Do Instream Structures Enhance Salmonid Abundance? A Meta-Analysis

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Abstract: Despite the widespread use of stream restoration structures to improve fish habitat, few quantitative studies have evaluated their effectiveness. This study uses a meta-analysis approach to test the effectiveness of five types of instream restoration structures (weirs, deflectors, cover structures, boulder placement and large woody debris) on both salmonid abundance and physical habitat characteristics. Compilation of data from 211 stream restoration projects showed a significant increase in pool area, average depth, large woody debris and percent cover as well as a decrease in riffle area following the installation of instream structures. There was also a significant increase in salmonid density (mean effect size of 0.51, or 167%) and biomass (mean effect size of 0.48, or 162%) following the installation of structures. Large differences were observed between species, with rainbow trout showing the largest increases in density and biomass. This compilation highlights the potential of instream structures to create better habitat for and increase the abundance of salmonids, but the scarcity of long-term monitoring of the effectiveness of instream structures is problematic.

Key Words:

Hydraulic structure, river, enhancement, improvement, fish habitat

Introduction

It is widely acknowledged that humans are negatively affecting the aquatic systems on which our survival depends (Richter et al. 1997; Ricciardi and Rasmussen 1999; Lake et al. 2007). In response to this degradation, the number of stream restoration projects has grown exponentially since the 1980s (Kondolf and Micheli 1995; Bash and Ryan 2002) and spending on restoration in the United States alone exceeds U.S.$1 billion per year (Bernhardt et al. 2005; Roni et al. 2008). Despite over a century of restoration activity, many unanswered questions remain regarding the effectiveness of various restoration approaches, which is in part due to the lack of project monitoring, and inconsistent results from studies that have been monitored (Bernhardt et al. 2005).

A number of literature reviews conclude that salmonid abundance typically increases following restoration (Bayley 2002; Roni et al. 2002; 2008), even if some case studies were not successful (e.g. Johnson et al. 2005; Rosi-Marshall et al. 2006; Klein et al. 2007). However, traditional literature reviews, while qualitatively describing the results of many individual case studies, do not allow statistical testing of overall trends (Roberts et al. 2006). Meta-analysis overcomes this problem by allowing the formal combination of results from a large number of case studies (Gates 2002). In a recent meta-analysis of instream structures, Stewart et al. (2009) found only equivocal evidence of their effectiveness at increasing salmonid abundance and significant variability in success among projects. Their commendable use of strict inclusion criteria required that all projects include some inherent replication or pseudoreplication, which resulted in only 17 studies and 38 data points in their analysis. Their small sample size prevented a comparison between structure types or fish species and limits the conclusions that can be drawn from this study.

Instream structures, such as weirs, deflectors, cover structures, boulder placements and large woody debris (LWD), are a common method of restoring habitat in rivers (Wesche 1985; Hey 1996; Roni et al. 2008). These structures act to alter flow and scour patterns, resulting in a more diversified physical habitat (Champoux et al. 2003; Thompson 2006). The installation of instream structures is typically carried out with the expectation that improved physical habitat will result in increases in the abundance and biomass of economically and culturally important salmonids (Roni et al. 2008). However, the number of projects that monitor physical habitat changes remains low; Bash and Ryan (2002) observed that twice as many restoration projects monitored salmonid populations compared to those that conducted physical habitat assessments. Furthermore, to the best of our knowledge, there has been no meta-analysis on the geomorphological impacts of these structures on key habitat characteristics such as pool area, depth or cover.

The objective of this study is to conduct a meta-analysis of the effectiveness of five types of instream restoration structures (weirs, deflectors, cover structures – which provide protection from overhead predators, boulder placement and LWD) using a sufficiently large number of case studies to test the impact of each type of structure on both salmonid abundance and physical habitat characteristics. Our extensive analysis, which includes a larger number of target species and types of restoration structure, compliments the more focussed study of Stewart et al. (2009).

Methods

Literature search

A literature search was conducted by performing key word searches on major biological and environmental science catalogues. ISI web of knowledge, Scopus and JSTOR were searched using keywords “trout OR salmo* AND river OR stream AND restor* OR enhance* OR improve* AND habitat” (where * represents a wildcard). The abstracts and references of articles that appeared relevant were examined. Searching through the reference lists of these articles turned up additional articles and reports. Only studies that provided salmonid density of at least a treated reach and a control reach were included in the meta-analysis. Time series studies, site comparisons and Before-After, Control-Intervention (BACI) studies were included. Projects needed to have installed one of more of the following: weirs, deflectors, cover structures, boulder placements, and LWD. A total of 51 reports met our criteria (see references with asterisk and Appendix A). Some reports were compilations of many different projects, thus providing a total of 211 stream projects for our analysis.

For each project, we recorded information about the restoration project (year of completion, type of structure installed, cost, length of the restored reach), project monitoring (number of years and type of monitoring - pre-and post restoration and/or treatment and control), and on the species and size classes of salmonids. When available, biomass data and physical habitat data were recorded for the pre- and post-restoration and/or the treatment and control sections. Physical habitat data consisted of the percent pool and riffle areas, mean stream width, number of pieces of LWD, percent cover and mean stream depth. It is possible that differences exist in how physical habitat data were measured among studies. However, in each report the overall change was used to assess the impact of restoration, which makes it unlikely that different definitions of LWD or cover between projects biased our overall results. For each species and size class of fish, the density (no.•m^{-2} or no.•m^{-1}) and biomass (g•m^{-2}) were recorded, or calculated, for the pre- and post-restoration and/or the treatment and control sections. No distinction was made between projects that collected density data via electro-fishing versus snorkelling. Although there is evidence that each method of estimating fish abundance has limitations (Peterson et al. 2004), the method used was consistent within each project and should not bias our results.

Data analysis

Effect size (L) was calculated for each study using the log response ratio

$$L = \ln(x_{tr} / x_c) \qquad (1)$$

where x_{tr} is the treatment mean and x_c the control mean (Hedges et al. 1999). The log response ratio was chosen because it measures the proportional change of important ecological variables caused by the treatment (Janetski et al. 2009). We did not use Cohen's D effect size (Stewart et al. 2009), because it requires a measure of the standard deviation of the response, which is not available for many single-site restoration projects. For BACI data the change in the treated reach served as the treatment value and the change in the reference reach served as the control. When BACI data were unavailable, the mean difference was used for the control and treatment sites, or for before and after restoration.

Data were available for 8 species of salmonids: brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), rainbow and steelhead trout (*Oncorhynchus mykiss*), cutthroat trout (*Oncorhynchus clarki*), Coho salmon (*Oncorhynchus kisutch*), Atlantic salmon (*Salmo salar*), Chinook salmon (*Oncorhynchus tshawytscha*) and arctic grayling (*Thymallus arcticus*). However, fewer than 10 studies monitored densities of Chinook salmon or arctic grayling, so these were not included in the comparison of individual species. Because steelhead trout are anadromous, whereas rainbow trout remain in fresh water throughout their lives, these two forms were analysed separately.

Three size classes of salmonids were created based on the most common size classification used in the analysed reports: (1) <10cm in length, which included fish aged 0+ and those classified as fry; (2) 10-15 cm in length, which included fish aged 1+ and those classified as parr; and (3) >15cm, which included age 2+ and 3+ fish and all fish classified as smolts or adults.

Effect size was calculated for total salmonid density in all cases, and for each of the following variables when available: total salmonid biomass, pool area (%), riffle area (%), width, depth, cover (%), and the number of pieces of LWD (pieces per 100m). For each project the density effect size was also calculated separately for each species, size class and year of monitoring. In order to assess overall project effectiveness, data for the last monitored year were used, to prevent projects with many years of monitoring from being over represented.

One-sample t-tests were used to determine if the mean effect sizes were significantly different than 0 at $\alpha=0.05$. ANOVAs were used to test whether there were significant differences ($\alpha=0.05$) between changes in density based on fish species, fish size class, the use of one structure type or multiple structure types, project age and publication type. Multiple regression analysis was used to determine the effect of changes in physical habitat factors on changes in salmonid density. Differences among structure types, on both biotic and abiotic variables, were also investigated through ANOVAs: these tests only included projects that used a single structure type.

Results

Physical effects

Fifty-three percent of studies installed only one type of structure, 28% used a combination of two structures, 13% combined three structures, 1% combined all 5 structures and 4% did not specify the type of structure(s) installed. The most common instream structures used were cover structures (88), followed by deflectors (87), weirs (69), LWD (46), and boulder placements (41). In 113 projects (54%), at least one physical habitat characteristic was monitored in addition to salmonid density and 78 (37%) projects reported biomass data as well as density data.

The installation of instream structures had significant effects on the physical habitat characteristics of the streams. Overall, there was a significant increase in pool area (mean effect size = 0.65; T_{72} = 5.56, P < 0.0001; Fig. 1a), a corresponding decrease in riffle area (mean effect size = -0.52; T_{38} = -4.87, P < 0.0001), an increase in the number of pieces of LWD in the river (mean effect size = 0.73; T_{14} = 3.21, P =0.006; Fig. 1b) (LWD projects were not included in the analysis of the overall LWD effect size), an increase in channel depth (mean effect size = 0.29; T_{37} = 2.93, P = 0.006; Fig. 1c), and an increase in percent cover (mean effect size = 1.14; T_{25} = 4.67, P < 0.0001; Fig. 1d).

Fig.1

However, the presence of instream structures had no significant effect on stream width (mean effect size = -0.01; T_{75} = -0.11, P = 0.91).

Projects with multiple structures increased pool area more than projects with only one type of structure (ANOVA, $F_{[1,73]}$= 38.5, P < 0.0001; Fig. 1a). For all other physical variables, however, there were no significant differences between the effect sizes for projects with multiple and single structures (ANOVA, all p-values > 0.08).

To investigate whether the five structure types had different effects on the physical habitat of streams, we compared the effect sizes for only single-structure projects (i.e. the light grey bars in Fig. 1). Effect size did not differ significantly between structure types for any of the six abiotic variables (ANOVA, all p values > 0.4; Fig.1). Fig. 1 also illustrates the mean effect size with 95% confidence intervals for all structure types, regardless of whether they were used alone or in combination (dark grey bars).

Effects on salmonids

Overall, average salmonid density and biomass increased following instream structure restoration, with mean effect sizes of 0.51 (T_{210} = 6.86, P < 0.0001) and 0.48 (T_{77} = 5.85, P < 0.0001) respectively (Fig. 2a and b). However, 56 projects (27%) showed a decrease in density following restoration and 10 showed a decrease in biomass (13% of those that monitored biomass). There was no significant difference between density or biomass effect size for projects that installed only one type of structure compared to those that installed multiple structure types (ANOVA, $F_{[1,199]}$ = 2.34, P = 0.128 and $F_{[1,32]}$ = 2.73, P = 0.11), nor was there a significant difference in density or biomass effect among structure types (ANOVA, $F_{[4,108]}$ = 0.64, P = 0.63 and $F_{[4,17]}$ = 1.10, P = 0.39 respectively).

Fig. 2

The density effect size varied significantly between species of salmonid (ANOVA, $F_{[6,327]} = 5.20$, $P < 0.0001$) (Fig.3). Based on a Tukey-Kramer post-hoc test, the effect size was largest for rainbow trout (1.48, n = 11), and smallest for steelhead trout (0.15, n = 50; Fig. 3). Ninety-five percent confidence intervals indicate that all species except brook trout and steelhead trout responded positively to the restoration efforts. Size classes responded differently to restoration, with an increasing linear trend among the three salmonid size classes (ANOVA, $F_{[2,319]} = 2.93$, $P = 0.055$; Fig. 4). Fig. 3&4

Backward stepwise regression was used to investigate the relationship between change in the 6 abiotic variables (pool area, riffle area, width, LWD, depth and cover) and biotic variables (density and biomass). Depth effect size was the only significant predictor of density effect size, although the R^2 value was low (0.11, n = 38, $P = 0.037$; Fig. 5a). Similarly, pool area effect size was the only significant predictor of biomass effect size ($R^2 = 0.51$, n = 8, $P = 0.046$; Fig. 5b). Fig. 5

Monitoring programs

The number of projects monitored decreased with increasing project age: 86 projects were monitored 1-year post construction while fewer than five projects were monitored 10 years post construction (Fig. 6a). None of the projects were monitored for over 20 years and 45% of all projects were only monitored once. The results for projects over 5 years post construction were combined due to small sample sizes. There was a significant difference in salmonid density effect size based on project age (ANOVA, $F_{[4,188]} = 2.59$, $P = 0.04$). The mean density effect size was greatest in projects monitored 2 years after completion (Fig. 6b). Fig. 6

Project cost was only reported in 24% of studies (51 out of 211). The mean cost of a project, indexed to the dollar value in 2000, was USD $127 490 while the median cost was $36 295. The average cost per metre of restored river length was $34.85 with some projects spending less than $5 per metre of stream restored and others upwards of $100. There was no relationship between total project cost, or project cost per metre of stream restored, and change in salmonid density (n = 54, $P = 0.52$ and n = 49, $P = 0.74$ respectively). Out of the total of 211 analysed projects, 148 (70%) came from the grey literature. A comparison of results published in the primary literature and in the grey literature revealed a slightly larger mean effect size of instream structures on salmonid density in the primary literature (0.55 compared to 0.49), but this difference was not significant (ANOVA, $F_{[1,209]} = 0.06$, $P = 0.81$).

Discussion

Meta-analysis of a large number of restoration projects showed that 73% of projects resulted in increased local salmonid densities and 87% in increased biomass, with an average effect size of 0.51 (167%) and 0.48 (162%), respectively. These findings are in agreement with the qualitative findings of previous studies (e.g. Hunt 1988; Keeley et al. 1996; McCubbing and Ward 1997). The 27% of projects that showed a decrease in overall salmonid density and 13% of projects that recorded a decrease in biomass following restoration did so for a number of reasons. Poor study design (e.g. badly chosen reference reach, short monitoring program), unexpected physical changes (e.g. decreased depth, decreased spawning gravel) and unexpected events (e.g. 100 year flood, fish kill, settling pond blowout) were listed as potential reasons for decreased density (Olsen et al. 1984; Thorn and Anderson 2001; Johnson et al. 2005). Structural failure was reported for only 4 of 56 projects that showed reduced salmonid density (Linløkken 1997; Reeves et al. 1997), however that does not mean that more projects did not experience any structural problems, only that they were not reported in relation to the salmonid response to restoration. Increased fishing pressure in the restored reaches was occasionally considered the cause of poor study outcomes (Hunt 1988; Avery 2004), but was usually not measured. A number of studies reported that though overall salmonid density decreased, the density of large fish had increased and that the larger decrease in fish under 10cm was responsible for the overall trend (Avery 2004; Rosi-Marshall et al. 2006). This trend may explain why a lower proportion of studies failed to increase salmonid biomass compared to density. However, the majority of studies that showed decreased salmonid densities following restoration provide no reason for this outcome. The large variation in how salmonids responded to stream restoration is in agreement with previous observations (Roni et al. 2008; Stewart et al. 2009).

In contrast to our results, Stewart et al. (2009) concluded that the "widespread use of in-stream structures for restoration is not supported by the current scientific evidence base" (p. 939). Stewart et al. (2009) also conclude that instream structures are more effective on small streams (<8m in width), whereas our analysis showed no difference in density effect size between streams of different widths; in fact streams over 8m in width had a larger mean density increase following restoration than smaller streams (L=0.59, 95% C.I.= 0.28 – 0.90, n=56 compared to L=0.41, 95% C.I.=0.24 – 0.58, n=108). A re-analysis of Stewart et al.'s (2009) data using L (eq. 1) as the measure of effect size was conducted to reconcile these different findings. Note that we have removed from the dataset the four projects in which either engineered instream structures were not used or no measure of abundance was reported (Mesick 1995; Scruton et al. 1998; Wu et al. 2000; Wang et al. 2002). We have also corrected a few errors in their data set: the treatment and control sections were reversed in Binns (2004); the n value listed corresponded to fish counted rather than river reaches in Linløkken (1997); and not all data from Gargan et al. (2002) were used. The results of our reanalysis show a clear positive effect size of 1.1 for instream structures (T_{28}= 4.90, P<0.0001), markedly larger than the average effect size in this study (0.51).

It is difficult to distinguish between increased fish abundance due to increased recruitment, survival or growth and increases caused by immigration and redistribution within the reach (Gowan and Fausch 1996). In order to measure changes in population size, the spatial and temporal scale of the study must be fairly large (Stewart et al. 2009). Unfortunately, many studies that attempt to determine the effect of instream structures on salmonid abundance are of short duration and at the reach rather than watershed scale. We excluded studies that specifically measured habitat preference, but did include studies measuring changes in abundance at the reach scale or for only a year following restoration. It is likely, therefore, that some of the studies reporting an increase in salmonid density are due to redistribution of fish. However, as Gowan and Fausch (1996) point out, immigration to preferred habitat is likely to increase the watershed-wide trout population, since it implies an increase in stream habitat capacity.

As expected, the installation of instream structures resulted in significant changes to the physical stream habitat. An increase in pool area, volume or frequency is a typical goal in instream structure installation (Roni et al. 2008). Our analysis indicated that all types of instream structures have the potential to increase pool area in a stream. Cover, which is a key salmonid habitat variable (Lewis 1969), can obviously be improved by cover structures but also by weirs and deflectors (the increase for boulder structures was not significant). Surprisingly, none of the projects analysed in this study measured the change in cover following the installation of LWD structures, despite the fact they are often installed to increase cover (Cederholm et al. 1997). Increased mean channel depth is another common restoration goal; deflectors, cover structures and boulder placements were all found to significantly increase depth while weirs showed a non-significant increase in depth. These physical characteristics are closely linked: increased pool area implies deeper channels and more cover since deep water functions as shelter from predators (Lozarich and Quinn 1995).

We found no significant effect of structure type on the observed change in salmonid density. Other studies that have directly compared different structure types have obtained conflicting results. Some studies suggest that deflectors outperform other structure types (e.g. Ward and Slaney 1981; Hunt 1988), others that boulder placements improve salmonid densities more than deflectors or weirs (e.g. Olsen et al. 1984), and yet others have concluded that weirs are preferable (e.g. Van-Zyll-De-Jong et al. 1997). We found evidence that weirs tended to be installed in steeper sloped streams while deflectors and cover structures were more frequently implemented on shallower slopes (< 0.5%). There is unfortunately not enough evidence to determine whether failure is more likely for a given type of structure on streams of different slopes. As different structures target different aspects of habitat quality, the best structure for increasing salmonid densities will be the one that best ameliorates the physical habitat deficiencies in an individual stream. It is therefore difficult to provide general recommendations without thorough knowledge of the specific problem. Our results imply that stream restoration practitioners are adept at picking the correct restoration technique, to create the correct habitat for the particular stream, but no one approach will work for all streams.

Surprisingly, despite the clear effect of instream structures on both physical habitat variables (see Fig. 1) and salmonid density (see Fig. 2a), change in habitat variables are not good predictors of changes in salmonid density, which raises the question: “what causes changes in salmonid density?” In order to increase salmonid abundance the restoration work must increase habitat that is limiting the population (Rosenfeld and Hatfield 2006). Determining these bottlenecks requires careful study by trained restoration practitioners, and even then mistakes are made (Hicks and Reeves 1994). Furthermore if multiple factors are co-limiting then several habitat changes would be required to provide adequate salmonid habitat. As for structure type, habitat variables that contribute to increased salmonid density likely vary from project to project, making it very difficult to establish a causal relationship from a large database which includes rivers in diverse environments.

There were significant differences between individual species density responses to the addition of instream structures. There is some evidence that instream structures are more effective for resident than for anadromous fish (Hicks and Reeves 1994), presumably because resident fish are larger and spend more time in the stream. Our observation that the effect size was higher for rainbow trout than for steelhead was consistent with this finding, whereas the stronger response by juveniles of anadromous Atlantic salmon than by resident brook and brown trout was not. Because older juvenile Atlantic salmon prefer deeper habitats (Armstrong et al. 2003), our analysis suggests that deeper habitats may have been limiting densities in those streams chosen for restoration. Similarly, the biomass of brook and brown trout responded more strongly than density (Whiteway, unpublished data), suggesting that restoration projects were more beneficial for larger than smaller fish (see below).

The observation that larger salmonids respond most strongly to instream structures suggests that they provide habitat that is particularly suited to adult salmonids. Previous studies have similarly documented better responses of larger fish to instream structures (e.g. Hunt 1988; Gowan and Fausch 1996) and many studies specifically seek to increase legal (often over 15cm) size trout (Burgess 1985; Hunt 1988). Energy intake is predicted to be higher in deeper water, meaning that the larger a fish's energy requirement (a function of size), the deeper the required habitat (Rosenfeld and Taylor 2009). Smaller trout do not show a strong preference for pool habitat (Bisson et al. 1988), which is likely why density increases are lower for these size classes. The observation that changes in pool area and biomass were more strongly correlated than pool area and density also suggests that increased pool area results in preferable habitat for larger salmonids.

Instream structures are typically designed to last at least 20 years (Frissell and Nawa 1992) though different structures have varying rates of structural failure during this time (Roni et al. 2002). While there is a consensus that more long-term monitoring on the effect of instream structures is needed (Frissell and Nawa 1992; Kondolf and Micheli 1995; Roni et al. 2008), the duration of monitoring projects remains short, averaging only 3 years. There are significant problems with determining project effectiveness when monitoring is done for only 1 or 2 years post-restoration as it may take up to 5 years after restoration work is completed before the full effect on salmonids can be seen (Hunt 1976, Kondolf 1995). Surprisingly our results show that the mean density effect size is largest for projects that have been in place for 2 years, and that the projects that monitor for 5 years or longer show a significantly lower density increase. It is possible that this is the result of gradual failure of the structures, however very few projects reported on the stability of the evaluated structures, which prevented us from drawing any conclusions about structural failure rates over time. Kondolf and Micheli (1995) recommend at least 10 years of post-restoration monitoring to measure physical changes in the river channel, since low recurrence floods are likely to alter the channel and because geomorphological adjustments following the installation of instream structures may take some time. The length of monitoring should also be determined based on the size and dynamic nature of the channel since it takes longer for geomorphological adjustments to take place on large rivers.

The median cost of the projects in our analysis was $36 295, almost double the $20 000 median cost of over 6000 instream habitat improvement projects compiled by Bernhardt et al. (2005). Costs were lower for projects that were able to use volunteer labour or readily available construction material. Higher costs can be expected for projects on inaccessible river reaches and projects that require the use of heavy machinery. There is, however, no evidence to suggest that higher spending leads to higher project success, as measured by increased salmonid density.

There is often a concern that successful restoration projects are more likely to be reported in the primary literature than unsuccessful projects (Kondolf and Micheli 1995). While it is impossible to analyze projects that have not been reported in any literature, comparing results that were published in the grey literature with those published in the primary literature allowed us to discount this potential bias.

This meta-analysis suggests that stream restoration projects are generally successful at improving salmonid habitat, salmonid density and total salmonid biomass in streams. While it is recommended that the installation of instream structures be used primarily as a temporary tool while larger scale watershed changes are made (Roper et al. 1997), for example reforesting riparian zones to provide natural LWD, the success of these structures remains an important consideration.

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Figure captions

Fig. 1. Effect of different types of instream structures on the mean (+ 95% confidence interval) effect size ($L = \ln(x_{tr} / x_c)$) of a) pool area, b) pieces of LWD, c) stream depth and d) cover. Within the “all” bars, the black all bar represents the average effect for all structure types, the white bar for projects that utilized only one type of structure and the striped bar for projects that used 2 or more structure types. Within each structure type the dark grey bar represents the mean for all projects that used that structure (whether or not another type of structure was used) and the light grey represents the mean for projects that only used that type of structure.

Fig. 2. The effect of structure type on the mean effect size (+ 95% C.I.) of a) salmonid density and b) biomass. Within the “all” bars, the black all bar represents the average effect for all structure types, the white bar for projects that utilized only one type of structure and the striped bar for projects that used 2 or more structure types. Within each structure type the dark grey bar represents the mean for all projects that used that structure (whether or not another type of structure was used) and the light grey represents the mean for projects that only used that type of structure.

Fig. 3. The effect of instream structures on the mean density effect size (+ 95% C.I.) of different salmonid species. Similar letters indicate that the mean does not differ significantly between species.

Fig. 4. The effect of instream structures on the mean density effect size (+ 95% C.I.) for salmonids of different size (< 10cm, between 10 and 15 cm, and > 15cm).

Fig. 5. Linear regression of a) salmonid density effect size against depth effect size ($y=0.612x+0.341$, $r^2=0.112$) and b) salmonid biomass effect size against pool area effect size ($y=0.306x+0.202$, $r^2=0.510$).

Fig. 6. Project monitoring a) number of projects monitored in each year following restoration, separated into projects monitored only once (in dark grey) and those monitored more than once (in pale grey) and b) salmonid density mean effect size (+ 95% C.I.) of projects monitored at different ages.

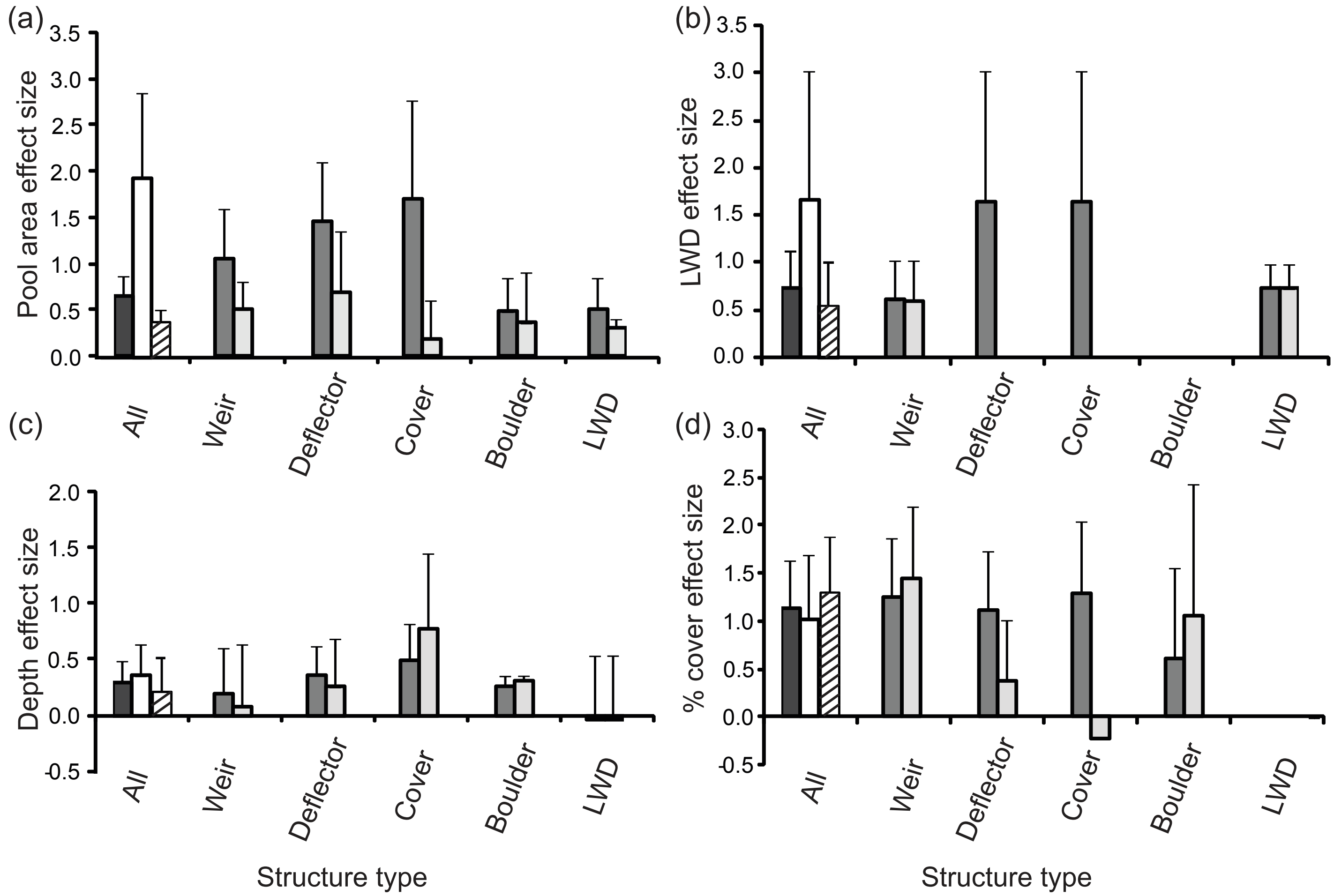
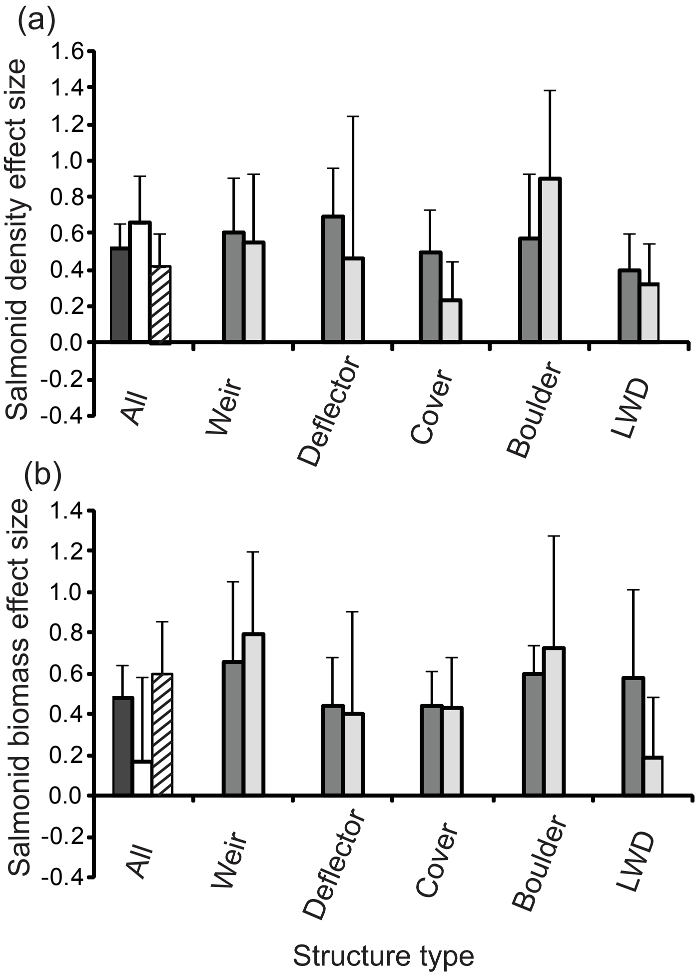


Figure 1. Whiteway et al.



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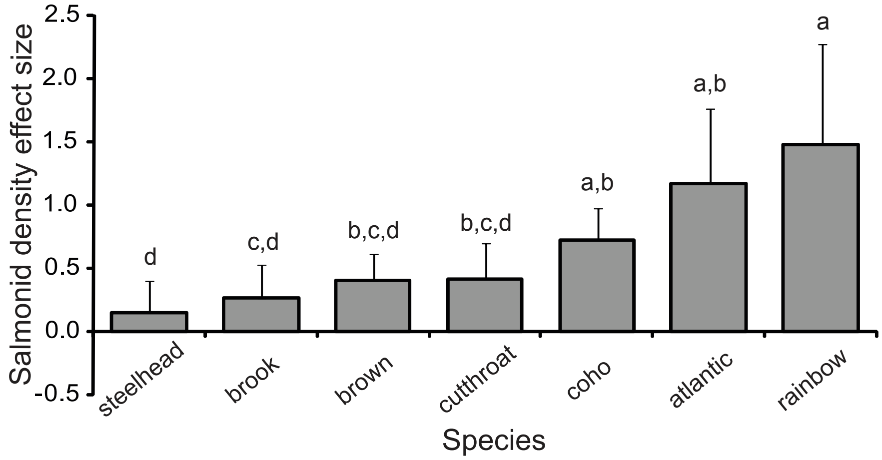

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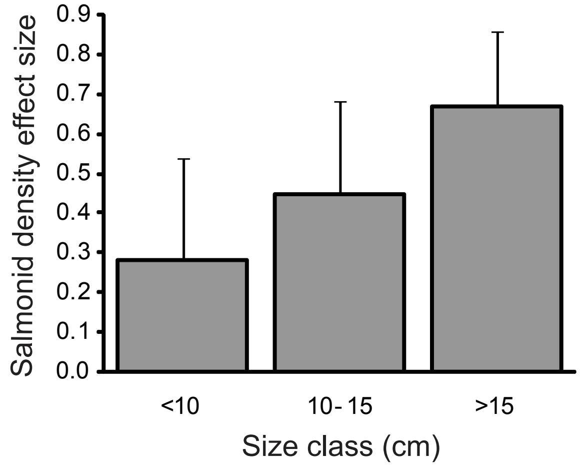

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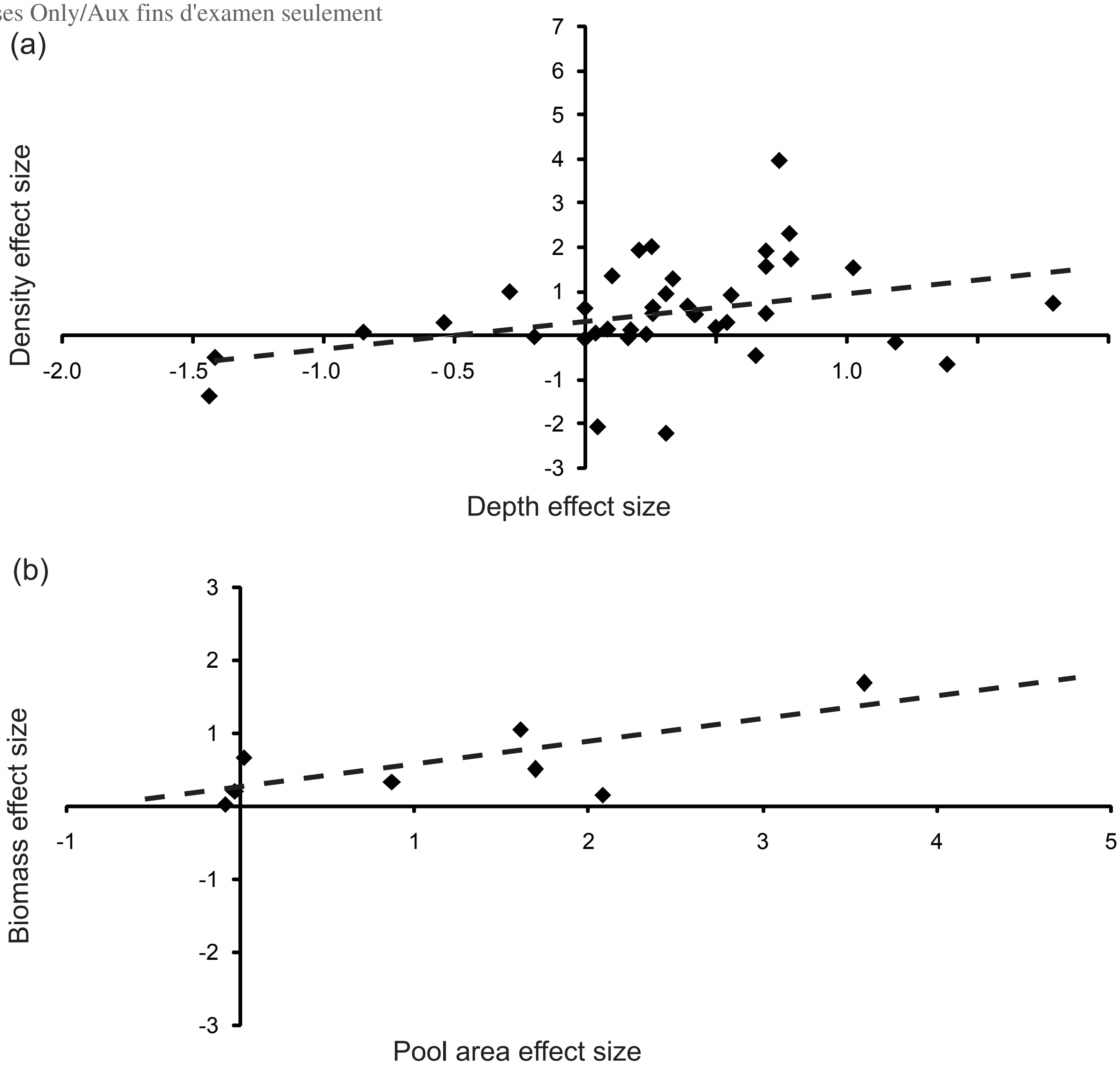

Figure 5. Whiteway et al.

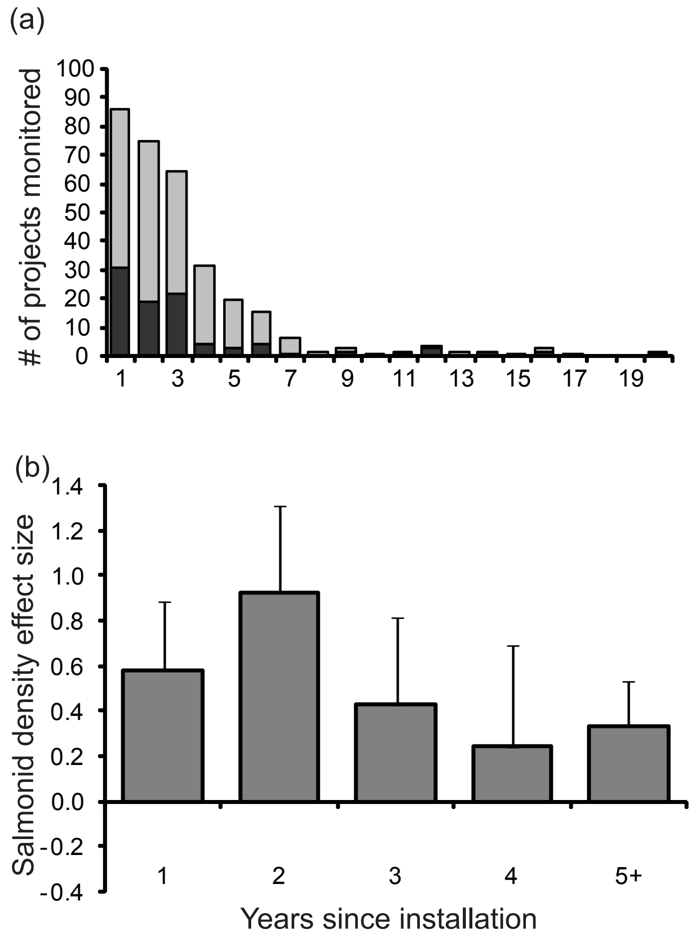

Figure 6. Whiteway et al.

Appendix A

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