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Abundance and body condition of sculpin (*Cottus* spp.) in a small forest stream following recolonization by juvenile Coho Salmon (*Oncorhynchus kisutch*)

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Review

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9 3 Abundance and body condition of sculpin (*Cottus spp.*) in a small forest stream following
10 4 recolonization by juvenile Coho Salmon *Oncorhynchus kisutch*
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16 6 S.M. Naman^{1, 2}, P.M Kiffney^{1, 3}, G.R Pess¹, T.W Buehrens⁴, T.R Bennett¹
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34 13 Keywords: Sculpin, Pacific salmon, recolonization, streams
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6 32**Abstract**

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9 33 Recolonization by native species following reintroduction can affect resident species through a
10 34 variety of processes. We examined the effects of natural recolonization by coho salmon
11 35 *Oncorhynchus kisutch* on sculpin (*Cottus rhotus* and *C. gulosus*), a small benthic fish, in a small
12 36 forest stream in Western Washington, USA. Provision fish passage around a small dam allowed
13 37 coho access to habitat which had been inaccessible for over 100 years. We found that density (g
14 38 m^{-2} and number m^{-2}) was unchanged and body condition (the slope of the relationship between
15 39 length and weight) of sculpin tended to increase from before relative to a five year period
16 40 following recolonization. The proportion of sculpin comprising the total fish assemblage
17 41 decreased after coho colonization relative to before but remained stable for a five year period
18 42 after coho reintroduction while coho density increased over five fold. Additionally, we used
19 43 Akaike's Information Criteria to evaluate the relative importance of physical and biological
20 44 variables to predict sculpin density in pool habitats during the initial coho recolonization period.
21 45 Physical microhabitat variables had little support for predicting sculpin density, while there was
22 46 a significant support for stream temperature; cutthroat trout (*O. clarki*) density and year were the
23 47 most important predictors of sculpin density. Coho density was not significant in any model. Our
24 48 results indicate coho introduction and subsequent recolonization has to date had minimal
25 49 individual or population level effects on sculpin therefore demonstrating that species
26 50 reintroductions into their native range can have no measurable effect on resident organisms.

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Introduction

Many species are being actively reintroduced or are recolonizing naturally into areas where they were locally extirpated. In the Pacific Northwest ecoregion of the United States, recent restoration effort has focused on reconnecting freshwater migration networks for species of anadromous salmonids (family Salmonidae- e.g. salmon, trout, and char) many of which have been federally listed as threatened or endangered under the United States Endangered Species Act (1970; NRC, 1996). A large amount of historic spawning and rearing stream habitat is currently inaccessible due to impassible barriers such as dams and culverts (Roni *et al.*, 2002). Therefore, removal or circumvention of barriers to allow fish passage and recolonization of previously inaccessible habitat has emerged as an important conservation strategy that is likely to increase in the future (Roni *et al.*, 2002; Kiffney *et al.*, 2009).

Species introductions and reintroductions can influence community structure and function in a variety of ways (Abrams, 1996). For example, juvenile salmonids establishing new populations in previously inaccessible habitats may interact with resident fishes and potentially compete for limited resources. Juvenile coho salmon *Oncorhynchus kisutch* may have particularly strong impacts on resident fish communities. Coho are territorial and aggressive to conspecifics and other species and have a long freshwater residence period relative to other juvenile anadromous salmonids (Bisson *et al.*, 1988; Reeves *et al.*, 1989). In contrast, there may be positive effects of Pacific salmon reintroductions as anadromous species are relatively fecund, producing large numbers of eggs which may serve as an important resource, providing energetic benefits to species that consume them (Willson and Halupka, 1995). Examining the impacts of fish colonization on resident biota and ecosystem processes is a critical area of research regarding the biological effects of species reintroductions; unfortunately, there has been little study in this area (Kiffney *et al.*, 2009).

Resident fish assemblages in low order streams of the Pacific Northwest ecoregion are dominated by some combination of anadromous and resident (typically coastal cutthroat trout *O. clarki clarki*) salmonids and non game species, particularly sculpin (*Cottus* spp.) (Roni, 2002; Wydoski and Whitney, 2003). Sculpin are widespread throughout the region and may be numerically dominant over salmonids in some areas (Eggers *et al.*, 1978; Reeves *et al.*, 1998). A large body of literature has focused on the effects of habitat and interactions on the distribution

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2
3 89 of Salmonids in streams (Hearn, 1987; Rosenfeld *et al.*, 2000; Quinn, 2005); however, the
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5 90 importance of interactions between sculpin and salmonids has received considerably less
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7 91 attention. Given the continuing shift toward community level metrics for conservation and
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9 92 restoration efforts (Roni, 2003; Adams and Schmetterling, 2007), further understanding of
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11 93 sculpin – habitat relationships and biological interactions will likely be beneficial in prioritizing
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13 94 future conservation efforts.

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15 95 We explored individual and population level effects of reintroducing anadromous
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17 96 salmonids on a sculpin including, riffle sculpin *Cottus gulosus* and torrent sculpin *C. rhotus*
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19 97 (which were numerically dominant), in a small forest stream. We used data collected
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21 98 opportunistically as part of a larger study examining the dynamics of natural recolonization by
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23 99 coho salmon in a river system where an impassible dam was modified (Kiffney *et al.*, 2009; Pess
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25 100 *et al.*, 2011; Kiffney *et al.*, 2011). The installation of a fish ladder facility at Landsburg
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27 101 Diversion Dam in 2003 on the upper Cedar River (see study site description in methods)
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29 102 provided access for anadromous salmonids to 32 km of habitat which had been inaccessible since
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31 103 1900. We examined temporal trends of sculpin density, length-weight relationships (a surrogate
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33 104 for body condition) and the proportion of sculpin comprising the total fish assemblage before the
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35 105 installation of the fish ladder (2000-2001) and for five years after initial coho recolonization
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37 106 (2005-2009). To our knowledge this is one of the first case studies to document the potential
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39 107 effects of Pacific salmon colonization on resident sculpin populations.

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41 108 The recolonization of coho has the potential to affect sculpin in a variety of ways.
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43 109 Previous research on direct sculpin-salmonid interactions has provided mixed results. Some
44
45 110 investigators have suggested the potential for interspecific competition as sculpin and salmonids
46
47 111 have been shown to consume similar prey (Patten, 1975; Ruetz *et al.*, 2003; Zimmerman and
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49 112 Vondercek, 2006b; Zimmerman and Vondercek, 2007a). However, other evidence suggests
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51 113 sculpin – coho interactions may be weak as they partition resources through differences in
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53 114 habitat use and foraging strategies (Moyle, 1977; Glova, 1986). If resource overlap between the
54
55 115 species is high, we may expect a negative response in sculpin populations (density) and
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57 116 individual condition (length-weight relationships) as a result of direct competition. Conversely,
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59 117 sculpin may also benefit from salmon recolonization as some studies have shown sculpin can
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3 118 consume juvenile salmonids and eggs (Patten, 1975; Quinn *et al.*, 2012) which could result in
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5 119 increased densities and individual condition of sculpin.
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8 120 The response of sculpin to coho recolonization may not be predictable from other
9
10 121 systems however. Given that coho and sculpin have historically co-occurred across their native
11 122 range, it is plausible to hypothesize that they have evolved respective methods to partition
12
13 123 resources and decrease the potential for competitive interactions. However, the local extirpation
14
15 124 of coho from Rock Creek for over 100 years complicates predictions as interactive dynamics
16
17 125 may differ after species have experienced a period of isolation. Sinclair (1998) suggested that
18
19 126 interspecific interactions with resident biota may be exacerbated during early stages of species
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21 127 introductions. This hypothesis was supported by Ward *et al.* (2008) who found abundance of
22
23 128 sculpin to be a key determinate of Atlantic salmon *Salmo salar* survival following reintroduction
24
25 129 above an anadromous barrier. However, a key difference in our study is that coho naturally
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27 130 entered Rock Creek as juveniles (Pess *et al.*, 2011) as opposed to being artificially stocked as fry
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29 131 (i.e. Ward *et al.*, 2008). Coho entering Rock Creek were likely to be of sufficient size to where
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31 132 we hypothesized morphological and behavioral differences would minimize their interactions
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33 133 with sculpin during the initial period of recolonization.

34 134 To additionally examine what biological and environmental factors influenced sculpin
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36 135 during early stages of coho recolonization, we modeled the relationship between sculpin density
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38 136 in pool habitats and a variety of predictors which included biological and physical variables.
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40 137 Abundance of torrent and riffle sculpin has been shown to respond positively to microhabitat
41
42 138 conditions such as substrate (Brusven and Rose, 1981; Brown, 1991), gradient and velocity
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44 139 (Kiffney and Roni, 2007). Additionally, predation and competition from trout has been shown to
45
46 140 have negative effects on sculpin abundance (Ruetz, 2003; Ramirez, 2011). There are few studies,
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48 141 however, which have integrated abiotic and biotic correlates to examine sculpin microhabitat
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50 142 abundance, especially over larger spatial and temporal scales (but see Grossman *et al.*, 2006).
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52 143 We used an information theoretic approach (Burnham and Anderson, 2002; Kiffney and Roni,
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54 144 2007) to estimate the relative importance of abiotic vs. biotic factors in predicting sculpin density
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56 145 during coho colonization in pool habitats in Rock Creek from 2005- 2009. We hypothesized that
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58 146 coho recolonization would have minimal individual and population level effects on sculpin in
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60 147 Rock Creek and biotic factors (potential competition with coho and trout) would be less

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3 148 important in predicting sculpin abundance than physical microhabitat features or ambient
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5 149 environmental conditions such as stream temperature.
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Methods

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153 *Study Site*

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18 154 Rock Creek is a tributary to the Cedar River (the largest tributary to Lake Washington)
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20 155 located in the Cedar River Watershed, which is a 36,644 hectare municipal watershed managed
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22 156 as a conservation area by the Seattle Public Utilities (Kiffney *et al.*, 2009). Landsburg Diversion
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24 157 Dam at river kilometer 35 (distance from Lake Washington) was installed in 1900 to divert
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26 158 drinking water to the greater Seattle area (Figure 1) and blocked upstream migration of fish until
27
28 159 2003 when the installation of a fish passage facility allowed anadromous and adfluvial fish,
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30 160 including Pacific salmon and trout, upstream migration access to 20 km of mainstem and 13 km
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32 161 of tributary habitat in the upper Cedar River. Rock Creek, the first major upstream tributary to
33
34 162 the upper Cedar River and enters the mainstem 3.4 km upstream of Landsburg. Rock Creek is the
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36 163 only tributary in the upper Cedar River watershed with most of its drainage area accessible to
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38 164 salmon (Anderson *et al.*, 2006).

39 165 Rock Creek has a rain dominated hydrograph with peak flows in winter and spring
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41 166 (~November to May) and low flows in summer. Due to the management of the watershed as a
42
43 167 drinking water source for the City of Seattle and as a de facto nature reserve, habitat conditions
44
45 168 and processes in the watershed are largely intact. In addition to torrent and riffle sculpin (Tabor
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47 169 *et al.*, 2007), the fish assemblage in Rock Creek following the fish ladder installation consists of
48
49 170 coho and low densities of Chinook salmon *O. tshawytscha*, cutthroat trout, as well as speckled
50
51 171 dace *Rhinichthys osculus* and one to three species of lamprey (*Lampetra* spp.). A small number
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53 172 of resident rainbow trout *O. mykiss*, which are abundant in the mainstem Cedar River have also
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55 173 been observed in Rock Creek.

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176 *Study Design*

177 In summer of 2000 and 2001, fish were collected from pool habitats (n=6) in the lower
178 2.6 km reach of Rock Creek to establish baseline conditions for fish populations of the Cedar
179 River and its tributaries before the introduction of anadromous salmon above the fish ladder (see
180 Riley *et al.*, 2001; Kiffney *et al.*, 2002). In 2005-2009 fish were collected from approximately 30
181 pools over a period of 3-7 days in midsummer across the same 2.6 km reach of Rock Creek (Pess
182 *et al.*, 2011). Only pool habitats were sampled because coho were the focal species of the study
183 (Pess *et al.*, 2011). An effort was made to sample a consistent set of habitat units across years;
184 however, a variable number of habitat units were sampled each year due to their elimination and
185 creation resulting from changes in channel morphology from flood events or inputs of large
186 wood. Fish were collected by three pass electrofishing (see Pess *et al.*, 2011 for additional
187 information on fish collection and processing) measured to the nearest mm (total length for
188 sculpin, fork length for coho and trout) and weighed to the nearest 0.1 g using an OHAUS
189 Scout[®] field balance. Population size was estimated using a maximum likelihood procedure
190 adapted to three pass electrofishing (Rosenfeld *et al.*, 2000; Pess *et al.*, 2011). Fish density was
191 determined by summing the total weight (g) or total number of fish captured in a pool habitat
192 unit and dividing by pool surface area (average wetted length * average wetted width).

193 Prior to fish collection, habitat surveys were conducted to quantify physical
194 characteristics of each pool habitat unit. At each pool, wetted length and width were measured
195 using a handheld laser rangefinder, and maximum and minimum depth (depth at pool crest) was
196 determined using a stadia rod. Average current velocity was quantified by measuring velocity at
197 three (top, middle and bottom) points of a pool using a handheld velocity meter (Table 1). We
198 visually estimated dominant substrate types and categorized them using the following criteria:
199 fines (diameter \leq 0.062 mm), sand (0.062 – 2 mm), gravel (2 – 64 mm), cobble (64 – 256 mm),
200 and boulder (> 256 mm) (Pess *et al.*, 2011). Stream temperature was measured using
201 continuously using data loggers (HOBO[®] Pendant temperature data loggers, Onset Corporation),
202 deployed at three locations within our sampling reach.

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3 203 Sculpin were not identified to species in the field due to logistical constraints resulting
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5 204 from the large effort needed to process coho, morphological similarities of torrent and riffle
6
7 205 sculpin and the large number of sculpin captured during electrofishing (see Pess *et al.*, 2011). A
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9 206 concurrent study in Rock Creek has indicated that torrent sculpin are the dominant species in
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11 207 pools in our study reach with riffle sculpin making up less than 3 % of the total population (T.P.
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13 208 Quinn Unpublished data), results that were in agreement with earlier work (Riley *et al.*, 2001).
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15 209 These finding were also supported by inspection of a subset of lethally sampled individuals from
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17 210 2007 and 2009 which showed that torrent were the dominant species (S. Naman unpublished
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19 211 data).

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23 213 *Statistical analysis*24
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28 215 All statistical analysis was performed using R version 2.13.1 (R Core Development
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30 216 Team, 2011). Data were tested for normality using a Shapiro Wilkes normality test; a log (n + 1)
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32 217 transformation was used when data deviated from a normal distribution. For proportion data, an
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34 218 Arcsine transformation was used to achieve homogeneity of variances (Zar, 1999). Pools
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36 219 sampled consistently from 2000-2001 to 2005-2009 (n = 6, hereby known as “before-after
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38 220 analysis”) were analyzed separately from the full set of pools sampled from 2005-2009 (n = 17-
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40 221 32, hereby known as “trend analysis”). For both analyses, we used a one way analysis of
41
42 222 variance (ANOVA) to examine differences in density and the relative proportion of sculpin
43
44 223 comprising total fish density across years. We also used ANOVA to test for differences in the
45
46 224 overall mean length and weight of sculpin captured in each year.

47 225 To examine temporal trends in body morphology of sculpin we tested for differences
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49 226 among years in the slope of the length weight relationship using analysis of covariance
50
51 227 (ANCOVA; Roni, 2002; Gray *et al.*, 2002). The slope parameter *b* was estimated using linear
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53 228 regression of the form

54
55 229 $\ln(\text{Weight}) = \ln(a) + b * \ln(\text{length})$

230 Where a is the Y axis intercept and b is the slope of the equation. A significant difference in the
231 slope coefficients indicated weight varied among years for sculpin of a given length.

232 To predict sculpin density (g m^{-2}) within pools in Rock Creek from 2005-2009, a set of a
233 *priori* candidate models were constructed using physical (measured at the microhabitat level),
234 environmental and biotic variables (Table 2). Physical microhabitat variables included residual
235 pool depth (the maximum depth minus the depth at its crest), current velocity (m s^{-1}), and
236 dominant substrate type. Coho and cutthroat trout biomass density (g m^{-2}) were included as
237 biotic variables. We partitioned trout into two size classes of greater and less than 100 mm fork
238 length. It has been suggested that these classes represent two distinct trophic groups and may be
239 a threshold for piscivory (Raggon, 2010; Kiffney *et al.*, 2011). We defined stream temperature as
240 the daily mean averaged across the summer (June 1-September 31) of each year. This metric was
241 used due to temporal gaps in temperature data (i.e. loggers were not recording consistently across
242 all years), and it provided the most unbiased estimate of relative temperature differences among
243 years. To account for annual variation in density, we included year as a fixed effect covariate in
244 the model set. Because all habitat covariates were not measured in 2000 and 2001, we did not
245 include baseline sculpin data from 2000 and 2001 in the modeling analysis.

246 The set of plausible candidate models were based on existing literature relevant to
247 sculpin-habitat relationships and biological interactions, exploratory analysis (i.e., correlation
248 matrix), and our own experience with the study system (Table 2). Due to the large number of
249 potential explanatory covariates and to the exploratory nature of our analysis, we only considered
250 additive, linear models and did not include any interaction or nonlinear terms in the model set.
251 The consideration of only simple additive models also limited the size of the model set, which
252 averted a common problem with model selection analysis of the consideration of too many
253 models (Burnham and Anderson, 2002).

254 Models were evaluated using the information theoretic approach of Burnham and
255 Anderson (2002). Candidate models were fit to the data using linear regression (Franklin *et al.*,
256 2000; Grossman *et al.*, 2006). The fit of each model was assessed using a bias corrected version
257 of Akaike's Information Criterion for small sample sizes (AIC_c). AIC_c measures the amount of
258 information loss in each candidate model, with better approximating models receiving lower
259 AIC_c scores. ΔAIC_c values were computed for each model as $\Delta\text{AIC}_c = \text{AIC}_{c,i} - \text{AIC}_{c, \min}$,

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3 260 where $\Delta AIC_{c,i}$ is the ΔAIC_c value for the i th model in the candidate model set and $\Delta AIC_{c, \min}$ is
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5 261 the minimum ΔAIC_c value among the candidate models. Models were ranked according to their
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7 262 ΔAIC_c weight values (w_i) which range from 0 (complete information loss) to 1 (no information
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9 263 loss). We considered a candidate model to have substantial empirical support and report
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11 264 parameter estimates and coefficients if the w_i value for a given candidate model was within 10%
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13 265 of the model with the highest w_i value (i.e. the best fitting model) (Burnham and Anderson,
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15 266 2002). We also estimated the relative importance of predictor variables within the most plausible
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17 267 candidate models by summing the w_i values of all models in the set where the given predictor
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19 268 variable occurred. The higher the sum, the more important variable i is relative to other variables
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21 269 (Burnham and Anderson, 2002).

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Results

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Temporal trends in density and body condition

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33 275 In the before-after analysis, we found no difference in sculpin biomass (g m^{-2}) or numerical
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35 276 density (ANOVA, $P > 0.6$, Figure 2a and 2b) from before (2000-2001) compared to after (2005-
36
37 277 2009) coho recolonization. In the trend analysis (full set of pools from 2005-2009), both
38
39 278 numerical and biomass density tended to increase over time but the slope was not different from
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41 279 zero ($P = 0.13$ and 0.16 respectively, Figure 3a and 3b). The mean proportion of sculpin
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43 280 comprising the fish assemblage was higher in 2000 and 2001 for both biomass (Figure 4a, $P <$
44
45 281 0.001) and numerical (Figure 4b, $P < 0.001$) density. The decrease in the proportion of sculpin
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47 282 comprising total fish density was largely due to the addition of coho which increased five-fold.
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49 283 However, in the trend analysis, the proportion of sculpin comprising the comprising the fish
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51 284 assemblage did not vary from 2005-2009 for both biomass and numerical density ($P > 0.2$,
52
53 285 Figure 5a and 5b).

54 286

55 287 The mean size of sculpin captured increased after coho recolonization. Sculpin were
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57 288 significantly shorter ($P < 0.001$) and lighter ($P < 0.01$) in 2000 and 2001 than in other years.

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3 289 Mean lengths and weights did not significantly change among years 2005-2009 (Table 3).
4
5 290 Sculpin length-weight relationships also increased over time in Rock Creek (Table 3). Sculpin
6
7 291 were significantly lighter for a given length in 2001 than other years (ANCOVA, $P < 0.001$) and
8
9 292 the slope of the relationship increased from 2005-2009 although the trend was not significant (P
10 293 > 0.20)
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15 295 *Predicting sculpin density in pools*

16
17 296 The best approximating model included an effect of year, small and large trout, coho density and
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19 297 residual pool depth (Table 4) while the second ranked model did not include coho density. The
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21 298 third best approximating model included an effect of mean daily temperature but was 5 times
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23 299 less likely than the two higher ranked models (Table 4). Year, small trout and large trout density
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25 300 were the most important predictor variables as indicated by their relative importance values
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27 301 (Figure 6) and were 1.2x and 2x the relative importance value of residual depth and coho density
28
29 302 respectively (Figure 6). Mean daily temperature was 10x less important than the most important
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31 303 predictors.

32
33 304 Slope coefficients for the effect of coho density and residual pool depth on sculpin
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35 305 biomass density were negative but not significantly different from zero in the three most
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37 306 plausible models (Table 5). Small and large trout densities were positively related to sculpin
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39 307 biomass density and were significantly different from zero in all models ($P < 0.01$). In the third
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41 308 most plausible model, mean daily temperature was negatively associated with sculpin biomass
42
43 309 density and did not overlap zero ($P = 0.07$, Table 4).
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46 311 **Discussion**

47 312 48 49 313 *Temporal patterns in density and body condition*

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52 314 The results from our study provide several lines of correlative evidence that
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55 315 recolonization by coho had no measurable effect on sculpin populations in Rock Creek. First, the
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3 316 before-after analysis revealed no change in sculpin density following coho recolonization
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5 317 relative to before. We recognize that the low number of pools sampled across years of the study
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7 318 potentially limited our power to detect effects; however, the trend analysis further supports these
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9 319 findings as it showed no change in numerical or biomass density of sculpin from 2005-2009
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11 320 despite the increase of coho densities over fivefold. (Pess *et al.*, 2011). The addition of coho
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13 321 rapidly increased total fish density, and therefore decreased the proportion of sculpin overall
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15 322 from before (2000-2001) to after (2005-2009) coho recolonization. However, the lack of change
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17 323 in the proportion of sculpin comprising the total fish assemblage in 2005-2009 provides further
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19 324 reinforcement to our conclusion and indicates that the sculpin population in Rock Creek
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21 325 remained stable during initial coho recolonization.

22 326 These results are additionally supported at the individual level as a positive trend in
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24 327 length-weight relationship was observed. We hypothesize that the increasing trend in individual
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26 328 size and length-weight relationships of sculpin likely reflect inter-annual differences in
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28 329 environmental conditions in Rock Creek. Increasing body condition of sculpin may be a possible
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30 330 response to salmon recolonization. Direct predation by sculpin on emerging coho fry (e.g. Patten,
31
32 331 1975) or eggs (Quinn *et al.*, 2012), or indirect bottom up food web effects as a result of marine
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34 332 derived nutrient additions (Wipfli *et al.*, 1998) are possible pathways which may lead to
35
36 333 increased condition of sculpin as well as other benthic consumers. However, we do not
37
38 334 hypothesize these mechanisms are operating in Rock Creek because adult coho were not
39
40 335 spawning in Rock Creek during the study (P.M Kiffney unpublished data) and juvenile coho
41
42 336 entering Rock Creek were typically large enough to escape potential predation from sculpin.

43 337 When combined, our results provide evidence to support the hypothesis that coho
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45 338 reintroduction and subsequent colonization has had minimal population and individual level
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47 339 effects on sculpin in Rock Creek. Our findings are especially notable when considering that
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49 340 during the relatively short duration of this study juvenile coho in Rock Creek approached
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51 341 densities comparable to other similar systems in the region with uninterrupted histories of
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53 342 anadromy (Pess *et al.*, 2011). These findings are relevant in a management context as there is
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55 343 concern that species reintroduction into new or previously inaccessible habitat has the potential
56
57 344 to produce negative effects on resident communities and species (e.g., Ricciardi and Simberloff,
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1
2
3 345 2009) and that negative interactions with resident organisms may limit reintroduction success
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5 346 (Sinclair *et al.*, 1998).

7 347 Current understanding of the biological effects of species reintroduction and
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9 348 recolonization on resident communities and processes is hindered in part by the lack of empirical
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11 349 case studies. This knowledge gap is deepened further as non-game organisms such as sculpin are
12
13 350 often ignored or treated as background species in many studies. Investigations have thus far
14
15 351 yielded ambiguous results. Ward *et al.*, (2008) found evidence of negative interactions between a
16
17 352 reintroduced population of Atlantic salmon (*Salmo salar*) and resident slimy sculpin (*C.*
18
19 353 *cognatus*) in the Connecticut River basin which resulted in reduced survival for Atlantic salmon.
20
21 354 In contrast, in a tangentially related study, Glova (1987) found no difference in sculpin density
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23 355 when comparing habitats above (allopatric cutthroat stocks) and below (sympatric cutthroat and
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25 356 coho) an anadromous barrier, suggesting the presence of coho did not affect sculpin at the
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27 357 population level.

28 358 Our results agree with the latter study and we hypothesize the lack of any measurable
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30 359 effect of coho recolonization on sculpin in Rock Creek was primarily due to minimal interactions
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32 360 between the species. This may be partly a result of resource partitioning (e.g. Schoener, 1974)
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34 361 and reflective of morphological and ecological differences between the species. Coho feed
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36 362 primarily on terrestrially derived insects falling on the surface and drifting invertebrates (Fausch,
37
38 363 1993) while sculpin, which lack a swim bladder, primarily forage for benthic invertebrates at the
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40 364 stream bottom (Moyle, 1977; Wydoski and Whitney, 2003). Given the morphological and
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42 365 ecological differences between the species, the lack of effect is not surprising; nevertheless, it
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44 366 provides important information to managers contemplating barrier removal or species
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46 367 reintroduction.

47 368 Additionally, it is important to place our results into a historic context. Stream fish
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49 369 assemblages have evolved sympatrically and likely occurred at greater densities than currently
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51 370 observed in the region (Lackey, 2009). With this in mind, it is intuitive that interspecific density
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53 371 dependence may be a less important mechanism regulating populations during initial periods of
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55 372 Pacific salmon reintroductions when habitats may not be saturated. In addition to the observed
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57 373 results with sculpin in this study, previous investigations in the Cedar River and Rock Creek
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59 374 have shown no evidence of negative population or individual level effects of coho recolonization
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3 375 on rainbow or coastal cutthroat trout (Kiffney *et al.*, 2009; Buehrens, 2011), species shown to
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5 376 have potentially high resource overlap with coho (Bisson *et al.*, 1988; Rosenfeld *et al.*, 2000).
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7 377 While the potential for negative interactions due to species reintroductions remains a concern for
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9 378 fish conservation and management efforts (Brenkman *et al.*, 2008), our results suggest that
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11 379 interspecific interactions between reintroduced Pacific Salmon and resident fish assemblages
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13 380 may be less important than other factors in limiting populations of resident species during the
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15 381 early stages of reintroductions.
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18 19 383 *Predicting sculpin density in pools*

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22 384 Although coho density was included in two of the three best approximating models, it
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24 385 was not a significant predictor of sculpin density providing further support that coho had little
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26 386 measurable effect on sculpin. An unexpected finding from this analysis was the strong positive
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28 387 relationship between small and large trout and sculpin density. Other studies have demonstrated
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30 388 predator facilitation (when the foraging action of one predator causes prey to become more
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32 389 available to a predator with a different foraging mode) between benthic feeding sculpin and drift
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34 390 feeding trout (Douglas *et al.*, 1994; Miyasaka and Nakano, 1999); however, these relationships
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36 391 may be a function of unmeasured habitat or environmental variables and controlled experiments
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38 392 are necessary for us to attribute this mechanism to modeled trout-sculpin associations in Rock
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40 393 Creek.

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42 394 No physical microhabitat variables were statistically important in predicting sculpin
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44 395 density. Previous studies have suggested abundance of both torrent and riffle sculpin was
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46 396 positively associated with high gradient, high velocity habitats with intermediate substrate
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48 397 composition (Hawkins, 1983; Kiffney and Roni, 2007; Wydoski and Whitney, 2003). The lack
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50 398 of significant microhabitat variables in predicting sculpin density was consistent with other
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52 399 studies however. Roni (2002) found that small scale physical habitat features had little influence
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54 400 on sculpin densities in streams across western Washington and Oregon. The inclusion of
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56 401 temperature as a significant predictor suggests the importance of ambient environmental
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58 402 conditions in predicting temporal patterns of abundance for sculpin in Rock Creek. This result
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3 403 was supported by other findings which have related sculpin abundance to precipitation (Roni,
4 404 2002) and stream discharge or landscape variables such as drainage area (Grossman *et al.*, 2006).

7 405 An important caveat to our study was that only pool habitats were sampled. Given that
8 406 many sculpin species occur at higher densities in other fast water habitat types (e.g. riffles or
9 407 glides; Roni, 2002), the non-significant effects of microhabitat variables in explaining sculpin
10 408 density may be attributed to not sampling across all habitat types. Despite this limitation we
11 409 believe our results offer some relevance because coho densities are typically much higher in
12 410 pools relative to other habitat types (Lonzarich and Quinn, 1995; Rosenfeld *et al.*, 2000; Roni,
13 411 2002); therefore, during summer low flow conditions, pools are the habitat type in which coho
14 412 are most likely to interact with sculpin and other fish species.

22 413 Given their widespread distribution, relatively high abundance, and low mobility there is
23 414 clearly a need for more effort quantifying factors that influence sculpin populations and their role
24 415 in aquatic food webs. In the context of anadromous salmon reintroductions, sculpin may be an
25 416 important indicator for the response of resident fish assemblages or ecosystem processes. For
26 417 example, our results indicate sculpin distribution in pool habitats was not likely affected by a
27 418 rapid increase in juvenile coho abundance. Moreover, the increasing emphasis on community or
28 419 ecosystem level approaches to management will likely place greater importance on the ecology
29 420 of non-commercial species such as sculpin. Case studies, such as ours are critical to advance our
30 421 understanding of this ecologically important yet understudied group of species. We recognize
31 422 that due to the observational nature of our study, we are hindered in our ability to explicitly
32 423 demonstrate mechanisms which may have contributed to the resilience of sculpin to Coho
33 424 reintroduction. Future research should incorporate experimental approaches to gain greater
34 425 understanding of explicit ecological and evolutionary factors that may influence the response of
35 426 extant species or assemblages to reintroductions.

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Figure Legends

Figure 1. Map of Cedar River watershed from Landsburg Diversion facility to Cedar Falls. Dotted lines indicate physical barriers to anadromous migration (from Kiffney *et al.*, 2009)

Figure 2. Mean biomass (a) and numerical (b) densities of sculpin in consistently sampled pool habitat (n=6) units in Rock Creek from 2000-2001 through 2005-2009. Whiskers represent the minimum and maximum density in each year.

Figure 3. Mean biomass (a) and numerical (b) densities of sculpin (\pm SE) in the full set of pool habitat units (n=17-32) from 2005-2009. Whiskers represent the minimum and maximum density in each year.

Figure 4. The mean relative proportion of sculpin comprising the total fish density in pools for consistently sampled set of habitat units from 2000-2001 through 2005-2009. Whiskers represent the minimum and maximum proportion in each year.

Figure 5. The mean relative proportion of sculpin comprising the total fish density for all pools sampled 2005-2009. Whiskers represent the minimum and maximum proportion in each year.

Figure 6. Relative importance values for covariates used to predict sculpin density (g m^{-2}) in pools. Values were obtained by summing AIC weights for all models in which a given covariate occurred.

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Table 1. Number of pools sampled and mean (\pm SD) values of physical microhabitat variables and fish densities measured from 2005-2009

Variable	Year				
	2005	2006	2007	2008	2009
Pools (n)	32	34	31	17	22
Residual depth (m)	0.37 (0.08)	0.31 (0.13)	0.35 (0.15)	0.34 (0.14)	0.37 (0.14)
Velocity (m s ⁻¹)	0.22 (0.15)	0.29 (0.12)	0.27 (0.23)	0.33 (0.19)	0.25 (0.20)
Temperature (°C)	10.6 (6.4)	12.6 (5.1)	13.4 (5.2)	11.7 (4.9)	12.4 (7.10)
Trout (g m ⁻²)	1.65 (1.32)	0.83 (1.01)	0.22 (0.18)	0.52 (0.27)	0.79 (0.35)
Coho (g m ⁻²)	0.29 (0.25)	0.59 (0.57)	0.64 (0.56)	0.72 (0.97)	1.79 (2.91)

Table 2. *a priori* candidate models including the number of parameters (k) used to predict sculpin density in pools from 2005-2009.

Model	Form	k
<i>Microhabitat</i>		
Residual depth	$SC = a + b(\text{RHD})$	2
Substrate	$SC = a + b(\text{SUB})$	5
Residual depth, Velocity	$SC = a + b(\text{RHD}) + b(\text{VEL})$	3
Residual depth, Velocity, substrate	$SC = a + b(\text{RHD}) + B(\text{VEL}) + b(\text{SUB})$	7
Year	$SC = a + b(\text{year})$	5
Year, Substrate	$SC = a + b(\text{year}) + b(\text{SUB})$	9
Year, Residual depth	$SC = a + b(\text{year}) + b(\text{RHD})$	6
Year, Residual depth, Velocity	$SC = a + b(\text{year}) + b(\text{RHD}) + b(\text{VEL})$	7
<i>Biological Interactions</i>		
Small trout	$SC = a + b(\text{TRT})$	2
Large trout	$SC = a + b(\text{LRGTRT})$	2
Coho	$SC = a + b(\text{CO})$	2
Year, Small trout	$SC = a + b(\text{year}) + b(\text{TRT})$	6
Year, Large trout	$SC = a + b(\text{year}) + b(\text{LRGTRT})$	6
Year, Coho	$SC = a + b(\text{year}) + b(\text{CO})$	6
Year, Small trout, Large trout	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{LRGTRT})$	7
Year, Small trout, Coho	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{CO})$	7
Year, Large trout, Coho	$SC = a + b(\text{year}) + b(\text{LRGTRT}) + b(\text{CO})$	7
Year, Small trout, Large trout, Coho	$SC = a + b(\text{year}) + b(\text{TRT}) + B(\text{LRGTRT}) + b(\text{CO})$	
<i>Physical and Biological Interactions</i>		
Year, Small trout, Temperature	$SC = a + b(\text{year}) + b(\text{TEMP}) + b(\text{TRT})$	7
Year, large trout, Temperature	$SC = a + b(\text{year}) + b(\text{TEMP}) + b(\text{LRGTRT})$	7
Year, Coho, Temperature	$SC = a + b(\text{year}) + b(\text{TEMP}) + b(\text{CO})$	7
Year, Small trout, Large trout, Temperature	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{LRGTRT}) + b(\text{TEMP})$	8
Year, Small trout, Large trout, Coho, Temperature	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{LRGTRT}) + b(\text{CO}) + b(\text{TEMP})$	9
Year, Small trout, Residual depth	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{RHD})$	7
Year, Large trout, Residual depth	$SC = a + b(\text{year}) + b(\text{LRGTRT}) + b(\text{RHD})$	7
Year, Coho, Residual depth	$SC = a + b(\text{year}) + b(\text{CO}) + b(\text{RHD})$	7
Year, Small trout, Large trout, Residual depth	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{LRGTRT}) + b(\text{RHD})$	8
Year, Small trout, Large trout, Coho Residual depth	$SC = a + b(\text{year}) + b(\text{TRT}) + b(\text{LRGTRT}) + b(\text{CO}) + b(\text{RHD})$	9

Table 3. Number of sculpin captured each year and their mean lengths with minimum and maximum lengths in parenthesis, asterisk indicate lengths were significantly different ($P < 0.001$). Slope (β) and intercept (α) parameters for the relationship of total length (mm) to weight (g) in each sampling year. Equations were of the form $\ln(\text{weight}) = \alpha + \beta \ln(\text{length})$. All equations within years were highly significant ($P < 0.001$). An asterisk indicates a year where the slope is significantly different ($P < 0.01$). To illustrate the effect of changing parameters, the predicted weight (g) is shown for each year for a sculpin which is 75 millimeters long.

Year	n	Length	Slope	Intercept (\pm SE)	Predicted weight at 75mm
2000	80	58 (20, 98)*			
2001	75	65 (10, 115)*	2.037*	-7.312(0.218)	4.4 g
2005	65	73 (27, 125)	2.996	-11.325(0.086)	5.0 g
2006	90	73 (41, 136)	3.209	-12.225(0.079)	5.1 g
2007	205	73 (45, 151)	3.038	-11.459(0.075)	5.3 g
2008	129	70 (43, 120)	3.067	-11.568(0.084)	5.3 g
2009	272	70 (19, 143)	2.871	-10.738(0.058)	5.3 g

Table 4. AIC_c scores for the highest ten ranked candidate models predicting sculpin density. Models are ranked from lowest AIC_c score (most plausible model) to highest (least plausible).

Model	k	AIC _c	ΔAIC _c	w _i
Year, Small trout, Large trout, Coho, Residual depth	9	982.78	0.00	0.44
Year, Small trout, Large trout, Residual depth	7	982.90	0.12	0.41
Year, Small trout, Large trout, Coho, Temperature	9	986.30	3.51	0.08
Year, Small trout, Large trout, Coho	8	987.93	5.15	0.03
Year, Small trout Large trout	7	988.67	5.89	0.02
Year, Small trout, Large trout, Temperature	8	989.00	6.22	0.02
Year, Large trout	6	996.04	13.26	0.00
Year, Large trout, temperature	7	996.98	14.20	0.00
Year, Large trout, Coho	7	997.19	14.41	0.00
Small trout, Large trout	3	1003.84	21.07	0.00

35

36

Table 5. Parameter estimates (\pm SE) of the three highest ranked AIC models (from Table 5) with w_i values within ten percent of best fitting model. An asterisk indicates that the parameter estimate is significantly different from zero ($P < 0.05$).

40

41

Model Rank

42

Parameter

1

2

3

43

Intercept

0.132 (0.214)

0.173 (0.213)

1.689 (0.921)

44

Year 2006

0.586 (0.192)*

0.539 (0.195)*

0.993 (0.267)*

45

Year 2007

0.876 (0.199)*

0.805 (0.194)*

1.383 (0.323)*

46

Year 2008

0.080 (0.224)

0.039 (0.233)

0.038 (0.245)

47

Year 2009

0.915 (0.250)*

0.793 (0.237)*

1.290 (0.393)

48

Small trout

0.089 (0.024)*

0.082 (0.023)*

0.088 (0.024)*

49

Large trout

3.304 (0.616)*

3.301 (0.618)*

3.461 (0.613)*

50

Coho

-0.088 (0.059)

-0.134 (0.061)

51

RHD

-0.516 (0.379)

-0.061 (0.375)

52

Temperature

-0.165 (0.086)*

53

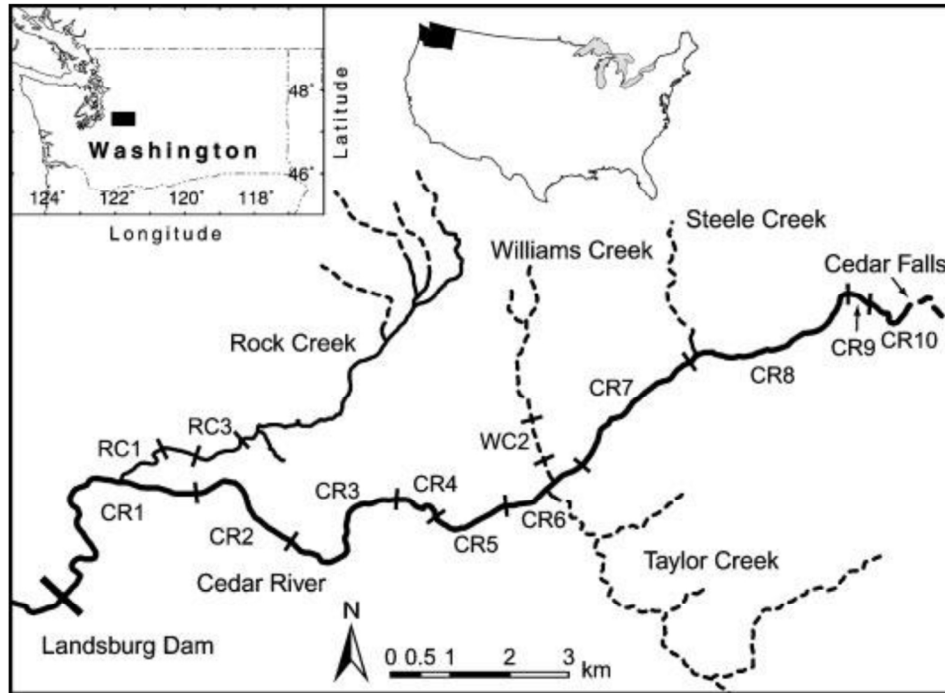
R²

0.31

0.30

0.30

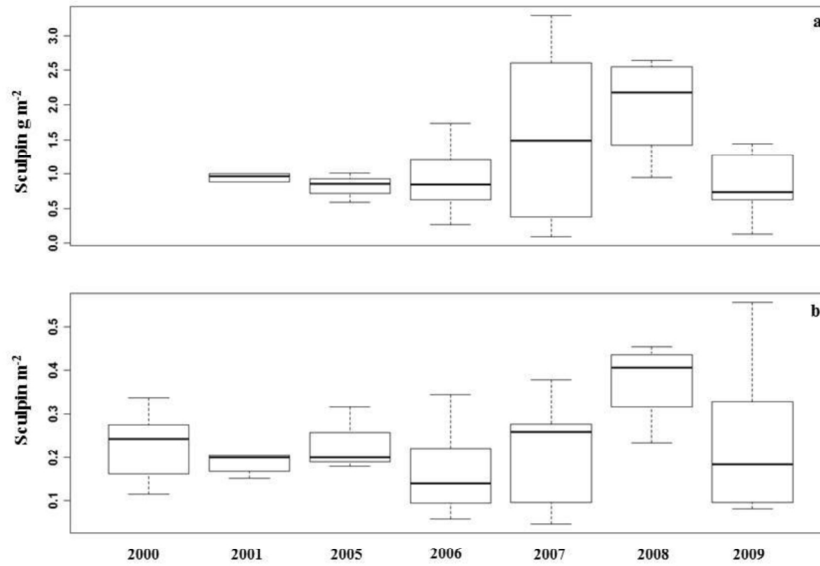
Naman *et al.* Figure 1.



138x114mm (300 x 300 DPI)

view

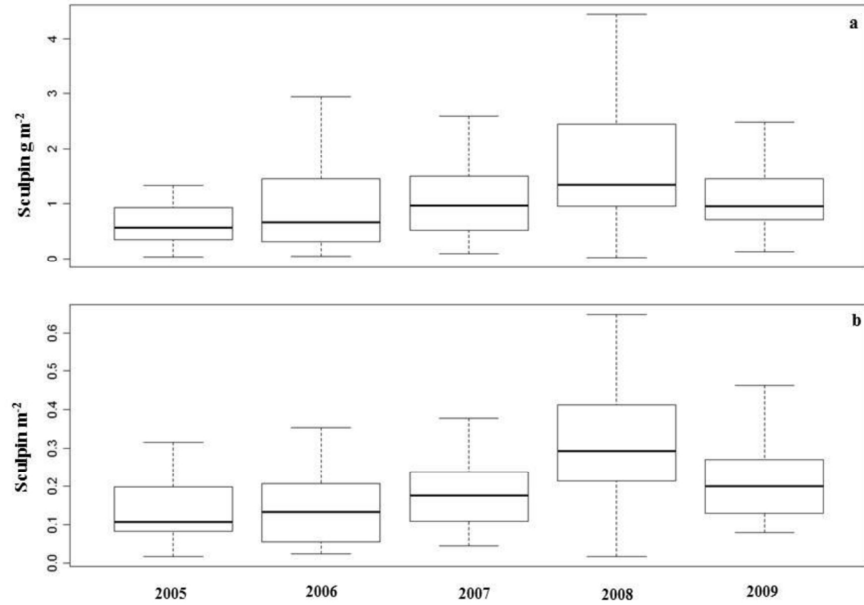
Naman *et al.* Figure 2.



151x119mm (300 x 300 DPI)

Review

Naman *et al.* Figure 3.



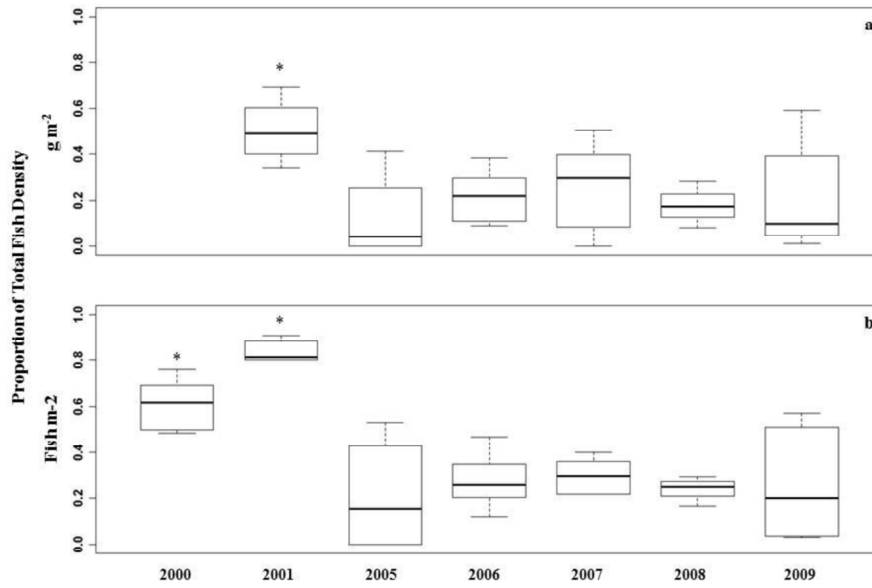
158x133mm (300 x 300 DPI)

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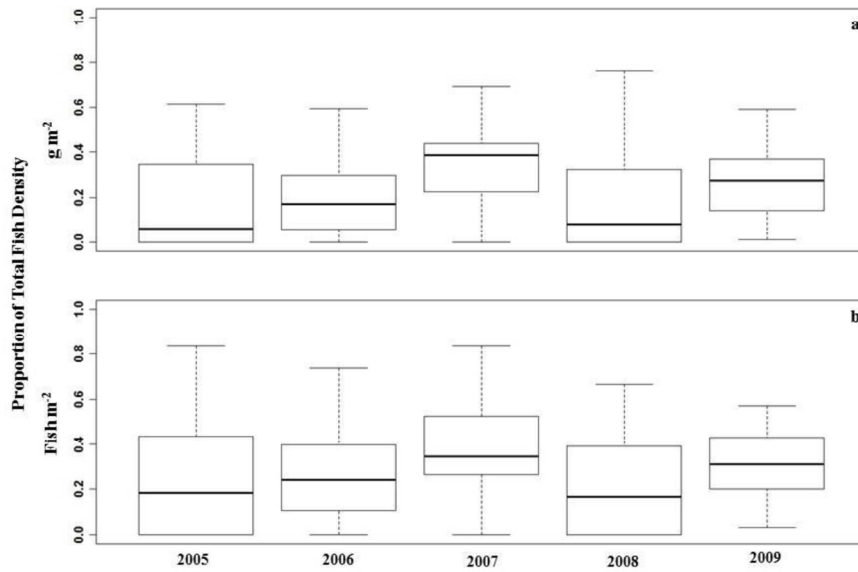
Naman *et al.* Figure 4.



150x125mm (300 x 300 DPI)

view

Naman *et al.* Figure 5.



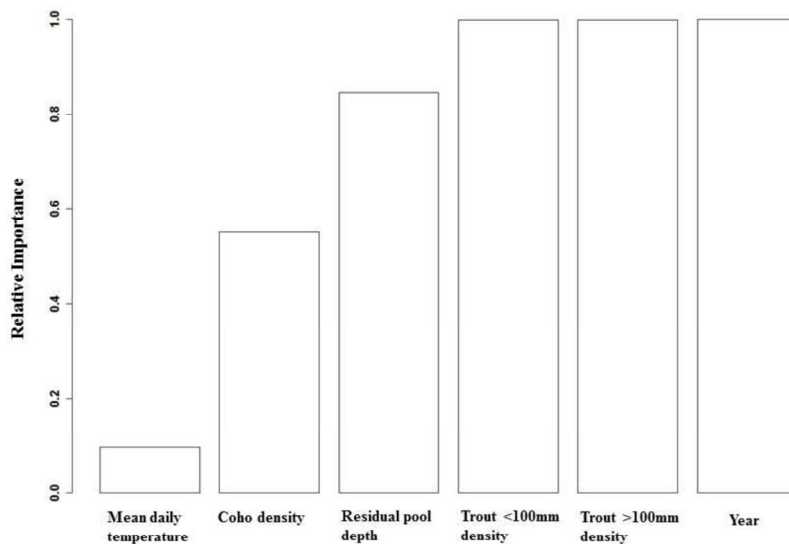
150x125mm (300 x 300 DPI)

view

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Naman *et al.* Figure 6.



146x123mm (300 x 300 DPI)

view