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ARTICLE

Habitat Associations of Juvenile Burbot in a Tributary of the Kootenai River

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Abstract

Burbot *Lota lota* in the lower Kootenai River, Idaho, have been the focus of extensive conservation efforts, particularly conservation aquaculture. One of the primary management strategies has been the release of Burbot into small tributaries in the Kootenai River basin, such as Deep Creek. Since 2012, approximately 12,000 juvenile Burbot have been stocked into Deep Creek; however, little is known about the habitat use of stocked Burbot. The objective of this study was to evaluate habitat associations of juvenile Burbot in Deep Creek. Fish and habitat were sampled from 58 reaches of the creek. Regression models suggested that Burbot moved little after stocking and were associated with areas of high mean depth and coarse substrate. This study provides additional knowledge on habitat associations of juvenile Burbot and suggests that managers should consider selecting deep habitats with coarse substrate for stocking locations.

Conservation of native species and freshwater ecosystems is an important goal for resource managers. To achieve successful restoration, conservation, and management of native species and freshwater ecosystems, an understanding of habitat requirements, species distributions, and species–habitat relationships is critical (Bond and Lake 2003; Rice 2005; Sindt et al. 2012). For example, Bond and Lake (2003) investigated species–habitat relationships in several streams of north-central Victoria, Australia, and used the results to inform habitat restoration activities in the system (Bond and Lake 2005). The subsequent habitat manipulations had a positive effect on fish abundance and were important in the conservation of fishes (Bond and Lake 2005). An understanding of habitat relationships may also be important for conservation aquaculture practices. For example, juvenile Atlantic Salmon *Salmo salar* require high-gradient streams with boulder or cobble substrate to maximize survival (Huntsman 1944; Caron and Talbot 1993; Scruton and Gibson 1993). As such, efforts to reintroduce Atlantic Salmon into Lake Ontario have focused on stocking fry and parr in streams with high gradients and large rocky substrate (Stanfield and Jones 2003). Despite the importance of species–habitat relationships, predicting and understanding these relationships are often difficult because fish species occurrence and relative abundance are influenced by a combination of abiotic and biotic factors acting across large and small spatial scales (Rahel and Hubert 1991; Lammert and Allan 1999; Marsh-Matthews and Matthews 2000; Quist et al. 2005).

Relationships between fish assemblages and habitat features measured at small scales are well documented (Gorman and

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Karr 1978; Lobb and Orth 1991; Rahel and Hubert 1991). The influence of instream, small-scale physical habitat features (e.g., depth, substrate composition, and cover) is easy to conceptualize, quantify, and study (Fischer and Paukert 2008; Sindt et al. 2012; Bakevich et al. 2013). Thus, many studies have used small-scale habitat variables to explain the distribution and abundance of fishes (e.g., Rahel and Hubert 1991; Gido and Propst 1999; Sindt et al. 2012). Although features at large spatial scales (e.g., elevation, temperature, and gradient) are often able to explain substantial variation in the occurrence and abundance of fishes (Rahel and Hubert 1991; Marsh-Matthews and Matthews 2000), an understanding of habitat at a small scale may be more useful because many management and conservation activities, such as stocking and habitat restoration, occur at small spatial scales.

Common approaches to restoration of a species include habitat restoration, harvest moratoria, and conservation aquaculture. One species that has been the focus of extensive restoration efforts is the Burbot *Lota lota*. Some Burbot populations, especially in the southern portion of the species' distribution, have experienced substantial declines (Paragamian et al. 2008; Stapanian et al. 2010). Reasons for the decline include alterations to habitat, overexploitation, interactions with nonnative species, and barriers to movement (Paragamian 2000; Stapanian et al. 2008, 2010). In Idaho, Burbot are native only to the Kootenai River and its tributaries (Simpson and Wallace 1982; Wallace and Zaroban 2013). Like many large rivers in North America, the Kootenai River in British Columbia, Montana, and Idaho has been highly altered since European settlement. These alterations include construction of levees and ditches on the lower portion of the river and organization of the floodplain into drainage districts (Northcote 1973; Partridge 1983; Richards 1997). However, construction of Libby Dam in 1972 near Libby, Montana, has arguably had the greatest influence on the Kootenai River, altering the river's thermal, hydrologic, and nutrient regimes (Paragamian et al. 2000). As a consequence, a shift in fish assemblage structure and deleterious effects on native riverine fishes have been documented (Paragamian et al. 2000, 2001; Paragamian 2002). The Burbot population of the lower Kootenai River has been in decline since 1959 (Partridge 1983), and the rate of decline has increased since the 1970s (Paragamian et al. 2000). Historically, Burbot in the lower Kootenai River supported subsistence, recreational, and commercial fisheries (Paragamian and Hoyle 2003; Ireland and Perry 2008). During the 1990s, the fisheries for Burbot in Idaho and British Columbia were closed (Paragamian et al. 2000). Despite closure of the fisheries, Burbot continued to decline; in 2004, the Burbot population in the lower Kootenai River was estimated at 50 adults and was considered functionally extinct (Paragamian et al. 2008). It was believed that Burbot would be extirpated from the lower Kootenai River system in less than a decade without intervention (Paragamian and Hansen 2009).

Restoration efforts for Burbot in the Kootenai River have been primarily in the form of conservation aquaculture, with the development of both intensive and extensive techniques (Jensen et al. 2008; Paragamian and Hansen 2009, 2011; Paragamian et al. 2011). Although a variety of stocking strategies (i.e., manipulation of fish size, quantity, timing, or location) has been employed, one strategy of particular interest is the release of Burbot into small tributary streams. Burbot spawning has been documented in the main-stem Kootenai River, but other data suggest that Burbot in the Kootenai River, Idaho, and Kootenay Lake, British Columbia, display an adfluvial life history, moving freely between Kootenay Lake and the Kootenai River to use small tributaries in the basin for spawning (Paragamian 1995). The goal of releases in small tributaries is to re-establish spawning runs in those habitats (Hardy and Paragamian 2013). Previous studies have sought to evaluate the release of juvenile Burbot in small tributaries (Neufeld et al. 2011; Stephenson et al. 2013), but the studies investigated dispersal behavior rather than the habitat associations of Burbot. An understanding of habitat use can aide in selecting Burbot stocking locations and help ensure efficient and effective stocking practices. Thus, the objective of this research was to model the occurrence and relative abundance of juvenile Burbot in Deep Creek, Idaho, a small tributary of the Kootenai River.

METHODS

Study area.—The Kootenai River has an international watershed of approximately 45,600 km^2 and is one of the largest tributaries to the Columbia River. The Kootenai River originates in Kootenay National Park, British Columbia, and flows southward into Montana and then Idaho before returning to British Columbia, where it joins the Columbia River. Many small tributaries contribute to the Kootenai River; one of the largest is Deep Creek, a third-order stream that originates east of White Mountain, Idaho, with a watershed area of about 480 km^2. Deep Creek is impounded approximately 10 km from its headwaters to form McArthur Lake (Figure 1). The creek flows 33 km north from McArthur Lake to its confluence with the Kootenai River, 5 km west of Bonners Ferry, Idaho. Our study area included the portion of Deep Creek between the McArthur Lake impoundment and a PIT tag antenna that was installed by the Idaho Department of Fish and Game (IDFG) approximately 7 km from the Deep Creek–Kootenai River confluence to monitor movement of stocked juvenile Burbot. In 2011, IDFG began stocking Burbot at two locations in Deep Creek (FIGURE 1). Since 2012, approximately 3,000 PIT-tagged juvenile Burbot (6 months old) have been stocked per year at the two stocking locations. To date, few Burbot have been detected at the PIT antenna.

Downstream of McArthur Lake, Deep Creek averages between 8 and 12 m in width and is dominated by cobble and gravel substrates. However, directly downstream of the McArthur Lake Dam, Deep Creek is dominated by deep pools and fine substrates. The impoundment has a major influence on water quality in the creek, which was included on Idaho's Clean Water Act Section 303(d) list of impaired waters due to excessive sediment and elevated temperatures (IDEQ 2006). Deep Creek

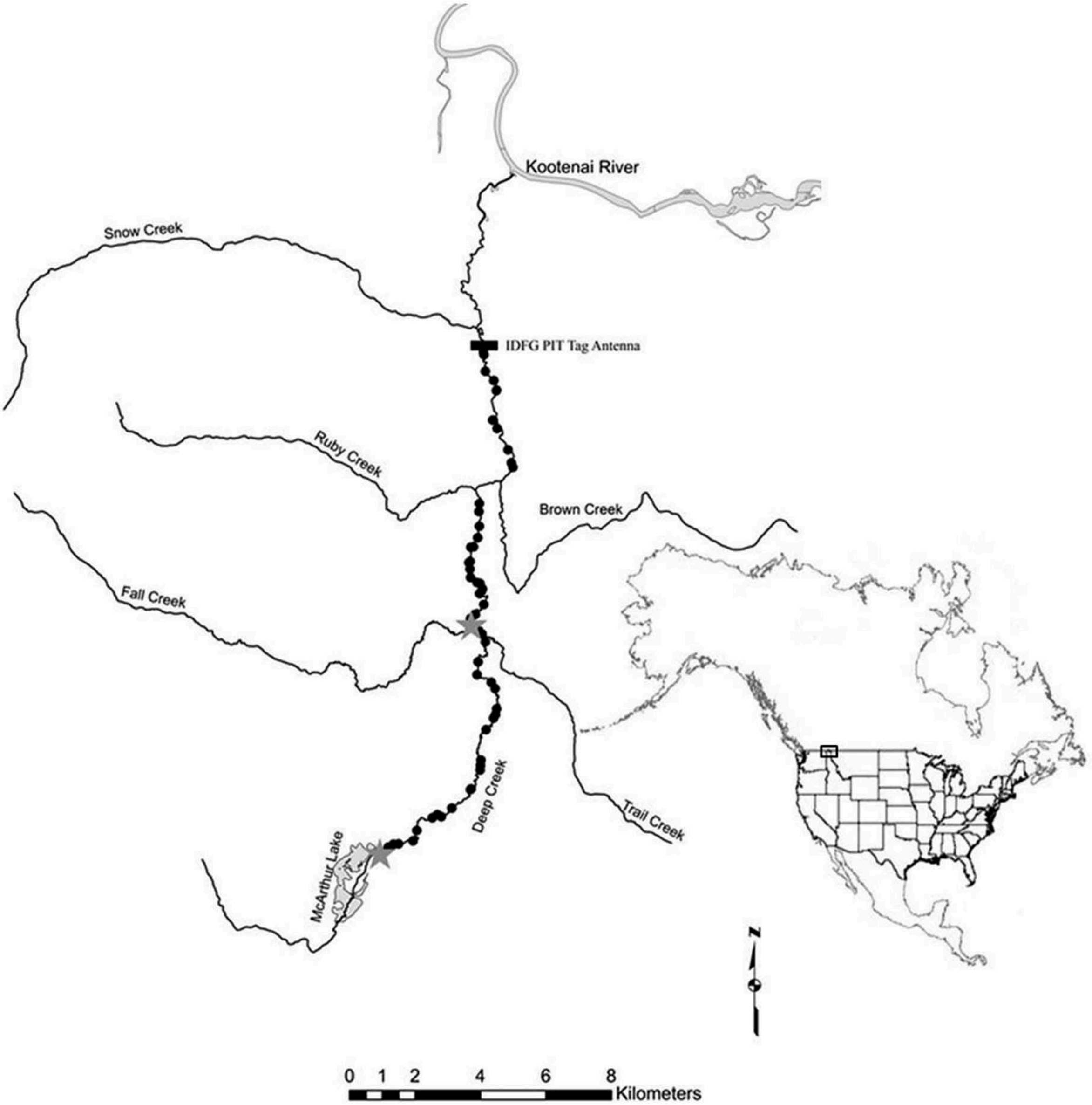

FIGURE 1. Map of the Deep Creek watershed, showing the main channel of Deep Creek, its five major tributaries (i.e., Trail, Fall, Ruby, Brown, and Snow creeks), and the Idaho portion of the Kootenai River. Each circle represents 1 of the 58 sampled stream reaches. Gray stars represent the two Burbot stocking locations. The black rectangle represents the PIT tag antenna installed by the Idaho Department of Fish and Game (IDFG).

has five major tributaries (Brown, Fall, Ruby, Snow, and Trail creeks). All tributaries except Snow Creek enter Deep Creek within the study area. Land ownership in the watershed is mixed. The U.S. Forest Service, Idaho Department of Lands, Forest Capital, and Stimson Lumber Capital all manage forest lands, mostly in the upper portions of the watershed. The lower portions of the Deep Creek watershed are generally privately owned and include areas of wetland, agriculture, residential development, and forest (IDEQ 2006).

Fish and habitat sampling.—Fishes and small-scale physical habitat characteristics were sampled from 58 stream reaches in Deep Creek (Figure 1) during the summer (June–August) in 2014 and 2015. In 2014, 25 reaches were randomly selected from Deep Creek. In 2015, 29 reaches were randomly selected. Four additional reaches were selected in 2015 based on known Burbot locations (i.e., detected with a portable PIT tag reader as part of another study). Each reach was 35 times the mean stream width (Lyons 1992; Simonson et al. 1994) up to a maximum length of 300 m and was delineated into macrohabitats (i.e., pools, riffles, runs, and off-channel units; Quist et al. 2003; Sindt et al. 2012). Fishes were sampled in each reach by using single-pass DC electrofishing (Model 15-C POW Backpack Electrofisher;

Smith Root, Inc., Vancouver, Washington; Simonson and Lyons 1995). For all electrofishing, one netter used a 6.4-mm-mesh dip net to collect fish. Each macrohabitat was sampled separately. Seconds of electrofishing were recorded for each macrohabitat and were used to calculate the CPUE (fish per minute of electrofishing). All fish were identified to species, and TLs were recorded. All Burbot were scanned for PIT tags.

Habitat was quantified by measuring physical habitat features in each macrohabitat. Total length of each macrohabitat was measured along the thalweg. If the macrohabitat length was 30 m or less, two transects at 25% and 75% of the length were established; if the macrohabitat length was over 30 m, transects were established at 25, 50, and 75% of the length (Quist et al. 2003). At each transect, wetted stream width, depth, current velocity, and substrate particle size were measured at four equidistant points and at the midpoint (i.e., 20, 40, 50, 60, and 80%; Platts 1983). Both benthic and mean current velocities were taken with a portable velocity meter (Hach FH950 Handheld Flow Meter; Hach Company, Loveland, Colorado). Benthic velocity was measured at 0.03 m above the substrate. Mean current velocity was measured at 60% of the depth when depths were 0.75 m or less; velocity was measured at 20% and 80% of depth and averaged when depths were greater than 0.75 m (Buchanan and Somers 1969). Substrate was visually estimated at each point and classified as wood, clay (<0.004 mm), silt (0.004–0.063 mm), sand (0.064–2.000 mm), gravel (2.001–16.000 mm), coarse gravel (16.001–64.000 mm), cobble (64.001–256.000 mm), boulder (>256 mm), and bedrock (i.e., modified Wentworth scale; Cummins 1962; Sindt et al. 2012). Canopy cover (%) was estimated at each transect by using a concave densiometer, facing each bank at the stream margin and facing upstream and downstream at the midpoint of the channel (Sindt et al. 2012). Distance from each bank to the nearest anthropogenic disturbance was visually estimated at each transect (<10 m from the bank, ≥10 m from the bank, or no disturbance). Bank characteristics, including the percent coverage of woody vegetation, nonwoody vegetation, roots, boulders, eroding ground, and bare ground, were visually estimated for both banks at each transect. All instream cover at least 0.3 m in length was quantified by taking one length measurement, three width measurements, and three depth measurements. Instream cover was classified as undercut bank, overhanging vegetation, branch complex, log complex, rootwad, boulder, aquatic vegetation, and other (Quist et al. 2003).

For each macrohabitat, area was estimated by multiplying the thalweg length by the mean width. Mean depth, current velocity, canopy cover, and bank coverage percentages were calculated for each macrohabitat unit. The coefficient of variation (CV) was also calculated (CV = 100 × [SD/mean]) for depth, wetted stream width, mean current velocity, and canopy cover. The proportions of each substrate, the distance to anthropogenic disturbance category, and the instream cover type were also quantified for each macrohabitat unit. Habitat characteristics were averaged across macrohabitats. Averaged values were weighted by the proportion of the total reach area represented by that macrohabitat (Sindt et al. 2012). Weighted values were summed to quantify habitat characteristics for the entire reach. Additional variables were created by summing two or more habitat variables (e.g., $\text{Substrate}_{Coarse}$; Table 1). In addition to physical habitat measures, the distance from the nearest Burbot stocking location was calculated along the midpoint of the channel for each reach.

Burbot-specific habitat relationships.—Burbot-specific habitat relationships with presence–absence data and CPUE data were evaluated by using a hurdle regression modeling approach (Martin et al. 2005; Smith et al. 2016). Hurdle

TABLE 1. Means (SE in parentheses) of habitat variables measured from 58 reaches of Deep Creek, Idaho, during 2014 and 2015.

Variable	Description	Mean (SE)
Depth	Mean depth (m)	0.40 (0.14)
Depth_{CV}	Mean coefficient of variation (CV) of depth	43.7 (15.9)
Vel_{Mean}	Mean current velocity (m/s)	0.18 (0.11)
Vel_{CV}	Mean CV of current velocity	77.7 (28.4)
CanopyCover	Mean canopy cover (%)	14.5 (13.3)
CanopyCover_{CV}	Mean CV of canopy cover	144.1 (86.5)
$\text{Substrate}_{Coarse}$	Proportion of substrate that was coarse (coarse gravel, cobble, and boulder)	0.59 (0.35)
Cover_{Woody}	Proportion of reach area with branch complexes, log complexes, or rootwads as cover	0.04 (0.04)
Cover_{Veg}	Proportion of reach area with aquatic macrophytes or overhanging vegetation as cover	0.17 (0.21)
DistAnt	Proportion of banks with no anthropogenic disturbance	0.38 (0.39)
Width_{CV}	Mean CV of wetted channel width	16.5 (8.5)
Pool	Percentage of reach area as pool	3.5 (6.2)
DistStock	Minimum distance to the nearest Burbot stocking location (m)	4,458.3 (3,849.2)

models consisted of two submodels. One submodel used logistic regression to predict the probability of species presence for all reaches. The other submodel investigated relationships among species-specific CPUE and habitat characteristics (negative binomial error distribution) for reaches with at least one individual of the focal species (Maunder and Punt 2004; Martin et al. 2005).

Hurdle submodels were constructed using the “glm” (R Development Core Team 2008) and “zerotrunc” (Zeileis and Kleiber 2015) functions in program R. Species-specific models were created for Burbot. Model fit was assessed using McFadden’s pseudo-R^2 (McFadden 1974; Hosmer and Lemeshow 1989). McFadden’s pseudo-R^2 was calculated as 1 minus the difference in log likelihoods between a model with an intercept plus explanatory variables and the intercept-only model (McFadden 1974). McFadden’s pseudo-R^2 values vary from 0.0 to 1.0, with values greater than 0.20 indicating a good fit (Hox 2010; Mujalli and De Ona 2013).

Spearman’s correlation coefficient (r_S) was used to investigate relationships among habitat characteristics to reduce the risk of multicollinearity. Variables with an r_S value of 0.70 or greater were considered highly correlated. When two variables were highly correlated, the most ecologically important or interpretable variable was retained for consideration in a priori candidate models (Sindt et al. 2012; Smith et al. 2016). For example, mean current velocity was highly correlated ($r_S \geq 0.70$) with mean benthic velocity, the proportion of riffle macrohabitat, and the proportion of run macrohabitat. Mean current velocity was deemed the most ecologically important variable and was retained in candidate models; the other variables were removed. Eight candidate models were generated a priori for each submodel. Candidate models were ranked by using Akaike’s information criterion adjusted for small sample size (AIC$_c$; Burnham and Anderson 2002). Models having AIC$_c$ values within 2 units of the best-performing model were considered to have equal support and were retained for interpretation (Burnham and Anderson 2002).

RESULTS

In total, 7,127 individual fish representing 18 species and 7 families were sampled from 58 reaches (249 separate macrohabitats). Redside Shiner *Richardsonius balteatus* was the most abundant species, followed by Longnose Dace *Rhinichthys cataractae*, Largescale Sucker *Catostomus macrocheilus*, Torrent Sculpin *Cottus rhotheus*, Brook Trout *Salvelinus fontinalis*, and Rainbow Trout *Oncorhynchus mykiss*. Twenty-eight Burbot varying from 126 to 357 mm TL were sampled from 10 different reaches. Most Burbot were sampled in reaches within 1 km of a stocking location. The mean distance (±SD) to Burbot stocking locations from reaches where Burbot were sampled was 0.5 ± 0.8 km. The maximum distance between a sampled Burbot and a stocking location was 2.6 km, and the minimum distance was 0.06 km.

Logistic regression models indicated that the presence of Burbot was negatively related to the distance from the nearest Burbot stocking location and positively related to mean depth (Table 2). The second stage of the hurdle regressions (i.e., CPUE) indicated that Burbot catch rates were related to habitat characteristics in a manner similar to that observed for presence–absence data (Table 2). Burbot relative abundance was positively related to the proportion of pool macrohabitats and inversely related to the distance from the nearest Burbot stocking location. Models with coarse substrate were nearly included in the top set of models (i.e., those within 2 AIC$_c$ units of the best model) for both presence–absence and relative abundance of Burbot.

TABLE 2. Candidate models investigating Burbot presence–absence and relative abundance (number captured per minute of electrofishing) among stream reaches ($n = 58$) sampled in Deep Creek, Idaho, during 2014 and 2015 (variable abbreviations are defined in Table 1). Akaike’s information criterion adjusted for small sample size (AIC$_c$) was used to rank the candidate models. Delta AIC$_c$ (ΔAIC$_c$) is the difference between the AIC$_c$ value of the given model and the AIC$_c$ value of the top model. The total number of parameters (K), model weight (w_i), and McFadden’s pseudo-R^2 are also presented. Direction of effect for each habitat covariate is indicated (positive [+], negative [–]).

Model	AIC$_c$	ΔAIC$_c$	K	w_i	R^2
Presence–absence					
+ Depth – DistStock	29.04	0.00	3	0.53	0.55
– DistStock	30.80	1.76	2	0.22	0.47
+ Depth – Substrate$_{Coarse}$ + DistStock	31.35	2.31	4	0.17	0.55
+ Pool – DistStock	32.98	3.93	3	0.07	0.47
+ Depth + Vel$_{Mean}$ – Substrate$_{Coarse}$ – DistStock + Pool	36.23	7.19	6	0.01	0.55
+ Vel$_{Mean}$	50.82	21.78	2	0.00	0.07
+ Depth	51.41	22.37	2	0.00	0.06
– Substrate$_{Coarse}$	52.62	23.58	2	0.00	0.03
+ Pool	54.06	25.02	2	0.00	0.01
Relative abundance					
+ Pool	42.16	0.00	3	0.34	0.08
– DistStock	42.20	0.04	3	0.33	0.08
+ Vel$_{Mean}$	44.34	2.18	3	0.12	0.02
+ Depth	44.74	2.58	3	0.09	0.01
+ Substrate$_{Coarse}$	44.83	2.67	3	0.09	0.01
– DistStock + Pool	48.07	5.91	4	0.02	0.12
+ Depth – DistStock	49.37	7.21	4	0.01	0.08
+ Depth – DistStock + Substrate$_{Coarse}$	59.83	17.67	5	0.00	0.13
+ Depth + Vel$_{Mean}$ – DistStock + Substrate$_{Coarse}$ + Pool	154.87	112.71	7	0.00	0.15

DISCUSSION

Few Burbot were sampled from Deep Creek. Low occurrence and relative abundance may suggest poor survival. However, additional research indicates that survival of

Burbot released into Deep Creek is higher than that of Burbot released into the main-stem Kootenai River (Beard 2016; IDFG, unpublished data). Burbot were only sampled near release locations, and their occurrence and relative abundance were negatively associated with distance from the nearest stocking location. Stephenson et al. (2013) found that age-1 and younger Burbot remained in the Goat River (British Columbia), Boundary Creek (Idaho), and the Moyie River (Idaho) for an average of 1 year after stocking. Those authors also provided evidence that age-1 and younger Burbot had significantly shorter dispersal distances and longer dispersal times than age-2 and older Burbot. These results are similar to our findings and suggest that juvenile Burbot are slow to disperse after stocking.

With regard to habitat characteristics, Burbot were most commonly sampled and abundant in deep habitats situated close to Burbot release locations. Although not included in the suite of top models, a positive relationship between coarse substrate and Burbot relative abundance did have support. It is not surprising that Burbot occurrence and/or relative abundance was positively associated with mean depth and the proportion of coarse substrate, as these are often cited as important habitat characteristics for Burbot (Dixon and Vokoun 2009; Eick 2013; Klein et al. 2015). For example, Klein et al. (2015) showed that coarse substrate was an important predictor of Burbot occurrence and catch rates in the Green River, Wyoming. Similarly, Dixon and Vokoun (2009) found that Burbot occurrence was primarily correlated with coarse substrate, substrate embeddedness, and depth in Connecticut streams. Eick (2013) reported that Burbot preferentially used habitat with coarse substrate and high depth in laboratory experiments. Several studies have concluded that the interstitial spaces between coarse substrate provide refugia for Burbot (McMahon et al. 1996; Fischer 2000; Hoffman and Fischer 2002). Dixon and Vokoun (2009) suggested that substrate was most important for Burbot occurrence and that the importance of depth was conditional on the substrate type. Our data indicated the opposite, as Burbot in Deep Creek were more likely to occur in deep habitats regardless of substrate, but if coarse substrate was present, Burbot tended to occur in higher densities. Although stocking location was the most important factor associated with Burbot occurrence and relative abundance, this factor likely does not explain the associations with depth and coarse substrate. Burbot were generally sampled near stocking locations, but they still moved to areas of deep habitat. Additionally, release locations were selected in part because they possessed habitat similar to that identified as important in previous studies of Burbot habitat use. Thus, it may be that the release sites already provided suitable habitat, thereby reflecting the close association of distance to stocking location and Burbot occurrence and relative abundance.

An important consideration is that we used single-pass backpack electrofishing to sample fishes in the present study. Capture efficiencies estimated from single-pass backpack electrofishing vary among species and in relation to stream characteristics (Price and Peterson 2010; Meyer and High 2011). A failure to account for differences in capture efficiency may have resulted in underestimation or overestimation of the strength of Burbot–habitat relationships (Meyer and High 2011). Stream characteristics were generally conducive for backpack electrofishing, with depths typically less than 0.5 m and low proportions of instream cover (Table 1). Therefore, we argue that any bias in sampling was likely consistent throughout the study and that the data are adequate for evaluating general patterns in Burbot occurrence and relative abundance.

Our research illustrated that Burbot were only sampled near stocking locations, but depth and coarse substrate also influenced Burbot occurrence and relative abundance. Similar results were observed for other age-1 and younger Burbot released in different tributaries of the Kootenai River, suggesting that Burbot are slow to disperse after stocking. Based on these data, managers should consider focusing their Burbot stocking efforts on deep habitats with coarse substrate.

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