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Title: Impact of in-stream restoration structures on salmonid abundance and biomass: an updated meta-analysis.

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Abstract

Due to declines in salmonid populations, in-stream restoration structures have been used for over 80 years to increase abundance of fish. However, the relative effectiveness of these structures remains unclear for some species or regions, partly due to contrasting conclusions from two previous meta-analyses. To update and reconcile these previous analyses, we conducted a meta-analysis using data available from 1969 to 2019 to estimate the effect of in-stream structures on salmonid abundance (number and density) and biomass. Data from 100 stream restoration projects showed a significant increase in salmonid abundance (effect size 0.636) and biomass (0.621), consistent with previous reviews and studies, and a stronger effect was found in adults than juvenile fish. Despite a shift towards using more natural structures (wood and boulders) since the 1990s, structures have not become more effective. However, most projects monitor for less than 5 years, which may be insufficient time in some systems for channel morphology to adjust and population changes to be apparent. Process-based techniques, which give more space for the river, allow more long-term self-sustaining restoration.

Key words: process-based, restoration, fish habitat, deflectors, weirs, large wood, boulders, cover

Introduction

Freshwater fish are in decline worldwide due to multiple factors including habitat degradation, overexploitation, pollution, climate change, and water extraction (Strayer and Dudgeon 2010), making them the most threatened group of vertebrates in the world (Reid et al. 2013). In response to these declines, a wide range of approaches have been used over the past 100 years to increase fish populations and restore rivers to a more natural state (Bernhardt et al. 2007; Roni et al. 2008; Bernhardt and Palmer 2011). It is increasingly recognized that process-based principles are needed to restore river ecosystems, such as leaving more space around rivers for fluvial processes to operate (Piégay et al. 2005; Kline and Cahoon 2010; Ollero 2010; Biron et al. 2014; Buffin-Bélanger et al. 2015). However, most restoration actions continue to focus on the symptoms of degradation by adding structures or modifying channel forms that are perceived to provide good habitat for fish (Roni et al. 2002; Beechie et al. 2010). The major target are stream salmonids, due to their economic and cultural importance, and the realisation that many are becoming threatened or endangered (Bash and Ryan 2002; Rosi-Marshall et al. 2006; Baldigo and Warren 2008).

Often, these restoration projects involve artificial structures (in-stream structures), first developed on a large scale for trout in Wisconsin and Michigan in the 1930s (Nickelson et al. 1992; White 1996; Thompson 2006). The structures used over the years have been based on early designs, despite little evidence in the pre-1980 literature that in-stream structures increased fish populations (Roni and Quinn 2001; Roni et al. 2002, 2008, 2015; Thompson 2006); few early studies measured salmonid abundance or biomass in response to structure installation (e.g. Hale 1969; Latta 1972; Hunt 1976; Ward and Slaney 1981; Hartzler 1983; Thompson 2006). The purpose of in-stream structures, both historically and currently, include enhancing habitat complexity and diversity (Bilby and Likens 1980; Swales 1994; Louhi et al. 2016; van Zyll de Jong and Cowx 2016), increasing pool habitat (Hunt 1976; Keller and Swanson 1979; House and Boehne 1985; Roni et al. 2010), providing spawning habitat, increasing cover (Hunt 1976; Gowan and Fausch 1996; Solazzi et al. 2000), increasing macroinvertebrate resources for fish (Kail et al. 2015), and particularly more recently, restoring channel morphology (Davidson and Eaton 2013). The overarching aim is to increase the productive capacity of habitat for fish (Mitchell et al. 1998; Roni et al. 2010). Although few restoration projects evaluate their success (Bash and Ryan 2002; Bernhardt et al. 2007; Bernhardt and Palmer 2011), recent reviews have concluded that in-stream structures can lead to an increase in salmonid abundance (Roni et al. 2008; Roni 2019).

Meta-analysis provides a statistical framework for comparing the results of multiple independent studies that test the same hypothesis (Harrison 2011) and is particularly valuable in ecological studies where statistical power is often low, due to small sample sizes and high variability (Hillebrand 2008; Stewart 2010). Meta-analyses compute a quantitative average estimate or effect across different studies and a measure of uncertainty for that effect (Hillebrand 2008; Harrison 2011). Furthermore, effect sizes can be calculated for different groups within the data (Hillebrand 2008), for example by species.

Two previous meta-analyses (Stewart et al. 2009; Whiteway et al. 2010) on the effectiveness of in-stream structures to increase salmonid abundance used different methodologies and gave contrasting results. However, the conclusion by Stewart et al. (2009 p. 931) that "effectiveness of in-stream devices is equivocal" was based on data that included errors (Whiteway et al. 2010). These errors included reversing treatment and control reaches for one study and including projects that were outside of the scope of their review: did not use in-stream structures; used multiple restoration techniques; or combined data for species outside their target range. On the other hand, Whiteway et al. (2010) included more studies than Stewart et al. (2009), (51 vs 31), by relaxing the criteria for inclusion and omitting a variance estimate in each study. Variance is considered a 'hallmark' of meta-analysis, providing a weighting to average and compare effect sizes and minimise bias from studies with small sample sizes (Lajeunesse 2011, 2015). Without it, erroneous conclusions may be drawn when pooling effect sizes (Lajeunesse 2011).

In this study, we examined how the use of in-stream structures has varied over time and conducted a meta-analysis to obtain a current estimate of the effect of in-stream structures on salmonid abundance and biomass. We updated the previous meta-analyses both published a decade ago (Stewart et al. 2009; Whiteway et al. 2010) and applied a similar meta-analysis methodology as Stewart et al. (2009) whilst correcting their errors and adding biomass as a response variable. We measured whether effectiveness differed between the types of structures installed, different salmonid species and age classes, and seasons that fish were sampled. We also tested whether the effectiveness of restoration varied with the size of the stream restored. Additionally, we examined how the types of in-stream structures used (more artificial – deflectors and weirs, or more natural – boulders and large wood) varied over time, to test whether the theoretical paradigm shift towards process-based restoration since the 2000s is reflected in practice.

Methods

Search strategy and study selection

To obtain evidence and data for the meta-analysis, systematic review methodology was followed using guidelines from the Collaboration for Environmental Evidence (CEE 2018) (see Table S1 for further details on the methods used). Review questions were formulated among the authors and criteria required for data included in the meta-analysis was generated (Table 1). Our search strategy was focused to include literature in peer-reviewed journals and grey literature (e.g. conference proceedings, government or organisation reports, and theses) published from 1969 to December 2019. First, studies used in the previous two meta-analyses (Stewart et al. 2009; Whiteway et al. 2010) and several reviews (Roni et al. 2002, 2008, 2015; Roni 2019) were assessed for relevant criteria (Table 1). A literature search was then conducted in English for peer-reviewed articles and grey literature publications in electronic databases, online search engines, and library catalogues of relevant organisations. Electronic article databases included Web of Science, Google Scholar, Scopus, and ProQuest Dissertations & theses. The following key words were used wherever possible: (trout OR salmo*) AND (river OR stream OR channel OR reach OR watershed OR catchment) AND (restor* OR enhanc* OR improv* OR rehabilit* OR structure OR placement OR weir OR deflector OR cover OR boulder OR log OR wood OR LWD) AND (habitat OR population OR abundance OR densit* OR biomass), where * denotes a wildcard that can represent any collection of characters. Fifteen other databases and specialist organisations were searched using a simplified search string (see Table S1 for details). Searches were conducted in December 2018, April 2019 and March 2020.

Results from the searches were screened in three stages: (*i*) title, (*ii*) abstract, and (*iii*) full text. At each stage, publications were searched for inclusion criteria for the analysis (Table 1) and further references were found from the literature cited of included articles. The criteria for inclusion limited the number of suitable publications from the search to 63. Some publications were compilations of multiple restoration projects that used different restoration techniques and were counted as different projects where monitoring data were reported separately, resulting in 100 projects overall. The number of publications used increased from the previous meta-analyses which included 51 (Whiteway et al. 2010) and 32 (Stewart et al. 2009) publications. Out of the 63 publications used in this analysis, 25 were not used in either of these previous analyses, 7 were used only by Stewart et al. (2009), 15 were used only by Whiteway et al. (2010), and 16 were used by both authors. The remaining publications in Whiteway et al. (2010) had no measure of standard deviation and/or sample size, and eight publications in Stewart et al. (2009) did not fit the assessment criteria (Table 1).

In-stream structures were categorised into five types within two categories: artificial structures - weirs (including v-dams or wedge dams), deflectors, and cover; and, natural structures - boulders, and large wood (also referred to as large woody debris (LWD)). Structure types used followed Whiteway et al. (2010) and are common in-stream restoration techniques for salmonid rivers (Hunt 1993). We defined artificial structures as those consisting of material, either naturally present or brought to the site, organised into a specific shape (e.g. V-shape or perpendicular) and can be made of boulders, wood or logs, metal, wire, or other artificial materials. Natural structures are boulders or woody material that have been placed in the stream to replicate natural accumulations of wood or rocks. Often, multiple structure types were used within a project. Projects were classified as Artificial if only artificial structures were used, Natural if only natural structures were used, and Both if both types were used. The types of structures in each category, the functions they provide, and the number of projects that include each structure are listed in Table 2. Projects used three general types of monitoring design, with considerable variation within each: Before/After, Control Impact (BACI) designs (Stewart-Oaten et al. 1986; Stewart-Oaten and Bence 2001); Before After (BA) designs tested differences between treatments (impact sites) before and after; and, Control Impact (CI) or extensive post-treatment designs compared treatments (or impacts) to adequate controls (Hicks et al. 1991; Roni and Quinn 2001; Roni et al. 2006). Within these designs, controls were selected in multiple ways: upstream or downstream from impact sites, on different streams of similar characteristics in the region, or even next to impact sites (i.e. stream is divided into two and one side is treated and one is not).

Data extraction and analysis

The location of restoration projects, year of restoration, project monitoring time, and use of different types of structures was recorded for each study. Change in the use of structures over time was tested with a chi-squared test. Five studies that did not report a single restoration year were excluded from this analysis, because either multiple projects were completed in different years that could not be extracted separately, or the year of restoration completion was not reported. The maximum monitoring time was calculated from the last year that data were extracted (7 projects that did not report monitoring time were excluded).

To obtain a measure for the effect of restoration structures on fish (termed effect size), the change in mean abundance (see below) at impact and control sites was extracted for BACI studies, and the mean for CI and BA studies. To calculate an effect size for a study *i*, a mean or change in mean (impact, m_{1i} and control, m_{2i}), standard deviation (SD_{1i} and SD_{2i}) and sample size were required for two groups (Deeks et al. 2001). The total sample size of a study *i* (N_i) is:

(1) $$N_i = n_{1i} + n_{2i}$$

where and n_{1i} and n_{2i} are the samples sizes from the impact and control groups, respectively. The pooled SD of the two groups (S_i) is:

(2) $$S_i = \sqrt{\frac{(n_{1i}-1)SD_{1i}^2 + (n_{2i}-1)SD_{2i}^2}{N_i - 2}}$$

When fish data were reported for multiple years before or after restoration, data were averaged across years. If multiple seasons were presented, data were extracted separately if possible, or averaged if less than three years of data were provided or no SD was given (at least three data points were needed to calculate SD). If SDs were not given, they were calculated from available data using methods recommended for meta-analyses (Higgins et al. 2019). Methods used to extract and calculate means and SDs were consistent for both groups within each project. Data from different restoration treatments, species, and age classes (if reported separately) within the same study were considered as independent data points, for which an effect size was calculated.

In each study, variables of salmonid abundance were measured in different units, such as total number in a reach, density (fish m^{-2}), or biomass (g m^{-2}). The method used was consistent within each project and should not bias the results, but was standardised using a common effect size (Deeks et al. 2001). Effect sizes were calculated for each species and age distribution reported, resulting in 198 individual effect sizes for number or density measures (termed abundance hereafter) and 52 for biomass measures. The Hedge's *g* standardised mean difference method (Hedges 1981) was used to calculate the effect size of each measure from the impact and control means, SDs, and sample sizes. In each measure, the size of the impact effect (difference in means) relative to the variability observed in that trial (pooled SD S_i) was assessed and an adjustment was included to correct for small sample bias (Deeks et al. 2001). Hedge's *g* is calculated as in Deeks et al. (2001):

(3) $$g_i = \frac{m_{1i} - m_{2i}}{s_i}\left(1 - \frac{3}{4N_i - 9}\right)$$

with standard error:

(4) $$SE(g_i) = \sqrt{\frac{N_i}{n_{1i}n_{2i}} + \frac{g_i^2}{2(N_i - 3.94)}}$$

The small sample bias approaches zero when N_i is large (over 10) but can be substantial when N_i is small (Hedges 1981), ensuring that studies with small sample sizes do not have equal weighting in the overall effect size. Individual effect sizes for abundance and biomass were each combined into a weighted average, termed an overall treatment effect, using the DerSimonian and Laird random effects model (DL) with a Hartung-Knapp adjustment (DerSimonian and Laird 1986; Deeks et al. 2001). Analysis was conducted using the 'meta' package (Schwarzer 2007) in R 3.5.1. (R Core Team 2019). Abundance and biomass were also combined to obtain an overall effect size for all studies. We used a linear regression to test for a relationship between effect size and stream width before restoration. There was no differentiation made between different stream width measurements used (i.e. bankfull or wetted width), as it was reported inconsistently across studies or the type of width measured was not stated.

An overall effect size was calculated by structure type, species, age class, and season that fish were sampled. An effect size was calculated for each structure type whether or not other structures were present, and for projects that installed only one structure type at the restoration site (i.e. no other structure interventions were used) to test the effect of that particular structure type. The number of effect sizes calculated for each species and age class is reported in Table 3. Ages were grouped into adults (generally fish > 15 cm in length but may include older parr and pre-smolts), young juveniles (0+), older juveniles (1+ and some 2+), unspecified juveniles (age of juveniles not specified or fish < 15 cm where the juvenile stage was not identified), or population (all ages grouped together or age not specified). Because the majority of biomass effect sizes were for the whole population (42), biomass effect sizes were not described for different age classes. If fish were sampled in more than one season, effort was made to extract data separately, but this was not always possible if SDs were not provided. When fish data from more than one season were combined, their effect size was calculated in a combined season category. Differences in effect sizes were tested with the Kruskal-Wallis nonparametric test (chi-squared values), Wilcoxon pairwise comparisons, or within the DL random effects model. Only species with four or more individual effect sizes were included in the analysis of species differences. T-tests were used to measure differences in the effectiveness of structures and in individual species responses between abundance and biomass measures.

The importance of the effect size seems to be consistent among disciplines, with an effect size of 0.2 considered as small, 0.5 as moderate, and 0.8 as large (Cohen 1988; Bayliss et al. 2015). Below zero, the effect would be negative. In impact and control studies, a negative effect size may not mean that abundance or biomass decreased, but that increases could have been larger in control reaches, indicating that structures had no effect on the increase. The significance level for all analyses was $P < 0.05$.

Assessment of bias

Publication bias (bias towards publishing papers with positive effects) could be the biggest potential source of type 1 error in a meta-analysis (Hillebrand 2008; Harrison 2011) and could overestimate the effectiveness of a treatment (Stewart et al. 2009). While the review methodology employed aimed to include grey literature and minimise bias (Pullin and Stewart 2006; Stewart 2010; Collaboration for Environmental Evidence 2018), it is inevitable that literature will be missed. Bias was identified through a funnel plot of effect size versus standard error (Sterne and Egger 2001). The effect sizes should be symmetrically distributed around the true effect size; asymmetry in the plot is suggestive of bias (Sterne and Harbord 2004; Harrison 2011). Studies with lower variance, and greater statistical power, will be centred around the top of the plot. Symmetry was tested by the Egger Test using a linear regression (Egger et al. 1997). If the Egger Test was significant (i.e. there was asymmetry), Duval and Tweedie's trim-and-fill method (Duval and Tweedie 2000) was used to determine the number of unpublished studies needed to correct the bias and the true effect size without the bias was estimated using the R 'meta' package (Schwarzer 2007). The presence of publication bias was only considered for the abundance and biomass grouped values and not for other groups (e.g. species, number of structures etc.).

Results

Restoration projects

Most of the studies were carried out in North America (78 projects from 48 publications), with 22 projects from 15 publications in Europe (Fig. 1). Table S2 reports the species targeted from each study and the structure type installed, along with the number of projects and effect sizes extracted from those studies. The number of projects that reported sufficient data to be included in the meta-analysis has declined in the past decades (Table 4). Of the 100 total projects, 39 were BACI designs, 39 were CI, and 22 used a BA design. Of the 93 projects that reported the monitoring time after restoration, 78% monitored for less than 5 years and only 5 projects (from three publications) monitored for 15 years or more. The mean monitoring time was highest in the 1990s, while the median was higher in the previous and following decades (Table 4).

The use of different types of restoration structures has changed over time (Fig. 2). Before 1990, weirs, deflectors, and cover structures were commonly used for habitat restoration (Fig. 2A). With time, a shift towards more natural structures was observed (Fig. 2B), and since the early 1990s, LWD clearly dominated with a marked declined in the use of deflectors and cover structures (Fig. 2A). Comparing the use of structures post-1990 with pre-1990, there was a significant increase in the number of projects that consisted only of natural structures and a decline in those that used only artificial structures (chi-square = 14.611, df = 2, p = < 0.001), while there was no significant difference in the number of projects that used both types (Fig. 2B).

Overall effect sizes

Both salmonid abundance (Fig. 3) and biomass (Fig. 4) significantly increased with in-stream structures (DerSimonian and Laird random effects model (DL) abundance = 0.636, 95% confidence interval (CI) (0.496, 0.776), p < 0.001, N = 198; biomass DL = 0.621 (0.395, 0.847), p < 0.001, N = 52). Combining abundance and biomass effect sizes resulted in an overall effect size of 0.637 (CI = 0.516, 0.757, p < 0.001, N = 250). Of the 198 abundance effect sizes, 158 were positive (an effect size greater than zero) of which 36 were significant, 40 were negative of which four were significantly negative, and 158 did not significantly differ from zero. Forty-two biomass effect sizes were positive, of which six were significant, ten were negative but none of these were significant, and 46 did not significantly differ from zero. There was no difference in abundance and biomass effect sizes (t-test = 1.1485, df = 51.572, $p = 0.256$), nor were there differences in effect sizes by the time period installed for abundance, biomass, or both combined (Fig. 5) (all $p > 0.1$). Only three studies measured biomass in the years 2000-2016 and were not included in the analysis.

Stream size did not appear to influence the outcome of restoration projects, with no relationship between effect size and stream width (abundance: $R^2 = 0.011$, df = 119, $p = 0.247$; biomass: $R^2 = 0.0076$, df = 28, $p = 0.648$). Stream widths reported for 50 projects ranged from 2.5 m to 33 m (median = 5.7 m). However, most of the projects were carried out on small streams, with only three studies on streams larger than 10 m wide, and as mentioned previously different methods for measuring width were used across studies. Fish sampling was mostly conducted in the summer (Table 5). Projects that measured fish in the spring had the highest effect sizes among seasons, followed by summer, while projects that measured fish in autumn and winter had non-significant effect sizes (Table 5).

Structure type

Projects often used a combination of structure types making it difficult to distinguish the effects of the individual structures. In-stream structures had a positive effect on salmonid abundance ($p < 0.001$) except when cover structures were the only intervention used ($p = 0.276$) (Fig. 6). In-stream structures also had a positive effect on salmonid biomass when multiple structure types were installed ($p < 0.05$), however, when considering studies with one structure type installed, only LWD had a positive effect ($p = 0.026$) (Fig. 7). Two structure types used alone (boulders and deflectors) were excluded in the biomass measures due to low sample sizes (N = 3). The type of structure installed did not make a significant difference to abundance or biomass effect sizes, whether only one type of structure was installed ($p > 0.1$) or multiple ($p > 0.05$). There was also no difference between projects classed as Artificial, Natural, or Both ($p > 0.1$) and effect sizes were significant for all these combinations for abundance and biomass ($p < 0.05$). The number of different types of structures installed did not significantly change the effectiveness of restoration on abundance or biomass (p >

0.1). While abundance effect sizes were all significant regardless of the number of different structures installed, there were no significant effects on biomass of projects with four types of structures (Table 6).

Species

The dominant species targeted in restoration was brown trout (*Salmo trutta*) (Table 3). Abundance increased for five species and salmonids combined ($p < 0.05$), but did not increase for cutthroat trout ($p = 0.128$, $N = 10$), Chinook salmon ($p = 0.067$, $N = 5$), or brook trout ($p = 0.110$, $N = 17$) (Fig. 8). There were no significant differences between species abundance effect sizes ($p = 0.352$), or between stream resident species (brown, brook, rainbow and cutthroat trout) and anadromous species (Atlantic, coho and Chinook salmon and steelhead trout) (chi-squared = 0.0029, df = 1, $p = 0.957$). There were also no differences in biomass studies between species or anadromous vs. not ($p > 0.1$), even though only brook and brown trout showed increases in biomass ($p < 0.05$) (Fig. 8). There were no differences in effect sizes by species between abundance and biomass measures ($p > 0.1$ for all with sufficient data to compare).

Structures and species

When the effects of structures by species were considered, brown trout showed positive responses to the greatest variety of structures than other species, followed by coho salmon (Table A1). Brown trout abundance and/or biomass increased with all structures and when artificial and natural structure types were used. There were no structures that appeared to benefit all species. However, many relationships between specific species and structures were not analysed due to small sample sizes (less than 4).

Age classes

Abundance increased in all age classes with the largest increases found in adults (Fig. 9). There were significant differences in abundance effect sizes between adults and young juveniles and unspecified juveniles ($p < 0.05$), but not between other age classes. Analysis between age classes of each species could not be carried out due to small sample sizes in each group (Table 3). Most of the adults were brown trout (73%) while mostly juveniles were sampled for the majority of other species (Table 3).

Assessment of bias

The abundance effect sizes showed more than expected positive studies, shown in the asymmetry in the grey squares (original effect sizes plotted) in the funnel plot (Fig. 10) ($t = 2.198$, $df = 196$, $p = 0.029$). This asymmetry indicates there is bias towards positive studies and should be corrected. Bias was corrected with Duval & Tweedie's trim-and-fill procedure (Duval and Tweedie 2000), where 39 dummy studies (black circles in Fig. 10) were added to the plot to compensate for the bias. The corrected effect size of 0.378 (0.210, 0.535; $p < 0.001$), indicated that in-stream structures still increased salmonid abundance, but the effect was small. The test for asymmetry for biomass meta-analysis showed no publication bias ($t = 1.659$, $df = 50$, $p = 0.103$) (Fig. 11), so no correction was needed.

Discussion

Overall effect sizes

Our meta-analysis showed that in-stream restoration structures increased both salmonid abundance (DL = 0.636) and biomass (DL = 0.621). With the exception of Stewart et al. (2009), these findings agree with most previous reviews and meta-analyses: that in-stream restoration structures increase fish abundance, although there is a large variation in responses (Roni et al. 2008; Whiteway et al. 2010; Roni 2019). While there was publication bias towards positive studies for abundance measures in our analysis, the corrected effect size for this bias was still significantly positive (DL = 0.378), albeit at a lower level according to Cohen's (1988) criteria. This bias may indicate that significant studies are more likely to be published (Hillebrand 2008; Kemp 2010; Harrison 2011). Suspected publication bias led Stewart et al. (2009) to conclude that the effect of in-stream structures on salmonid abundance was equivocal, even though their meta-analysis did produce a statistically significant positive result. Our analysis did include a much higher proportion of grey literature than Stewart et al. (2009) (29% vs. 9%) and there was no publication bias detected in our biomass effect size. Furthermore, there were very few significantly negative effect sizes in our analysis (four for abundance and none for biomass). For several studies in Stewart et al.'s (2009) analysis, we extracted data for longer time periods or from different experimental sections, including: Brusven et al. (1986); Hvidsten and Johnsen (1992); Linløkken (1997); Mitchell et al. (1998); Giannico (2000); Zika and Peter (2002); Johnson et al. (2005); and Sweka and Hartman (2006). We found errors with data used by Stewart et al. (2009) including reversing treatment and control sections, using sites that had not used in-stream restoration structures, and over-stating sample sizes. Given the extent of these errors, the results reported by Stewart et al. (2009) should be considered with caution. Whiteway et al. (2010) re-analysed Stewart et al.'s (2009) data using a log response ratio with some of the errors corrected and found a clear positive effect size of 1.1, larger than Whiteway et al.'s (2010) own effect size of 0.51. Additionally, in contrast to Whiteway et al. (2010), we examined study variance, resulting in a more robust estimate of effect size.

Structure type

The large variation in response to different in-stream structures in our study suggests that there are no broad guidelines for all salmonid species (Fig. 6 and 7). As suggested by previous reviews (Roni et al. 2002; Whiteway et al. 2010; Roni 2019), we found no significant differences in effectiveness between structure types. This result may be due partly to insufficient sample sizes. Overall, cover structures on their own (when no other structures types were installed) appeared to have no significant effect on abundance or biomass; however, individual effectiveness of these structures was highly variable, and in some projects they appeared to be very successful (e.g. Brusven et al. 1986; Höjesjö et al. 2014). Deflectors were often reported to be the most successful structures in fish rehabilitation projects (Ward and Slaney 1981; Mitchell et al. 1998; Thompson 2002). Deflectors had one of the highest effects sizes for salmonid abundance in our analysis, but had a very small effect on biomass. . A higher response rate was detected for brook trout and Atlantic salmon numbers when boulder clusters and v-dams (classed as weirs in this study) were installed, compared to half log covers in a Newfoundland stream (van Zyll de Jong and Cowx 2016), while a combination of LWD and cover structures resulted in a positive response to adult brown trout in Minnesota, compared to no significant effect with just cover structures (Thorn and Anderson 2001). Our analysis suggested that boulders were highly effective for increasing both abundance and biomass. Boulder clusters or placements are effective at creating habitat complexity and hiding places (Kennedy et al. 2014) and increasing pool volumes and instream cover (Näslund 1989; House 1996; van Zyll de Jong and Cowx 2016). Several studies have documented the high stability of boulder structures (House 1996; Kennedy et al. 2014), even after 20 years of placement (van Zyll de Jong and Cowx 2016). In the right context, boulders can provide cost-effective (Kennedy et al. 2014), and sustainable long-term improvements in stream conditions (van Zyll de Jong and Cowx 2016). Whether structures are artificial or natural, fish are responding to the effect of the structure, rather than the structure itself (Crispin et al. 1993; House 1996; Roni and Quinn 2001; Roni et al. 2008; Floyd et al. 2009; Clark et al. 2019). While artificial structures, such as log weirs and deflectors can be effective creating habitat, and thus increasing fish numbers, often these effects do not last as long as more naturally placed structures such as LWD and boulders (Roni et al. 2002, 2015).

Species and age classes

Previous evidence suggests that in-stream structures may be more effective for stream-resident, and larger fish, than for juveniles of anadromous populations (Hunt 1988; Hicks and Reeves 1994; Whiteway et al. 2010). Other studies indicate that Pacific salmonid species and life stages that prefer pools, such as juvenile coho salmon, Chinook salmon, and cutthroat trout, may benefit more from in-stream restoration (Roni et al. 2008). Stream residents may prefer slower water velocity than migrants (Morinville and Rasmussen 2008), and deeper habitats may be more preferred for larger fish (Fausch 1993; Mäki-Petäys et al. 1997; Horan et al. 2000; Armstrong et al. 2003). For example, brown trout avoid shallow pools (less than 60 cm deep) and habitats that lack cover (Dieterman et al. 2018). The indication that species that prefer pool habitats do better after in-stream restoration is not surprising given that the purpose of many in-stream structures is to increase pool habitat and availability. Multiple studies document an increase in pool area, number of pools, and/or increased depth with the addition of structures, particularly LWD (e.g. Näslund 1989; Gowan and Fausch 1996; Cederholm et al. 1997; Roni and Quinn 2001; Antón et al. 2011; O'Neal et al. 2016). While our analysis does not suggest that structures were better for different species or migratory phenotypes, the sample sizes were probably not sufficient to detect differences between species. Furthermore, combining all life-stages together may overlook important differences in habitat preferences at different life-stages within species. Our analysis indicated that while salmonid abundance increased over all age-classes, there were significant differences in effectiveness between adults and young juveniles ($p = 0.020$) and unspecified juveniles ($p = 0.002$). The effect size for adults was higher than other life-stages, agreeing with previous suggestions that in-stream restoration favours larger fish.

Reasons for structure or project failure

Previous research has suggested that many restoration projects fail to result in improvements in fish density or biomass because the habitat is inadequate (Nickelson et al. 1992; Rosenfeld and Hatfield 2006) or the underlying causes of decline are not addressed (Cochran-Biederman et al. 2014; Roni et al. 2014; van Zyll de Jong and Cowx 2016). Habitat requirements will be different at each site or stream, for species targeted, and at each life stage (Bjornn et al. 1991; Armstrong et al. 2003) and will depend on the time of day (Mitchell et al. 1998) and season (Nickelson et al. 1992; Roni and Quinn 2001). Habitat requirements at each stage must be understood so that restoration can be targeted for the desired species (Nickelson et al. 1992; Armstrong et al. 2003). However, most salmonid restoration projects focus on providing summer habitat, even if habitat is more limiting in other seasons (Nickelson et al. 1992). For example, coho salmon smolt production is thought to be limited by the availability of winter habitat in many coastal Oregon streams (Nickelson et al. 1992). Of the projects included in our study, 53% sampled fish only in the summer, whereas several studies show high seasonal variability in habitat use (Nickelson et al. 1992; Cederholm et al. 1997; Mäki-Petäys et al. 1997; Bramblett et al. 2002; Zika and Peter 2002; Mollenhauer et al. 2013; Polivka et al.

2015). If most restorations focus on summer habitat, then they will appear to be less effective if sampling is in the fall, when fish are beginning to move to over-wintering habitat. In our study, the highest effects on abundances and biomass were found in the spring, while the lowest were in autumn.

There are very few long-term (greater than 5 years) assessments of restoration outcomes (Champoux et al. 2003; Roni et al. 2008). Despite research suggesting that at least four to eight years are needed post-restoration to determine the full fish population responses (Hunt 1976; Binns and Remmick 1994), monitoring time for projects in our analysis has not significantly increased over time (Table 4) and over three quarters of all projects monitored for less than 5 years. In some catchments, years to decades may be needed to rebuild fish populations to sustainable levels (Luhta et al. 2012). More than a year or two is needed for populations to expand into existing and new habitats (Binns and Remmick 1994; O'Neal et al. 2016), and multiple generations may be required to detect adult responses (Roni et al. 2002). First, adequate flows are required to change stream morphology to create the required habitat; fish then respond to this habitat change (Roni and Quinn 2001; Roni et al. 2006). Monitoring for only one or two years may be confounded by natural variations in fish stocks (Binns and Remmick 1994). Additionally, in some environments, morphological changes may not be apparent after only two years (Kondolf and Micheli 1995). Therefore, it is recommended that 10 years or more of continuous monitoring is required to detect a sustained response to restoration (Bisson et al. 1992, 2003; Kondolf and Micheli 1995; Reeves et al. 1997).

Installed structures are not always appropriate for the stream characteristics or cause the intended effects. In early restoration projects, failure was often attributed to applying techniques developed for low gradient streams (weirs and deflectors) to streams of high gradient and energy (Platts and Rinne 1985; Frissell and Nawa 1992). The move towards using LWD instead of artificial structures in some ways attempts to limit structural failure (Bisson et al. 2003; Roni et al. 2006) and buffer the effects of drought and flooding events (Sweka and Hartman 2006). Three of the projects with a significant negative effect size in our analysis were due to droughts and floods resulting in structural failure and decreased fish abundance (Reeves et al. 1997; Vehanen et al. 2010).

Many artificial structures are not built to last, or last much less time than intended (Champoux et al. 2003). An early assessment of habitat structures in Oregon and Washington over a 20-year period found that woody debris and individual boulder placement were the only structures that did not fail in more than half the cases; many projects lasted less than 15 years because of wash out from floods (Frissell and Nawa 1992). Just two years after construction in Meadow Creek, Oregon, fewer than 20% of structures were still functioning (Miller 1997), while in several Wisconsin streams, a lack of maintenance caused structures built to last up to a century deteriorate in 25 years (White 1996). In other earlier projects, artificial structures had deteriorated or were no longer working almost 20 years after installation (Ehlers 1956; White 1972). Bank deflectors placed in Lawrence Creek, Wisconsin, failed in one area but were still mostly functioning in another 36 years later, highlighting the need for knowledge of the geomorphological context when placing structures (Champoux et al. 2003). For example, Champoux et al. (2003) postulated that deflectors were well adapted for narrow and sinuous channels, rather than wide, steep, and dynamic stretches with coarse sediments, resembling the area where they failed. Other studies have noted much lower failure rates (Roper et al. 1998; Schmetterling and Pierce 1999; Roni et al. 2008, 2015). In an assessment of almost 4,000 structures (logs, boulders, and gabions), Roper et al. (1998) found that more than 80% remained in place after floods with return intervals greater than 5 years; structures in larger streams and with higher magnitude floods were more likely to fail. Naturally placed structures seem to last longer than those anchored in place (Roni et al. 2008, 2015). In a stream in western Montana, 85% of rock and wood structures remained stable after a 50-year recurrence interval flood (Schmetterling and Pierce 1999). Likewise, in a summary of natural and placed wood structures, Roni et al. (2015) reported that less than 20% failed.

In other projects, added material may not cause the intended effect (Flannery et al. 2017). For instance, LWD is often used to create pool habitats. In central Appalachians streams, however, LWD did not form pools in the highest gradient streams and only 4% of the added LWD created pool habitat (Sweka and Hartman 2006). Installing effective and long-lasting stream restoration structures likely requires skill and experience (Baril et al. 2019). However, restoration projects are managed by groups with a broad range in expertise (but are often lacking hydrogeomorphological expertise) (Baril et al. 2019), likely contributing to the high variability in responses and failure rates. For a longer-term solution, restoring natural processes is a better restoration approach. Unmodified streams with naturally dynamic hydrological regimes and processes are more resistant to the negative effects of flooding and drought (Poff et al. 1997; Elosegi et al. 2011), and if natural processes are restored, habitats that are limiting to native fish may more likely be created or restored (Florsheim et al. 2008; Beechie et al. 2010).

Process-based restoration

In recent years, there has been a stronger emphasis on restoring natural fluvial processes at the watershed scale (Roni et al. 2008; Beechie et al. 2010; Biron et al. 2017), instead of focusing on manipulating in-stream habitats and specific morphologies that are targeted by in-stream structures (Roni et al. 2002). This shift in restoration approach has influenced the use of in-stream structures, reflected in the use of more large wood in restoration since the 1990s and an increase in projects that use only natural structures and a decrease in the use of artificial structures (Fig. 2). Restoration projects in western North America have also shown this trend, where the understanding of wood processes has led to more restoration projects using techniques that allow wood movement, rather than anchoring wood structures in place (Bisson et al. 2003). An added benefit of more natural wood structures is that they are often much cheaper than anchored techniques (Carah et al. 2014). This shift is also clear in the use of bank stabilisation over time. Prior to 1995, several restoration projects incorporated bank stabilisation structures such as riprap into restoration projects in an attempt to limit erosion and the lateral migration of channels (Hvidsten and Johnsen 1992) as well as narrow channels to create pool habitat (Hunt 1992). Bank erosion was perceived as a hazard and something that needed to be controlled (Piégay et al. 2005). Often riprap was employed for public safety or economic purposes, not for restoration benefits (Florsheim et al. 2008). More recently, allowing a river to move freely and maintaining a natural sedimentation regime is recognized as important for ensuring healthy river ecosystems (Florsheim et al. 2008; Choné and Biron 2016; Williams et al. 2020). The ecological significance of erosion in providing riparian habitats and natural bank habitats providing large woody debris are becoming more obvious for river restoration practitioners (Benda et al. 2004; Piégay et al. 2005; Florsheim et al. 2008). A naturally moving river acts as a passive restoration approach, as a natural sedimentation regime creates habitats that many in-stream structures aim to replicate (Choné and Biron 2016; Biron et al. 2017; Williams et al. 2020).

The shift in structure use was not associated with an increase in the effect size of in-stream structures over time (Fig. 5). This suggests that the type of in-stream structure is not important for generating overall positive benefits in fish response, also indicated by other authors (Roni et al. 2008; Whiteway et al. 2010; Roni 2019). However, for effective restoration it is suggested that in-stream enhancement techniques should only be used where short-term improvements are needed or used alongside process-based restoration (Roper et al. 1998; Roni et al. 2002) and that focus should instead be on protecting and connecting habitat, and restoring habitat-forming processes (Roni et al. 2008; Cramer 2012). This can be achieved by adopting river management approaches based on "erodible corridor" (Piégay et al. 2005), "fluvial territory" (Ollero 2010), "river corridor" (Kline and Cahoon 2010), or "freedom space" (Biron et al. 2014; Buffin-Bélanger et al. 2015; Choné and Biron 2016). While these approaches are often aimed at larger watersheds, their principles of allowing natural erosion and hydrological processes to be restored can be implemented in smaller streams, where most restoration projects are carried out. Such passive restoration approaches are more likely to succeed in the long term, even if their impacts may be difficult to quantify in the short term.

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Tables

Table 1. Criteria for inclusion in the meta-analysis.

Criteria	Include	Exclude
Ecosystem	River and streams, artificial channels that are connected to natural ones	Lakes and coastal waters
Species	Salmonid species	Non-salmonid species
Location	Global	None
Intervention	In-stream restoration structures: deflectors, weirs, cover structures, boulders, large woody debris. Examples of these structures are provided in Table 2.	Fish passage devices, comparison of degraded rivers to natural rivers, in-stream structures that were not used for restoration (e.g. for flood control or erosion control unless specifically for restoration purposes), spawning gravel
Monitoring design	Before/After, Control/Impact (BACI); control (or reference) and impact (or treatment, intervention, or experimental sections) (CI), also called extensive-post treatment; before and after treatment (BA)	No monitoring data, no appropriate control, modelled outcomes
Monitoring data	Mean, standard deviation, and sample size for at least a treatment group and control group	No quantitative data, inability to compute standard deviation
Time period	Any length of study published between 1969 and 2019	Articles published outside of these limits
Measures	Total number, density and/or biomass for at least two comparable groups (i.e. impact and control, before and after). Any method of collection.	Survivability, condition, growth, movement of fish
Outcome	Irrespective of effects	None

Table 2. Structure types, functions, and use in projects included in the meta-analysis.

Structure type (number of projects)	Examples	Functions (references that describe function of structures)
		Engineered structures
Weirs (29)	Can be made from logs, boulders, and gabions; v-notch weirs, v-dams	• Creates pools (House and Boehne 1985; Gowan and Fausch 1996; Mitchell et al. 1998) • Reduces stream bed gradients (Klassen and Northcote 1986) • Impounds beds with spawning gravel (Klassen and Northcote 1986; House 1996) • Restore incised streams (Conner et al. 2016) • Creates flow heterogeneity (Näslund 1989) • Increase oxygen content in water by creating turbulent flow (Mitchell et al. 1998)
Deflectors (25)	Can be made from boulders, logs and gabions; bank deflectors, wing deflectors, vanes, groins, spur dikes, abutments	• Concentrates and accelerates stream current (Saunders and Smith 1962; Linløkken 1997; Mitchell et al. 1998) • Modifies flow and creates flow heterogeneity – important for fish development and spawning (Champoux et al. 2003) • Enhances scour to create pools (Ward and Slaney 1981; Pagliara and Kurdistani 2017) • Improves sinuosity (Hunt 1976; Mitchell et al. 1998) • Narrows and deepens channels (Hunt 1976), creating pools (Thompson 2006)

Cover (30)	Bank cover structures, half-logs, cribs, single logs, can be made from artificial and natural material	• Provides protection from predators, competitors, or unfavourable conditions such as high velocities (Hartzler 1983; Boss and Richardson 2002) • Provides visually isolated shady areas (Dolloff 1986; Fausch 1993) • Creates good feeding position of low velocity adjacent to a switch current (Dolloff 1986; Fausch 1993; Inoue and Nakano 1998)
		Natural structures
Boulders (24)	Boulder groups and clusters, natural or placed boulders, cobble addition	• Creates shelter from current (Fausch 1993; Giannico 2000) • Provides cover (Bjornn et al. 1991; Fjellheim et al. 2003; Dieterman et al. 2018) • Increases turbulence and modifies flows (Näslund 1989; Miller 1997) • Creates habitat diversity (van Zyll De Jong et al. 1997)
LWD (43)	Natural and added large wood, structures, bundles, brush, fine woody debris, log dams (includes engineered if structured in a way to replicate natural placement), rootwads, beaver dam analogues (if	• Facilitates gravel deposition for spawning (Sweka and Hartman 2006; Antón et al. 2011) • Retains sediments, gravel, and organic matter (Bilby and Likens 1980; Roni et al. 2015) - affects benthic invertebrate communities which are food for fish (Inoue and Nakano 1998; Kratzer 2018) • Creates and scours pools (Keller and Swanson 1979; Berg et al. 1998) • Provides cover (Culp et al. 1996; Flebbe 1999; Giannico and Hinch 2003) • Increases nutrients through carcass retention (Johnson et al. 2005)

made from natural materials)	• Increases bank stability (Keller and Swanson 1979) • Provides shelter from current (Fausch 1993), especially during floods (Lehane et al. 2002) • Increases structural complexity (Giannico 2000; Zika and Peter 2002) and diversity of habitats (Bilby and Likens 1980; Roni 2001; Lehane et al. 2002) • Raises water table and repairs incised streams (Pollock et al. 2014)

Table 3. Number of effect sizes (ES) calculated by salmonid species (abundance and biomass) and age class (abundance).

Species	Abundance ES	Biomass ES	Number of abundance effect sizes by age-class				
			Adult	Young juvenile	Older juvenile	Unspecified juvenile	Population
Salmo trutta	67	21	25	15	10	8	9
S. salar	12	4		2	5	1	4
Oncorhynchus kisutch	36	-	1	19	3	13	
O. mykiss (steelhead)	30	-		3	16	10	1
O. mykiss (rainbow)	5	3		1			4
O. clarkii	10	3		1	5	1	3
O. tshawytscha	6	-		2		4	
Salvelinus fontinalis	17	9	4	2	1	3	7
Thymallus arcticus	-	1					
Salmonids (combined or unspecified)	15	11	4	6	1	3	1
Total	198	52	34	51	41	43	29

Table 4. The number of projects completed over time and average monitoring times. Data is from abundance and biomass studies calculated from the maximum number of years that projects were monitored after installing structures.

Time period	Mean (median) monitoring time (years)[1]	Number of projects completed[2]
Pre-1980	2.92 (2)	13
1980-1989	3.7 (3)	34
1990-1999	4.2 (2)	28
2000-2016	3.8 (3)	20
All years	3.7 (3)	95

[1]Projects that reported monitoring time after restoration (93 in total).

[2]Projects that reported a date for restoration completion (one project during 1980-1989 and two during 1990-1999 time periods did not report monitoring times).

Table 5. Effect sizes by the season of fish sampling (significant effect sizes in bold).

Season	Abundance effect size (95% CI)	N (p-value)	Biomass effect size (95% CI)	N (p-value)
Summer	**0.615 (0.441, 0.788)**	**105 (<0.001)**	**0.770 (0.369, 1.170)**	**25 (<0.001)**
Autumn	-0.232 (-0.769, 0.306)	23 (0.381)	0.402 (-0.287, 1.090)	8 (0.211)
Winter	1.393 (-0.085, 2.872)	5 (0.059)	-	-
Spring	**1.134 (0.816, 1.452)**	**14 (<0.001)**	**1.158 (0.237, 2.079)**	**6 (0.023)**
Combined (fish sampled in two or more seasons)	**0.676 (0.399, 0.953)**	**25 (<0.001)**	**0.466 (0.090, 0.841)**	**8 (0.022)**
Unspecified	**0.784 (0.406, 1.162)**	**26 (<0.001)**	0.381 (-0.628, 1.389)	5 (0.354)

Table 6. Effect sizes by the number of different structure types installed at a site (significant results in bold).

Number of structure types	Abundance effect size (95% CI)	N (P-value)	Biomass effect size (95% CI)	N (P-value)
1	**0.693 (0.509 - 0.877)**	112 (< 0.001)	**0.650 (0.348 – 0.952)**	26 (0.001)
2	**0.493 (0.148 – 0.839)**	50 (0.006)	**0.776 (0.026 – 1.525)**	14 (0.044)
3	**0.675 (0.453 – 0.896)**	20 (<0.001)	**0.310 (0.077 – 0.543)**	6 (0.019)
4	**0.841 (0.500 – 1.183)**	16 (<0.001)	0.571 (-0.181 – 1.323)	6 (0.108)

List of Figures

Fig. 1. Location of restoration studies included in the meta-analysis (black circles) in North America (A) and Europe (B). Maps made in ArcGIS 10.4. Sources: Baselayer for A and B: Esri (2011). State and province boundaries in A: Esri (2009). Points: approximate locations of studies used in the meta-analysis were estimated from location data in each study.

Fig. 2. Changes in the use of structure types over time. Proportions are calculated within each time period. A: Proportion of projects that individual structures are used. Proportions do not add to 100% as more than one structure could be used per project. B: Proportion of projects that use artificial, natural, or a combination of both artificial and natural structures.

Fig. 3. Forest plot showing the effect of in-stream structures on salmonid abundance. Black circles represent the effect size of individual measures and the black diamond at the bottom is the overall standardised effect size for all studies (0.636). Studies with multiple effect sizes calculated are shown on the same line. Grey error bars are 95% confidence intervals. Note that some confidence intervals extend beyond the axis.

Fig. 4. Forest plot showing the effect of in-stream structures on salmonid biomass. Black circles represent the effect size of individual measures and the black diamond is the overall standardised effect size (0.621) for all studies. Studies with multiple effect sizes calculated are shown on the same line. Grey error bars are 95% confidence intervals.

Fig. 5. Changes in the effect sizes by structure installation period for abundance, biomass and both (All) effect size groups. Grey bars are 95% confidence intervals.

Fig. 6. Abundance effect sizes (mean +/- 95% CI) by structure type. The top panel (One and more) shows the effect sizes for projects with each structure installed, whether or not another structure type is present, the middle panel (Only one) are effect sizes for projects with just one structure type, and the bottom panel (Combined) are effect sizes for projects with only artificial, only natural, or a combination of structures (both).

Fig. 7. Biomass effect sizes (mean +/- 95% CI) by structure type. See Fig. 6 caption for further explanation.

Fig. 8. Effect sizes (mean +/- 95% CI) by species for abundance or biomass measures. Other included unspecified or combined trout or salmon or species that had less than four studies.

Fig. 9. Effect sizes (mean +/- 95% CI) by age classes for abundance measures. All species were combined.

Fig. 10. Trim and fill funnel plot showing the abundance effect sizes against their standard errors. The original effect sizes in the meta-analysis are the grey squares whereas the black circles are the added points to correct for the bias. Significance levels are shown (p values) with significant studies lying in the $p < 0.05$ and $p < 0.01$ areas. The dotted line is the pooled estimate of the effect size and SE for all abundance measures. A: All effect sizes are shown. B: Only the top of the plot (y axis 0 to 5) is plotted to show more detail. The large variance in standard error of grey squares in A and asymmetry of studies (grey squares) around the mean (dotted vertical line) in B is indicative of bias.

Fig. 11. Funnel plot for the biomass effect sizes against their standard error. Significance levels are shown (p values) with significant studies lying in the $p < 0.05$ and $p < 0.01$ areas. The dotted line is the pooled estimate of the effect size and SE for all biomass measures.

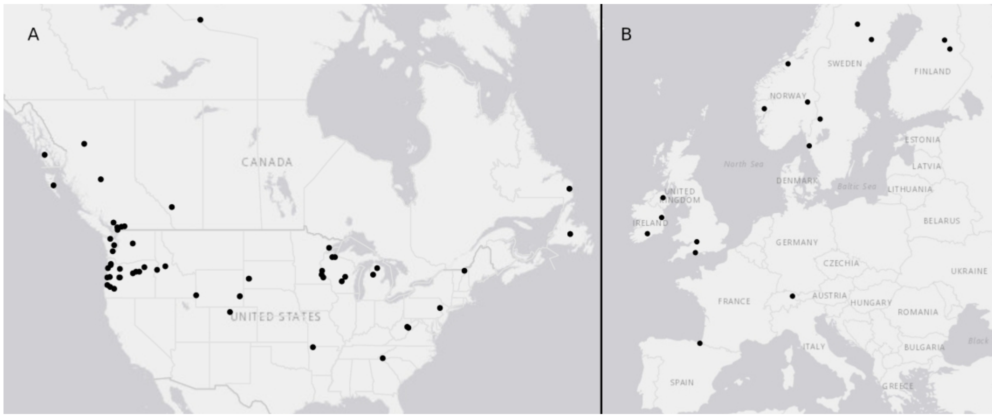

Fig. 1. Location of restoration studies included in the meta-analysis (black circles) in North America (A) and Europe (B). Maps made in ArcGIS 10.4. Sources: Baselayer for A and B: Esri (2011). State and province boundaries in A: Esri (2009). Points: approximate locations of studies used in the meta-analysis estimated from location data in each study.

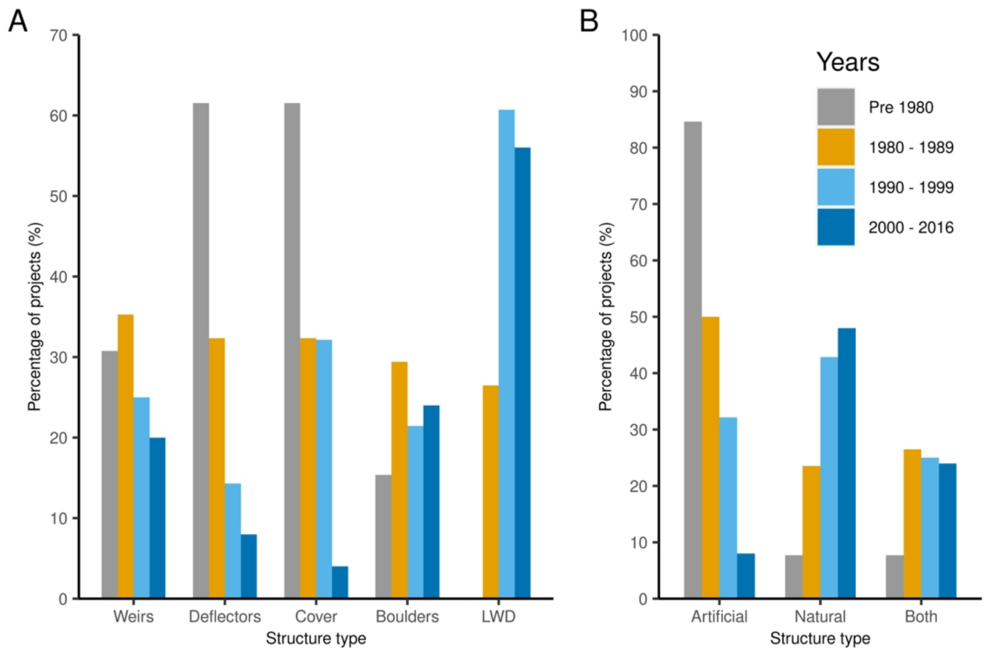

Fig. 2. Changes in the use of structure types over time. Proportions are calculated within each time period. A: Proportion of projects that individual structures are used. Proportions do not add to 100% as more than one structure could be used per project. B: Proportion of projects that use artificial, natural, or a combination of both artificial and natural structures.

152x101mm (300 x 300 DPI)

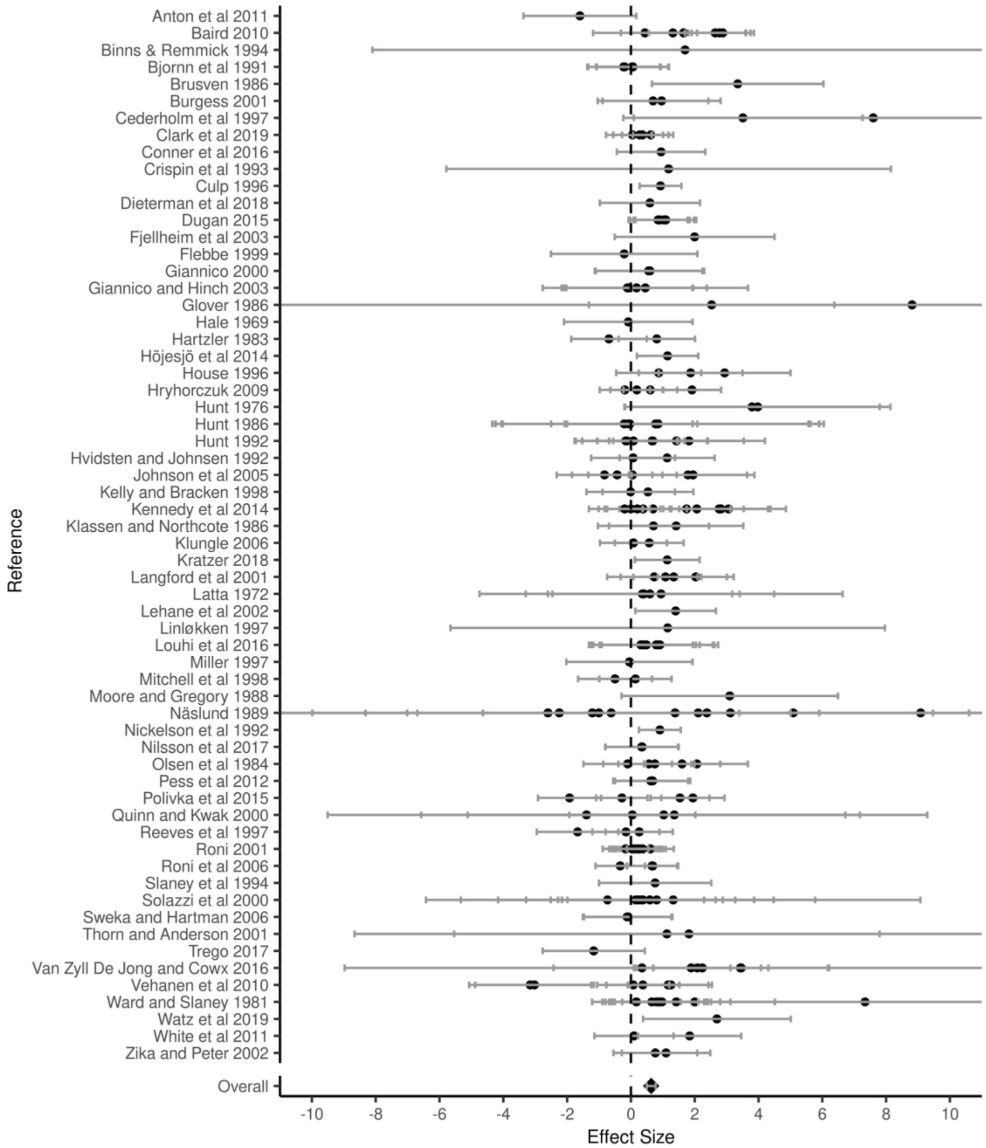

Fig. 3. Forest plot showing the effect of in-stream structures on salmonid abundance. Black circles represent the effect size of individual measures and the black diamond at the bottom is the overall standardised effect size for all studies (0.636). Studies with multiple effect sizes calculated are shown on the same line. Grey error bars are 95% confidence intervals. Note that some confidence intervals extend beyond the axis.

152x177mm (300 x 300 DPI)

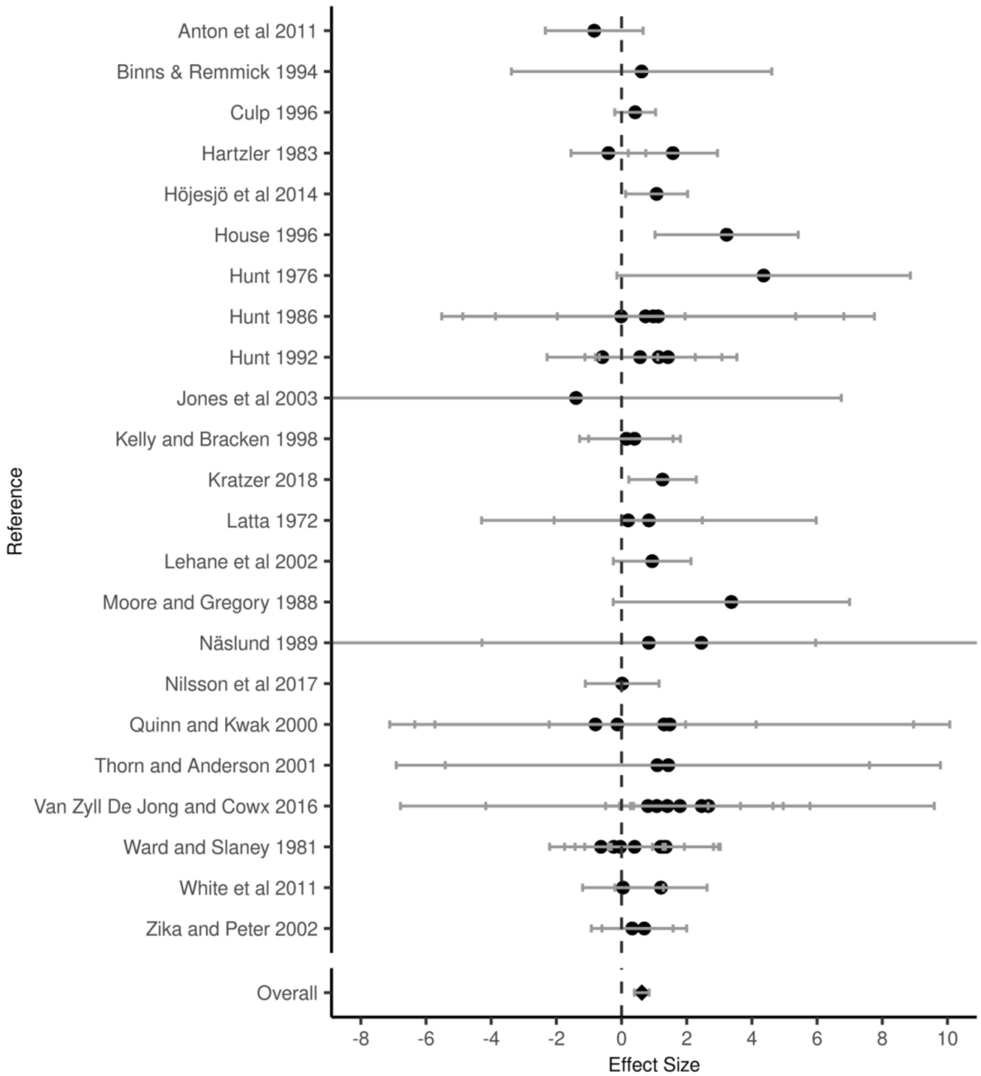

Fig. 4. Forest plot showing the effect of in-stream structures on salmonid biomass. Black circles represent the effect size of individual measures and the black diamond is the overall standardised effect size (0.621) for all studies. Studies with multiple effect sizes calculated are shown on the same line. Grey error bars are 95% confidence intervals.

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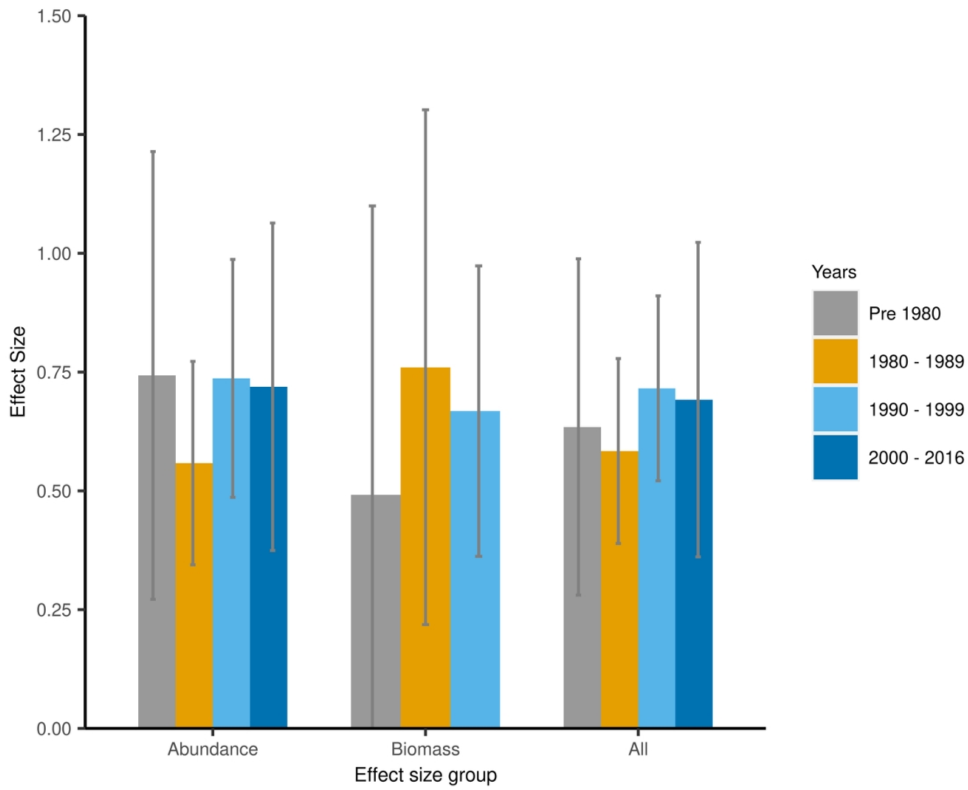

Fig. 5. Changes in the effect sizes by structure installation period for abundance, biomass and both (All) effect size groups. Grey bars are 95% confidence intervals.

127x101mm (300 x 300 DPI)

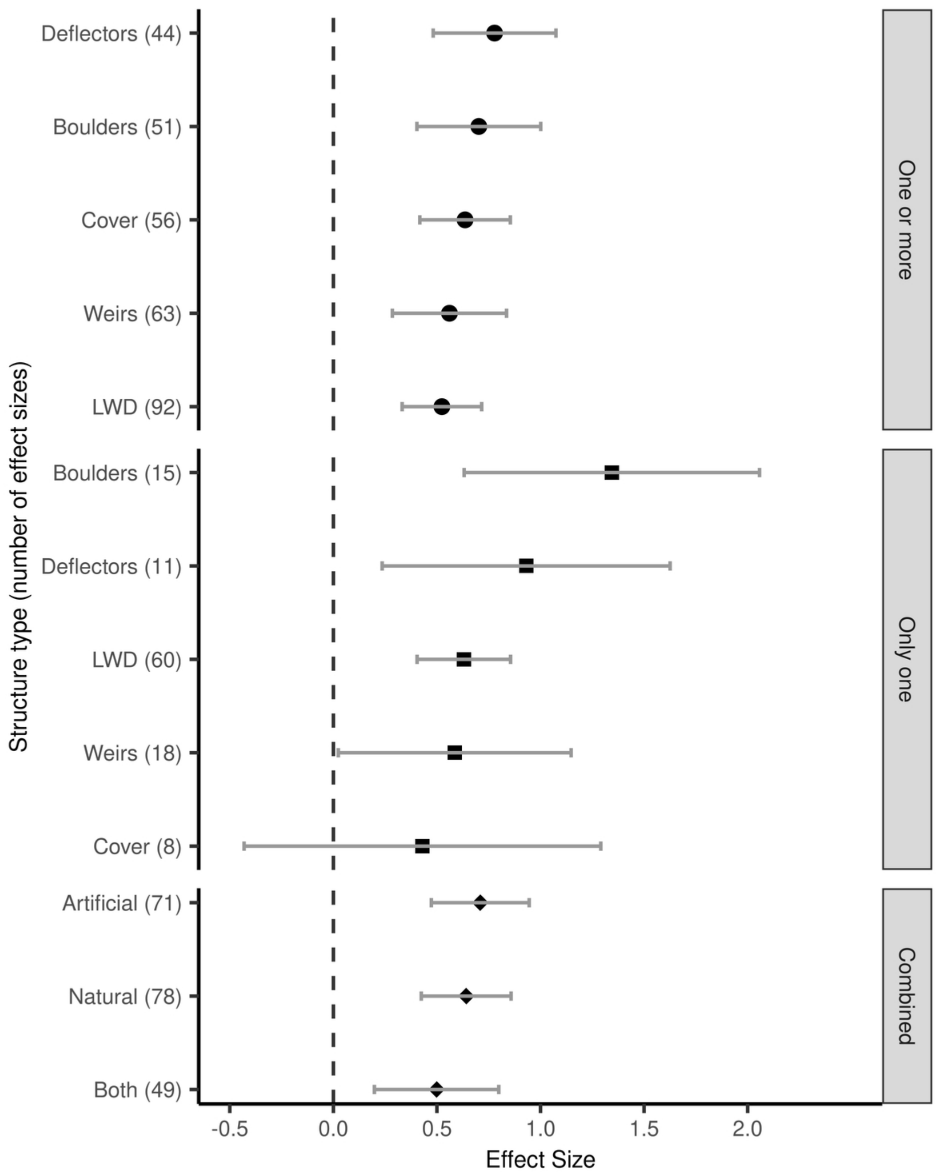

Fig. 6. Abundance effect sizes (mean +/- 95% CI) by structure type. The top panel (One and more) shows the effect sizes for projects with each structure installed, whether or not another structure type is present, the middle panel (Only one) are effect sizes for projects with just one structure type, and the bottom panel (Combined) are effect sizes for projects with only artificial, only natural, or a combination of structures (both).

101x127mm (300 x 300 DPI)

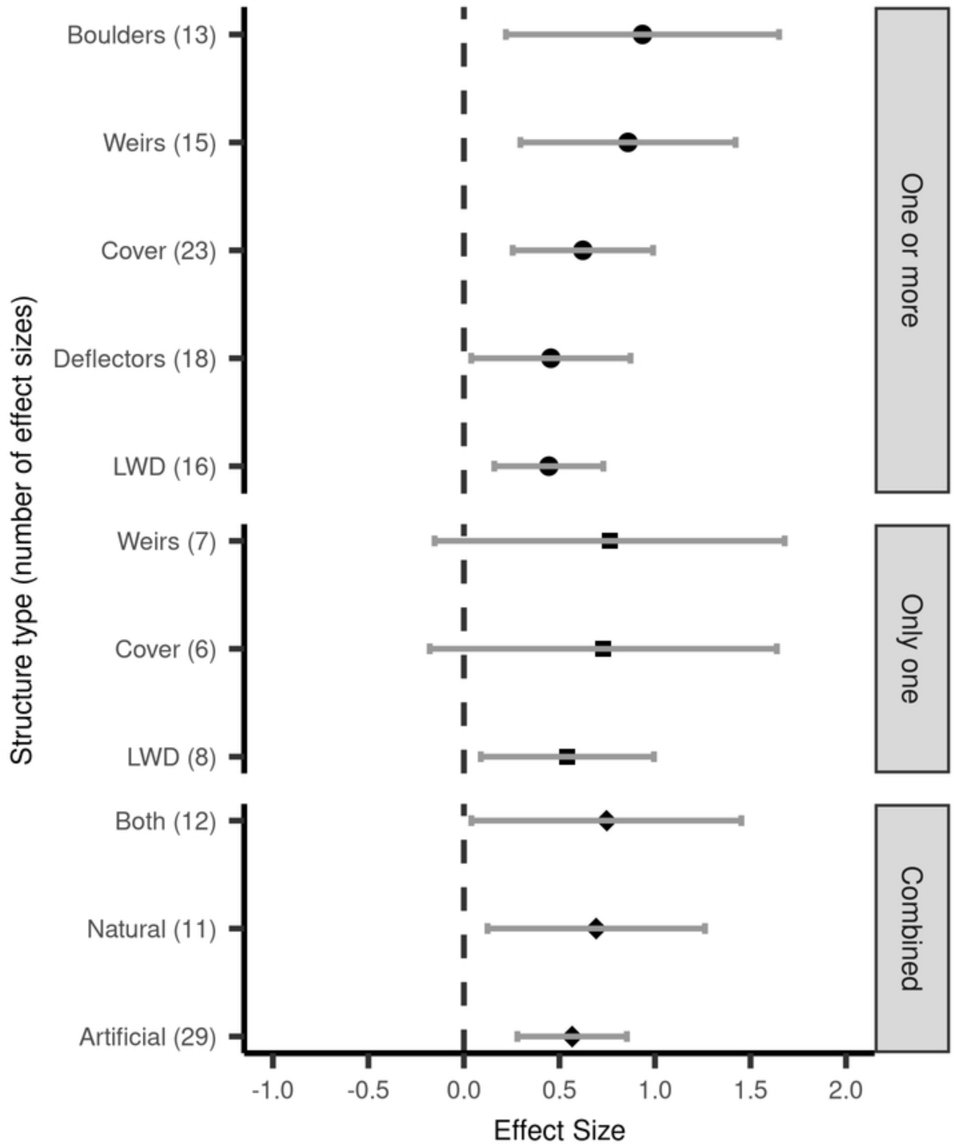

Fig. 7. Biomass effect sizes (mean +/- 95% CI) by structure type. See Fig. 6 caption for further explanation.

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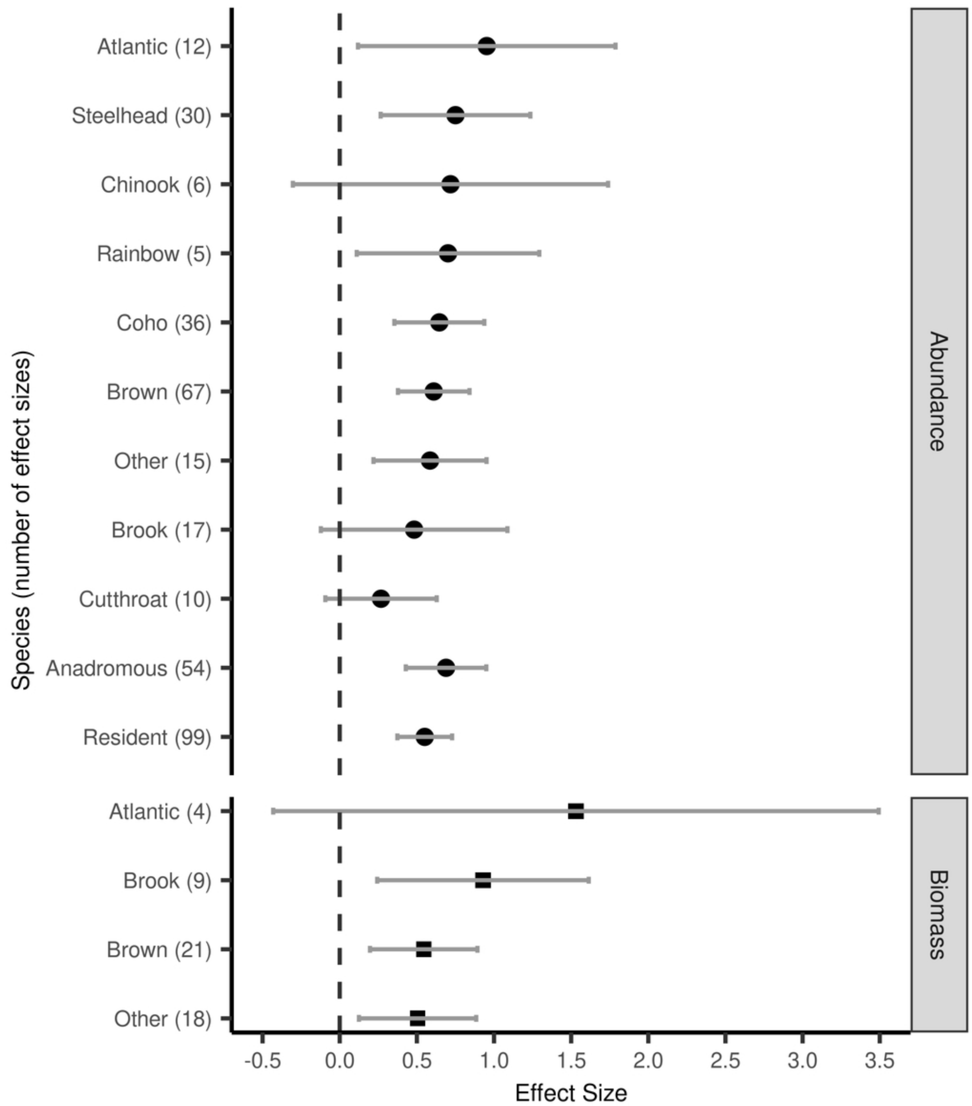

Fig. 8. Effect sizes (mean +/- 95% CI) by species for abundance or biomass measures. Other included unspecified or combined trout or salmon or species that had less than four studies.

88x101mm (300 x 300 DPI)

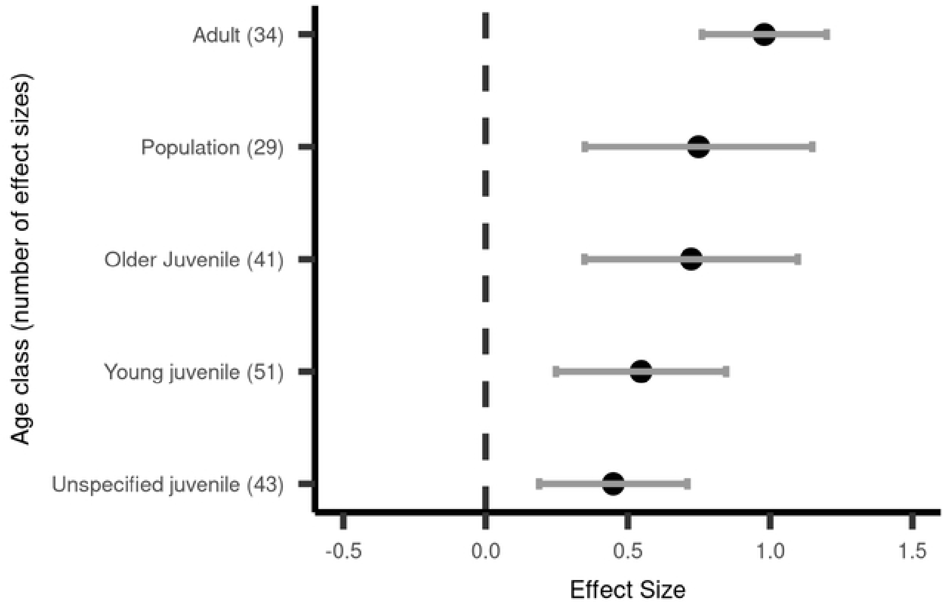

Fig. 9. Effect sizes (mean +/- 95% CI) by age classes for abundance measures. All species were combined.

57x38mm (300 x 300 DPI)

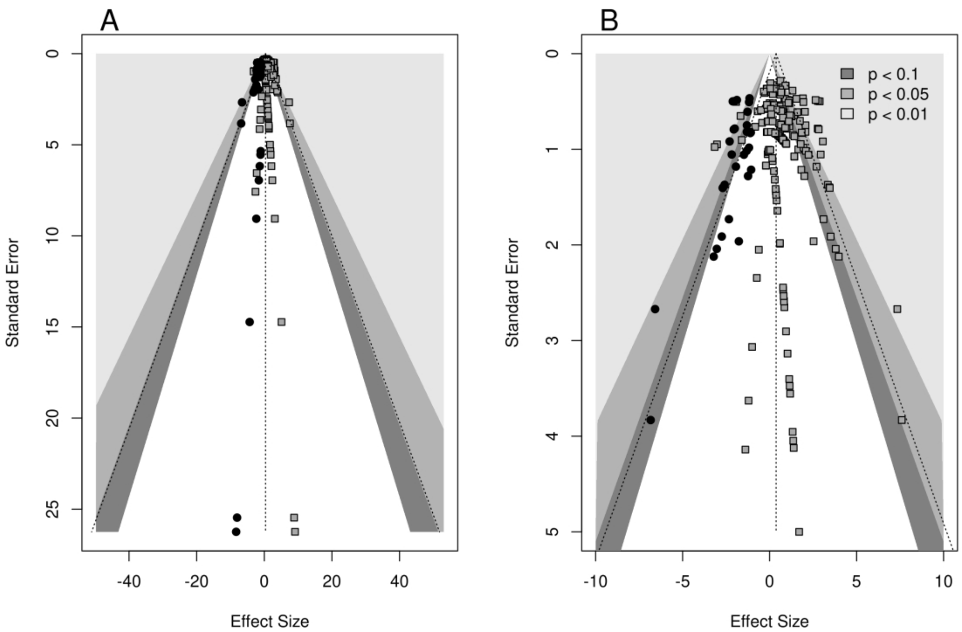

Fig. 10. Trim and fill funnel plot showing the abundance effect sizes against their standard errors. The original effect sizes in the meta-analysis are the grey squares whereas the black circles are the added points to correct for the bias. Significance levels are shown (p values) with significant studies lying in the p < 0.05 and p < 0.01 areas. The dotted line is the pooled estimate of the effect size and SE for all abundance measures. A: All effect sizes are shown. B: Only the top of the plot (y axis 0 to 5) is plotted to show more detail. The large variance in standard error of grey squares in A and asymmetry of studies (grey squares) around the mean (dotted vertical line) in B is indicative of bias.

254x177mm (300 x 300 DPI)

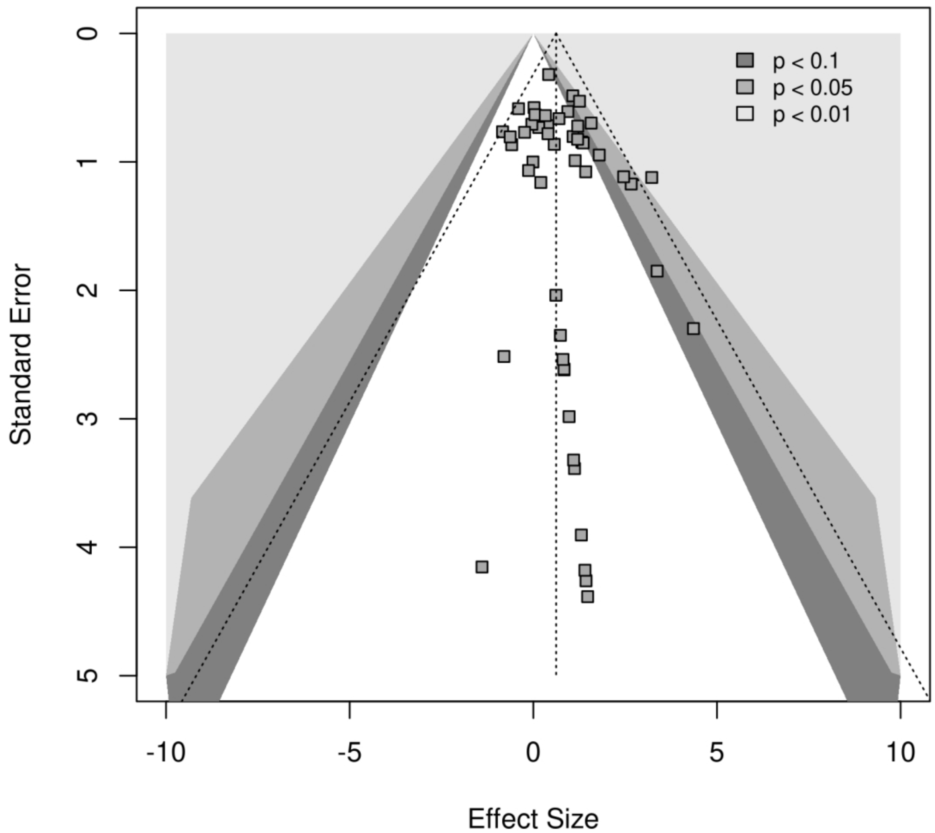

Fig. 11. Funnel plot for the biomass effect sizes against their standard error. Significance levels are shown (p values) with significant studies lying in the p < 0.05 and p < 0.01 areas. The dotted line is the pooled estimate of the effect size and SE for all biomass measures.

152x152mm (300 x 300 DPI)

Appendix 1

Table A1. Effect size of in-stream structures on individual salmonid species (significant effects are in bold). Effect sizes were only estimated for species with four or more studies in each structure category.

Structure type	Species (number of effect sizes)	Effect size (95% CI)	P-value
	Abundance measures		
Weirs	Brown trout (23)	0.221 (-0.311, 0.752)	0.399
	Brook trout (6)	0.536 (-0.400, 1.471)	0.201
	Coho salmon (6)	**0.855 (0.113, 1.598)**	**0.032**
	Cutthroat trout (4)	**0.742 (0.229, 1.255)**	**0.019**
	Steelhead trout (11)	0.642 (-0.214, 1.499)	0.126
Deflectors	**Brown trout (20)**	**0.927 (0.610, 1.245)**	**<0.001**
	Brook trout (10)	0.114 (-0.667, 0.895)	0.750
	Atlantic salmon (4)	0.730 (-1.473, 2.938)	0.369
Cover	**Brown trout (21)**	**0.700 (0.398, 1.002)**	**<0.001**
	Brook trout (11)	0.334 (-0.298, 0.966)	0.266
	Coho salmon (7)	0.241 (-0.209, 0.691)	0.238
	Steelhead trout (5)	0.851 (-1.857, 3.559)	0.432
Boulders	**Brown trout (27)**	**0.658 (0.331, 0.985)**	**<0.001**
	Coho salmon (5)	0.625 (-0.727, 1.978)	0.285
	Steelhead trout (6)	0.777 (-0.771, 2.324)	0.253
	Atlantic salmon (6)	0.900 (-0.622, 2.422)	0.189
LWD	**Brown trout (20)**	**0.632 (0.302, 0.962)**	**<0.001**
	Coho salmon (24)	**0.652 (0.245, 1.058)**	**0.003**
	Cutthroat trout (7)	0.186 (-0.157, 0.529)	0.232

	Steelhead trout (18)	0.498 (-0.166, 1.162)	0.132
Artificial	**Brown trout (26)**	**0.567 (0.098, 1.036)**	**0.020**
	Brook trout (11)	0.507 (-0.214, 1.227)	0.148
	Coho salmon (9)	**0.513 (0.024, 1.002)**	**0.042**
	Steelhead trout (8)	0.902 (-0.023, 1.827)	0.054
	Atlantic salmon (7)	0.970 (-0.114, 2.053)	0.071
Natural	**Brown trout (18)**	**0.705 (0.210, 1.201)**	**0.008**
	Brook trout (4)	0.754 (-1.386, 2.894)	0.344
	Coho salmon (23)	**0.626 (0.248, 1.005)**	**0.002**
	Cutthroat trout (4)	0.195 (-0.477, 0.866)	0.424
	Steelhead trout (15)	**0.761 (0.066, 1.456)**	**0.034**
Both	**Brown trout (23)**	**0.623 (0.293, 0.954)**	**<0.001**
	Coho salmon (4)	1.177 (-0.887, 3.241)	0.167
	Cutthroat trout (5)	0.756 (-0.309, 1.821)	0.120
	Steelhead trout (7)	0.418 (-0.948, 1.784)	0.482
		Biomass measures	
Weirs	**Brown trout (4)**	**1.369 (0.631, 2.109)**	**0.010**
Deflectors	**Brown trout (6)**	**0.725 (0.037, 1.412)**	**0.042**
	Brook trout (4)	0.371 (-1.907, 2.649)	0.640
Cover	**Brown trout (11)**	**0.734 (0.199, 1.269)**	**0.001**
	Brook trout (6)	0.305 (-0.925, 1.536)	0.552
LWD	Brown trout (9)	0.260 (-0.288, 0.807)	0.306
Artificial	**Brown trout (10)**	**0.818 (0.247 1.389)**	**0.010**
	Brook trout (6)	0.520 (-0.597, 1.637)	0.285
Natural	Brown trout (6)	0.312 (-0.478, 1.013)	0.356
Both	Brown trout (5)	0.099 (-0.351, 0.550)	0.574