; Schiff, Benoit, & Macbroom, 2011). However, restoration of shorter reaches may be sufficient to facilitate some biotic effects, like reduced NH_4 uptake length (Gabriele, Welti, & Hein, 2013) or increase macrophyte species richness and cover (T. C. M. Pedersen, Baattrup-Pedersen, & Madsen, 2006). Also, other unremediated stressors can moderate the effects of these restoration measures (Klein, Clayton, Alldredge, & Goodwin, 2007; Muotka & Syrjanen, 2007; Northington et al., 2011; Pierce, Podner, & Jones, 2015).

Although longer multichannel reaches have not been restored, restorations of side or secondary channels, have exhibited positive effects. Some native fish increased in isolated side channels of the Provo River (Utah, USA), while restorations have improved macroinvertebrates and fish in side or secondary channels of the Rhine (Netherlands), Rhone (France), Danube (Germany) and Missouri (Missouri, USA) Rivers (Belk, Billman, Ellsworth, & Mcmillan, 2016; Besacier-Monbertrand, Paillex, & Castella, 2014; Billman et al., 2013; Castella et al., 2015; De Vaate et al., 2007; Pander, Mueller, & Geist, 2015; Starks, Long, & Dzialowski, 2016).

Nutrient retention may be increased at smaller scales, but the extent of nonpoint nutrient inputs is generally sufficient that an extensive area of restored channel may be needed to affect overall nutrient concentrations. For example, the studies of two-stage ditches exhibited potential for nitrogen removal, particularly during inundation of the created floodplain, but concluded that this type of restoration needs to be applied to longer stream reaches to substantially reduce nitrogen export (Davis, Tank, Mahl, Winikoff, & Roley, 2015; Griffiths, Tank, Roley, & Stephen, 2012; Mahl, Tank, Roley, & Davis, 2015; Roley, Tank, Griffiths, Hall, & Davis, 2014; Roley, Tank, Stephen, et al., 2012; Roley, Tank, & Williams, 2012).

The scale of projects using Natural Channel Design (Rosgen, 1996) may be insufficient to affect biota when a short stream reach is restored. Baldigo, Ernst, Warren, & Miller (2010) observed only slight improvements in fish assemblages of reaches ranging from 0.34 to 1.1 km, less improvement in the shorter reaches (0.34−0.5 km) and no significant effects on macroinvertebrates (Ernst, Warren, & Baldigo, 2012). Natural Channel Design does not change other stressors, such as increased stream temperature, decreased riparian cover (Klein et al., 2007), elevated specific conductance and total dissolved solids (Northington et al., 2011), or the presence of introduced parasites (Pierce et al., 2015).

Observations of reference conditions can supply a benchmark for how much the biotic assemblage might be expected to change in response to a restoration. Although unrestored or control conditions may be easily sampled, reference conditions may be more difficult to observe. To observe potential reference conditions, the Kissimmee River restoration in part included sampling of sand-bottom rivers in coastal Georgia (Koebel et al., 2014), while some of the restorations of streams altered for log drives in Scandinavia used smaller headwater stream reaches that were never used for log drives (Muotka & Laasonen, 2002). If a restoration makes only small changes to physical conditions in a stream, biotic assemblages are unlikely to respond (Shields, Knight, & Cooper, 2000).

3.5 | Multiple restoration measures

Projects that use several restoration techniques generally attempt to address more than one stressor in the streams or more than one source (Bergfur, Demars, Stutter, Langan, & Friberg, 2012; Yu, Huang, Wang, Brierley, & Zhang, 2012). Of the studies reviewed, five and three showed positive or minor improvements in measured biotic effects, respectively (see Table 5 for examples and see Supplementary Tables 1 and 6 for details of the reviewed studies).

In relatively short reaches (mean = 1.1 to 1.5 km), multiple restoration measures that included re-establishing meandering or multiple channels, adding large woody material and removing weirs when present often had only minor increases in fish or macrophyte richness, but generally had no effects on macroinvertebrates (Haase, Hering, Jaehnig, Lorenz, &

Sundermann, 2013; Lorenz, Stoll, Sundermann, & Haase, 2013). Two direct tests of the effect of increased linear extent of restoration (i.e., 0.05–26 km) and time since the restoration (0.5 – 6 years), Schmutz et al. (2014; 2016) found positive effects on fish assemblage richness and density metrics.

The reviewed studies suggest that combining restoration techniques may or may not further improve the outcomes for biotic assemblages or function (Bergfur et al., 2012; Yu et al., 2012). As for other restoration measures, increasing the longitudinal extent of the restoration or the time of monitoring since a restoration did produce greater effects (Bergfur et al., 2012; Schmutz et al., 2016; Schmutz et al., 2014). Individual alterations, such as increased stream width and addition of instream habitat, appear to have the greatest effects on specific subgroups, particularly centrarchids or primary producers (Kupilas et al., 2016; Shields, Knight, & Cooper, 2007).

4 | DISCUSSION

Two of the restoration types reviewed here: channel restoration and dam removal had 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. Some of the channel restoration and dam studies were done over large lengths of stream or catchment areas. The channel and dam restoration studies did not entail treating water contamination but addressed mainly hydrological alterations to the streams and rivers.

Although dams can have various effects on streams or rivers depending on their construction and how they route water flows, they tend to be localized sources of these alterations, and therefore, dam removal, if the individual dam is not one of a series of dams, can be a localized solution to these effects, which is reflected in the positive results of the dam removal studies. As more limited, usually stressor specific, restoration measures, changes in dam operation or installation of fish passage structures have more variable effects on the biotic assemblages.

Many of the channel restoration studies dealt with undoing large-scale (i.e., longitudinal) direct human alterations of streams, such as channelization, while some dealt with decreased channel stability using natural channel design approaches. When the restoration is reversing direct human alterations, such as recreating multiple or meandering channels, an increasing longitudinal extent of the restoration appears to contribute to restoration effectiveness. However, when channel stability degrades, the restoration may need to more directly address the hydromorphological -causes of such degradation, such as changes in sediment loads or water flows, rather than the smaller scale changes in channel morphology (Wohl, Lane, & Wilcox, 2015). This would reinstate mechanisms that would sustain the restoration. The studies of instream habitat modifications illustrate this and suggest that such relatively small longitudinal-scale restoration measures may not significantly improve biotic assemblages or their function unless the restoration supplies more specific habitat requirements, such as cover for fish or stable substrates for benthic macroinvertebrates. Because riparian zones are closely aligned with stream channels, their longitudinal integrity affects biotic assemblages and functions by affecting inputs of fine sediment, water temperature, availability of light, the quality of allochthonous organic inputs and even factors like nutrients that are more related to integrity at the even larger whole-catchment scale (Sponseller & Benfield, 2001; Sponseller, Benfield, & Valett, 2001). In forested biomes, woody material can be an important component of streams (Benke & Wallace, 2010; Dolloff & Warren, 2003; Wohl & Goode, 2008), although in many regions, this relationship has been extensively altered by forest harvesting and conversions to non-forest land uses (Krankina & Harmon, 1994). Forest regrowth requires a long temporal scale, while wood addition is a comparatively short-term solution. In the interim, the developing forest can increase shading and provide other riparian functions.

In many of the stream or river restoration studies examined, biotic assemblages did not satisfactorily improve when the restoration did not address other stressors, caused additional damage to stream habitats, or did not allow time for biotic recovery before monitoring. Identification of the important stressors affecting the biotic assemblage or function at the stream site is important. Causal assessments may be used to more clearly identify the likely direct and indirect causes of degradation at stream sites and determine the scale of the sources (S. B. Norton, Cormier, & Suter, 2015; Suter, Norton, & Cormier, 2010).

Although some studies suggest a relatively long recovery times of 12.5 to 30 years after a restoration (Favaro, Moore, Reynolds, & Beakes, 2014; Hansen & Hayes, 2012; Schmutz et al., 2016), others suggest much shorter periods (Lüderitz, Jupner, Muller, & Feld, 2004; M. L. Pedersen et al., 2007; Tuckerman & Zawiski, 2007). The period of recovery may vary among assemblages if their metacommunity dynamics differ (Parkyn & Smith, 2011; Swan & Brown, 2017) or if the connectivity of sites to other reaches differs (Lüderitz et al., 2004; McManamay, Orth, Dolloff, et al., 2013). The level of biotic organization (i.e., population, community, ecosystem) measured may also affect the period of recovery. Studies that monitor biotic effects almost immediately following the restoration are probably allowing an insufficient time for recovery (Keckeis, 2014). Moreover, such short-term changes may not be necessarily indicative of the longer-term outcomes of the restoration measures.

Other considerations when planning the biomonitoring of a restoration would include identifying reference sites from which one can define the extent to which the biotic assemblage or function at stream site may recover. Several studies reviewed here acknowledge that the restored site was not very impaired (Rosi-Marshall et al., 2006; Schiff et al., 2011; Shields et al., 2000), while other studies lacked any information about likely maximum possible improvements. The difficulty of identifying reference sites varies with the stressors and restoration measure. For example, finding a reference site may be difficult for changes in land use that affect riparian zones, but easier for small dam removals, where it has been often possible to sample an upstream reach that was not directly influenced by the dam, although in these cases, the dam still affects the connectivity of the upstream reach with contiguous stream reaches (Kornis et al., 2015).

Planning for stream restorations should include a comparison of the scale of the planned restoration to the scale of the stressors and their causes and sources at the site of the stream restoration. In the reviewed studies, we found three and four generalized differences in the spatial and temporal scales, respectively, affecting stream restorations and monitoring to assess biotic outcomes (Table 6). The geospatial context of stream restoration can readily be evaluated beyond the site-scale given the availability of spatially explicit variables in datasets such as StreamCat (Hill, Weber, Leibowitz, Olsen, & Thornbrugh, 2016). At least for the coterminous USA, variables from a variety of national datasets, National Land Cover Data, STATSGO, US census and so forth, are calculated for each of 2.65 million catchments and the areal extent surrounding each reach in National Hydrography Dataset (NHD) Plus Version 2 (Hill et al., 2016). Additionally, the variables are also expressed cumulatively for the catchments upstream of each reach, and some variables are also calculated within a 100-m buffer of National Hydrography Dataset streams. A comparison of the catchment to its corresponding watershed metrics may identify whether the proposed restoration in the catchment is likely to be effective given the conditions upstream. In current applications, two indices, an Index of Watershed Integrity and an Index of Catchment Integrity, are used to quantify and compare integrity at whole-watershed scale with that of the more local drainage of individual stream segments to assess the scales of stressors in the individual segments (Johnson, Leibowitz, & Hill, 2019; Kuhn et al., 2018; Thornbrugh et al., 2018).

Another available tool, Recovery Potential Screening, can assist with strategic planning and priority-setting in restorations (D. J. Norton et al., 2009). This tool can access data at the watershed level for most states and territories of the United States to calculate ecological, stressor and social indicators and compare watersheds in terms of larger-scale characteristics that are relevant to the potential for improvements from restorations (USEPA, 2018).

5 | CONCLUSIONS

Comparing studies within and among different types of stream restorations, we found that there is evidence that better matching of the scale of the stressors and of the restoration is likely to improve the outcomes of stream restorations in terms of the biotic assemblages or functions. While different reasons for the lack improvements in biotic assemblages or functions can be identified for specific examples, such as the existence of conditions or other stressors not affected by the restoration or the lack of unimpacted reference sites that would provide a metric for the potential improvements, an important overarching consideration is that of temporal and spatial scale. Consideration of temporal scale includes consideration of the time needed for the recovery of the biotic assemblages in the restored stream and of their biotic functions following restoration. This also includes recognizing that maturation of ecosystems, such as restored riparian zones, or the readjustment of natural stream geomorphology takes time. Consideration of spatial scale particularly includes consideration of the scale of the stressors (i.e., reach-level versus watershed-level alterations) and whether the restoration is at a similar scale or whether there are barriers to the recolonization of the restored stream by biotic assemblages.

Supplementary Material

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TABLE 1.

Details and results of selected examples of studies where the restoration was dam removal, changes in dam operation, or fish passage structures. The examples include a restoration with mixed effects, a space-for-time substitution study showing biotic improvements with time, a restoration with positive effects and a restoration with no or minimal effects.

Citation	Details	Results
Bushaw-Newton et al. (2002), Thomson et al. (2005), Velinsky, Bushaw-Newton, Kreeger, and Johnson (2006)	This study removed an approximately 200-year-old mill dam, but the impoundment had dredged <30 years before the removal. The dam and its 500-m long impoundment were within a town and <500 m of the confluence with a larger river.	No significant upstream-downstream or before-after differences were observed in metrics describing algal or macroinvertebrate assemblages, such as the number of diatom species, chlorophyll *a*, and total macroinvertebrate density. While % abundance of the dominant diatom species (*Cocconeis placentula* or *Nitzschia inconspicua*) and number of EPT taxa decreased and % abundance of Chironomidae increased downstream; these differences did not change between before and after dam removal. Riffle-dwelling fish abundance was initially depressed downstream from the dam, but by the summer of the next year, these abundances increased. Similarly, riffles were quickly established in the channel where the impoundment had been; riffle-dwelling fish abundance was similar to that of the upstream and downstream sites by the summer of the next year.
Hansen and Hayes (2012)	Compared the macroinvertebrate assemblage in riffles and runs upstream and downstream from the sites of eight dams in Michigan and Wisconsin removed 0, 3, 6, 14, 14, 19, 33 and 40 years, respectively, before the study.	Bray-Curtis taxonomic similarity of macroinvertebrates in sites both in riffles and runs within the reach downstream from the former dam site and within the former impoundment increased in relation to the years since dam removal, nearing an asymptote after 15 to 30 years. Percent difference in richness also approached 0 to 5%.
Hatten et al. (2016)	Compared spawning of Tule fall chinook salmon in the lower 1.7 km of the White Salmon River—downstream of the dam—following the explosive breach of Condit Dam. A drain tunnel dug at the base of the dam was designed to allow a maximum flow of 293 m^3/s and drain the reservoir in about 6 h, but the reservoir drained in less than 2 h, flows exceeding 400 m^3/s and more sediment flushed than expected.	While some spawning of Tule Fall Chinook salmon has occurred upstream of the dam site (<10% of Tule Fall Run), as important were the effects on spawning habitat downstream of the dam, 0.96–3.37 rkm from the Columbia River confluence. Much of the silt, sand and clay released by dam breach were deposited downstream of the spawning habitat (rkm 0–0.6), and coarser cobble and gravels were deposited from rkm 0.61–1.7, raising bed elevation, decreasing water depths and increasing water velocities in spawning habitat that has been enlarged by added habitat from rkm 0.61–0.96. Parr escapements have increased. Dams can starve the downstream reach of mobile sediments, causing channel downcutting or bed armoring (Poff & Hart, 2002). The area of added spawning habitat had been a large pool.
Orr, Rogers, and Stanley (2006), Orr et al. (2008)	Some sediment had accumulated behind the upper dam, but the channel was not visibly altered, and its pool was only 85 m^2. The lower dam had a 190 m^2 pool. The study focused on the effects of dam removal on a 90-m reach immediately downstream from the lower dam. This reach was compared with a 90-m reference reach established upstream of the influence of the upper dam.	While the % fine sediment increased in the downstream reach from 15–30% to 50–75% immediately after dam removal, it decreased to less than that in the reference reach (20–25% vs. 38–48%) by the second summer. SRP did not differ between the reference and downstream sites before or after dam removal, and TSS increased only briefly following dam removal. While uptake lengths for SRP increased immediately following the dam removal, they had decreased to lengths similar to that before dam removal by the next summer. Periphyton chlorophyll *a* increased following dam removal but remained lower than in the reference reach during both the first and second summers. Similarly, macroinvertebrate and ET abundances and assemblage similarities decreased in the downstream reach relative to the reference reach after the dam removal; these differences remained the following summer.

SFP, soluble reactive phosphorus; EPT, Ephemeroptera-Plecoptera-Trichoptera; rkm, river kilometers; ET, Ephemeroptera and Trichoptera, TSS, total suspended solids

TABLE 2.

Details and results of selected examples of studies where the restoration was in-stream habitat modification. The examples include a restoration with mixed effects, a restoration with positive effects and a restoration with no or minimal effects.

Citation	Details	Results
Haapala et al. (2003)	The restored stream was channelized to facilitate log transport, which ceased in the 1950s. Restoration was mainly for fishery purposes and involved installation of boulders, boulder dams, flow deflectors and other in-stream structures. No woody material had been added. Pebbles, cobbles and coarse gravel were added to create juvenile nursery habitats and spawning areas. The boulders were expected to increase OM retention. Monitoring included a leaf release experiment and sampling for benthic invertebrates in the restored stream, a channelized stream and a unchannelized, reference stream.	The restored stream increased in leaf retention from 25% to 75%, which was similar to retention in the reference stream. The most retentive feature in the restored stream were cobbles and boulders. In the retentive sites, more CPOM accumulated. In the restored stream, macroinvertebrate and detritivore densities were greater in the retentive sites and were greater than the reference stream. Shredder and predator densities were greater in the retentive sites in the restored stream but were not as great as in the reference stream or even the channelized stream, while scraper densities were generally much lower in the restored stream compared to the reference stream. Moss coverage in the restored stream was much lower than in the reference stream.
Rosi-Marshall et al. (2006)	As a pilot project to improve trout habitat in streams of Superior National Forest, three 100-m reaches were selected: one which was kept as an unrestored control, one where 2 k-dams, a wood and stone structure that deepens pools, were installed and one where skyboom structures were installed to mimic undercut banks. These enhancements are well known to provide areas of critical habitat for adult trout.	Water depths and the QHEI increased in both restored reaches. However, the reaches all scored as high quality both before and after the restoration. Wood and leaf retention increased in the k-dam reach but only wood retention increased in the skyboom reach. OM abundance was variable among years and reaches and did not differ among reaches. Periphyton chlorophyll *a* and AFDM also did not differ among reaches, and neither did macroinvertebrate density, diversity, or functional group composition. Macroinvertebrate density varied with substrate coarseness, which was not affected by the restorations. Only larger size classes of trout increased in the k-dam reach in the first year before the skyboom was installed, but after the skyboom was installed, larger trout increased only in the skyboom reach.
Scrimgeour et al. (2013)	During development of a diamond mine, a dam was created to drain two lakes, and a stream was constructed to divert the water to a downstream lake. An initial assessment characterized the constructed stream as a poor surrogate for natural streams identified as reference. The channel and nonvegetated riparian zone were underlain by blast rock, and the in-stream habitat was deep, having slow-flowing runs with 44% fine sediments. Substrates in natural, reference streams were only 14% fine sediments and were primarily cobble and boulder riffles. An effort to naturalize the stream involved installing engineered rock structures, including vanes, weirs, ramps and a single groin. However, only nine structures were constructed, each covering about 5 m of stream bed in a 3.4-km channel. This study sampled macroinvertebrates on engineered structures and unenhanced reaches of the stream channel.	In no case did any measured physical variables (i.e., water depth, water velocity, silt and sand, or gravel + cobble) differ between the unaltered controls and the sites with structures, but there were trends in decreased depth and proportion of silt and sand and decreased proportion of gravel + cobble. Moreover, there were no differences in epilithon, CPOM, woody material, bryophytes, macrophytes. the densities of total macroinvertebrates, Chironomidae, Simuliidae, Nematoda, Oligochaeta or Ephemeroptera, or the biomass of nonchironomid Diptera, NOT or EPH between the unaltered controls and sites without structures. There were differences in the biomass of total macroinvertebrates or Chironomidae, but biomass decreased at the sites with structures compared to the unaltered controls. Comparisons among the individual structures exhibited a lot of variability, but the comparisons that suggest that mean densities and biomass were similar between the reference streams and the individual structures are meaningless, unless density and biomass differed between the individual structures and the unaltered control reaches. The analyses that grouped all the sites with structures do not show such differences.

AFDM, ash-free dry mass; CPOM, coarse particulate organic matter; EPH, Ephemeroptera + Plecoptera + Hemiptera; NOT, Nematoda + Oligochaeta + Turbellaria; OM, organic matter; QHEI, qualitative habitat valuation index

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TABLE 3.

Details and results of selected examples of studies where the restoration was riparian restoration or in-stream addition of woody material. The examples include a restoration with positive effects, a restoration with mixed effects and a restoration with no or minimal effects.

Citation	Details	Results
Coe et al. (2009)	Both streams have had reduced recruitment of wood, the N. Fork Stillaguamish River from timber harvest and road building in the headwaters, and the Elwha by two upstream hydroelectric dams. In the Elwha, 16 ELJ with a combined surface area of about 12,000 m^2 were constructed in a 0.3 km reach. In the N. Fork Stillaguamish, five ELJ with surface areas >7000 m^2 were placed in a 0.7 km reach. Comparisons were made to cobbles on upstream reaches that generally lacked large woody material.	Periphyton biomass and autotrophic index were greater on wood than on cobbles, both in the treatment reach or the control reach. Similarly, total macroinvertebrate densities were generally greater on the wood than cobbles.
Howell et al. (2012)	This project added SWH to three riffles (three other riffles were used as controls) all within a 1.5-km reach of an approximately 5–6 km reach, the UHRRI reach. Three pools within the UHRRI reach also had SWH added, and additional control pools and riffles were established both upstream and downstream of the UHRRI. The riffles received multilayered deflector jams embedded into the banks, while the pools received prefabricated pool jams. The volume of wood ranged from 96–180 m^3 in each deflector jam and 0.009–0.018 m^3/m^2 was added to each riffle. The wood volume of the pool jams ranged from 2.3–5.4 m^3, and 0.0009– 0.0026 m^3/m^2 was added to each pool, generally doubling the pool wood.	In riffles, fish assemblage structure varied between the treatments and controls with greater abundances of an introduced species, *Gambusia holbrooki* (eastern mosquitofish) and the native *Retropinna semoni* (Australian smelt) in the treatment riffles. Species richness was not significantly different, and mean abundance was often greater at the treatment riffles after the SWH introduction, but mean biomass in the control riffles was either greater or not different from treatment riffles. In pools, fish assemblage structure did not vary between the treatments and controls, and fish metrics did not vary between the treatments and controls.
Keller et al. (2014)	In streams of the desert Southwest, colonization of stream banks and channel bars by the non-native shrub, *Tamarix*, stabilizes the stream channel, which in these streams is usually mobile, and causes steepening of banks and narrowing of the stream channel. The study was conducted in 90 km of the lower San Rafael River, Utah, below a diversion dam at Hatt's Ranch. From 2008–2009, *Tamarix* was removed from 424.5 ha of the riparian zone of the upper 24 km of the section. Some of the piles of *Tamarix* had been burned and others not before several overbank floods ending in a very large overbank spring flood in 2011. The fish data from the *Tamarix* removal sites was compared to sites above Hatt's Ranch.	Even though there were some increases in channel width and heterogeneity in the reaches where *Tamarix* was removed, there was no increase in the native fish species. Speckled dace (*Rhinichthys osculus*), flannelmouth sucker (*Catostomus latipinnis*), bluehead sucker (*Catostomus discobolus*) and round-tail chub (*Gila robusta*) did not increase, and non-native fish species, such as common carp, remained dominant.

ELJ. Engineered log jams; SWH, structural woody habitat; UHRRI, Upper Hunter River Rehabilitation Initiative

TABLE 4.

Details and results of studies where the restoration was channel restoration. The examples include a restoration with positive effects, a restoration with no or minimal effects and mixed effects.

Citation	Details	Results
Colangelo (2007), Jordan and Arrington (2014), Koebel et al. (2014)	Historically, a highly meandering 15–30 m wide, 2–3 m deep channel had been overlain by a wider (100-m), deeper (9-m), straightened flood control canal, leaving only stagnant, remnant river channel sections to either side of the canal. The restoration backfilled the canal and restored flow to the original channel.	Mean DO increased from near 2 mg/L before the restoration to about 3.5 (wet season) and 6.0 mg/L (dry season) after the restoration and was more similar to that of the reference reaches. River channel GPP, CR and P/R increased after the restoration; NDM was more negative, indicating that CR still exceeded GPP. Piscivorous fishes ingested more types of prey in the restored reach than those in the unrestored reaches. There were fewer fish and larval dragonfly species in the unrestored reaches. Fish prey comprised a smaller proportion of the diet of largemouth bass, *Pomoxis nigromaculatus* (black crappie) and *Amia calva* (bowfin) in the unrestored reaches. Individual bowfin consumed more prey than other predator fish and more prey in the restored pool than in the unrestored pools. This suggests food web structure was improved in the restored reach, particularly at the predator level. Restoration also increased midchannel current velocities from <0.01 to 0.38 m/s and decreased OM accumulations that resulted in exposure of the natural sand substrates. In benthic habitats, taxa richness increased from 20 to 26 in the restored reach versus an increase of 16 to 18 in the control reach, but relative abundances changed as the assemblage shifted from OM-dwelling to sand-dwelling species, as 11 of 19 taxa were sand-dwelling. On snags, taxa richness decreased slightly in both restored and control reaches, but shifts in relative abundances included a 500-fold increase in collector-filterers, such as *Rheotanytarus, Cheumatopsyche and Cyrnellus* in the restored reach. However, large predators, such as Odonata, were absent from snags in both reaches.
Jähnig et al. (2009a); Jähnig and Lorenz (2008)	About 200-m reaches of seven streams that historically had multiple channels were restored from single channels to multiple channels. In the active restorations, levees and other structures that confined the flow to a single channel were removed. In the passive restorations, maintenance of these structures was stopped, and the flow reoccupied the multiple channels. The rest of the stream length was maintained as largely uniform, single channels. The restored reaches each were paired with a control single channel reach nearby on the same stream.	The authors concluded that there were at most only minor differences in the macroinvertebrate assemblages on similar substrata in the unrestored and restored channel reaches. Although there was greater habitat diversity in the restored channel reaches, α-diversity had not increased.
Muotka and Laasonen (2002), Muotka and Syrjanen (2007)	Streams dredged in the 1950s to facilitate log transport were reconstructed to restore their normal heterogeneous bed. Undredged streams selected as reference streams had undisturbed heterogeneous beds. The restoration measures included boulder dams constructed of boulders set side-by-side with the inside partly filled with cobbles and pebbles, deflectors set across half the channel width and areas of sorted gravel in relatively swift currents for salmonid spawning habitat. Aquatic mosses are common and relatively important for OM retention in these streams, but the heavy equipment used in the restoration damaged much of the mosses. The riparian zones of most streams had been harvested for timber. The forest was drained during the 1950s to 1970s and had regrown for at least 30 years, and the streams had few woody material dams and little woody material.	Restored streams had greater substrate complexity than channelized streams and was similar to that of natural streams. Moss cover was lower in the restored streams compared with the channelized or natural streams. For artificial leaves, retention was greatest in the natural streams, intermediate in the restored streams and least in the channelized streams. Grazing macroinvertebrate densities increased in the restored streams compared with the channelized streams, while other detritivores changed little.

CR, community respiration; DO, dissolved oxygen; GPP, gross primary production; NDM, net daily metabolism; OM, organic matter; P/R, Production/Respiration ratio

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TABLE 5.

Details and results of studies in which multiple restoration measures were undertaken. The examples include a restoration with no or minimal effects, a restoration with positive effects and a study the shows a positive relationship with the linear extent and time since a restoration.

Citation	Details	Results
Haase et al. (2013)	This is a space-for-time substitution design without before sampling. The lengths of the restored reaches were 0.1–8 km (mean = 1.5 km), and the sites were sampled 1 to 12 years (mean = 4 years) since restoration. At each site, the restored reach and an upstream unrestored reach was sampled. Although variable between sites, the restorations increased physical heterogeneity, longitudinal connectivity, or river length; added woody material or flow deflectors; decreased entrenchment depth or bank fixation; added new water courses or multiple channels; or reconnected back waters. Fish, macroinvertebrates and macrophytes were sampled. Comparing the restored sites with their paired unrestored sites, the authors observed general improvements in hydromorphology.	There were small, statistically significant improvements in mean taxa richness for fish (restored/unrestored = 9.6/9.0) and macrophytes (5.0/3.3) and in mean number of endangered species per reach for fish (4.5/4.0). However, Simpson diversity, Bray-Curtis similarity of feeding types, Bray-Curtis or Jaccard similarity, or the number of indicator species did not change for either fish or macrophytes. Moreover, none of these metrics improved for macroinvertebrates, and an overall measure of assemblage status generally did not improve for macrophytes or macroinvertebrates. While fish improved at 11 of 24 sites, only 1 site achieved a characterization of good.
Lorenz et al. (2013)	Data from 46 restored reaches with paired unrestored reaches were compared. Restorations involved installing LWM and re-establishing meandering or multiple-channel patterns. When present, weirs were removed. The restored reach averaged 1.1 ± 1.4 km. Adult fish and YOY were sampled.	The number of fish per 100 m or ha was greater, and the number of adult fish or YOY fish per 100 m was greater. Total fish species richness increased from 9.5 to 10.8, while YOY fish species richness increased from 6.1 to 7.7.
Schmutz et al. (2014)	Regression and correlation are used to test whether fish metrics were affected by time and linear extent of rehabilitation classified as either in-stream habitat assessment, backwater habitat enhancement, or creation of dynamic bank development, particularly by removal of riprap. The length of restoration ranged from 50 m–9.7 km, while the time since restoration varied from 0.5 to 5 years.	The study found a positive relationship between time since restoration and linear length of the restoration and various fish metrics.

LWM, large woody material, YOY, young-of-year

TABLE 6.

Generalized differences in the spatial and temporal scales affecting stream restorations and monitoring to assess biotic outcomes and the restoration-type where examples were observed.

Scale	Type	Restoration-type
Spatial scale	Stream alterations at larger linear scales (i.e., more than reach) dealt with at only smaller (i.e., often reach) scales.	Instream-habitat restoration Riparian restoration Channel restoration
	Stressor sources occurring at larger (i.e., often watershed) scales dealt with at small (i.e., often point or reach) scales.	Acid-mine drainage restoration Urban restoration Instream-habitat restoration Channel restoration
	Instream metacommunity or metapopulation processes (i.e., particularly dispersal) leading to recovery of community structure and function affected by distance or connectivity.	Changes in dam operation Instream-habitat restoration Channel restoration
Temporal scale	Instream metacommunity or metapopulation processes (i.e., dispersal, recruitment) affecting the time required for recovery of community structure and function.	Dam removal Instream habitat restoration
	Reestablishment of stable geomorphological conditions following restoration of a stressor source that had altered geomorphological conditions.	Dam removal Channel restoration
	Reestablishment and growth of terrestrial communities (i.e., usually dominated by plants in riparian zones) to the extent that they fully influence the movement of water, materials (i.e., organic matter, nutrients and inorganic sediments) and energy into streams.	Riparian restoration or woody material addition
	Long-term persistence of restorations in the face of organic decay and instream geomorphic processes, such as storm flows.	Instream-habitat restoration Woody material addition Channel restoration