Beyond redistribution: In-stream habitat restoration increases capacity for young-of-the-year salmon and trout in the Entiat River, WA

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Suggested running head – Fish habitat restoration and capacity

Abstract:

Post-restoration effectiveness studies of fish habitat typically focus on whether observed density/abundance of fish is increased as a result of restoration. A common design for such studies is the before-after-control-impact (BACI) design, but often pre-treatment data are limited. Furthermore, studies using this design rarely examine whether an increase in density is due to an actual increase in habitat capacity for fish, or whether fish have simply moved in from nearby untreated habitats with no net gain in overall abundance in the study area. We tested a framework and associated field methodology to determine whether installation of engineered log jams (ELJs) in habitat restoration leads to an increase in habitat capacity for fish. Field surveys emphasize observations from unrestored habitat within a reach treated with ELJs compared to unrestored habitat in untreated reaches and compared to surveys in restored habitat in treated reaches. We made post-restoration fish observations in five reaches treated with ELJs in the lower

river basin, and in three reaches in the upper river basin of the Entiat River, Washington, USA. Each basin segment had a different number and type of reference reach available. We observed high densities of young-of-the-year Chinook salmon and steelhead in pools created or restored by ELJs relative to unrestored habitat in the same reach. Comparison of fish density or occupancy in unrestored habitat in treated reaches vs. control reaches indicated that ELJ-restored pools added to overall habitat capacity and ELJs added to reaches in the upper basin were good analogs for natural structures, although results varied slightly between species and between the upper and lower basin segments.

In the United States, public agencies, academic researchers, and private consultants are engaged in activities related to the management and restoration of aquatic habitat to recover valuable fisheries (Katz et al. 2007, Roni et al. 2008), involving millions of dollars of public and other funds, annually (Bernhardt et al. 2005, O'Neal et al. 2016). In the Pacific Northwest region, for example, managers have thus far implemented restoration projects in several hundred kilometers of stream habitat to improve rearing conditions for salmonid fishes. Use of large wood or other material to build in-stream habitat structures is one of the most common techniques to protect and recover salmon populations across the region (Roni et al. 2015, Hillman et al. 2016). Large wood structures (engineered log jams; ELJs) are intended to promote processes that restore channel morphology (Davidson and Eaton 2013), increase pool availability (Roni et al. 2010), provide cover (Bond and Lake 2003a), and stimulate production of macroinvertebrate prey resources for fish (e.g., Hilderbrand et al. 1997; Kail et al. 2007). An important component of this restoration is the evaluation of its efficacy by demonstrating a biological response using the most robust metrics available (Block et al. 2001).

Habitat modeling tools such as Ecosystem Diagnosis and Treatment (EDT; Lichatowich et al. 1995, Lestelle and Mobrand 1996) and Physical Habitat Simulation System (PHABSIM; Bovee 1982) use fish-habitat correlations to guide restoration actions based on the assumption that specific habitat improvements will increase fish abundance and effectively increase fish populations. However, correlative analyses (e.g., multiple regression, ordination methods) may lead to overestimation of the effect that manipulation of habitat variables will have on fish abundance due to weak correlation

coefficients, despite statistical significance (Shirvell 1989, Lammert and Allan 1999), complex interactions between habitat variables (Beechie and Sibley 1997), variable scales of inference (Feist et al. 2010), and model uncertainty (McElhaney et al. 2010). Furthermore, field surveys can show underutilization of “good” habitat, possibly due to responses to biological factors (e.g., escapement/seeding, food availability, predation risk, or competition) not measured as part of physical habitat surveys (Polivka 2005). Thus, fish restoration effectiveness studies that rely upon the observation of fish density restored habitat, relative to unrestored habitat (e.g., Bond and Lake 2003b, Roni et al. 2008) often show small or no effects of restoration (Roni et al. 2008, Whiteway et al. 2010, Stranko et al. 2012, Hillman et al. 2016). When positive numerical responses are observed, the distributional patterns can vary greatly by species, year, and time period within a rearing season (Pess et al. 2012, Polivka et al. 2015).

Increased capacity to support fishes is one hypothesized outcome of restoration, given the relatively consistent association between total habitat area and occupancy (Hankin and Reeves 1988, Solazzi et al. 2000, Connor et al. 2001). Whereas it is relatively straightforward to compare fish density in treated vs. untreated habitat using a BACI (before-after-control-impact; Bernstein and Zalinsky 1983) design, the observation of more fish in restored habitat may indicate that individuals have simply redistributed themselves within a reach from poor habitat to restored habitat (Roni 2019). Demonstration that habitat capacity to support more fishes has actually increased would be a more robust indicator of restoration success. Furthermore, BACI designs attempt to show effects of restoration on fish abundance at the reach scale. This can be problematic due to issues with different levels of replication among reaches (Stewart-Oaten et al.

1986, Underwood 1992) and the fact that the effects of in-stream structures can be very localized (Polivka et al. *in press*).

Within a reach, there are several patch types (e.g., pools, riffles, glides), commonly referred to as habitat units (Cramer and Ackerman 2009). Studies at the patch scale address the replication of habitat units in restoration treatments and can identify the extent to which the effects of restoration are localized (Polivka et al. 2015; Polivka et al. *in press*). Comparison of untreated habitat patch in a treated reach with the restored habitat patches (usually ELJ pools), and then with untreated habitat patches in untreated reaches can indicate apparent capacity increases (Polivka et al. 2015). Prior to treatment, stream reaches in which restoration is planned should have the same average density per habitat as the nearby unrestored reaches (Figure 1a), assuming that treatment and control reaches have similar geomorphic, hydrological and ecological characteristics. This assumption could be affected by seeding levels from spawning in the previous year, but can be verified for unrestored reaches sufficiently near the reach to be treated (Gowan and Fausch 1996, Kennedy et al. 2014). To identify a capacity increase when pre-treatment data are not available, post-treatment fish density in restored habitats is examined for one of these hypothetical patterns:

1) *Redistribution of fish within the treated reach*. Fish density is higher in restored habitats relative to unrestored habitats in a treated reach, but density is lower in unrestored habitats in the treated reach relative to all habitats in untreated reaches (which are, of course, also unrestored). This is due to the movement of fish to the restored habitats from unrestored habitats within the reach, such that total average reach density has not changed (Figure 1b).

2) *Type 1 capacity increase*. Fish density is higher in restored habitats relative to unrestored habitats in a treated reach, and remains equal across unrestored habitats in both treated and untreated reaches. In this case, the fish that occur in restored habitat add to the density observed in unrestored habitat and therefore represent an apparent capacity increase. (Figure 1c; Polivka et al. 2015).

3) *Type 2 capacity increase*. Fish density is higher in restored habitats relative to unrestored habitats in the treated reach, and higher in unrestored habitats in treated reaches relative to untreated reaches. In this case, restoration has increased capacity of *all* habitats in the treated reach (Figure 1d) and a density increase at the reach scale might be observed.

Here, we apply the hypotheses in Figure 1 to post-treatment patterns of density in eight reaches of the same study system in Polivka et al. (2015), over multiple post-treatment years. From this analysis, we determined whether restoration increased juvenile rearing capacity and which type of capacity increase was observed.

[A]Methods

[C]*Study system*.—The Entiat River is a tributary of the Upper Columbia River in Washington State, USA (Figure 2). Its headwaters are in the Cascade Mountains and the river flows ~ 69 river kilometers to its confluence with the Columbia River (at river km 778; 49.6567 N, 120.2244 W). Species targeted by restoration efforts include anadromous Chinook salmon *Oncorhynchus tschawytscha* and steelhead trout *O. mykiss*. The freshwater phase is an important component of the life cycle of these two species and in-stream habitat restoration is designed to enhance sub-yearling growth and survival.

Although some non-salmonid species, as well as age 1 steelhead, are present in each of our study reaches, we focused our sampling and analysis on sub-yearling Chinook salmon and steelhead. The Chinook sub-yearlings in the Entiat are a mix of spring-run and summer-run and overlap during the study season. Genetic testing gives moderate to good probabilities of being able to distinguish runs (Degroseillier et al. 2018), but with some hybridization and spatial overlap, run type cannot be assumed. During the study years, total estimated spawning escapement of spring and summer runs (mean ± sd) in the previous season was 338 ± 142 and 258 ± 85 adults, respectively (Fraser et al. 2018). There were two years in which Spring Chinook escapement and one year in which Summer Chinook escapement were slightly below 1 SD of the respective means. We use escapement observations for consideration as mechanisms for abundance patterns of sub-yearlings reported below.

The restoration projects studied here were implemented in the Entiat watershed as part of a systematic program of restoration, combined with post-treatment monitoring (Bennett et al. 2016). Treatment reaches occurred in two of three major geomorphic basin segments in the Entiat River (Godaire et al. 2009). The lower basin segment (river km ~ 0-26) is a high gradient segment with large cobble substrate and relatively fast-flowing water. There were five treated reaches located between river km 2.7 and 6.4 (Table 1). Four untreated reaches occurred among them with the most upstream reach at river km 11.4. The upper basin segment (river km ~ 32-42) is situated just upstream of a glacial moraine and is of similar gradient and substrate as the lower basin. Upstream of river km 42 is above the Okanogan-Wenatchee National Forest boundary and was not part of the restoration. Treated reaches in the upper basin occur between river km 34.6-39.9 with

only one available control reach ("Upper Control," UC; Table 1). The UC reach had naturally occurring log structures as well as untreated habitat and thus offered the opportunity to compare the effects on fish density of man-made ELJs with that of natural ELJs.

[C]*Study reaches*—In the Entiat River, reaches treated with ELJs were selected by conservation agencies based on a combination of landowner willingness and recommendations following the completion of reach assessments for hydrologic conditions that were best suited for favorable outcomes of restoration (e.g., Sixta 2010). All reaches with ELJs were included in this study as treated reaches (Figure 2, Table 1). Control reaches were the nearest available untreated reaches with public or private permissible access. The availability of suitable control reaches in post-treatment effectiveness studies can influence the optimal sampling approach and sometimes, as in this study, treatment reaches outnumber reference reaches.

The restoration schedule of habitats differed between the lower and upper basin segments (Table 1). The five lower basin reaches were treated with ELJs in July and August 2014, whereas the three upper basin reaches were treated in September 2012. Because fish activity and density in the Entiat River tend to be highest in July compared to other summer months (Polivka et al. 2015), post-treatment surveys in both upper and lower basins took place in July (lower basin 2015-2016; upper basin 2013-2016). The upper-basin reach "Tyee" not surveyed in 2016 due to a change in accessibility.

[C]*Sampling design*—The general approach in reaches of both the upper and lower basin was to establish a starting point at the upstream end of each reach regardless if treated or untreated. We selected 15 unrestored habitat patches (standardized to $15m^2$: 3 x 5 m) by

first using randomly generated distance measurements following the method of Polivka et al. (2015). Random distances had a minimum of 10 m and a maximum distance of total reach length divided by 15 (e.g., Kiosk reach length of 805 m/15 = 54 m maximum longitudinal distance between habitat surveys). This ensured that all randomly selected unrestored habitat patches occurred within and likely throughout the entire length of the designated reach. Patches typically consisted of pools or slower-flowing glides, but generally not riffles. Surveys proceeded from the upstream end of the reach and each randomly generated distance was used to determine the distance downstream to the next survey patch from the previous. The survey took place on either the left or right river margin, selected via coin toss. Only habitat patches occurring on the stream margin were surveyed because mid-channel surveys have been found to provide little additional information in other salmonid streams (Beechie et al. 2005) and the river margin is where ELJs occurred at this site. The 15 m^2 area was surveyed by snorkeling for visual identification and enumeration of all fishes present in unrestored habitat. In treated reaches, ELJ-restored pools were encountered during the movement downstream between unrestored habitat surveys. As ELJs were encountered, restored habitat surveys were conducted in the entire pool and total pool area was measured. The survey crew would then continue downstream following the random distances to the next unrestored habitat survey. In the upper basin, the survey protocol was identical, with the exception that in the control reach (UC), unrestored habitat surveys were conducted along with surveys at the naturally occurring log structures.

Turbidity did not affect detection of fish during the study season; visibility for snorkeling was always >2–3 m, which permitted snorkelers to completely see the survey

area. Through snorkel observations during seine captures, Polivka et al. (2015) showed that not only was detectability very high, but few fish escaped capture. However, structures vary in their complexity and it is possible that fish could be completely obstructed from view, resulting in underestimation of density at restored habitats. For all snorkel surveys, we recorded depth, current velocity, dissolved oxygen, temperature and (for restored pools) pool area.

[C]*Data analysis*—There were three categories of fish density observations: 1) in pools created by or restored with ELJs, 2) in randomly selected unrestored habitat in treated reaches, and 3) in randomly selected unrestored habitat in untreated reaches. Comparison of datasets 1) vs. 2), i.e., post-treatment fish abundance in pools restored with structures compared with unrestored habitat in the same reach, was made using generalized linear models (GLM, Core Team, 2018) on fish counts (i.e., abundance). Because exploratory analysis resulted in residuals with high overdispersion, we used the GLM assumption of a negative binomial distribution with a log-link function. Separate analyses were performed for Chinook and steelhead, and for Lower and Upper Entiat basin segments. Pool area (log-transformed) was included in all models because of varying sizes of restored pools, and thus treats fish numbers between habitat types in terms of density. Data included all years in which post-treatment surveys took place (2 years of data from Lower Entiat reaches, 4 years from Upper Entiat reaches) and therefore survey year was added as a term in the GLMs. Because data from all reaches were combined, we used an interaction term, reach x habitat type (restored vs. unrestored), to determine whether there was a difference in restoration effects among the treated reaches. To identify which reaches

showed significant effects of restoration on fish density, we used post-hoc interaction means tests with the *P*-value adjusted for multiple comparisons (Holm 1979).

To compare datasets 2) with 3), i.e., randomly selected unrestored habitats in treated vs. untreated reaches, we had to modify our response variable to fish presence/absence because there were a substantial number of observations of zero fish, coupled with concentrations of fish in small pools. We had attempted to apply zero-inflated models, both mixture and two-part, but the high proportion of zeros made it difficult for the model to optimize (zeros were not predictable) and the wide range of fish counts meant that variances were very high relative to the means, both of which resulted in poor model fit and residuals (Zuur et al. 2009). We thus determined that analysis of presence vs. absence would simplify the model. We again assumed a negative binomial logit-link distribution in GLMs comparing the frequencies of fish presence. Separate analyses were performed for Chinook and steelhead, and for Lower and Upper Entiat basin segments. Survey year was included as a term in all GLMs. In the Lower Entiat analyses, a reach type designation (5 treated vs. 4 untreated reaches) was included as a GLM term. The interaction term of reach type x year was initially included in the model, but dropped as it was not significant for either Chinook or steelhead. In the Upper Entiat analyses, reach was included as a GLM term, comparing the three treated reaches to the single untreated reach (UC). The lack of multiple untreated reaches made testing for reach x year interactions impossible. Pool area was not included in these models because all unrestored habitats were of equal area (15 m^2).

Below, we report the statistical results of all GLM analyses described above, whether for abundance or for presence/absence; however, for simplicity and clarity, we

graphically display the data as mean and standard error of fish density at restored and unrestored habitat in each study reach. Means also represent density for all study years combined despite making statistical comparisons among years via a term in the GLMs. Differences among years are described separately.

[A]Results

[B]Lower basin: restored vs. unrestored habitat in treated reaches

At each of the five treated reaches, fish densities in habitats restored with ELJs were generally higher than fish densities in randomized surveys of unrestored habitats, although there was some variation among species (Figure 3). Chinook density at ELJ restored habitats were significantly greater than at unrestored habitats in all reaches (Table 2a), but steelhead abundance was significantly greater at only three reaches (HSC, KEY, and YAK). At the PUD reach, unrestored habitat had slightly, but not significantly (P=0.96), higher mean steelhead abundance than restored habitats. Chinook abundances were similar between the two survey years, but steelhead abundance was higher in 2015 than in 2016.

[B]Upper basin: restored vs. unrestored habitat in treated reaches

At each treated reach, Chinook density in ELJ restored habitats was significantly greater than in unrestored habitats (Figure 4, Table 2b), but steelhead density was only significantly greater at one of the three treated reaches (DW), but contributing to a marginally significant (p = 0.07) reach x habitat interaction term Table 2b. The same term for Chinook was non-significant, indicating that the difference in fish density between ELJ pools in the three treated reaches and natural log jam pools in UC was

indistinguishable. In general, Chinook were more abundant in 2013 and 2015 than in 2014 or 2016. Steelhead abundance was very low at all sites, regardless of habitat and especially in 2016.

[B]Lower basin: unrestored habitat in treated vs. untreated reaches

Chinook and steelhead were observed significantly more frequently at unrestored habitats in treated reaches than in untreated reaches (Figure 5, Table 3a). This, combined with results from restored habitat in treated reaches (Figure 3), demonstrates that the addition of structures appeared to result in a Type 2 capacity increase where density in all habitats in the treated reaches increased as a result of ELJ placement. In general, both fish species were observed more often in 2015 than in 2016 in all reaches, regardless of treatment status.

[B]Upper basin: unrestored habitat in treated vs. untreated reaches

Unrestored habitat in all upper basin study reaches had identical occupancy rates of both species averaged over all post-treatment years (Figure 6, Table 3b) with one exception: steelhead occupancy of unrestored habitat at the “Tyee” reach was lower than in the other reaches, though it is uncertain whether this was affected by lack of data for that reach in 2016. This ensures that the difference in Chinook density between restored and unrestored habitat at all three treated reaches was, on average, equal to the difference between density at pools naturally occurring structures and unrestored habitat at the “UC” reach (Figure 4), indicating that ELJs are good analogs for naturally occurring woody debris. This most likely indicates a Type 1 capacity increase, but with no accessible nearby reaches lacking natural and man-made log structures available, it is difficult to totally rule out a Type 2 increase. Chinook had higher occupancy of unrestored habitat in

2013 and 2015 than in 2014 or 2016, whereas steelhead occupancy was greater in 2013 than all other years.

[A]Discussion

Our data demonstrate the different outcomes possible in post-restoration observations of fish distribution and abundance as evaluations of restoration efficacy. Each of the treated study reaches clearly contained a higher density of fish in restored habitats than in unrestored habitats. Higher density, however, may simply be due to redistribution of fish from poorer habitats (Figure 1a), which may occur at spatial scales ranging from habitats within reaches to within and among whole reaches. Given their high degree of mobility and the dependence of spatial scale on their relationships with habitat characteristics (Feist et al. 2003, 2010), salmonids have high potential for redistribution in response to restoration. The inability to distinguish between redistribution and capacity increase is a possible explanation for the inconsistent response of fish density to restoration across different study systems, even with multiple treated and control reaches (Smokorowski and Pratt 2007, Whiteway et al. 2010). In many cases, small or no effects of restoration are apparent (Roni et al. 2008, Stranko et al. 2012, Hillman et al. 2016), and positive responses often vary greatly by species, year, and time period within a rearing season (Pess et al. 2012, Polivka et al. 2015).

Because our study design included surveys of unrestored habitat in treated reaches, we can distinguish between redistribution of fish and capacity increases in situations where pre-treatment data are not available. What happens in unrestored habitat gives information about how to interpret observed density increases in restored habitat.

This study design also requires suitable control reaches, which may not be available under some restoration programs or in some watersheds. Our reference reach for the upper basin segment was UC, which had natural structures present. On the one hand we couldn't be certain of a Type 1 vs. Type 2 capacity increase in the treated reaches in the upper basin because we were not able to sample reaches without man-made or natural structures. Nevertheless, the similarity in distribution among habitats between the treated reaches and UC suggests that the ELJs are, at least, good analogs for natural structures.

There are behavioral theories governing the settlement of habitat that might affect comparisons of untreated habitat among treated and untreated reaches. Ideal free distribution theory (Fretwell and Lucas 1970, Morris 1988) describes how habitat quality affects relative immigration of individuals into habitats at varying densities and immigration/recruitment has been shown in brown trout to contribute significantly to survival at later life stages (Dieterman and Hoxmeier 2011). Polivka (in press) showed density dependent immigration for sub-yearling Chinook suggesting applicability of these habitat settlement rules. Depending on whether the difference among habitats is qualitative or quantitative, at low total density, the density difference between higher-quality habitats and lower quality habitats may be larger than when total population density is higher. (Morris 1988). If this were the case here among restored and unrestored habitats, the relative magnitude of the difference between restored and unrestored habitat in both restored and unrestored reaches would be greater in years with lower density populations. We did not find a significant reach type x year interaction term, indicating that among-year differences in density did not contribute to the magnitude of density differences among habitats. It is nevertheless possible that use of tools from behavioral

ecology might clarify this question, given identification of density depending immigration into pools in Polivka (*in press*). Variation in escapement can result in different levels of "seeding" that might lead to such differences for Chinook. Although we combined years for statistical analysis, Chinook density only showed year effects in the upper basin, suggesting that the differences in Chinook escapement that exist among our study years (Fraser et al. 2018) were not biologically meaningful in terms of habitat distribution.

Despite apparent improvement of habitat capacity observed here and in Polivka et al. (2015), a given density increase associated with ELJ restoration can be difficult to observe at the reach scale. In a focused study at three of the five lower basin reaches studied here, Polivka et al. (*in press*) demonstrated that observed density increases at ELJ-restored pools are very localized. This does not necessarily negate the overall capacity increases observed here; it merely demonstrates that there can be a large difference between restored and unrestored habitat in a treated reach. Monitoring programs often emphasize a reach scale perspective (Bennett et al. 2006) based on earlier recommendations (Roni et al. 2013). From extensive surveys here and in Polivka et al. (2015), it is clear that detection of reach-scale capacity increase vs. basic density increase requires: 1) separate surveys of treated and untreated habitat *within* a treated reach and 2) availability of and access to physically and biologically similar untreated reaches. The first stipulation suggests a smaller spatial scale for monitoring studies than the reach scale, also indicated by the very localized effect of ELJs found in Polivka et al. (*in press*). The second stipulation means that the characteristics of control reaches should thus be a consideration both for restoration site selection and for post-restoration effectiveness

monitoring designs. Land ownership and local physical conditions can result in the situation we encountered in the upper basin. Statistical issues may also result from low replication of structures. This was particularly an issue at the Dillwater (DW) reach in which few structures were installed to begin with, some of which were above the water line during lower river flows characteristic of much of the summer growing season for juvenile salmonids in this study system.

When there are relatively few restored reaches, some form of a before-after-control-impact monitoring design is often appropriate (Roni et al. 2018a, 2018b). Our approach somewhat compensates for the cases when a BACI design is intractable, such as when pre-treatment data are sparse owing to the fact that site selection occurred only 1-2 years in advance of restoration. Thus, equal density in both control reaches and reaches planned for treatment, as indicated in Figure 1a, relies on assumption based on comparison of untreated habitat after treatments have been implemented. If pre-treatment data were available, our design would become a BACI design, with the improvement that there would be an extra set of data to compare untreated habitats between reaches. Nevertheless, our comparison of untreated habitat patches in treated reaches with untreated habitat in untreated reaches yields robust conclusions about the effects of restoration on fish density and capacity and is possibly less labor intensive than a BACI design, owing to the fact that it can be done without collection of pre-treatment data..

We have shown that juvenile salmonid habitat capacity increased, but our field survey approach is not designed to evaluate capacity in terms of a fitness correlate such as sub-yearling growth or survival through any subsequent life stage. Studies that are more mechanistic in nature are required to support observations of capacity increases

here and in related studies (Beechie et al. 2015). In this study system, mechanistic studies of movement (Polivka *in press*) and growth of individuals (Polivka et al. *in review*) partially support that occupancy of restored pools is associated with fitness-correlated benefits. Restoration in spawning streams is implemented to augment habitat for young-of-the-year, an important stage for the "stream-type" life history, in which considerable growth and development take place in streams (Groot and Margolis 1991). The capacity increase described here for the upper and lower geomorphic basin segments of the Entiat River indicate that restoration in this sub-basin is benefitting Chinook salmon and, to a less definite extent steelhead, early in their life history.

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Table 1**.** Lower and upper basin study reaches, their upstream reach location along the river, reach length, treatment type, and the number of wood structures (ELJ or natural) surveyed per reach. Unrestored habitat (N=15/reach) was also surveyed at each untreated and treated reach. Reach names were designated by restoration project sponsors and usually refer to a landmark or sponsoring entity.

Basin segment	River (km)	Reach length (m)	Reach name	Treatment (year)	Structures (#)
Lower Entiat R.					
	2.1	805	Kiosk (KIO)	Untreated	
	2.7	322	Keystone (KEY)	Treated (ELJ 2014)	6
	4.3	322	Yakama (YAK)	Treated (ELJ 2014)	8
	5.1	644	Asher/Milne (ASH)	Treated (ELJ 2014)	7
	5.6	483	PUD	Treated (ELJ 2014)	3
	6.4	322	Harrison S.C. (HSC)	Treated (ELJ 2014)	6
	7.2	644	Harrison ref. (HRN)	Untreated	
	10.3	644	Roaring Crk. (ROA)	Untreated	
	11.4	644	Hatchery (HAT)	Untreated	
Upper Entiat R.					
	32.8	483	Upper Control (UC)	Untreated (natural)	5
	34.6	483	Dillwater (DW)	Treated (ELJ 2012)	3
	36.2	805	Tyee (TYE)	Treated (ELJ 2012)	5
	39.9	1287	3D	Treated (ELJ 2012)	7

Table 2. Results of GLM analysis of Chinook and steelhead density in restored versus unrestored habitat in treated reaches in the A) lower and B) upper Entiat basin. Survey area was $\log_e$-transformed. The year term indicates whether significant differences exist among study years (2015-2016 in the lower basin; 2013-2016 in the upper basin). The interaction term, reach x habitat type (restored vs. unrestored), and post-hoc interaction means tests (see Table 1 for site name abbreviations) are used to identify differences in restoration effects among treated reaches.

Basin			Chinook		Steelhead	
segment	GLM variable	df	χ2	*P*-value	χ2	*P*-value
A) Lower Entiat R.						
	Area	1	9.01	0.0027	22.01	<0.0001
	Year	1	0.37	0.5415	57.68	<0.0001
	Habitat	1	66.03	<0.0001	45.58	<0.0001
	Reach	4	37.02	<0.0001	26.52	<0.0001
	Reach x Habitat	4	13.66	0.0085	13.68	0.0084
	Residual	196				
Reach x Habitat (restored vs. unrestored) means comparison						
	ASH		6.10	0.0271	4.87	0.0547
	HSC		19.24	<0.0001	10.81	0.0040
	KEY		32.97	<0.0001	27.64	<0.0001
	PUD		3.85	0.0498	0.00	0.9639
	YAK		7.22	0.0217	10.25	0.0041

B) Upper Entiat R.

Area	1	0.02	0.8951	0.82	0.3651
Year	3	10.74	0.0010	40.08	<0.0001
Habitat	1	19.41	<0.0001	6.89	0.0086
Reach	3	11.34	0.0100	6.87	0.0760
Reach x Habitat	3	2.30	0.5126	6.99	0.0723
Residual	283				
Reach x Habitat (restored vs. unrestored) means comparison					
3D		5.71	0.0338	3.87	0.1170
DW		5.27	0.0338	11.96	0.0022
TYE		11.61	0.0020	4.26	0.1170
UC		13.29	0.0011	4.16	0.1170

Table 3. Results of GLM analysis of Chinook and steelhead presence/absence in unrestored habitats of treated versus untreated reaches in the A) lower and B) upper Entiat basin. In the Lower Entiat analyses, a reach-level treatment type variable was included, comparing five treated to four untreated reaches. In the Upper Entiat analyses, reach itself was included, comparing unrestored habitats in three treated reaches to one untreated reach (UC). The year term indicates whether significant differences exist among study years (2015-2016 in the lower basin; 2013-2016 in the upper basin). Survey area was not included in GLMs because unrestored habitats were of equal area (15 m^2).

Basin	GLM		Chinook		Steelhead	
segment	variable	df	$\chi 2$	*P*-value	$\chi 2$	*P*-value
A) Lower Entiat R.						
	Treatment	1	12.79	0.0003	8.16	0.0043
	Year	1	6.89	0.0086	59.02	<0.0001
	Residual	15				
B) Upper Entiat R.						
	Reach	3	2.72	0.4365	10.01	0.0185
	Year	3	18.04	0.0004	36.35	<0.0001
	Residual	8				

Figure captions

Figure 1. Theoretical fish densities at restored habitats (“Structures”) and unrestored habitats (“Untreated”) within a treated reach, relative to unrestored habitats within untreated reaches. A) Pre-treatment fish density at unrestored habitats in treated reaches and in untreated reaches. Post-treatment observations can involve: B) re-distribution of fish in the treated reach from unrestored habitat to restored habitat, C) additional fish in restored habitat, but no decrease in density in unrestored habitat relative to unrestored habitat in all untreated reaches, or D) increased density in restored habitat, and unrestored habitat in the treated reach is also increased relative to unrestored habitat in all untreated reaches.

Figure 2. Locations of the treated reaches (with ELJs indicated by black diamonds) and untreated (control) reaches [natural log jams (Upper Control) or no wood at all (Hatchery, Roaring C., Harrison Ref., Kiosk)] in the upper and lower basin segment of the Entiat River. The upstream end of each control reach is indicated (white square). Additional details on reaches and treatments are given in Table 1.

Figure 3. Summarized observations of A) Chinook and B) steelhead juvenile density (mean ± 1 SE) in ELJ restored and unrestored habitats from treated reaches in the lower basin segment of the Entiat River. For display purposes, densities from 2015-2016 were averaged.

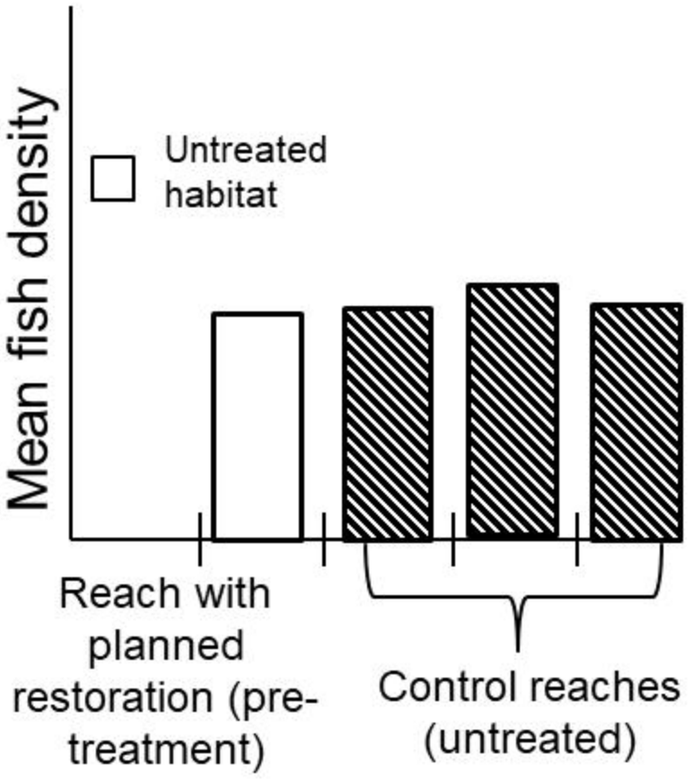
Figure 4. Summarized observations of A) Chinook and B) steelhead juvenile density (mean ± 1 SE) in ELJ restored, natural log jam (UC), and unrestored habitats from treated and natural reaches in the upper basin segment of the Entiat River (site names explained in Table 1). For display purposes, densities from 2013-2016 were averaged. See Table 2b for identification of differences among restored and unrestored habitats within and among reaches.

Figure 5. Proportion of unrestored habitats (mean ± 1 SE) with juvenile Chinook and steelhead presence in treated (ASH, HSC, KEY, PUD, YAK) and untreated (KIO, HRN, ROA, HAT) reaches in the lower basin segment of the Entiat River. For display purposes, proportion of habitat patches occupied from 2015-2016 were averaged. Total unrestored habitat occupancy across all reaches by both species was greater in 2015 than 2016 (Table 3a).

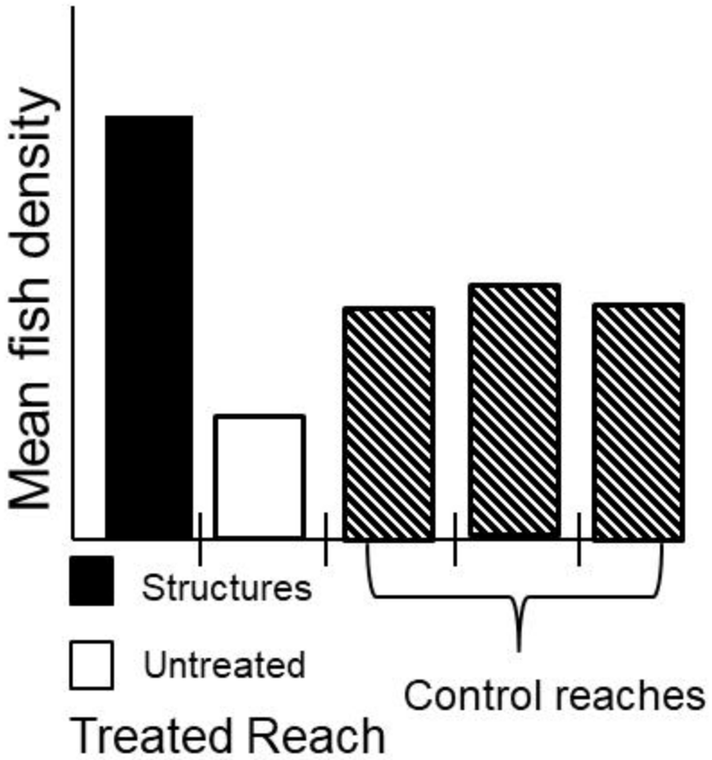
Figure 6. Proportion of unrestored habitats (mean ± 1 SE) with juvenile Chinook and steelhead presence in treated (ELJs: 3D, DW, TYE) and untreated (natural log jams: UC) reaches in the upper basin segment of the Entiat River. For display purposes, proportion of habitats occupied from 2013-2016 were average. Among year differences are indicated in Table 3b and described in the text.

Alternative hypotheses:

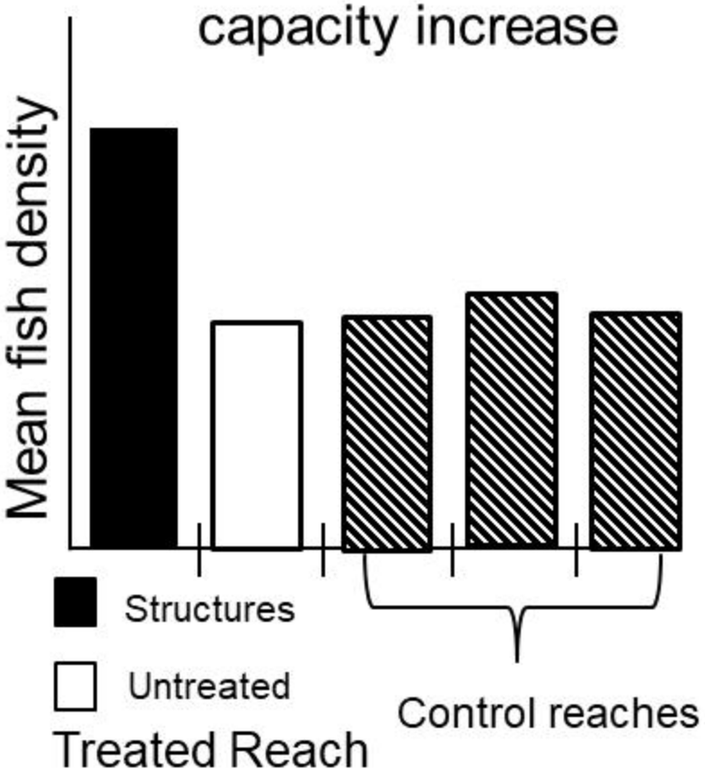
a) Prediction: Equal density, untreated habitat in planned treatment + control reaches.



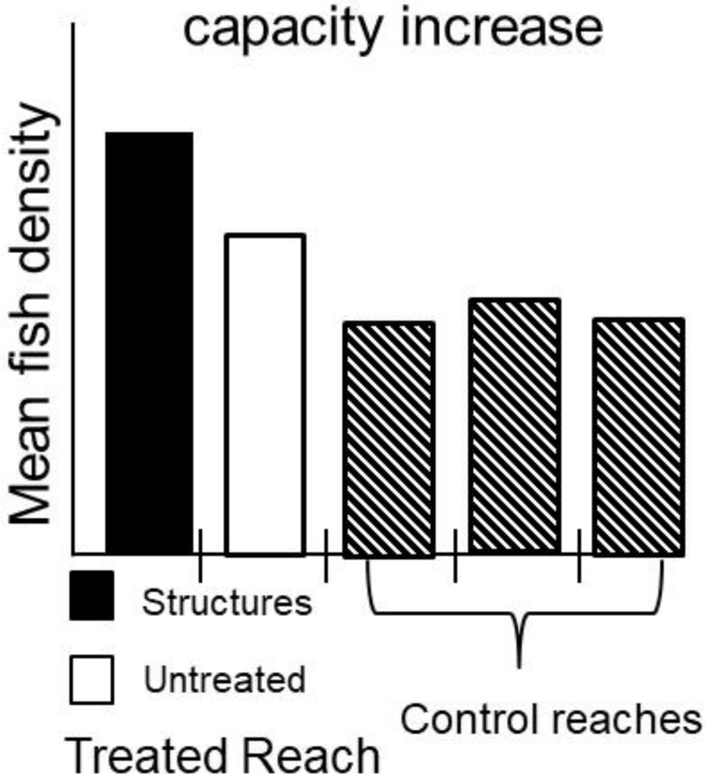
b) Depletion of poorer habitat within reach ("redistribution")

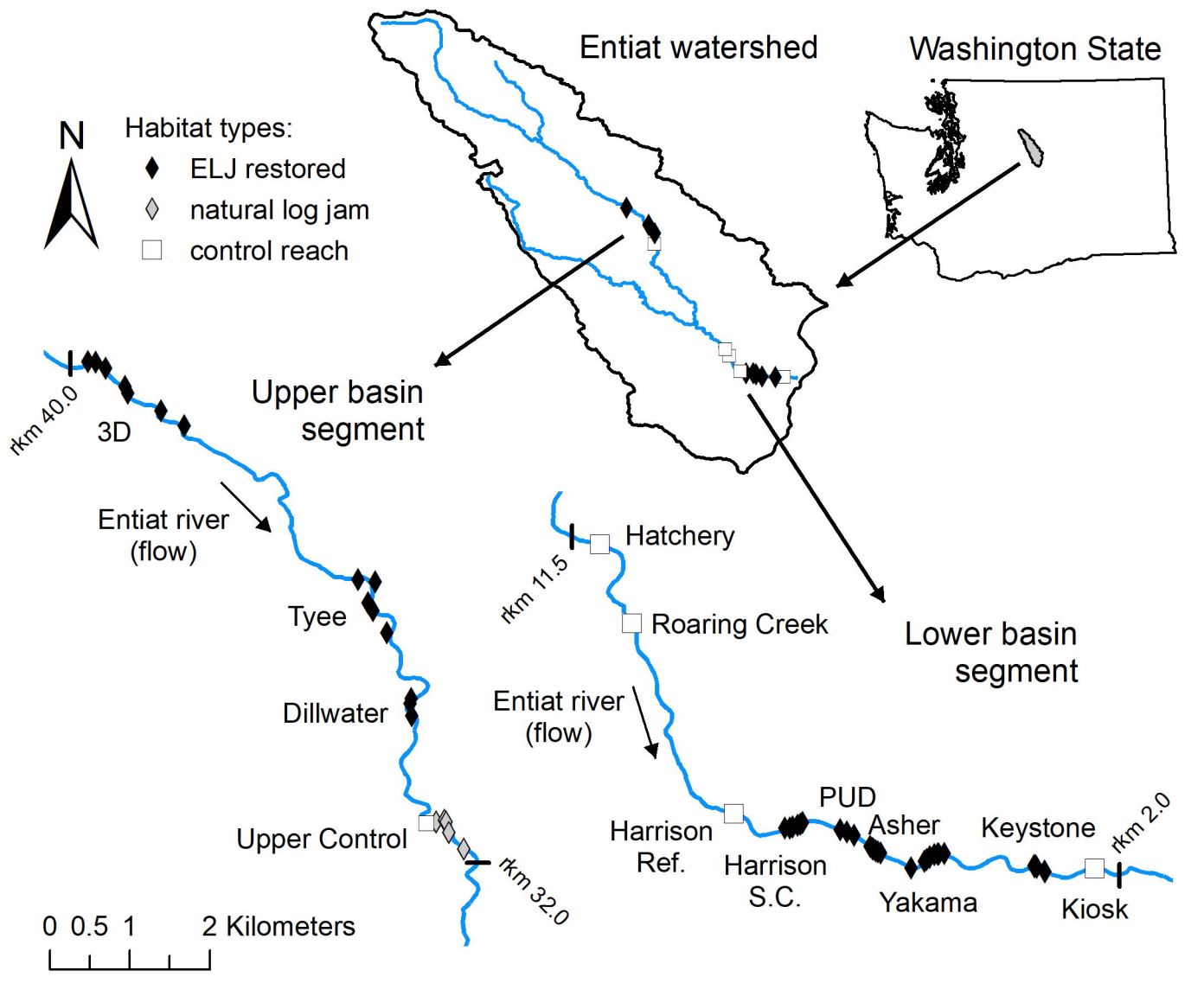


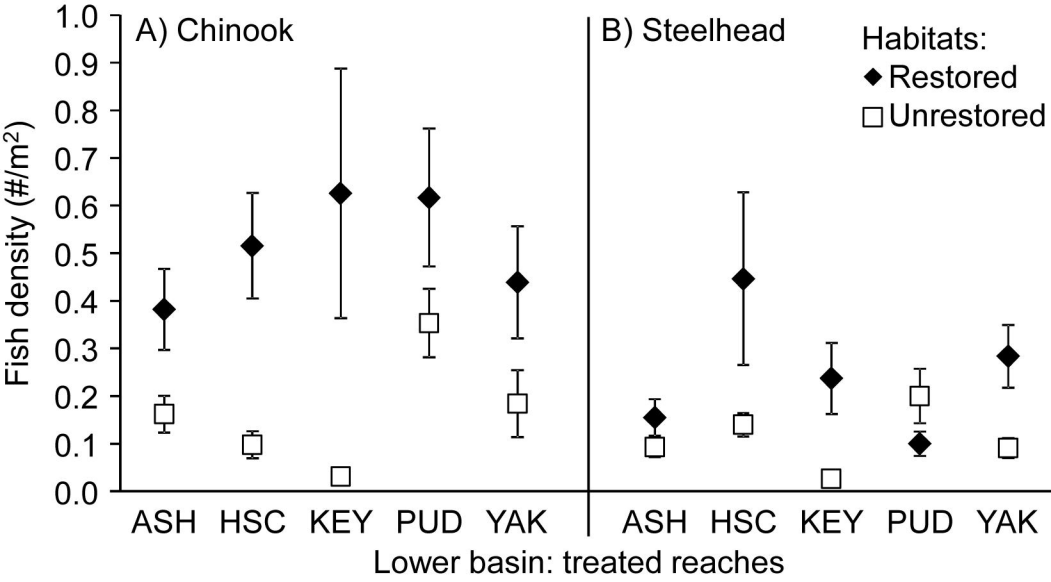
c) Higher density in restored habitats of treated reach: TYPE 1 capacity increase

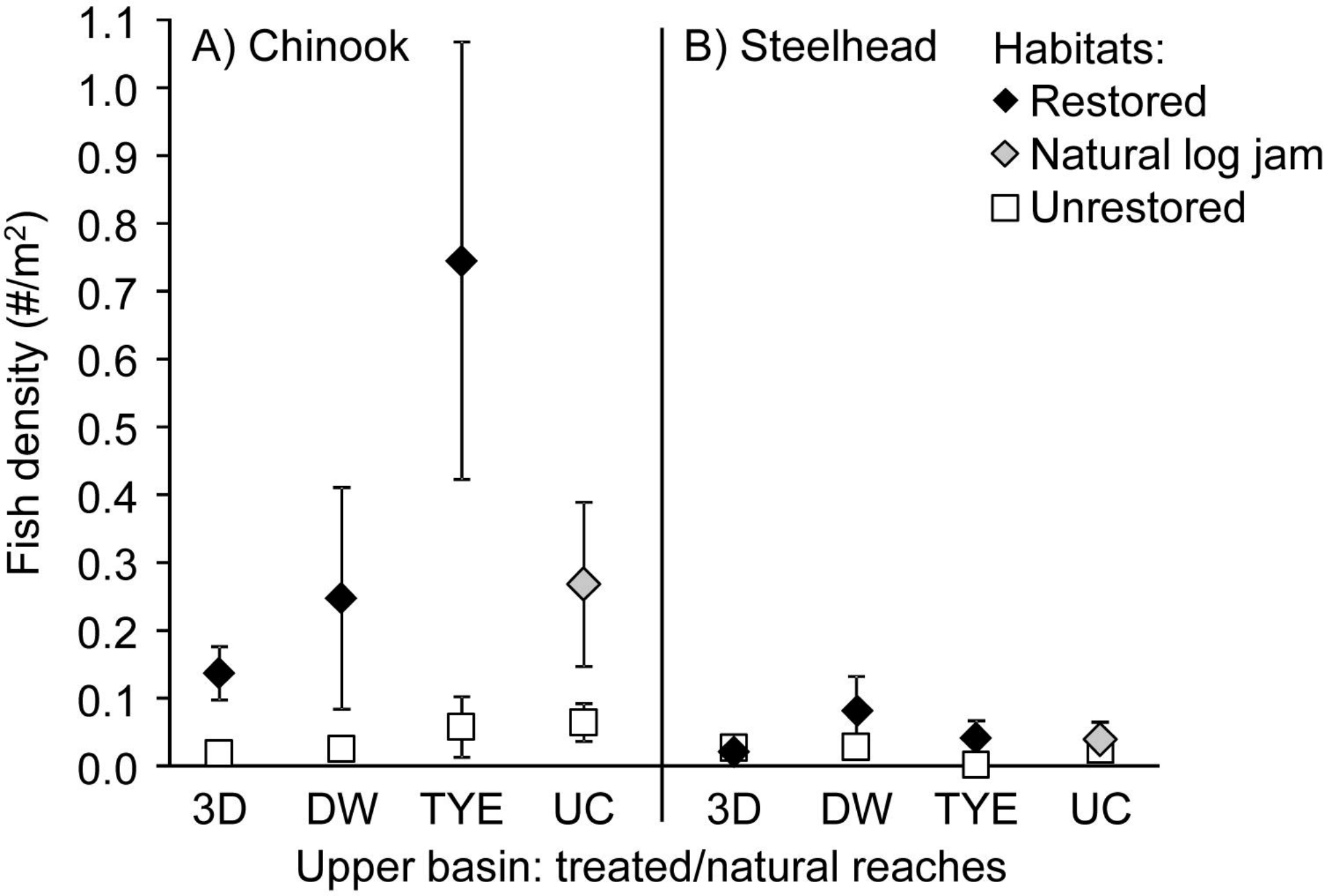


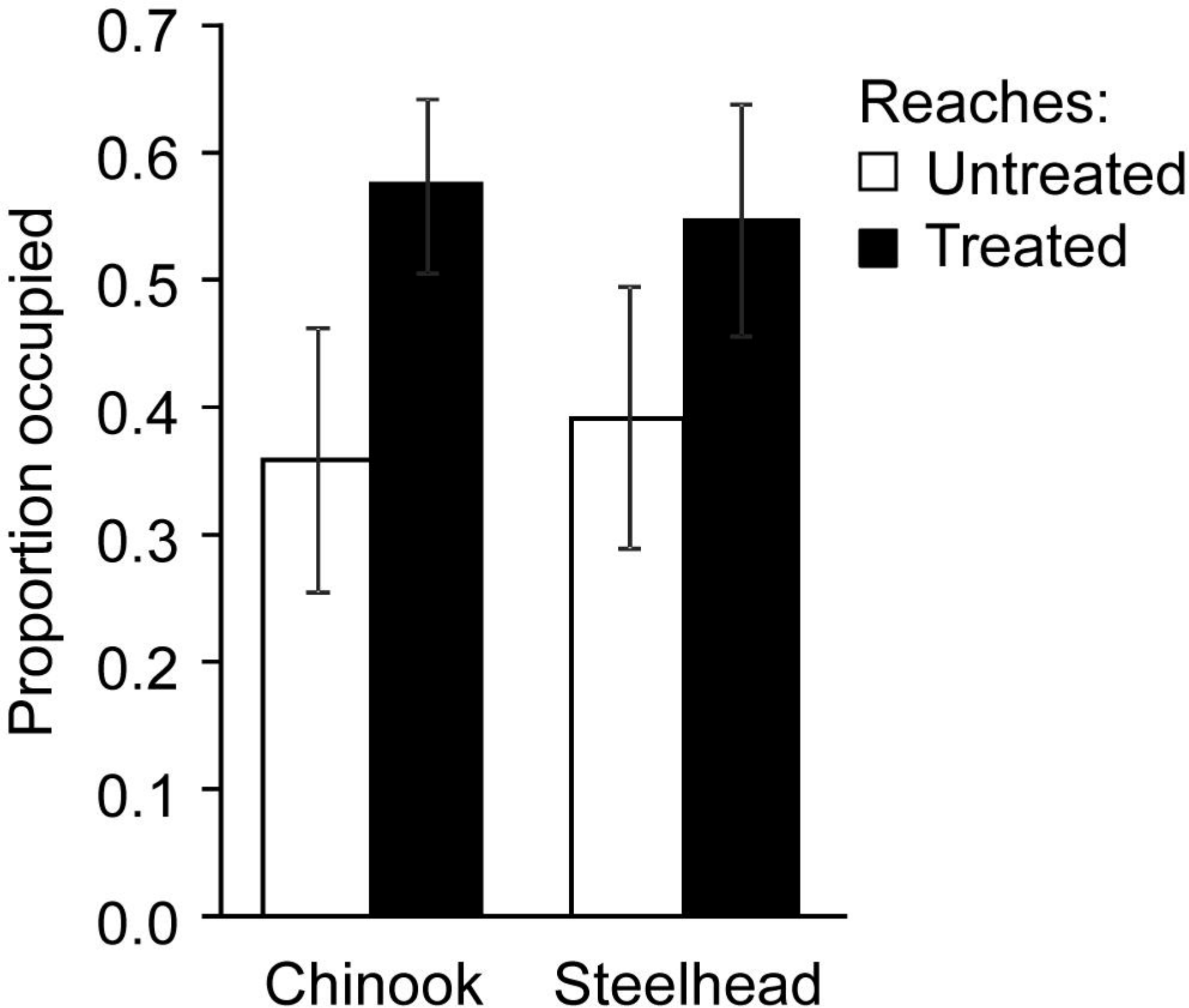
d) Higher density in all habitats of treated reach: TYPE 2 capacity increase

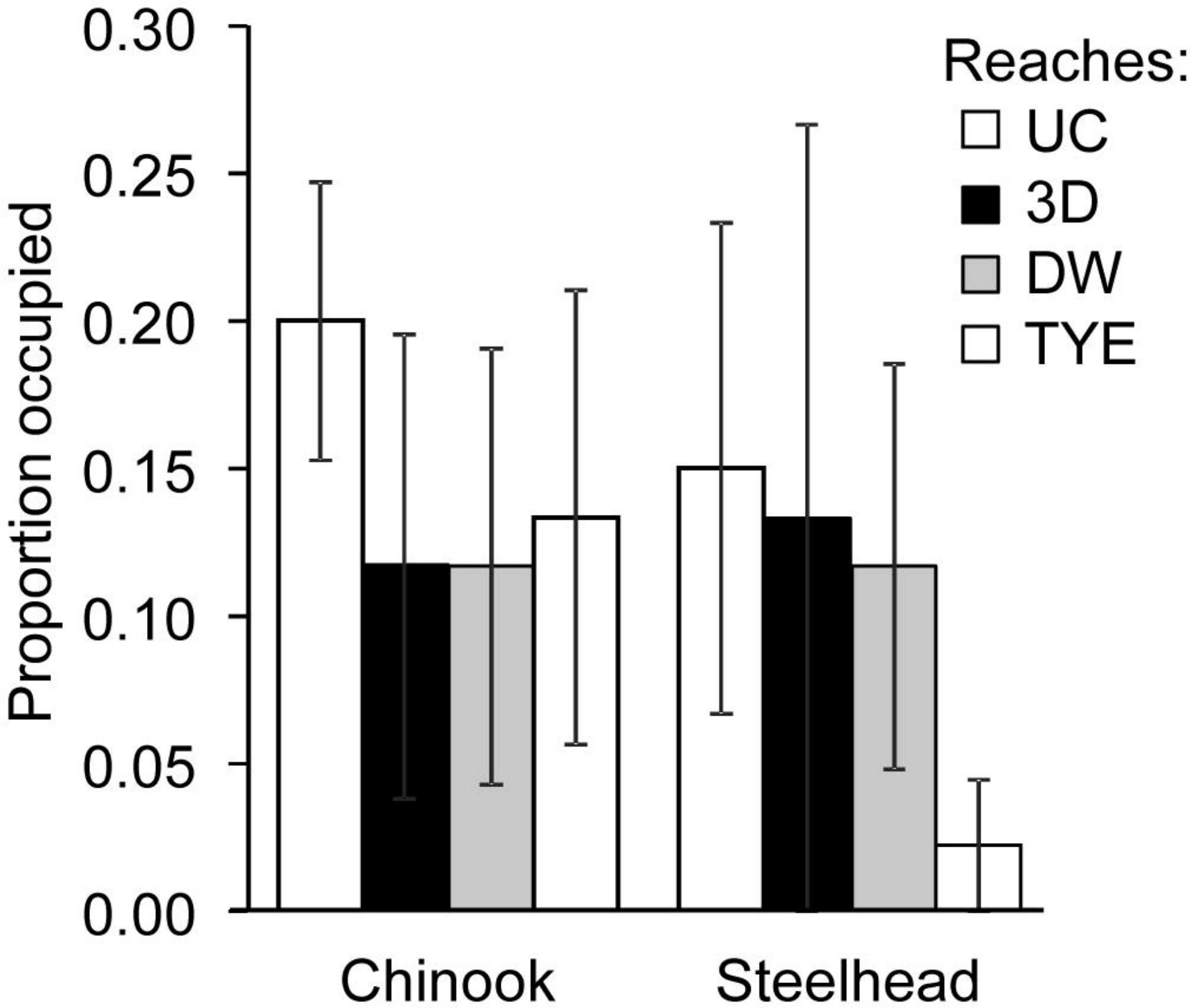



Entiat watershed
Washington State
N
Habitat types:
ELJ restored
natural log jam
control reach
Upper basin segment
Lower basin segment
rkm 40.0
3D
Entiat river (flow)
Tyee
Dillwater
Upper Control
rkm 32.0
rkm 11.5
Hatchery
Roaring Creek
Entiat river (flow)
Harrison Ref.
Harrison S.C.
PUD
Asher
Yakama
Keystone
Kiosk
rkm 2.0
0 0.5 1 2 Kilometers


A) Chinook
B) Steelhead
Habitats:
Restored
Unrestored
Fish density (#/m2)
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
ASH
HSC
KEY
PUD
YAK
ASH
HSC
KEY
PUD
YAK
Lower basin: treated reaches


A) Chinook
B) Steelhead
Habitats:
Restored
Natural log jam
Unrestored
Fish density (#/m2)
1.1
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
3D
DW
TYE
UC
3D
DW
TYE
UC
Upper basin: treated/natural reaches


Proportion occupied
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
Chinook
Steelhead
Reaches:
Untreated
Treated


Proportion occupied
0.30
0.25
0.20
0.15
0.10
0.05
0.00
Chinook
Steelhead
Reaches:
UC
3D
DW
TYE