Survival, Movement Dynamics, Distribution, and Habitat and Species Associations of Juvenile Burbot in a Tributary of the Kootenai River

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Natural Resources

in the

College of Graduate Studies

University of Idaho

by

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December 2016

Authorization to Submit Thesis

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Abstract

Burbot *Lota lota* in the lower Kootenai River have been the recent focus of extensive conservation efforts, particularly the release of juvenile Burbot into small tributaries. Since 2012, approximately 12,000 juvenile Burbot have been released into Deep Creek, a small tributary of the Kootenai River. However, few Burbot have been detected at a passive integrated transponder (PIT) tag antenna on Deep Creek, thereby raising questions about the fate of Burbot released in the system. This research sought to evaluate survival, movement, distribution, and habitat and species associations of Burbot in Deep Creek. Survival of juvenile Burbot was estimated using mark-recapture analyses. Movement and distribution of Burbot were evaluated using stationary and mobile PIT tag antennas. I also identified habitat characteristics that were most closely related to the presence and relative abundance of Burbot. Results of this research will contribute to conservation and management of Burbot in the lower Kootenai River.

Acknowledgements

First, I would like to thank my advisor, Michael Quist. Mike's patience and guidance has ensured that I will be ready for a career as a fisheries professional. I also thank my committee members Daniel Schill, Christopher Caudill, and Timothy Johnson for their guidance and helpful suggestions. I owe a great deal of thanks to Ryan Hardy and T.J. Ross for their helpful suggestions, support, and guidance throughout my time in graduate school. I am very thankful to Cathy Gidley, Jeff Johnson, Landon Schofield, Mackenzie Miner and Shane Cossel for their assistance with field work. Without their assistance this work would not have been possible. Finally, I thank my fellow graduate students whose insight and support kept me focused, upbeat, and challenged me to rethink the how and why of my research.

Dedication

This work is dedicated to Karen Moritz and Doug Beard

Table of Contents

List of Tables

List of Figures

Chapter 1: General Introduction

Burbot *Lota lota maculosa* is the only freshwater member of the family Gadidae and has a circumpolar distribution. They occur across a diversity of lentic and lotic systems and function as top-level predators (McPhail and Paragamian 2000; Worthington et al. 2010). Burbot is a species of high conservation concern throughout its native distribution and many populations are in decline, especially in the southern portions of their distribution (Stapanian et al. 2010). Reasons for decline include alterations to habitat, overexploitation, interactions with nonnative species, and barriers to movement (Paragamian 2000; Stapanian et al. 2008; Stapanian et al. 2010). Despite the fact many Burbot populations are in decline, many populations are not actively managed (Paragamian and Willis 2000) and a lack of focused conservation actions is an issue worldwide (Maitland and Lyle 1990, 1996; Keith and Allardi 1996; Argent et al. 2000; Arndt and Hutchinson 2000; Paragamian et al. 2000). In systems where populations are stable, efforts have focused on maintaining populations, especially in systems where Burbot support important recreational fisheries (Quinn 2000). In systems where Burbot populations are in decline, substantial attention has focused on improving Burbot populations (Paragamian et al. 2000: Dillen et al. 2008; Ireland and Perry 2008; Stapanian et al. 2010; Neufeld et al. 2011).

In Idaho, Burbot is native only to the Kootenai River and its tributaries (Simpson and Wallace 1982; Wallace and Zaroban 2013). The Kootenai River has been highly altered since European settlement beginning with the construction of levees on the lower portion of the river in the late nineteenth century (Northcote 1973). Despite the construction of drainage ditches and the organization of the floodplain into drainage districts (Partridge 1983; Richards 1997) construction of Libby Dam in 1972 near Libby, Montana has likely had the greatest influence on the Kootenai River. Libby Dam altered the river's thermal, hydrologic, and nutrient regimes (Paragamian et al. 2000), and as a consequence, a shift in fish assemblage structure and deleterious effects on native riverine fishes have been documented (Paragamian et al. 2000; Paragamian et al. 2001; Paragamian 2002). However, the decline of Burbot in the lower Kootenai River began in 1959, prior to the construction of Libby Dam (Partridge 1983). Although the decline of Burbot began before construction of Libby Dam, the rate of decline increased after its construction (Paragamian et al. 2000). Historically, Burbot in the lower Kootenai River supported subsistence, recreational, and commercial fisheries and were important both economically and culturally (Paragamian and Hoyle 2003; Ireland and Perry 2008). In response to declining numbers of adult Burbot, recreational and commercial fisheries for Burbot were closed in Idaho and British Columbia in the 1990s (Paragamian et al. 2000). Despite closure of the fisheries, Burbot continued to decline and it was thought they may be extirpated from the lower Kootenai River system without intervention (Paragamian and Hansen 2009).

The Burbot population in the lower Kootenai River is genetically distinct from nearby populations (e.g., Kootenay Lake, upstream of Kootenai Falls, Duncan Reservoir, Trout River; Powell et al. 2008). Due to the unique genetic composition and population declines, the lower Kootenai River Burbot population was proposed for listing under the Endangered Species Act as a distinct population segment in 2000. However, the U.S. Fish and Wildlife Service determined that listing was not warranted (U.S. Federal Register 2003). Despite this ruling, a multiagency coalition consisting of the Kootenai Tribe of Idaho (KTOI), the Idaho Department of Fish and Game (IDFG), and the British Columbia Ministry of Forest, Lands, and Natural Resource Operation has implemented restoration efforts for Burbot in the lower

Kootenai River system. Current restoration efforts have primarily focused on conservation aquaculture, with the development of both intensive and extensive techniques (Jensen et al. 2008; Paragamian and Hansen 2009; Paragamian and Hansen 2011; Paragamian et al. 2011). Conservation aquaculture activities by the KTOI and the University of Idaho have been practiced at a relatively small scale with ~ 73,000 juvenile Burbot (i.e., excluding larval Burbot releases) released since 2009 (University of Idaho, unpublished data). A large hatchery operated by the KTOI was opened in October 2014, with the goal releasing 125,000 age-0 Burbot (i.e., post-larval fish) annually by 2019. During its first year of operation, the hatchery released about 253,000 six-month-old juvenile Burbot into the Kootenai River system. Various stocking strategies (i.e., fish size, number of fish, timing, location) have been and will be employed in future years. However, one strategy of particular interest is the release of fish into small tributary streams. Data suggest that Burbot in the Kootenai River, Idaho, and Kootenay Lake, British Columbia, have an adfluvial life history, moving between Kootenay Lake and the Kootenai River to use small tributaries in the basin for spawning (Paragamian 1995). The goal of small tributaries releases is to reestablish spawning runs in small tributaries (Hardy and Paragamian 2013). Preliminary results from 2009 to 2012 suggested high survival from small tributary releases (Hardy and Paragamian 2013).

In 2012, IDFG implemented a project on Deep Creek, Idaho, to evaluate movement of stocked Burbot into the Kootenai River. The Idaho Department of Fish and Game constructed a fixed Passive Integrated Transponder (PIT) tag antennae array on Deep Creek in October 2012, approximately 7 km upstream of its confluence with the Kootenai River. Three thousand age-0 Burbot were implanted with PIT tags and released at two locations in Deep Creek. The first location was downstream of the town of Naples, Idaho, approximately 20 km from the mouth of Deep Creek. The second release location was at the outlet of MacArthur Lake, approximately 33 km from the mouth of Deep Creek. In 2013, 2,500 age-0 Burbot were released at the same two sites. From those releases, few Burbot have been detected at the array (IDFG, unpublished data). These data suggest that some fish survive, rear in the Deep Creek system, and then outmigrate to the Kootenai River. However, these data also raise a number of questions regarding the status of fish that have not been detected. Key questions include whether or not the remaining fish are alive and the characteristics of fish that died (e.g., age, effect of stocking location). Other important questions focus on the spatial distribution of survivors, their habitat use and selection, and outmigration patterns and characteristics. Understanding mortality rates, movement dynamics, habitat use, and species associations of Burbot released into tributaries is critical for ensuring that stocking practices are efficient and effective. To answer questions associated with introducing Burbot into Deep Creek, I had two objectives: (1) evaluate habitat use and species associations of stocked Burbot, and (2) describe the status, movement, and distribution of Burbot released in the Deep Creek System.

Thesis Organization

This thesis is composed of four chapters. The second chapter investigates habitat and species associations of juvenile Burbot and other fishes in Deep Creek. The third chapter evaluates survival, movement, and distribution of juvenile Burbot released in Deep Creek. The final chapter is a general conclusion that synthesizes the results of each chapter as they relate to conservation and management of Burbot in the lower Kootenai River.

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Chapter 2: Habitat and Species Associations of Juvenile Burbot and Other Fishes in a Tributary of the Kootenai River

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Abstract

Burbot *Lota lota* in the lower Kootenai River 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. Since 2012, approximately 12,000 juvenile Burbot have been stocked into Deep Creek, a small tributary of the Kootenai River; however, little is known about the habitat use of stocked Burbot. The objectives of this study were to evaluate habitat associations and species associations of juvenile Burbot and other fishes in Deep Creek. Fish and habitat were sampled from 58 reaches in Deep Creek. Species richness decreased with increased channel gradient. Nonmetric multidimensional scaling indicated that patterns in species richness were largely a function of channel gradient and associated habitat characteristics (e.g., current velocity). Both ordination and regression model results suggested that Burbot move little after stocking and were associated with areas with high mean depths and coarse substrate. Species-specific habitat relationships for other fishes in Deep Creek were generally reflective of the ecology of each species. This study provides insight on patterns of fish assemblage structure, as well as important information on the ecology of native and nonnative fishes in a western stream system. Lastly, this study provides additional knowledge on juvenile Burbot and suggests managers should consider selecting deep habitats with coarse substrate for stocking locations.

Introduction

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, understanding 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 in 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). Understanding habitat relationships may also be important for conservation aquaculture practices. 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 in Lake Ontario have focused on stocking fry and parr in streams with high gradients and large rocky substrate (Stanfield and Jones 2003). Despite their importance, predicting and understanding species-habitat relationships is often difficult because fish species occurrence is 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 Karr 1978; Lobb and Orth 1991; Rahel and Hubert 1991). The influence of instream, small-scale physical habitat features (e.g., depth, substrate composition, 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, gradient) are often able to explain substantial variation in fish assemblage structure (Rahel and Hubert 1991; Marsh-Matthews and Matthews 2000), understanding habitat at a small scale may be more useful because many management and conservation activities (e.g., stocking practices, habitat restoration) occur at smaller spatial scales.

Common approaches to restoration include habitat restoration, harvest moratoria, and conservation aquaculture. One species that has been the focus of extensive restoration efforts is Burbot *Lota lota*. Burbot is a species of high conservation concern throughout their native distribution (Stapanian et al. 2010). Alterations to habitat, overexploitation, interactions with nonnative species, and barriers to movement are cited as major factors contributing to their decline (Paragamian 2000; Stapanian et al. 2008; Stapanian et al. 2010). In Idaho, Burbot are native only to the Kootenai River and its tributaries (Simpson and Wallace 1982; Wallace and Zaroban 2013). Like most rivers in North America, the Kootenai River has been highly altered since European settlement. These alterations include the construction of levees and ditches on the lower portion of the river, and the organization of the floodplain into drainage districts (Northcote 1973; Partridge 1983; Richards 1997). However, construction of Libby Dam in 1972 near Libby, Montana has potentially had the greatest influence on the Kootenai River. Libby Dam has altered 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; Paragamian et al. 2001;

Paragamian 2002). Since 1959, the lower Kootenai River Burbot population has been in decline (Partridge 1983) and the rate of decline has increased since the 1970s (Paragamian et al. 2000). Traditionally, Burbot in the lower Kootenai River supported subsistence, recreational, and commercial fisheries (Paragamian and Hoyle 2003; Ireland and Perry 2008). Fisheries for Burbot were closed in Idaho and British Columbia in the 1990s (Paragamian et al. 2000). Despite closure of the fisheries, Burbot continued to decline and it was hypothesized 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 Kootenai River Burbot 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; Paragamian et al. 2011; Paragamian and Hansen 2011). Although a variety of stocking strategies (i.e., fish size, quantity, timing, location) have been employed, one strategy of particular interest is the release of fish into small tributary streams. Mainstem spawning in the Kootenai River has been documented, but other data suggest that Burbot in the Kootenai River in Idaho and Kootenay Lake, British Columbia have 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 reestablish spawning runs in those habitats (Hardy and Paragamian 2013). However, the stocking strategy was implemented with virtually no understanding of the habitat use and species associations of juvenile Burbot and other fishes in small tributaries. Understanding habitat use and species associations can aide in selecting stocking locations, and help ensure efficient and effective stocking practices. The objectives of this research were to investigate patterns in fish assemblage structure, and model the occurrence and relative abundance of juvenile Burbot and other fishes in Deep Creek, 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 south 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. Deep Creek is a 3^{rd}-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 2.1). Deep 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 passive integrated transponder (PIT) tag antenna that was installed by the Idaho Department of Fish and Game (IDFG; ~ 7 km from the confluence of Deep Creek with the Kootenai River) to monitor movement of stocked juvenile Burbot. The Idaho Department of Fish and Game began stocking Burbot in Deep Creek in 2011 at two stocking locations (Figure 2.1). Since 2012, approximately 3000 PIT-tagged juvenile Burbot have been stocked per year at the two stocking locations.

Downstream of McArthur Lake, Deep Creek averages between 8 m 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 Deep Creek. Deep Creek was listed on the Idaho §303(d) list of impaired waters for excessive sediment and elevated temperatures (IDEQ 2006). Deep Creek has five major tributaries (Brown, Fall, Ruby, Snow, and Trail creeks). All tributaries, except Snow Creek, enter Deep Creek in 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 Deep Creek are generally privately owned and include areas of wetlands, 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 2.1) during the summers (June – August) of 2014 – 2015. In 2014, twenty five reaches were randomly selected from Deep Creek. In 2015, 29 reaches were randomly selected and four reaches were based on known Burbot locations (i.e., portable PIT tag reader detection). Each reach was 35× 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 using single-pass DC electrofishing (Model 15-C POW 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 used to calculate catch per unit effort (CPUE; fish per minute of electrofishing). All fish were identified to species and total lengths were recorded. Fish that could not be identified in the field were preserved and transported to the laboratory for identification.

Habitat was quantified by measuring physical habitat features in each macrohabitat. Total length of each macrohabitat was measured along the thalweg. If the macrohabitat was ≤ 30 m, two transects at 25% and 75% of the length were established; if the macrohabitat was > 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 the midpoint (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, and at 20% and 80% of the depth when depths were > 0.75 m (Buchanan and Somers 1969). Substrate was 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 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, and no disturbance). Bank characteristics were visually estimated for both banks at each transect. Bank characteristics included the percent coverage of woody vegetation, nonwoody vegetation, roots, boulders, eroding ground, and bare ground. 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, root wad, 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 for depth, wetted stream width, mean current velocity, and canopy cover (CV = 100 × [standard deviation/mean]). The proportions of each substrate, distance to anthropogenic disturbance category, and instream cover type were also quantified for each macrohabitat unit. Habitat characteristics were averaged across macrohabitats to characterize habitat in each stream reach. Averaged values were weighted by the proportion of the total reach area represented by that macrohabitat. Weighted values were summed to quantify habitat characteristics for the entire reach. Additional variables were created by summing two or more habitat variables (Table 2.1). In addition to physical habitat measures, the distance to the nearest Burbot stocking location was calculated along the midpoint of the channel for each reach.

Fish assemblage—

An *a priori* investigation of the elevation profile of Deep Creek revealed three distinct sections with different average gradients: a lower section with low average gradient, a middle section with high average gradient, and an upper section with a moderate average gradient. Species richness (S) was calculated for each sample reach. Species richness values were plotted against the stream elevation profile of Deep Creek to visualize longitudinal patterns in species richness associated with channel gradient and stream section. Mean species richness values and standard deviations were calculated for each section of Deep Creek. Differences in species richness were compared among the three sections using a one-way analysis of variance (Ott and Longnecker 2010).

Fish assemblage relationships were investigated using non-metric multidimensional scaling (NMDS). Non-metric multidimensional scaling is an ordination technique that is widely used to describe fish assemblage patterns (e.g., Helms et al. 2005; Ruetz et al. 2007; Smith et al. 2015). Ordination stress was used to evaluate fit; final stress values less than 20.0 indicated a good fit of the ordination to the data (McCune and Grace 2002). All ordinations used a Bray-Curtis distance measure. Two separate NMDS ordinations were used to evaluate patterns in the fish assemblage. The first ordination used presence-absence data and the second used reach-specific CPUE data. Differences in fish assemblage structure among stream sections (i.e., lower, middle, upper) were evaluated with a permutational multivariate analysis of variance (PERMANOVA). If a significant difference ($P \leq 0.05$) among stream sections was observed, habitat vectors were fit onto the NMDS ordination with rotational vector fitting (Faith and Norris 1989). Spearman's correlation coefficient was used to test for significant correlations between NMDS axis scores for fish presence-absence and fish CPUE and stream habitat characteristics. A Bonferroni correction was used to maintain the family-wise error rate (Benjamini and Hochberg 1995); habitat variables that were significant ($P \leq 0.002$) with at least one ordination axis were displayed on the ordination. Ordinations were constructed using metaMDS in the vegan package in R (Oksanen et al. 2015). Permutational multivariate analysis of variance tests were performed separately for presence-absence and relative abundance ordinations using the ADONIS function in the vegan package in R.

Species-specific habitat relationship—

In addition to investigations of fish assemblage structure, species-specific habitat relationships with presence-absence data and CPUE data were evaluated using a hurdle regression modeling approach (Martin et al. 2005; Smith et al. 2015). Hurdle 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 all species found in at least 10% of the stream reaches (Smith et al. 2015; Watkins et al. 2015). Model fit was assessed McFadden's pseudo R^2 was also used to investigate model fit (McFadden 1974; Hosmer and Lemeshow 1989). McFadden's pseudo R^2 was calculated as one minus the difference in the log likelihood of a model with an intercept plus explanatory variables and the log likelihood of an 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 good fit (Hox 2010; Mujalli and de Ona 2013).

Spearman's correlation coefficient was used to investigate relationships among habitat characteristics to reduce the risk of multicollinearity. Variables with a correlation coefficient ≥ 0.70 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. 2015). For example, mean current velocity was highly correlated ($r \geq 0.70$) 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 retained in candidate models; the other variables were removed. Five to fifteen candidate models were generated *a priori* for each submodel. Candidate models were ranked using Akaike's Information Criterion adjusted for small sample size (AIC_c; Burnham and Anderson 2002). Models within two AIC_c values were considered to have equal support and retained for interpretation (Burnham and Anderson 2002).

Results

In total, 7,127 individual fishes 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 *Onchorhynchus mykiss* (Figure 2.2). In addition, 28 Burbot were sampled (Figure 2.2). Most Burbot were sampled in reaches within 1 km of a stocking location. The mean distance (± SD) to Burbot stocking locations for reaches where Burbot were sampled was 0.5 ± 0.8 km. The maximum distance away from a stocking location that a Burbot was sampled was 2.6 km and the minimum distance was 0.06 km. Species richness varied significantly ($F_{3,54} = 3.5$, $P = 0.02$) among stream sections with the lowest richness in the middle section (mean richness ± SE; 5.5 ± 0.5) followed by the upper (6.7 ± 0.2) and the lower sections (7.7 ± 0.5; Figure 2.3).

A stable NMDS ordination was fit to the species occurrence data (stress = 13.8; Figure 2.4). Permutational multivariate analysis of variance indicated that fish assemblage composition differed among stream sections with regard to species occurrence ($F_{2,55}$ = 10.7, P < 0.01). Based on occurrence, Northern Pikeminnow *Ptychocheilus oregonensis*, Largemouth Bass *Micropterus salmoides*, Black Bullhead *Ameiurus melas*, and Bluegill *Lepomis macrochirus* were most closely associated with the lower section (Figure 2.4). Mountain Whitefish *Prosopium williamsoni* and Torrent Sculpin were most common in the middle section, and Burbot, Tench *Tinca tinca*, and Brook Trout were most common in the upper section (Figure 2.4). In addition, the presence of Burbot was most closely associated with Tench and Brook Trout. Several habitat characteristics were significantly correlated with NMDS axis scores (Table 2.2). Mean depth, proportion of coarse substrate, proportion of instream woody cover, proportion of instream vegetative cover, and distance to the nearest Burbot stocking location were all significantly correlated with at least one axis of the fish presence-absence ordination (Table 2.2; Figure 2.4). Mean depth, proportion of instream woody cover, and proportion of instream vegetative cover were higher in reaches in the upper section, and the proportion of coarse substrate was higher in reaches in the middle section of Deep Creek (Figure 2.4).

Logistic regression models indicated that the relationship between probability of occurrence and habitat varied by species (Table 2.3). Moreover, the results corroborated patterns observed with the NMDS analysis. In general, logistic models appeared to have good fit and predict species occurrence for most species. The presence of Burbot was negatively related to distance to the nearest Burbot stocking location and positively related to mean depth. The presence of Largescale Sucker, Pumpkinseed, Redside Shiner, and Brook Trout was positively related to mean depth. The proportion of instream woody cover was positively related to the presence of Pumpkinseed and Brook Trout but negatively related to the presence of Rainbow Trout and Torrent Sculpin. The proportion of instream vegetative cover was positively related to the presence of Tench and negatively associated with the presence of Lake Chub. The presence of Torrent Sculpin and Lake Chub was positively associated with the proportion of coarse substrate. Brook Trout occurrence was negatively related to the proportion of coarse substrate.

A stable NMDS was fit to the CPUE data (stress = 10.8; Figure 2.5) and patterns were similar to those observed in the presence-absence ordination. Permutational multivariate analysis of variance indicated that assemblage composition differed among sections with regard to relative abundance ($F_{2,55} = 7.2$, $P < 0.01$). Ordinations of relative abundance indicated that Northern Pikeminnow were most abundant in the lower section (Figure 2.5). Longnose Dace and Torrent Sculpin were most abundant in the middle section; whereas, Burbot, Redside Shiner, Largescale Sucker, and Brook Trout were most abundant in the upper section. Habitat variables that were significantly correlated with at least one axis of the fish CPUE NMDS ordination included mean depth, mean current velocity, the proportion of coarse substrate, and the proportion of instream vegetative cover (Table 2.2; Figure 2.5). Both mean current velocity and the proportion of coarse substrate were highest in the middle section of Deep Creek. In addition, the relative abundance of species such as Longnose Dace and Torrent Sculpin were most closely associated with mean current velocity and coarse substrate (Figure 2.5). Mean depth and the proportion of instream vegetative cover were highest in the upper section of Deep Creek. The relative abundance of species such as Tench and Redside Shiner was most closely associated with deep habitats that contained vegetative cover (Figure 2.5).

The second stage of the hurdle regressions (i.e., CPUE) indicated that relationships with habitat characteristics varied among species (Table 2.4). Moreover, results corroborated relationships observed with the NMDS ordination. Burbot relative abundance was positively related to the proportion of pool macrohabitats and inversely related to the distance to the nearest Burbot stocking location. The relative abundance of Largescale Sucker, Pumpkinseed, Lake Chub, Redside Shiner, Tench, and Brook Trout was negatively associated with mean current velocity (Table 2.4). Instream vegetative cover was negatively associated with catch rates of Lake Chub, Longnose Dace, and Rainbow Trout; whereas, the relative abundance of Largescale Sucker, Pumpkinseed, Redside Shiner, and Tench was positively associated with instream vegetative cover. Torrent Sculpin, Longnose Dace, and Rainbow Trout catch rates were positively related to coarse substrate (Table 2.4).

Discussion

Both biotic (e.g., piscivores) and abiotic (e.g., gradient, temperature) characteristics influence fish assemblage structure and create discernible patterns in lotic systems (Rahel and Hubert 1991; Quist et al. 2005). For instance, many studies have shown a pattern of gradual addition of species over a longitudinal gradient (Kuehne 1962; Harrel et al. 1967; Evans and Noble 1979; Rahel and Hubert 1991; Quist et al. 2004). However, in Deep Creek we observed high species richness in the upper and lower sections, and low species richness in the middle section. Low species richness in the middle section was most likely due to the high channel gradient and related habitat characteristics. Patterns in fish assemblage structure associated with stream gradient have been commonly reported in lotic systems. For instance, Paller (1994) found that high species richness in headwater streams was related to a lack of steep elevations in coastal plain streams of South Carolina. Lyons (1996) found similar results where high-gradient streams in the Driftless Area of Wisconsin had many riffle-dwelling species and were distinguished from speciose low-gradient streams in the Southeastern Wisconsin Till Plains that had more pool-dwelling species. Maret et al. (1997) reported stream gradient as one of the landscape characteristics that was important for structuring the fish assemblages in the upper Snake River basin, Idaho and Wyoming. Waite and Carpenter (2000) found that fish assemblages were primarily structured by stream gradient, and that gradient-related patterns superseded patterns that were ecoregion specific for streams in the Willamette Basin, Oregon. Quist et al. (2004) also found patterns in fish assemblage structure associated with gradient in the Salt River basin, Idaho and Wyoming. Cutthroat Trout *Oncorhynchus clarkii* and Paiute Sculpin *Cottus beldingii* were located in stream reaches with high channel slopes and low species richness. Stream reaches with low gradient had high species richness. The middle section of Deep Creek had limited habitat diversity in comparison to the lower and upper sections.

It is important to consider that our study used single-pass electrofishing to sample fishes. Capture efficiencies estimated from single-pass backpack electrofishing vary by species and stream characteristics (Price and Peterson 2010; Meyer and High 2011). A failure to account for differences in capture efficiency may have resulted in under- or overestimation of the strength of fish-habitat relationships (Meyer and High 2011). However, we argue that the bias in sampling was likely consistent throughout the study and the data are adequate for evaluating general patterns in species occurrence and relative abundance. Patterns of species occurrence in different sections of Deep Creek reflect the ecology of each species. Species that occurred most frequently in the middle section of Deep Creek included Mountain

Whitefish and Torrent Sculpin. These species are generally considered fast-water species (Wallace and Zaroban 2013) and the middle section of Deep Creek was characterized by a high gradient, fast current velocities, and large substrate (i.e., riffles). In contrast, some native (i.e., Northern Pikeminnow and Redside Shiner) and non-native species (Tench, Largemouth Bass, Yellow Perch, Pumpkinseed) are generally more common in lentic systems or low-velocity habitats in lotic systems (Becker 1983; Zarkami et al. 2010; Wallace and Zaroban 2013). These species occurred most frequently either directly downstream of the McArthur Lake impoundment (i.e., upper section) or in the lower section of Deep Creek—areas characterized by low-velocity habitats. Brook Trout most frequently occurred in the upper section of Deep Creek, which was characterized by an intermediate gradient, deep habitats, fine substrates, and high proportions of instream vegetation. This observation was a bit surprising as Brook Trout have often been cited as most frequently occurring in high-gradient, cold streams (Maret et al. 1997; Walrath et al. 2016). However, the upper section contained many Beaver *Castor canadensis* pools that have been shown in other systems to provide excellent habitat for Brook Trout (Kozel and Hubert 1989; Behnke 2002).

Patterns in species occurrence were easily observed, but patterns associated with the relative abundance of fishes in Deep Creek were not as clear. Although the PERMANOVA indicated differences among sections, patterns in relative abundance were less obvious. For example, results of the NMDS ordination showed that species such as Torrent Sculpin and Longnose Dace were most abundant in the middle section of Deep Creek. These species are generally considered fast-water species (Wallace and Zaroban 2013), which likely explains their higher abundance in the middle section of Deep Creek. Species such as Redside Shiner, Largescale Sucker, Pumpkinseed, and Tench were most abundant in the upper section of Deep

Creek; a section characterized by deep habitats, fine substrate, aquatic vegetation, and low water velocities. These observations are concurrent with the results of other studies and contribute to our knowledge of the ecology of the species (Scott and Crossman 1973; Zarkami et al. 2010; Wallace and Zaroban 2013).

The patterns observed in fish assemblage structure are important for management of Deep Creek; however, we were particularly interested in understanding how these patterns related to Burbot. Burbot were only found in the upper section of Deep Creek and both their occurrence and relative abundance was negatively associated with distance to the nearest stocking location. Stephenson et al. (2012) 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 one year after stocking. The 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 were similar to our findings and suggest juvenile Burbot are slow to disperse after stocking. Burbot were most common at sites with Brook Trout and Tench. Because Tench were only found directly downstream from McArthur Lake, they are likely introduced on a regular basis from McArthur Lake. Brook Trout were only sampled in the upper section of Deep Creek. Thus, the combination of stocking location (both were in the upper section), position in the watershed, and introduction from McArthur Lake likely explains the observed biotic association with Burbot.

With regard to habitat characteristics, Burbot were most common and most abundant in deep habitats close to Burbot release locations. Additionally, the NMDS based are CPUE data provide evidence that Burbot abundance was associated with increasing proportions of coarse substrate. Moreover, coarse substrate as a predictor for Burbot relative abundance was within a delta AICc of three. 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 embeddness, and depth in Connecticut streams. Eick (2013) also 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 suggests the opposite in that Burbot were more likely to occur in deep habitats regardless of substrate, but if coarse substrate was present, Burbot tended to occur in higher densities. While stocking location was the most important factor associated with Burbot occurrence and relative abundance, it is unlikely that this explains the associations with depth and coarse substrate. While Burbot were generally sampled near stocking locations, they still moved 0.5 ± 0.8 km to areas of deep habitats.

Our research illustrated differences in fish assemblage structure among sections of Deep Creek. These differences appeared to be related to changes in stream gradient. A clear pattern of decreased species richness with increased gradient was apparent, with the lowest species richness observed in the middle section and the highest richness in the lower section of Deep Creek. In addition, species-specific habitat relationships indicated that habitat use varied by species, thereby suggesting that fish assemblage structure in Deep Creek is influenced by a diversity of abiotic characteristics. We would expect similar patterns in other western stream systems with similar fish assemblages. Burbot were only sampled near stocking locations, but depth and coarse substrate also influenced Burbot occurrence and relative abundance. These data suggest that managers might consider focusing their stocking efforts for Burbot on deep habitats with coarse substrate.

Acknowledgements

We thank J. Johnson, L. Schofield, L. Terraberry, G. Peck, V. Oliveras, M. Briggs, V. B. Nunez, A. Miera, K. Bush, and numerous Idaho Department of Fish and Game employees, particularly C. Gidley, for assistance with field work. We also thank the Kootenai Tribe of Idaho for providing the funding for raising Burbot and the Aquaculture Research Institute at the University of Idaho for rearing the Burbot stocked into Deep Creek. Funding for this project was provided by the Idaho Department of Fish and Game. Additional support was provided by the U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit. The Unit is jointly sponsored by the U.S. Geological Survey, University of Idaho, the Idaho Department of Fish and Game, and Wildlife Management Institute. This project was conducted under the University of Idaho Institutional Animal Care and Use Committee Protocol 2014-8. The use of trade, firm or product names is for descriptive purposes only and does not imply endorsement by the U. S. Government.

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Table 2.1. Mean and standard error (in parenthesis) of habitat variables collected from 58 reaches on Deep Creek, Idaho during 2014 and 2015. Habitat variables are organized by section (upper, middle, lower).

		Section		
Variable	Description	Upper	Middle	Lower
Depth	Mean depth (m)	0.45 (0.13)	0.31 (0.12)	0.37 (0.09)
$Depth_{CV}$	Mean coefficient of variation (CV) of depth	42.9 (16.6)	41.4 (15.1)	52.7 (11.6)
Vel_{Mean}	Mean current velocity (m/s)	0.14 (0.09)	0.25 (0.13)	0.28 (0.07)
Vel_{cv}	Mean CV of current velocity	85.0 (30.2)	62.8 (23.2)	68.8 (9.0)
CanopyCover	Mean canopy cover (%)	17.0 (14.4)	12.9 (10.4)	4.8 (5.6)
$CanopyCover_{CV}$	Mean CV of canopy cover	151.6 (88.5)	149.8 (75.4)	93.1 (91.7)
$Substrate_{Coarse}$	Proportion of substrate that is coarse (coarse gravel, cobble, and boulder)	0.45 (0.33)	0.84 (0.25)	0.84 (0.13)
$Cover_{Woody}$	Proportion of reach area with branch complexes, log complexes, or root wads as cover	0.05 (0.05)	0.02 (0.02)	0.03 (0.03)
$Cover_{Veg}$	Proportion of reach area with aquatic macrophytes or overhanging vegetation as cover	0.25 (0.23)	0.05 (0.08)	0.01 (0.01)
DistAnt	Proportion of banks with no anthropogenic disturbance	0.45 (0.40)	0.27 (0.35)	0.20 (0.35)
$Width_{CV}$	Mean CV of wetted channel width	15.8 (8.3)	16.0 (9.5)	21.4 (6.7)
Pool	Proportion of reach area as pool	2.7 (5.8)	4.4 (5.6)	6.4 (8.5)
DistStock	Minimum distance (m) to the nearest Burbot stocking location	2,457 (2,024)	5,985 (2,919)	11,984 (1,224)

Table 2.2. Spearman's correlation coefficients (ordination *P*-values in parenthesis) between stream habitat characteristics and nonmetric multidimensional scaling (NMDS) axis scores for fishes sampled in Deep Creek, Idaho during 2014 and 2015. The NMDS analyses were conducted using fish presence-absence (P-A) and fish catch per unit effort (CPUE; fish per minute of electrofishing). Values in bold were significant using a bonferroni correction (i.e., $P \leq 0.002$).

Variable	Fish P-A		Fish CPUE	
	NMDS1	NMDS2	NMDS1	NMDS2
Depth	**-0.62 (<0.01)**	0.06 (0.67)	-0.39 (<0.01)	**-0.44 (<0.01)**
$Depth_{CV}$	-0.01 (0.95)	0.21 (0.11)	0.01 (0.98)	0.34 (0.01)
Vel_{Mean}	0.38 (<0.01)	0.23 (0.09)	**0.63 (<0.01)**	0.19 (0.15)
Vel_{CV}	-0.35 (0.01)	-0.04 (0.77)	-0.37 (<0.01)	0.09 (0.50)
CanopyCover	-0.22 (0.10)	-0.21 (0.11)	-0.03 (0.84)	0.07 (0.58)
$CanopyCover_{CV}$	-0.04 (0.74)	0.08 (0.56)	0.16 (0.24)	-0.17 (0.21)
$Substrate_{Coarse}$	**0.41 (<0.01)**	0.29 (0.03)	**0.59 (<0.01)**	0.18 (0.18)
$Cover_{Woody}$	**-0.42 (<0.01)**	-0.07 (0.58)	-0.36 (0.01)	-0.03 (0.80)
$Cover_{Veg}$	**-0.52 (<0.01)**	-0.31 (0.02)	**-0.59 (<0.01)**	-0.28 (0.03)
DistAnt	-0.30 (0.02)	-0.15 (0.26)	0.07 (0.58)	-0.35 (0.01)
$Width_{CV}$	-0.09 (0.52)	0.01 (0.97)	-0.06 (0.65)	0.22 (0.09)
Pool	0.04 (0.74)	0.33 (0.01)	0.36 (0.01)	-0.26 (0.05)
DistStock	0.37 (<0.01)	**0.42 (<0.01)**	0.15 (0.26)	0.26 (0.05)

Table 2.3. Top models from the candidate sets investigating species occurrence among stream reaches (n = 58) sampled in Deep Creek, Idaho during 2014 and 2015. Akaike's Information Criterion (AIC_c) adjusted for small sample size ranked the candidate models. The total number of parameters (K), model weight (w_i), and McFaddens pseudo R^2 are included. Direction of effect for each habitat covariate is indicated (positive [+], negative [-]).

Taxa	Model name	AIC_c	ΔAIC_C	K	w_i	R^2
Catostomidae						
Largescale Sucker	+ Depth	62.35	0.00	2	0.24	0.06
	+ Pool	64.06	1.57	2	0.10	0.03
	+ Depth + Substrate$_{Coarse}$	64.33	1.98	3	0.09	0.06
Centrarchidae						
Pumpkinseed	+ Depth	68.50	0.00	2	0.31	0.06
	+ DistAnt	69.44	0.94	2	0.20	0.05
	+ Cover$_{Woody}$	69.94	1.44	2	0.15	0.04
Cottidae						
Torrent Sculpin	- Depth + Substrate$_{Coarsse}$	39.61	0.00	3	0.23	0.38
	- Depth - Cover$_{Woody}$	39.75	0.14	3	0.21	0.38
	+ Vel$_{Mean}$	39.84	0.23	2	0.20	0.33
	- Depth	40.69	1.08	2	0.13	0.32
Cyprinidae						
Lake Chub	+ Substrate$_{Coarse}$	42.50	0.00	2	0.35	0.10
	- Cover$_{Veg}$	42.75	0.25	2	0.31	0.10
	- Vel$_{Mean}$ + Substrate$_{Coarse}$	44.19	1.69	3	0.15	0.12
Northern Pikeminnow	+ Vel$_{Mean}$	76.39	0.00	2	0.23	0.06
	+ Pool + Substrate$_{Coarse}$	76.87	0.48	3	0.18	0.09
	- Cover$_{Woody}$	77.41	1.02	2	0.14	0.05
	+ Pool + Depth + Substrate$_{Coarse}$	78.28	1.90	4	0.09	0.10
Redside Shiner	+ Vel$_{CV}$	28.81	0.00	2	0.45	0.28
	+ Depth	30.05	1.25	2	0.24	0.24
Tench	+ Cover$_{Veg}$	46.15	0.00	2	0.40	0.21
	- Vel$_{Mean}$ + Cover$_{Veg}$	47.98	1.82	3	0.16	0.22
	- CanopyCover + Cover$_{Veg}$	48.01	1.86	3	0.16	0.22
Gadidae						
Burbot	+ Depth - DistStock	29.04	0.00	3	0.53	0.55
	- DistStock	30.80	1.76	2	0.22	0.47
Percidae						
Yellow Perch	+ Depth	37.77	0.00	2	0.42	0.13
	+ Cover$_{Woody}$	38.48	0.71	2	0.30	0.11

Table 2.3 cont'd

Salmonidae						
Brook Trout	+ Depth - $\text{Substrate}_{\text{Coarse}}$ + $\text{Cover}_{\text{Woody}}$	53.94	0.00	4	0.67	0.43
	+ Depth + $\text{Cover}_{\text{Woody}}$	55.72	1.78	3	0.28	0.37
Rainbow Trout	- Width_{CV}	40.22	0.00	2	0.26	0.07
	- $\text{Cover}_{\text{Woody}}$	41.89	1.66	2	0.11	0.02

Table 2.4. Top models from the candidate sets investigating relative abundance of species in relation to habitat characteristics sampled from reaches (n = 58) in Deep Creek, Idaho during 2014 and 2015. Akaike's Information Criterion (AIC_c) adjusted for small sample size ranked the candidate models. The total number of parameters (K), model weight (w_i), and McFaddens pseudo R^2 are included. Direction of effect for each habitat covariate is indicated (positive [+], negative [-]).

Taxa	**Model Name**	**AIC_c**	**ΔAIC_C**	***K***	**w_i**	**R^2**
Catostomidae						
Largescale Sucker	- Pool	302.90	0.00	3	0.19	0.01
	+ $Width_{CV}$	303.62	0.72	3	0.13	0.01
	- Vel_{Mean}	304.25	1.35	3	0.10	0.01
	- $Cover_{Woody}$	304.50	1.60	3	0.09	0.01
	- $Cover_{Woody}$ + $Width_{CV}$	304.56	1.66	4	0.08	0.01
Centrarchidae						
Pumpkinseed	+ $Cover_{Veg}$	40.60	0.00	3	0.18	0.06
	+ CanopyCover	40.73	0.13	3	0.17	0.06
	- Vel_{Mean}	41.58	0.98	3	0.11	0.03
	+ Depth	41.61	1.01	3	0.11	0.03
	+ $Cover_{Woody}$	41.61	1.01	3	0.11	0.03
	+ $Velocity_{CV}$	41.73	1.13	3	0.10	0.03
	+ DistAnt	42.58	1.98	3	0.07	0.01
Cottidae						
Torrent Sculpin	+ $Substrate_{Coarse}$ - Vel_{CV}	287.10	0.00	4	0.29	0.03
	+ $Substrate_{Coarse}$	288.34	1.34	3	0.15	0.02
Cyprinidae						
Lake Chub	- Vel_{Mean}	25.80	0.00	3	0.46	0.10
	- $Cover_{Veg}$	26.69	0.89	3	0.29	0.03
	- $Substrate_{Coarse}$	27.04	1.24	3	0.25	0.01
Longnose Dace	- $Cover_{Veg}$	476.60	0.00	3	0.26	0.06
	- Depth - $Cover_{Veg}$	476.94	0.34	4	0.22	0.06
	- Depth + $Substrate_{Coarse}$	477.65	1.05	4	0.16	0.06
	- Depth + Vel_{Mean} - $Cover_{Veg}$ + $Substrate_{Coarse}$ + CanopyCover + $Width_{CV}$ + Vel_{CV} - Pool	477.71	1.11	10	0.15	0.09
Northern Pikeminnow	- CanopyCover	109.20	0.00	3	0.27	0.03
	+ Vel_{Mean}	110.44	1.24	3	0.15	0.02

Table 2.4 cont'd

Redside Shiner	+ $Cover_{Veg}$ - $Substrate_{Coarse}$	547.70	0.00	4	0.37	0.04
	- Vel_{Mean} + $Cover_{Veg}$	548.48	0.78	4	0.25	0.04
Tench	- Vel_{Mean}	38.30	0.00	3	0.48	0.09
	+ $Cover_{Veg}$	40.23	1.93	3	0.18	0.03
Gadidae						
Burbot	+ Pool	42.10	0.00	3	0.34	0.08
	- DistStock	42.14	0.04	3	0.33	0.08
Percidae						
Yellow Perch	- Depth	29.40	0.00	3	0.53	0.32
Salmonidae						
Brook Trout	- Vel_{Mean}	105.80	0.00	3	0.58	0.08
	- Vel_{Mean} + $Substrate_{Coarse}$	107.14	1.34	4	0.30	0.10
Rainbow Trout	+ $Substrate_{Coarse}$	265.00	0.00	3	0.17	0.01
	- $Cover_{Veg}$	265.48	0.48	3	0.13	0.01
	+ $Width_{CV}$	265.79	0.79	3	0.12	0.01
	+ Depth - $Cover_{Veg}$	266.63	1.63	4	0.08	0.01
	+ Depth + $Substrate_{Coarse}$	266.69	1.69	4	0.07	0.01
	- Vel_{Mean} +$Substrate_{Coarse}$	266.80	1.80	4	0.07	0.01

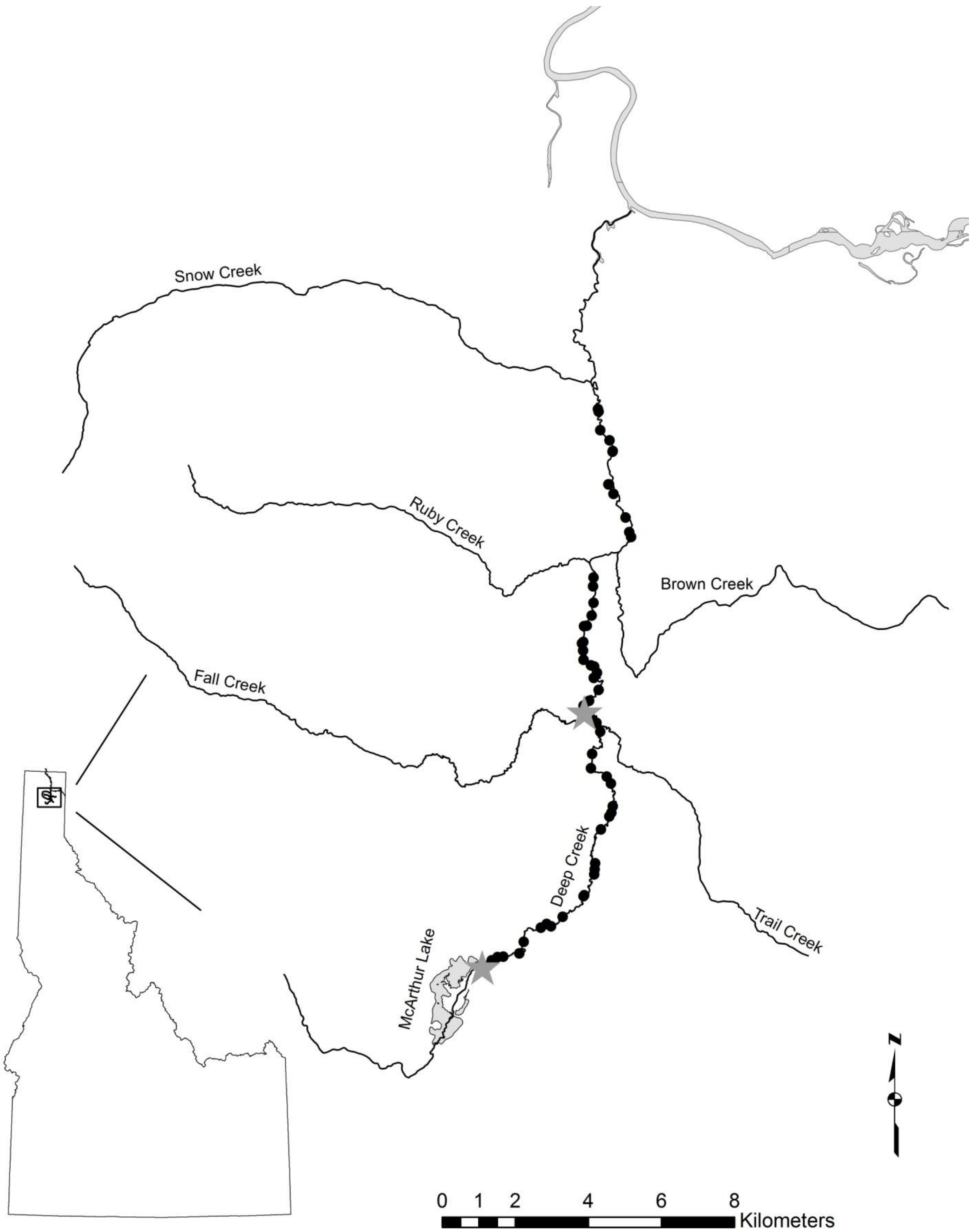

Figure 2.1. Deep Creek watershed showing the main channel of Deep Creek, Deep Creek's five major tributaries (i.e., Trail, Fall, Ruby, Brown, and Snow creeks), and the Idaho portion of the Kootenai River. Circles represent one of the 58 sampled stream reaches on Deep Creek. Stars represent the two Burbot stocking locations.

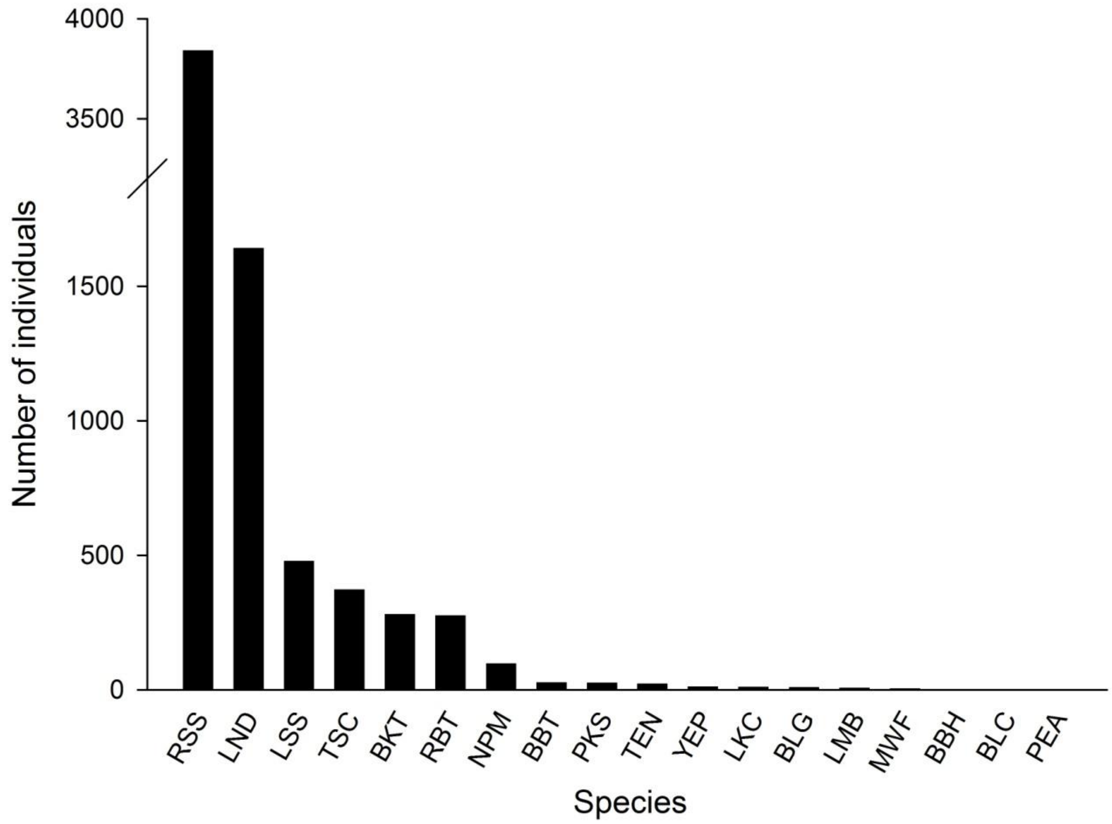

Figure 2.2. Total number of individuals of each species sampled from Deep Creek during the summers of 2014 and 2015. Taxa include Brown Bullhead (BBH), Burbot (BBT), Brook Trout (BKT), Black Crappie (BLC), Bluegill (BLG), Lake Chub (LKC), Largemouth Bass (LMB), Longnose Dace (LND), Largescale Sucker (LSS), Mountain Whitefish (MWF), Northern Pikeminnow (NPM), Peamouth (PEA), Pumpkinseed (PKS), Rainbow Trout (RBT), Redside Shiner (RSS), Tench (TEN), Torrent Sculpin (TSC), and Yellow Perch (YEP).

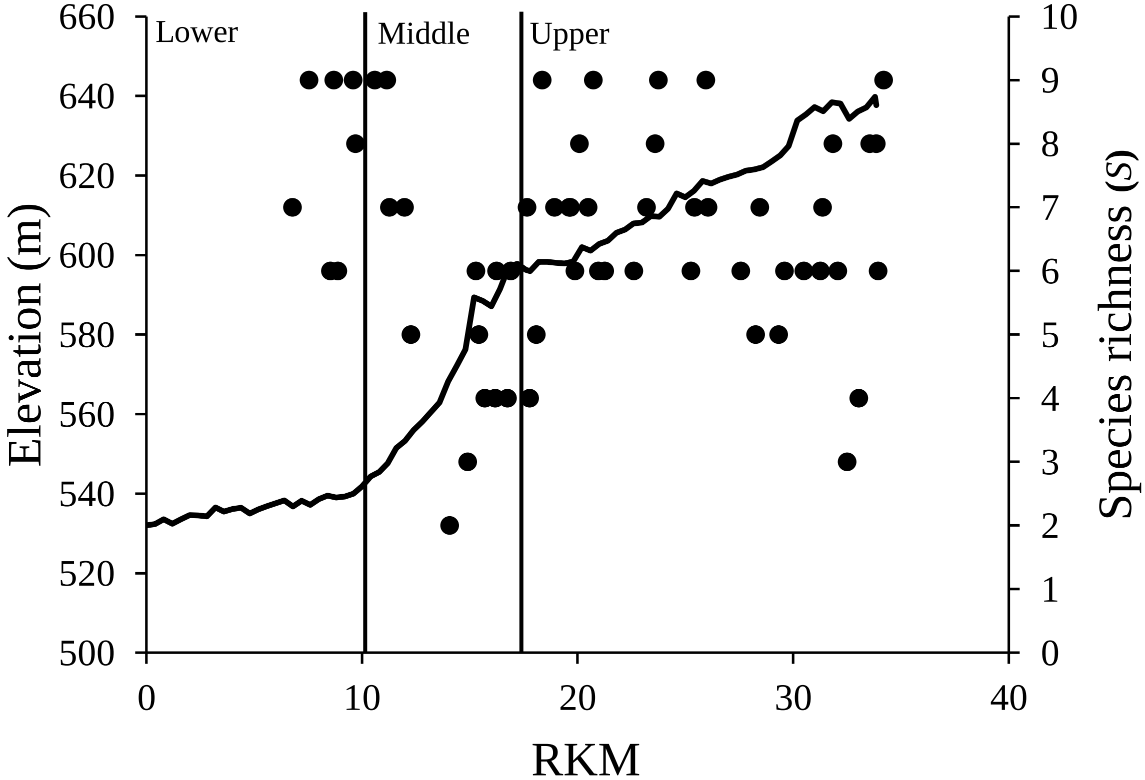

Figure 2.3. A stream elevation (m) profile of Deep Creek from the McArthur Lake impoundment to its confluence with the Kootenai River (RKM 0). Vertical bars represent the breaks between the lower, middle, and upper sections of Deep Creek. Circles represent the species richness value for each of the 58 sampled reaches in Deep Creek.

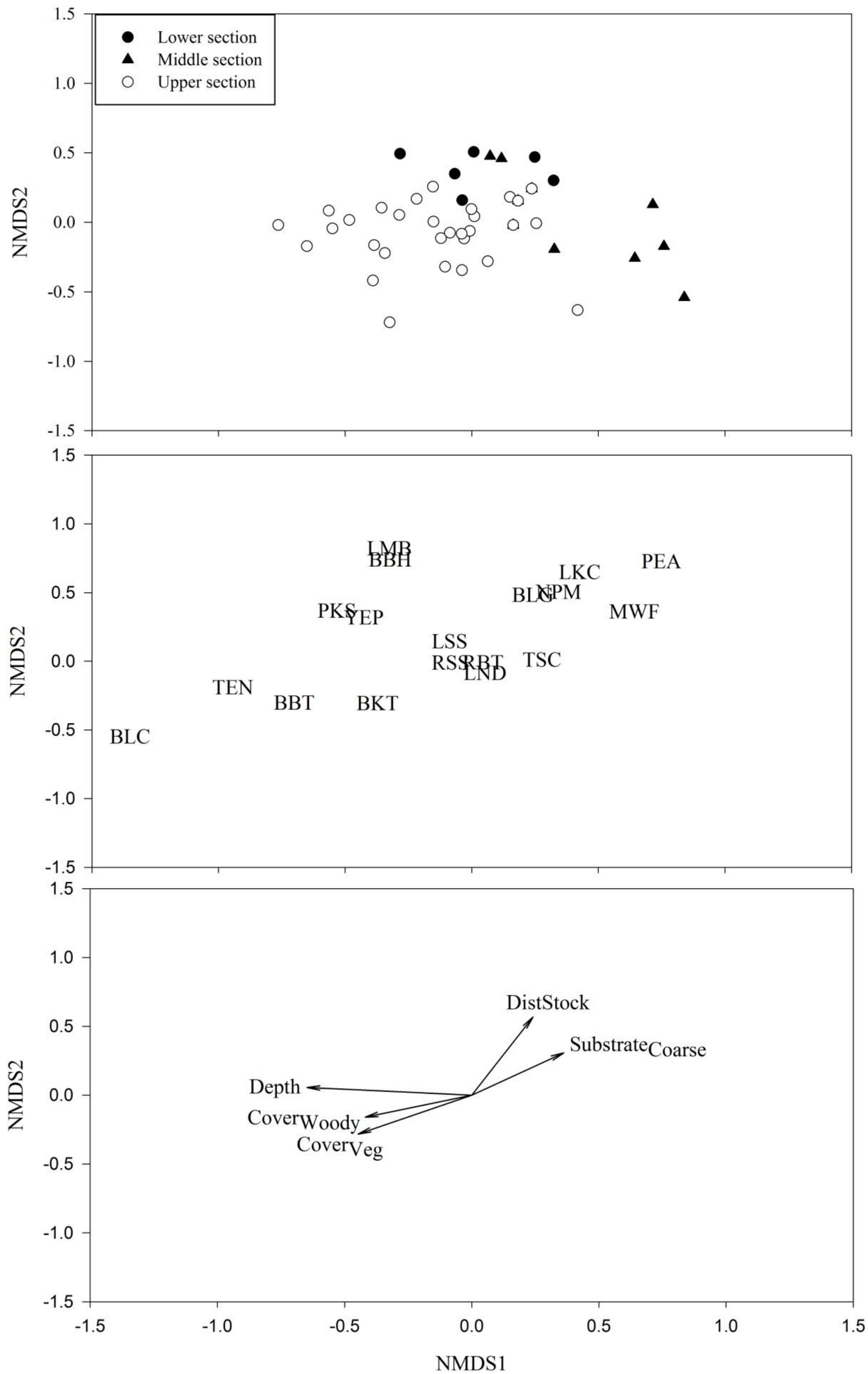
Lower section
Middle section
Upper section
NMDS2
LMB
BBH
PEA
LKC
BLG
NPM
MWF
PKS
YEP
LSS
RSS
RBT
LND
TSC
TEN
BBT
BKT
BLC
DistStock
Substrate Coarse
Depth
Cover Woody
Cover Veg
NMDS1
-1.5
-1.0
-0.5
0.0
0.5
1.0
1.5

Figure 2.4. Non-metric multidimensional scaling ordination (stress = 13.8) of reach-specific (*n* = 58) fish assemblage presence-absence data from Deep Creek organized by section (upper, middle, lower). Species scores are displayed in the middle figure, and taxa include Brown Bullhead (BBH), Burbot (BBT), Brook Trout (BKT), Black Crappie (BLC), Bluegill (BLG), Lake Chub (LKC), Largemouth Bass (LMB), Longnose Dace (LND), Largescale Sucker (LSS), Mountain Whitefish (MWF), Northern Pikeminnow (NPM), Peamouth (PEA), Pumpkinseed (PKS), Rainbow Trout (RBT), Redside Shiner (RSS), Tench (TEN), Torrent Sculpin (TSC), and Yellow Perch (YEP). Significant ($P < 0.002$) habitat vectors were fit to the ordination and include minimum distance to the nearest Burbot stocking location, proportion of coarse substrate, mean depth, proportion of instream woody cover, and the proportion of instream vegetative cover.

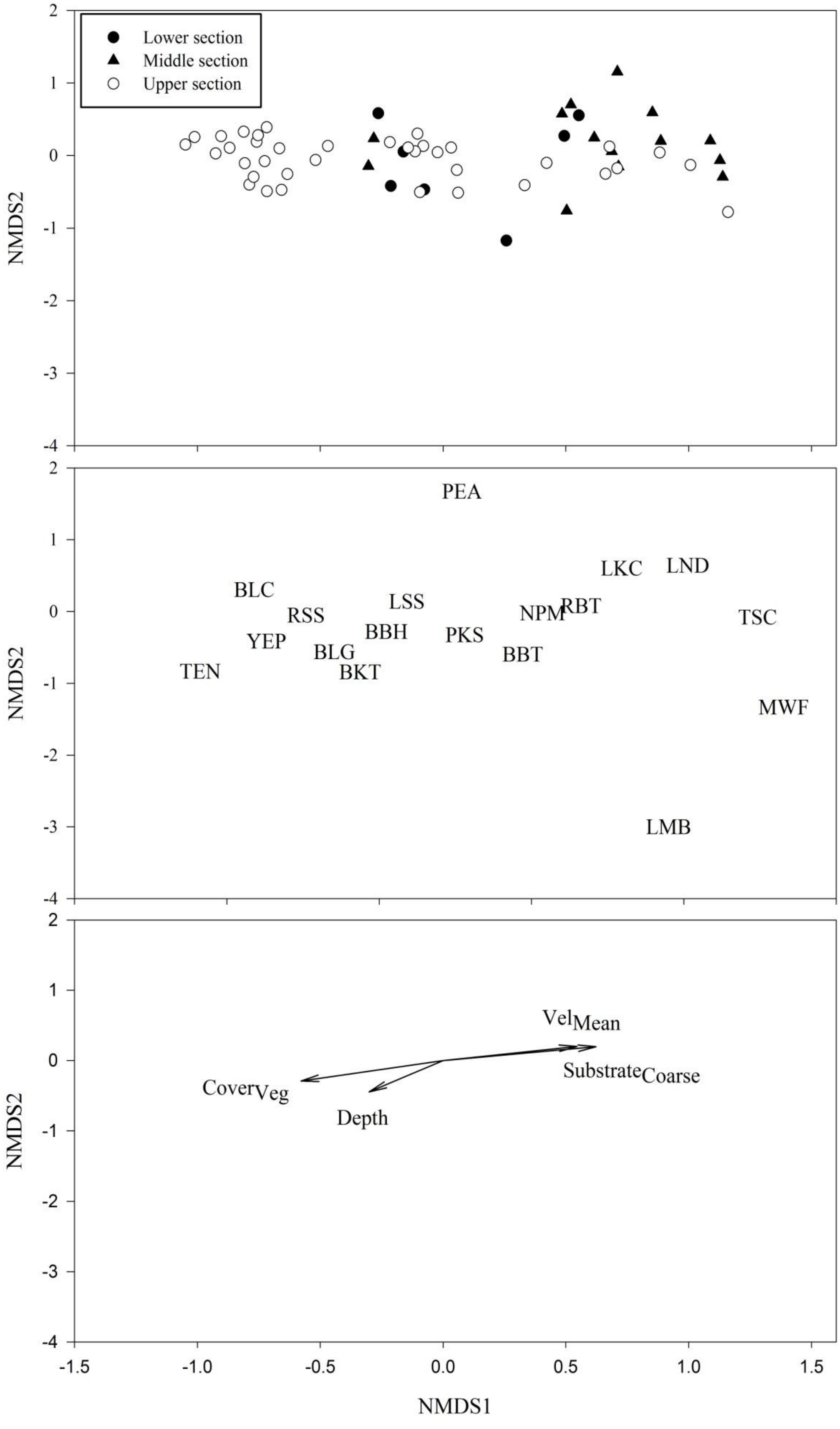
Lower section
Middle section
Upper section
NMDS2
PEA
LKC
LND
BLC
LSS
RSS
NPM
RBT
TSC
YEP
BBH
PKS
BLG
BBT
TEN
BKT
MWF
LMB
Vel Mean
Substrate Coarse
Cover Veg
Depth
NMDS1

Figure 2.5. Non-metric multidimensional scaling ordination (stress = 13.8) of reach-specific (n = 58) fish assemblage relative abundance data from Deep Creek organized by section (upper, middle, lower). Species scores are displayed in the middle figure, and taxa include Brown Bullhead (BBH), Burbot (BBT), Brook Trout (BKT), Black Crappie (BLC), Bluegill (BLG), Lake Chub (LKC), Largemouth Bass (LMB), Longnose Dace (LND), Largescale Sucker (LSS), Mountain Whitefish (MWF), Northern Pikeminnow (NPM), Peamouth (PEA), Pumpkinseed (PKS), Rainbow Trout (RBT), Redside Shiner (RSS), Tench (TEN), Torrent Sculpin (TSC), and Yellow Perch (YEP). Significant ($P < 0.002$) habitat vectors were fit to the ordination and include the proportion of coarse substrate, mean current velocity, mean depth, and the proportion of instream vegetative cover.

Chapter 3: Survival, Movement, and Distribution of Juvenile Burbot in a Tributary of the Kootenai River

Zachary S. Beard, Michael C. Quist, Ryan S. Hardy, Tyler J. Ross

Abstract

Burbot *Lota lota maculosa* in the lower Kootenai River have been the focus of extensive conservation efforts, particularly the release of hatchery-reared juvenile Burbot into small tributaries. The Idaho Department of Fish and Game installed a fixed passive integrated transponder (PIT) antenna on Deep Creek, a tributary of the Kootenai River, to evaluate movement of juvenile Burbot to the Kootenai River. Since then, approximately 12,000 juvenile Burbot have been PIT tagged and released into Deep Creek, but few Burbot have been detected at the antenna. The lack of detections raised questions about the fate of Burbot in Deep Creek. The objectives of this study were to evaluate survival, movement, and distribution of Burbot released into Deep Creek. In 2014, 3,000 age-0, 200 age-1, 16 age-2, and 16 age-4 Burbot were released at two different locations in Deep Creek. In 2015, 3,000 age-0 Burbot were released at six different locations (i.e., 500 per site) in Deep Creek. Additional stationary PIT tag antennas were installed on Deep Creek prior to stocking in 2014. Mobile PIT tag antennas were used to survey Deep Creek in 2015 and 2016. A Barker model in program MARK was used to estimate survival from mobile and stationary PIT tag antenna data. Few Burbot were detected at stationary PIT tag antennas. Mobile PIT tag antenna surveys relocated 758 tags, 88% of which were within 1 km of a release location. Mobile PIT tag antenna surveys of release locations in Deep Creek suggest poor dispersal from stocking locations. Survival estimates varied from (0.10 to 0.29) and did not

significantly differ between year or release location. Results of this study suggest that managers might consider releasing Burbot at multiple locations and lower densities.

Introduction

Burbot *Lota lota maculosa* are the only freshwater member of the family Gadidae and have a circumpolar distribution. In North America, Burbot are found throughout Canada, Alaska, and the northern tier of the continental United States. Burbot are a species of high conservation concern throughout their native distribution (Stapanian et al. 2010). In Eurasia, only four of twenty-four countries reported “secure” Burbot populations in a review of worldwide Burbot population status (Stapanian et al. 2010). Thirteen countries reported Burbot populations that were imperiled, declining, or vulnerable to extirpation, and Burbot have been extirpated from Belgium, the United Kingdom, and parts of Germany (Stapanian et al. 2010; Worthington et al. 2010). In the United States, eight of twenty-five states reported having “secure” Burbot populations, eleven states reported populations that were either imperiled or vulnerable to extinction, and Burbot have been extirpated from Kansas and Nebraska (Stapanian et al. 2010). Reasons for decline include alterations to habitat, overexploitation, interactions with nonnative species, and barriers to movement (Paragamian 2000; Stapanian et al. 2008; Stapanian et al. 2010).

In Idaho, Burbot are native only to the Kootenai River and its tributaries (Simpson and Wallace 1982; Wallace and Zaroban 2013). Like most rivers in North America, the Kootenai River has been highly altered since European settlement. Anthropogenic alterations began with the construction of levees on the lower portion of the river in the late 19th century (Northcote 1973). By 1935, over 90% of Idaho’s portion of the Kootenai River floodplain was organized into drainage districts (Partridge 1983; Richards 1997). However, construction of Libby Dam in 1972 near Libby, Montana has potentially had the greatest influence on the Kootenai River. Libby Dam has altered the river's thermal, hydrologic, and nutrient regimes (Paragamian et al. 2000) all of which have had deleterious effects on native riverine fishes (Paragamian et al. 2000; Paragamian et al. 2001; Paragamian 2002). Since 1959, the lower Kootenai River Burbot population has been in decline (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). Recreational and commercial fisheries for Burbot were closed in Idaho and British Columbia in the 1990s (Paragamian et al. 2000). Despite closure of the fisheries, Burbot continued to decline and it was thought that they may be extirpated from the lower Kootenai River system in less than a decade without intervention (Paragamian and Hansen 2009).

A multiagency coalition consisting of the Kootenai Tribe of Idaho (KTOI), the Idaho Department of Fish and Game (IDFG), and the British Columbia Ministry of Forest, Lands, and Natural Resource Operation has begun restoration efforts for Burbot in the lower Kootenai River. Intensive and extensive conservation aquaculture techniques have been developed and are the current focus of restoration efforts (Jensen et al. 2008; Paragamian and Hansen 2009; Paragamian et al. 2011; Paragamian and Hansen 2011). Conservation aquaculture activities by the KTOI and University of Idaho have been practiced at a relatively small scale with approximately 73,000 juvenile Burbot released since 2009 (University of Idaho, unpublished data). A hatchery operated by the KTOI was opened in October 2014, and has greatly increased the number of Burbot released into the system. In its first year of operation, about 253,000 juvenile Burbot were released into the Kootenai River basin. Although a variety of stocking strategies (i.e., fish size, number of fish, timing, location) have been and will be employed, one strategy of particular interest is the release of fish into small tributary streams. Data suggest that Burbot in the Kootenai River, Idaho, and Kootenay Lake, British Columbia, historically expressed a variant adfluvial life history, moving freely between Kootenay Lake and the Kootenai River to use small tributaries in the basin for spawning (Paragamian 1995). Additionally, previous work suggested that the standard operations of Libby Dam inhibits Burbot spawning migrations and made the mainstem Kootenai River less suitable for Burbot (Paragamian 2000; Paragamian et al. 2005; Paragamian and Wakkinen 2008) The goal of releases in small tributaries is to reestablish spawning runs in tributaries (Hardy and Paragamian 2013).

In 2012, IDFG implemented a project on Deep Creek, Idaho, to evaluate movement of stocked Burbot into the Kootenai River. The Idaho Department of Fish and Game constructed a fixed Passive Integrated Transponder (PIT) tag antennae array on Deep Creek in October 2012, near its confluence with the Kootenai River. Three thousand age-0 Burbot in 2012 and 2,500 age-0 Burbot in 2013 were implanted with PIT tags and released at two locations upstream of the IDFG PIT tag antenna. From those releases, 59 were detected at the array in 2012, 77 were detected at the array in 2013, and 33 were detected in 2014 (IDFG, unpublished data). These data raise a number of questions regarding the status of fish that have not been detected. Key questions include whether or not the remaining fish are alive and the characteristics of fish that died (e.g., effect of stocking location). Other important questions focus on the spatial distribution and movement of survivors. Understanding mortality rates, movement dynamics, and spatial distribution of Burbot released into tributaries is critical for ensuring that stocking practices are efficient and effective. Thus, our objectives were to estimate survival for Burbot stocked in Deep Creek, as well as describe their movement and spatial distribution in the system.

METHODS

Study area

The Kootenai River has an international watershed of approximately 45,600 km^2, primarily located within the province of British Columbia, with smaller portions located in Montana and Idaho (Knudson 1994). The Kootenai River originates in Kootenay National Park, British Columbia, and initially flows south into Montana before turning west into Idaho. From Idaho it flows back north into Canada where it enters the Columbia River. Many small tributaries contribute to the Kootenai River; one of the largest is Deep Creek. Deep Creek is a 3^{rd}-order stream that originates east of White Mountain, Idaho. Deep Creek is impounded approximately 10 km from its headwaters to form McArthur Lake (Figure 3.1). Deep Creek flows 33 km north from McArthur Lake to its confluence with the Kootenai River about 5 km west of Bonners Ferry, Idaho. The study area included the portion of Deep Creek between the McArthur Lake impoundment and a PIT tag antenna installed by IDFG (7 km from the confluence of Deep Creek with the Kootenai River; R5 in Figure 3.1).

Downstream of McArthur Lake, Deep Creek averages about 10 m in width and is dominated by cobble and gravel substrates. However, directly downstream of the dam that forms McArthur Lake, Deep Creek is dominated by deep pools and fine substrate. The impoundment has a major influence on water quality in Deep Creek. Deep Creek was listed on the Idaho §303(d) list of impaired waters for excessive sediment and elevated temperatures (IDEQ 2006). Deep Creek has five major tributaries (Brown, Fall, Ruby, Snow, and Trail creeks) that enter Deep Creek in the study area (except for Snow Creek). 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 Deep Creek are generally privately owned and include areas of wetlands, agriculture, residential development, and forest (IDEQ 2006).

Stocking

In 2014, 3,000 age-0, 200 age-1, 16 age-2, and 16 age-4 Burbot were implanted with half-duplex (HDX) PIT tags and released on October 30, at two different locations in Deep Creek (i.e., McArthur and Naples; Figure 3.1). Due to concerns regarding poor survival, the stocking strategy was altered in 2015. Three thousand age-0 Burbot (i.e., 500 per site) were measured for total length, implanted with HDX PIT tags, and released at six different stocking locations (i.e., McArthur, Shiloh, Naples, 2nd Bridge, Swimming Hole, Resort; Figure 3.1) on October 30, 2015. Stocking locations in 2015 were categorized as either a high- (i.e., Naples, 2nd Bridge, Swimming Hole; Figure 3.1) or moderate-quality habitat (i.e., McArthur, Shiloh, Resort; Figure 3.1). High-quality locations were those dominated by deep pools and large substrate within 1 km of the release location. Deep habitats with large substrate are commonly reported as important habitat characteristics for Burbot (Dixon and Vokoun 2009; Eick 2013; Klein et al. 2015, see Chapter 2). For example, Dixon and Vokoun (2009) found that Burbot occurrence was correlated with coarse substrate, substrate embeddness, and depth in Connecticut streams. Similarly, Klein et al. (2015) reported that coarse substrate was an important predictor of Burbot occurrence and catch rates in the Green River, Wyoming. Furthermore, Eick (2013) reported that Burbot used habitat with coarse substrate and high depth in laboratory experiments. Moderate-quality habitat locations lacked deep habitats with large substrate within 1 km of the release location.

Stationary antennas

Prior to stocking in 2014, five stationary half-duplex antennas were installed on Deep Creek (Figure 3.1). Antenna locations were selected to ensure a broad spatial coverage of the system and were partly constrained by stream characteristics, accessibility, and land owner cooperation. Each HDX array consisted of a 141.9 L cooler, inside of which were four 12-volt batteries (connected in parallel; 126 ampere-hours per battery; Interstate Batteries, Dallas, Texas) and the HDX PIT tag reader-data logger (Oregon RFID, Portland, Oregon). Each cooler was placed above the high water mark at each site. Twinax cable connected the reader to an antenna-tuning box (Oregon RFID, Portland, Oregon). On both sides of the stream, a 10.2 cm diameter wooden post was partially buried; the tuning box was attached to the top of one of the posts. Antenna wire exited the tuning box, formed a loop around the stream, and returned to the tuning box (i.e., pass-through design). All of the antennas consisted of a single loop of 6 American wire gauge (AWG) class K welding cable. Polypropylene rope was stretched between each wooden post to provide support for the top of each antenna loop. The antenna wire was run through 1.9 cm polyvinyl chloride (PVC) pipe to protect the bottom of the antenna loop. Polyvinyl chloride pipe was secured to the substrate using a combination of rebar stakes and duckbill anchors.

Antennas operated continuously from October 30, 2014 to February 5, 2015. On February 5, a high flow event damaged all five antennas and they were inoperable until May 11, 2015 when they were reinstalled. After reinstallation, antennas operated continuously until the end of the study on July 3, 2016. Upon reinstallation, 140-watt solar panels (Solartech Power, Inc., Ontario, California) were added and the number of 12-volt batteries was reduced from four to two (connected in parallel; 104 ampere-hours per battery; Sun Xtender, West Covina, California). In addition, PIT tag antennas of a pass-over design were installed in the fall of 2015 at each site to prevent damage from high-flow events. Pass-over antennas consisted of wire that exited the tuning box and formed a loop on the bottom of the stream before returning to the tuning box. Three pass-over antennas consisted of two loops of 12 AWG 19-strand thermoplastic high heat-resistant nylon-coated (THHN) wire. The remaining two pass-over antennas consisted of a single loop of 10 AWG solar photovoltaic wire. Antenna wire was run through 1.9 cm PVC pipe to protect the antenna and the PVC pipe was secured to the stream bottom using a combination of rebar and duckbill anchors. On December 15, 2015, antenna operation at all sites was changed from pass-through design antennas to the pass-over design antennas.

The efficiency of each antenna was thoroughly tested by conducting detection tests. For the pass-through antennas, a PIT tag was passed through each antenna at 50 cm intervals across Deep Creek on both a horizontal and vertical plane (Compton et al. 2008). The tag was passed through three times at each location; on the first pass it was positioned parallel to the antenna, on the second pass it was positioned at 45° to the antenna, and on the third pass it was oriented perpendicular to the antenna. For the pass-over antennas, a PIT tag was also passed over the antenna at different depths (i.e., 50 cm intervals; bottom of the water column, mid-water column, and at the surface). In addition, pass-through and pass-over antennas at each site were operated together for two weeks to provide another estimate of antenna efficiency. Antenna efficiency estimated from large-scale detection tests for pass-through designs varied from 74 to 100%. Efficiency estimates for pass-over design antennas from large scale detection tests varied from 56 to 97%. During the two weeks pass-through and pass-over antennas were operating continuously, all pass-over antennas with the exception of R2 (see Figure 3.1 for reader locations and abbreviations) had equal or greater efficiency than the pass-through antennas.

A temperature logger (Onset, Cape Cod, Massachusetts) was installed at each stationary antenna on October 23, 2014 and recorded temperature every hour for the duration of the study. The temperature logger at R1 (see Figure 3.1 for reader locations and abbreviations) was lost and only recorded water temperature through September 13, 2013. In addition, the temperature logger at R2 had several periods where it was out of the water; those data were removed from the analysis. Two water level data loggers (Onset, Cape Cod, Massachusetts) were installed on April 23, 2015 at R3 and at R5, and recorded water level every hour. Stream discharge was measured multiple times in 2015 and 2016 at each water level logger location and a regression between water level and stream discharge was used to estimate discharge for both sites (Bower 2005).

Mobile interrogation

Two mobile PIT tag surveys of Deep Creek were completed during 2015 and 2016. The first mobile survey was conducted from May 26 to June 23, 2015 and sampled Deep Creek from the McArthur Lake impoundment to the IDFG PIT tag antenna. The second mobile survey of Deep Creek was conducted over the same area from May 17-31, 2016. The second mobile survey of Deep Creek also included sampling the four major tributaries (i.e., Trail, Fall, Ruby, Brown Creeks) up to the first major barrier to fish passage. Both of these surveys are hereafter referred to as the "longitudinal distribution surveys". The first mobile antenna was Oregon RFID's pole antenna for their backpack reader (Oregon RFID, Portland, Oregon). The second mobile PIT tag antenna consisted of the antenna described by Fischer et al. (2012) mounted to an inflatable pontoon raft (The Creek Company, Steamboat Springs, Colorado). Both antennas were used to continuously scan the entire length of the study area in Deep Creek while operators waded in a downstream direction. When a tag was encountered, global positioning system coordinates were taken and an attempt was made to disturb the fish to determine its status (i.e., dead or alive). A Burbot was considered to be alive if (*i*) it was disturbed and observed alive, (*ii*) moved upstream from the last observation, or (*iii*) when a tag was disturbed and moved > 1 m (Breen et al. 2009). If the tag was continually relocated but did not meet any of these criteria, it was assigned a shed or dead fate. If the tag could not be relocated after being disturbed, it was assigned an unknown fate.

Data from the 2015 longitudinal distribution survey of Deep Creek suggested low survival and dispersal of Burbot from release locations. These findings led us to question how quickly mortality of stocked Burbot occurred (i.e., immediate or slowly over time), how Burbot disperse from the release locations over time, and how mortality and dispersal may differ by stocking location. As such, mobile PIT tag antenna surveys were conducted twice at each of the six 2015 release locations to monitor dispersal of Burbot after release. Mobile surveys at release locations were conducted from November 2-6 and January 21-27. Additional surveys were attempted but could not be conducted due to unsafe ice cover (winter) and high discharge (spring). Hereafter, both of these surveys will be referred to as "release location surveys". Release location surveys were 2 km in length centered on the stocking location. Two kilometer long reaches were chosen because data from the 2015 longitudinal distribution survey of Deep Creek indicated that most tags (88%) were located within 1 km of where they were stocked. Release location surveys were conducted in the same manner as the longitudinal distribution surveys of Deep Creek. Two mobile PIT tag antennas were used to continuously scan from the upstream beginning of the reach to the downstream end. One antenna was the Oregon RFID antenna described previously. The other antenna had a 25.4 centimeter diameter ring on the end (antenna) and used 18 AWG THHN wire enclosed in PVC casing. When a tag was encountered, the protocol from the longitudinal distribution surveys was followed.

Efficiency for longitudinal distribution surveys and release location surveys was estimated by attaching PIT tags to rocks and conducting blind searches (Bubb et al. 2002). For each blind search 30 PIT tags were placed beneath rocks in similar positions to where Burbot are normally found. Antenna operators, who had no prior knowledge of where tags were hidden, then scanned the reaches where the tags were hidden using the methods described for the longitudinal distribution and release location surveys. Blind searches were conducted five times during the 2015 longitudinal distributional survey, three times during the 2016 longitudinal distribution survey, and two times during the November 2015 and January 2016 release location surveys. The mean estimate of efficiency (± SD) for the 2015 longitudinal distribution survey was 60.9 ± 13.5% and 51.7 ± 5.8% for the 2016 longitudinal distribution survey. Efficiency for the November 2015 release location surveys was 63.5 ± 6.4% and 57.8 ± 5.5% for the January 2016 release location surveys.

Data analysis and summarization

To summarize movement, the distance moved upstream and downstream was calculated for each Burbot. Distance moved upstream was calculated as the furthest distance upstream that a Burbot was detected alive from its release location. Similarly, distance moved downstream was calculated as the furthest distance downstream a Burbot was detected alive from its release location. In addition, the total number of detections and individual Burbot detected at each antenna were summarized by stocking location. Burbot detected at R5 were considered to have outmigrated from the study area. The percentage of detections that occurred during the day and at night was also calculated for each reader. Night was defined as one half-hour after official sunset to one half-hour before official sunrise. The number of detections and individual Burbot detected per month was plotted against mean daily temperature for each stationary antenna to examine patterns in movement associated with temperature. Discharge estimated from the water level logger at R3 was plotted with the number of detections and individual Burbot by month for R1, R2, and R3. Discharge estimated from the water level logger at R5 was plotted with the number of detections and individual Burbot by month for both R4 and R5. The number of tags relocated per 1 km segment for each longitudinal distribution survey was depicted using maps to visualize movement and distribution of Burbot in the system. For data collected from the release location surveys (surveys of 2 km reaches centered on release locations), a time series of maps depicting the number tags relocated every 50 m for each stocking location reach was used to visualize how Burbot dispersed from release locations.

Relative survival estimates were calculated from both longitudinal distribution surveys of Deep Creek as the number of Burbot confirmed alive divided by the total number of tags found. Ninety-five percent confidence intervals were calculated for relative survival estimates using the simple confidence interval for sample proportions (Vollset 1993). The effect of total length at release on survival was evaluated by calculating the mean total length at release for Burbot released in 2015 (only fish in 2015 were individually measured prior to release) and recaptured during the 2016 longitudinal distribution survey. In addition to relative survival estimates, a Barker extension to the joint live-dead encounter model in Program MARK (Barker 1997; Barker 1999; White and Burnham 1999; Al-Chokhachy and Budy 2008) was used to estimate survival (S) for Burbot released in Deep Creek in 2014 and 2015. The Barker model can incorporate capture-recapture data from individual sampling occasions, as well as recapture data between sampling occasions, thereby improving the precision of estimates of survival over models that only incorporate recapture data from sampling occasions (Barker 1999). The Barker model can also provide estimates of recapture probability (p), the probability of resighting a dead animal (r), the probability of recapturing an animal between sampling intervals (R), the probability of recapturing an animal before the animal dies between sampling intervals (R'), the probability an animal at risk of capture in time i is at risk of capture at time $i + 1$ (F), and the probability an animal not at risk of capture at time i is at risk of capture at time $i + 1$ (F'; Barker 1999).

We used our stationary antenna data (pass-through and pass-over antenna recaptures), and the data from all mobile PIT tag antenna surveys (longitudinal distribution surveys and release location surveys) of Deep Creek for the Barker model analyses. We defined four detection events over the course of our study. The first detection event was the initial release in 2014 (October 30, 2014), the second detection event was the 2015 longitudinal distribution survey of Deep Creek (May 26 - June 23, 2015), the third event was the 2015 stocking in combination with the November 2015 release location surveys (October 30 - November 6, 2015), and the fourth detection event was the 2016 longitudinal distribution survey of Deep Creek (May 17-31, 2016). In addition, stationary antenna recaptures during the intervals between detection events were incorporated as live-resightings.

To evaluate *S* across stocking locations and release strategies, we established a model that included group (release locations) and age effects for *S* and *P* (Lebreton et al. 1992). Age effects resulted in estimates of *S* for the first 7 months Burbot were in Deep Creek. For Burbot released in 2014, an additional *S* parameter was estimated for the time period after the initial 7 months in Deep Creek. Other variables that were less pertinent to our analysis (*F*, *F´*, *R*, and *R´*) were estimated as time dependent (Lebreton et al. 1992). We used the likelihood function in Program MARK to estimate the slope (β) for all parameters and a logit link function to transform β estimates into interpretable estimates of *S* (Al-Chokhachy and Budy 2008). In addition, the Markov Chain Monte Carlo parameter estimation procedure in Program MARK was used to improve the precision of estimates of *S* and provide 95% credible intervals (White 2008).

Results

Overall numbers of detections and individuals at antennas were low, with the exception of R1 and R2, which were close to Burbot release locations. In total, R1 had the most detections from the most individuals followed by R2, R4, R3, and R5 (Table 3.1). Burbot released in 2014 were most commonly detected at antennas close to stocking locations. Burbot released at the McArthur release location in 2014 were detected primarily at R1 (50 m from the McArthur release location) with a few detections at the other antennas (Table 3.1). Burbot released at Naples in 2014 were most commonly detected at R4 (1,910 m downstream of the Naples release location) followed by R3 (1,000 m upstream of Naples release location; Table 3.1). Similar patterns were observed with detections for Burbot released in 2015 at both high- and moderate-quality release locations. For example, Burbot released at all three high-quality locations were detected most at R4 followed by R3 with few detections at other antennas. In addition, Burbot released at the Shiloh location were detected most at R2 (70 m from Shiloh release location) with few detections at other antennas. Few Burbot were detected at R5 out-migrating from Deep Creek (Table 3.1). While age-2 and older Burbot released in 2014 are not included in Table 3.1, 81% were detected out-migrating from Deep Creek during the first month. The mean maximum distance traveled upstream for Burbot released in 2014 was 135 m (± SD; ± 435 m) and downstream was 288 m (± 1,533 m). For Burbot released at high-quality locations in 2015, the mean maximum distance traveled upstream was 781 m (± 1,606 m) and downstream was 171 m (± 426 m). Burbot released at moderate-quality locations had a mean maximum upstream total distance of 301 m (± 1,046 m) and 106 m (± 468 m) downstream.

The majority of Burbot detections occurred at night. The percent of detections that occurred at night were 52% at R1, 43% at R2, 86% at R3, 66% at R4, and 74% at R5. Plots of detections and individuals against temperature did not show consistent patterns (Figure 3.2). However, for Burbot stocked in 2014, the number of detections at R1, R3, R4, and R5 increased with decreasing temperatures in the fall of 2015. In addition, Burbot released in 2015 were more commonly detected at all antennas as water temperatures were warming from January to March 2016 (Figure 3.2). Plots of detections and individuals against discharge showed different patterns between years (Figure 3.3). Burbot released in 2014 were more commonly detected at all antennas when discharge was low. However, Burbot released in

2015 were more commonly detected at R1, R4, and R5 in March, April, and May when flows were high (Figure 3.3).

During the 2015 longitudinal distribution survey, tags were only relocated in seven 1 km segments and all were within 3 km of a release location (Figure 3.4). In addition, 88% of all tags and 88% of tags with an alive fate were relocated within 1 km of a release location. During the 2016 longitudinal distribution survey, tags were relocated throughout Deep Creek (Figure 3.4). Nevertheless, 88% of tags and 91% of tags with an alive fate were relocated within 1 km of a release location. Additionally, three tags were relocated in tributaries of Deep Creek during the 2016 longitudinal distribution survey, all of which had a shed or dead fate.

During the November 2015 release location surveys, the majority of tags (85-96% of tags) relocated at high-quality locations and were within 200 m of the release location (Figure 3.5). Furthermore, during the January 2016 release location surveys and the 2016 longitudinal distribution survey at high-quality release locations, tags were evenly distributed throughout the 2 km reach. Patterns in distribution of relocated tags between high- and moderate-quality reaches from release location surveys and the 2016 longitudinal distribution survey were similar (Figure 3.6). For example, at moderate-quality release locations during the November 2015 release location surveys, the majority of tags (73-100% of tags) were relocated within 200 m of release locations. Additionally, during both the January 2016 release location surveys and the 2016 longitudinal distribution survey, the majority of relocated tags at two moderate-quality locations (i.e., Shiloh and Resort; Figure 3.6) were evenly distributed throughout the 2 km reach.

Data from the 2015 longitudinal distribution survey of Deep Creek provided relative survival estimates for Burbot released in 2014 of 0.15 (95% confidence interval; 0.09-0.22) for the McArthur release location and 0.07 (0.01-0.13) for the Naples release location (Figure 3.7). Relative survival estimates from the 2016 longitudinal distribution survey for Burbot released in 2014 were similar to those from the 2015 longitudinal distribution survey. Relative survival estimates from the 2016 longitudinal distribution survey for Burbot released in 2015 varied from 0.18 to 0.39 among release locations (Figure 3.7). Mean total lengths (± SD) at release for Burbot relocated with an alive fate during the 2016 longitudinal distribution survey was 100.8 ± 8.2 mm, 100.3 ± 7.4 mm for Burbot with a shed or dead fate, and 100.7 ± 7.7 mm for Burbot with an unknown fate.

Initial 7-month survival estimates from the Barker model for Burbot released in 2014 were 0.22 (95% credible interval; 0.17-0.29) and 0.17 (0.12 – 0.21) for the McArthur and Naples release locations (Figure 3.7). Recapture probabilities varied from 0.16 to 0.33 for Burbot released in 2014. After the first 7 months, survival improved to 0.56 (0.34 - 0.73) and 0.68 (0.36 – 0.88) for McArthur and Naples release locations, respectively. Initial 7-month survival estimates for Burbot stocked at the six locations in 2015 were similar and varied from 0.10 to 0.29 (Figure 3.7). Recapture probabilities for Burbot released in 2015 varied from 0.38 to 0.62.

Discussion

Results of the 2015 longitudinal distribution survey of Deep Creek suggested little movement of Burbot away from release locations. Low numbers of detections at stationary antennas also provided evidence that Burbot moved little after release into Deep Creek. The 2015 longitudinal distribution survey of Deep Creek provided further evidence that survival was relatively low. We hypothesized that the two release locations provided suitable habitat for juvenile Burbot and fish were not motivated to move great distances. We also hypothesized that high densities of fish at a stocking location may have attracted predators and (or) decreased per capita resource availability. As such, some release locations with a lack of high-quality habitat were selected in 2015 in an attempt to "force" Burbot to move. Additionally, Burbot were stocked in lower numbers across more release locations in 2015 to reduce the risk of predation and limited prey availability. Despite changes in release strategies from 2014 to 2015, Burbot released in Deep Creek continued to move short distances from release locations and experience similar survival rates (< 30%).

During the 2015 longitudinal distribution survey, all tags were relocated within 3 km of a release location and 88% of all tags were relocated within 1 km of a release location. Results from the 2016 longitudinal distribution survey were nearly identical, as 91% of tags with an alive fate and 88% of all tags were relocated within 1 km of a release location. While low detection efficiencies may partially explain patterns observed with movement of Burbot, results of the mobile surveys and detections at stationary antennas suggest that Burbot move little after being released into Deep Creek. Many tags were assigned a "shed or dead fate". Ashton et al. (2014) evaluated PIT tag retention in age-0 Burbot and found that retention rates were 99 ± 1%. As such, it is likely that the majority, if not all tags assigned a shed or dead fate in our study were from dead Burbot. Additionally, the proportion of tags with an "alive fate" that were relocated within 1 km of a release location was similar to the proportion of tags relocated within 1 km of a release location, suggesting similar distribution patterns for live and dead Burbot.

Reasons for the lack of movement by Burbot after release into Deep Creek remain unclear. One reason may be that habitat at all sites was suitable for juvenile Burbot and fish did not need to move far to locate suitable habitat. However, this explanation seems unlikely given the observation that movement of Burbot stocked into locations with varying habitat quality was similar. Another reason Burbot did not move after being released into Deep Creek may be that the life history of Burbot is such that they simply do not move much during their first year. Two previous studies have attempted to evaluate dispersal of Burbot from small tributary releases in the Kootenai River system (Neufeld et al. 2011; Stephenson et al. 2013). Neufeld et al. (2011) found that all age-2 and age-3 Burbot stocked in the Goat River, British Columbia out-migrated within 9 days. In contrast, Stephenson et al. (2013) reported 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 one year after stocking. Stephenson et al. (2013) also found that age-1 and younger Burbot had significantly shorter dispersal distances and longer dispersal times than age-2 and older Burbot. Results from Stephenson et al. (2013) are similar to our study in that age-0 Burbot in Deep Creek moved little and did not immediately out-migrate. As such, it may be that age-1 and younger Burbot simply remain near their release location and rear for a period of time before out-migrating. Whether this behavior is unique to hatchery-reared fish is unknown because similar studies with naturally produced Burbot have not been conducted.

Hatchery-reared fishes often fail to move long distances after release into lotic waters (Cresswell 1981; Helfrich and Kendall 1982; High and Meyer 2009). For example, Cresswell (1981) reported that studies evaluating post-stocking movements of Brook Trout *Salvelinus fontinalis*, Brown Trout *Salmo trutta*, and Rainbow Trout *Oncorhynchus mykiss* found a large proportion of fish recaptured within 4.5 km of the stocking location. High and Meyer (2009) reported that Rainbow Trout released in the Middle Fork Boise River, Idaho were almost always within 3 km of the release site and over half were observed within 1 km of the release site. In Big Stony Creek, Virginia, 75% of Rainbow Trout, Brook Trout, and Brown Trout were recovered within 1 km of their release location (Helfrich and Kendall 1982). Although a lack of movement is commonly observed with hatchery-reared fish, reasons for the behavior are unclear. For Burbot released in Deep Creek, one reason may be that the fish are the progeny of lake-origin broodstock (i.e., Moyie Lake, British Columbia; Powell et al. 2008; Hardy and Paragamian 2013). However, other data suggests that Burbot progeny from lake-origin broodstock are acclimating (e.g., good dispersal) to the Kootenai River system (Hardy et al. 2015).

While Burbot exhibited a general lack of movement, patterns related to movement direction and timing were observed in Deep Creek. For example, Burbot released in 2015 moved a greater distance upstream than downstream. However, this pattern differed for Burbot released in 2014. Changes in release strategy may explain this difference, as one of the two release locations in 2014 did not allow for upstream movement. Additionally, the majority of detections at all but one stationary antenna occurred at night. Burbot are more active and are often observed foraging at night (Lawler 1963; Boag 1989). Increased detections at night suggests that Burbot in Deep Creek express similar behavior.

Survival estimates from the relative returns of live and dead or shed tags (0.07-0.39), and from the Barker model (0.10-0.29) were similar. Results from release location surveys suggested that mortality of Burbot did not occur rapidly after release. Differences between relative survival estimates and Barker model estimates result from the Barker models ability to incorporate data from mobile PIT tag surveys and stationary PIT tag antennas. Relative survival estimates only incorporate data from mobile PIT tag surveys. Although comparable data are limited, survival of Burbot stocked in Deep Creek appears to be equal or higher than that reported for Burbot released into the mainstem Kootenai River and for other species. Recent estimates of survival for six-month-old juvenile Burbot released in the mainstem of the Kootenai River varied from 0.02 to 0.20 (IDFG unpublished data) and were generally lower than estimates of survival for Burbot released in Deep Creek. Estimates of survival for other fish species reared in a hatchery and released at similar sizes are similar to our estimates. For example, Margenau (1992) estimated that overwinter survival of fingerling Muskellunge *Esox masquinongy* stocked in four northern Wisconsin lakes averaged 0.19. Survival rates for Walleye *Sander vitreus* fingerlings released in three Iowa rivers varied from < 0.01 to 0.16 (Paragamian and Kingery 1992).

Several factors may affect survival of Burbot released into Deep Creek. Size at the time of release is one factor, but the effect of size is likely minimal in our study given Burbot relocated during the 2016 longitudinal distribution survey with alive, shed or dead, and unknown fates had similar mean total lengths at release. Another factor affecting Burbot survival may be predation. During the summer of 2015, a River Otter *Lontra canadensis* colony established a den 90 m downstream of the McArthur Lake release location. River Otter predation was investigated by scanning several River Otter trails within 100 m of the release location using the Oregon RFID mobile PIT tag antenna. While scanning River Otter trails, 16 individual tags were relocated in areas above the high water mark. This suggested that at least some predation by River Otters occurred. Moreover, Brook Trout and Largemouth Bass *Micropterus salmoides* are also present in Deep Creek and are known to be highly piscivorous (Becker 1983). In addition to Brook Trout and Largemouth Bass, predation by other Burbot may have contributed to low survival. Cannibalism is common in Burbot populations (Gallagher and Dick 2015). Furthermore, several piscivorous bird species (e.g., Great Blue Heron *Ardea herodias* and Belted Kingfisher *Megaceryle alcyon*) are common in the system and may prey upon Burbot.

Functional responses of predators to prey are well documented (Holling 1959; Peterman and Gatto 1978). Functional responses include a linear response where the amount of prey consumed is proportional to the prey density (Type I), a response where predators become saturated and the number of prey consumed remains constant even with increasing prey density (Type II), and an S-shaped response where predators increase their search activity and consumption with increasing prey density (Type III; Holling 1959). If predation is a major factor influencing survival of Burbot, a Type II or Type III response is unlikely because we would expect to see differences in survival between the 2014 and 2015 release strategies. Thus, the observed response is likely a Type I response given similar mortality estimates at different stocking densities. Understanding the Type of functional response may have specific management implications. With a Type I functional response, the number of Burbot lost to predation is proportional to the density at which they are released. If a fixed proportion of Burbot is lost to predation regardless of density, managers may consider stocking Burbot at high densities so a larger absolute number of Burbot survive.

Prey resources are commonly cited as a factor limiting survival in many populations, particularly for juvenile fishes (Cushing 1969; Cushing 1990; Schlosser 1991; Hoxmeier et al. 2006). Previous studies have found that juvenile Burbot (41.0-152.6 mm) primarily feed on macroinvertebrates such as Amphipoda, Ephemeroptera, Odonata, and Plecoptera (Beeton

1956; Ryder and Pesendorfer 1992; Fisher 2000). Low densities of macroinvertebrates in Deep Creek may have contributed to the survival observed for the initial seven-months after release into Deep Creek. Unfortunately, data on macroinvertebrate assemblage structure and density are unavailable. Once Burbot survived the first seven months in Deep Creek, survival improved to over 50%. Additionally, survival estimates for Burbot that survived the first seven months are similar to previous estimates for age-1 to age-3 Burbot released into the Kootenai River system (0.54-0.78; Paragamian et al. 2008; Stephenson et al. 2013), and those of wild adult Burbot in Lake Superior (0.57; Schram 2000) and Moyie Lake (0.53-0.80; Prince 2007; Neufeld 2008). An increase in survival may reflect Burbot reaching larger sizes and having the capacity to exploit a diversity of prey resources. Most fishes consume progressively larger and more diverse prey items as their gape increases (O'Brien 1979; O'Brien 1987; Schael et al. 1991). While Burbot may exploit a diversity of prey resources as they attain larger sizes, additional research is needed to understand prey availability in Deep Creek.

Our research provided evidence that age-0 Burbot move little after being released and are slow to disperse from release locations regardless of perceived habitat quality. This research also provided some of the first estimates of survival for age-0 hatchery-reared Burbot released in lotic environments. Both estimates of survival (longitudinal distribution surveys of Deep Creek and estimates from the Barker model) were similar across years and release locations. Survival estimates from the Barker model improved for Burbot that survived the first seven months in Deep Creek. Although releasing Burbot at multiple release locations and a lower density did not increase survival, this strategy may be beneficial to managers. Release of Burbot at multiple locations in low densities provides a buffer against localized events that may increase mortality and reduces competition for resources. Future research is needed to determine the amount and importance of predation on Burbot. Furthermore, quantifying the diversity and abundance of prey resources within the system may be important for understanding how resources influence survival of juvenile Burbot. Quantifying the amount of predation and abundance of available prey resources would provide additional insight that would aid in developing release strategies for Burbot in the Kootenai River system and beyond.

Acknowledgements

We thank L. Schofield, M. Miner, S. Cossel, P. Branigan, M. Dobos, M. Terrazas, M. Thomas, B. Lunger, J. Wenzel, and numerous Idaho Department of Fish and Game employees, particularly C. Gidley, for assistance with field work. We thank the Idaho Department of Lands, Susan Edwards, Patrick Edwards, Marc Redding, and Diane Redding for allowing construction of stationary antennas on their properties. We also thank the Kootenai Tribe of Idaho for providing the funding for raising Burbot and the Aquaculture Research Institute at the University of Idaho for rearing the Burbot stocked into Deep Creek. Funding for this project was provided by the Idaho Department of Fish and Game. Additional support was provided by the U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit. The Unit is jointly sponsored by the U.S. Geological Survey, University of Idaho, the Idaho Department of Fish and Game, and Wildlife Management Institute. This project was conducted under the University of Idaho Institutional Animal Care and Use Committee Protocol 2014-8. The use of trade, firm or product names is for descriptive purposes only and does not imply endorsement by the U. S. Government.

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Table 3.1. Number of detections at all five antennas (R1-R5: see Figure 3.1) for age-0 Burbot in 2014 and 2015 by release location. Numbers in parentheses represent the total number of individual Burbot. Release locations in 2015 are grouped by high-quality habitat release locations and moderate-quality release locations.

Release	R1	R2	R3	R4	R5
2014					
McArthur	262,061 (1,103)	9 (7)	5 (5)	5 (1)	8 (3)
Naples	0 (0)	14 (2)	497 (109)	727 (74)	34 (10)
2015					
		High quality			
Naples	0 (0)	0 (0)	162 (34)	223 (9)	0 (0)
2nd Bridge	0 (0)	7 (2)	158 (55)	1,407 (102)	0 (0)
Swimming	0 (0)	0 (0)	113 (35)	186 (50)	1 (1)
		Moderate quality			
McArthur	1,536,148 (436)	4 (2)	0 (0)	0 (0)	0 (0)
Shiloh	11,645 (26)	3,311 (282)	0 (0)	0 (0)	0 (0)
Resort	0 (0)	0 (0)	3 (2)	25 (9)	57 (7)

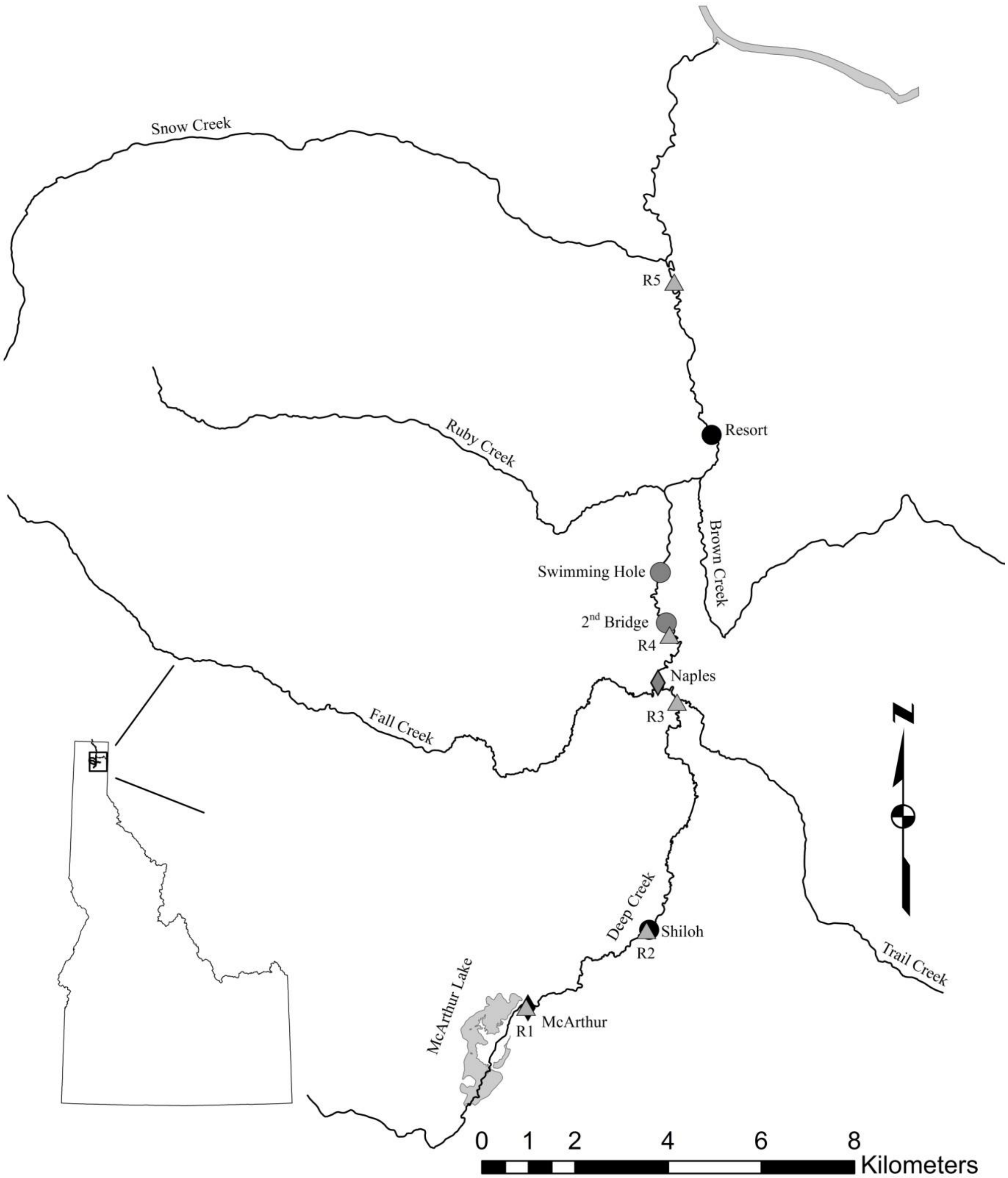


Figure 3.1. Deep Creek watershed showing the main channel of Deep Creek, Deep Creek's five major tributaries (i.e., Trail, Fall, Ruby, Brown, and Snow creeks), and a portion of the Kootenai River. Light gray triangles represent PIT tag antenna locations and are labeled R1 through R5 from upstream to downstream. Diamonds represent stocking locations in that were

the same in 2014 and 2015 (i.e., McArthur and Naples). Circles represent new stocking locations in 2015 (i.e., Shiloh, 2nd Bridge, Swimming Hole, Resort). Dark gray shading represents high-quality release locations and white shading represents moderate-quality release locations.

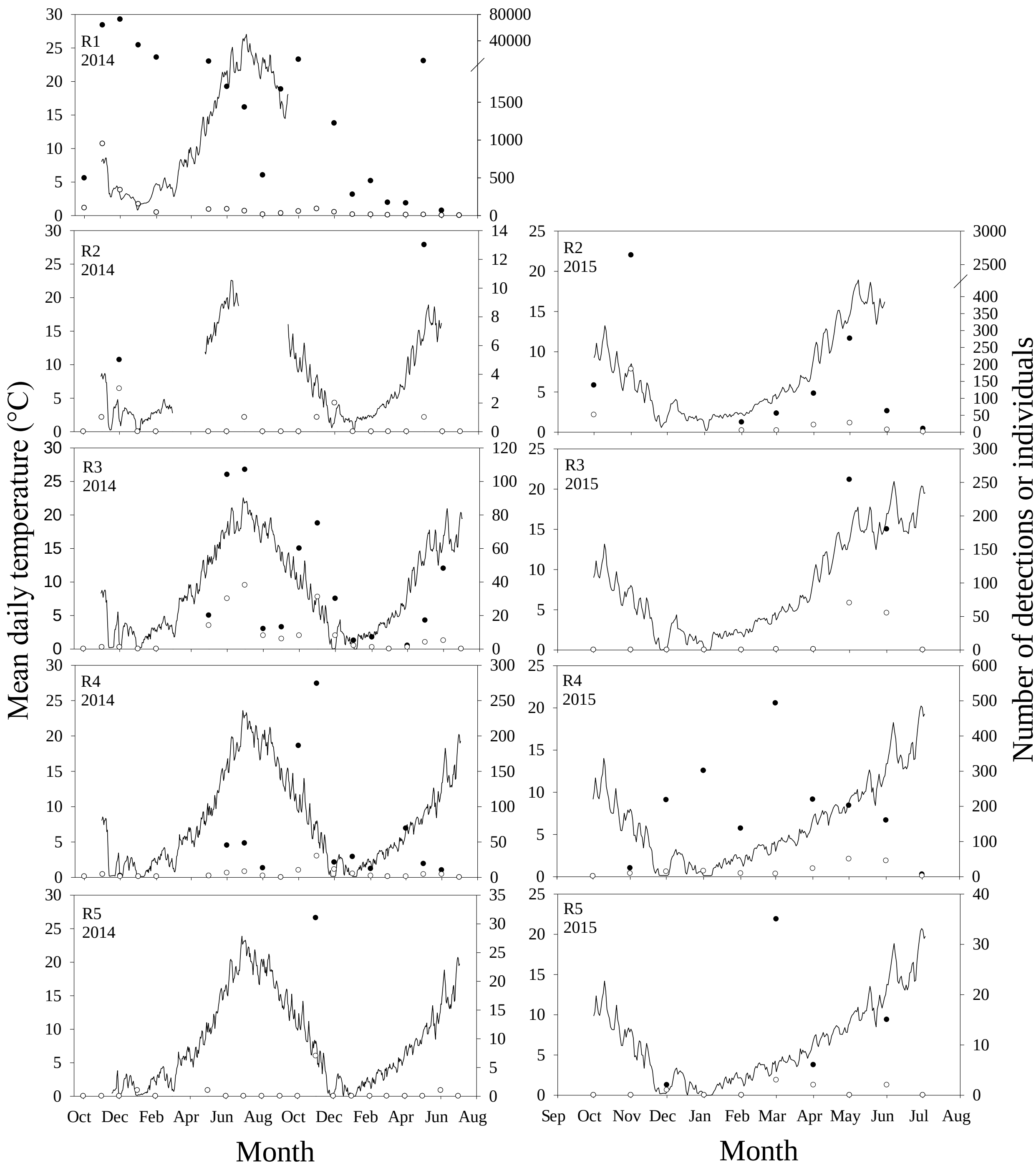


Figure 3.2. Mean daily temperature (°C) and the total number of detections and individuals by month at all five stationary PIT tag antennas for Burbot stocked in 2014 and 2015. Black circles represent the total number of detections and open circles represent the total number of individual Burbot. When only an open circle is visible the number of detections and individual Burbot are equal.

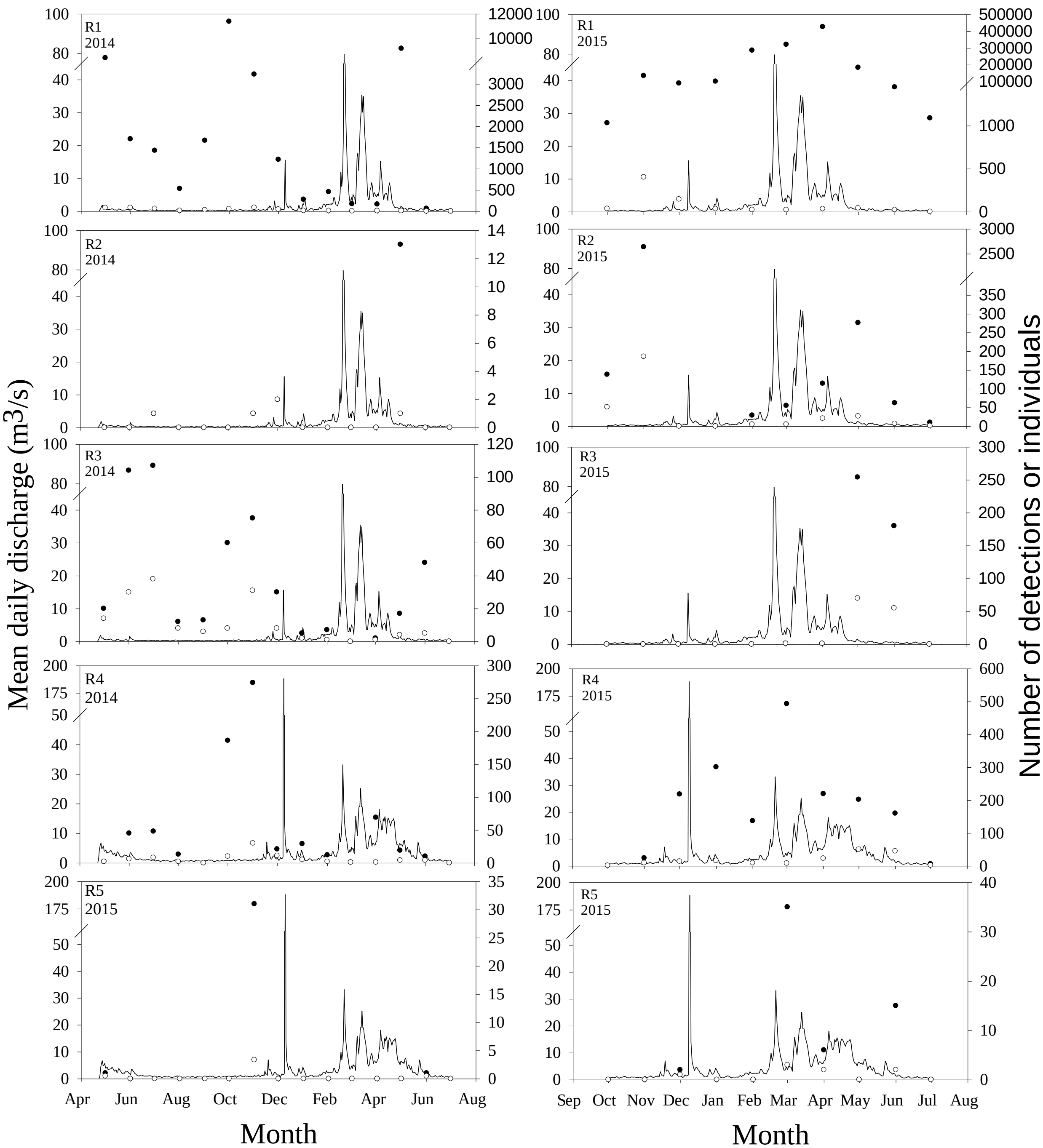


Figure 3.3. Mean daily discharge (m^3/s) and the total number of detections and individuals by month at all five stationary PIT tag antennas for Burbot stocked in 2014 and 2015. Black circles are the total number of detections and open circles are the total number of individual Burbot. When only an open circle is visible the number of detections and individual Burbot are equal.

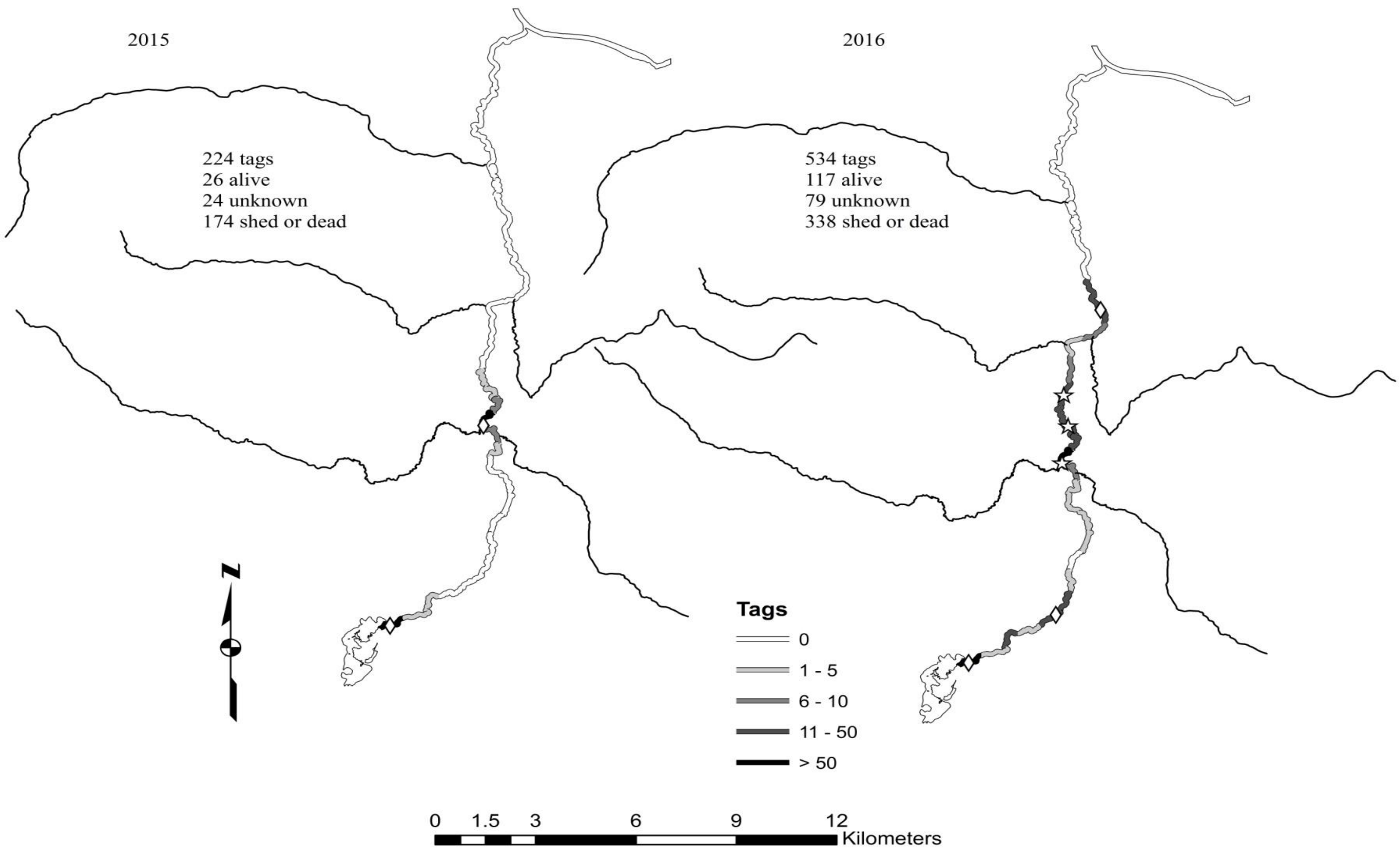


Figure 3.4. Maps of Deep Creek showing the locations and numbers of tags relocated during the 2015 and 2016 mobile passive integrated transponder tag surveys of Deep Creek. Diamonds represent the 2014 stocking locations and the moderate-quality stocking locations in 2015. Stars represent the high-quality stocking locations in 2015.

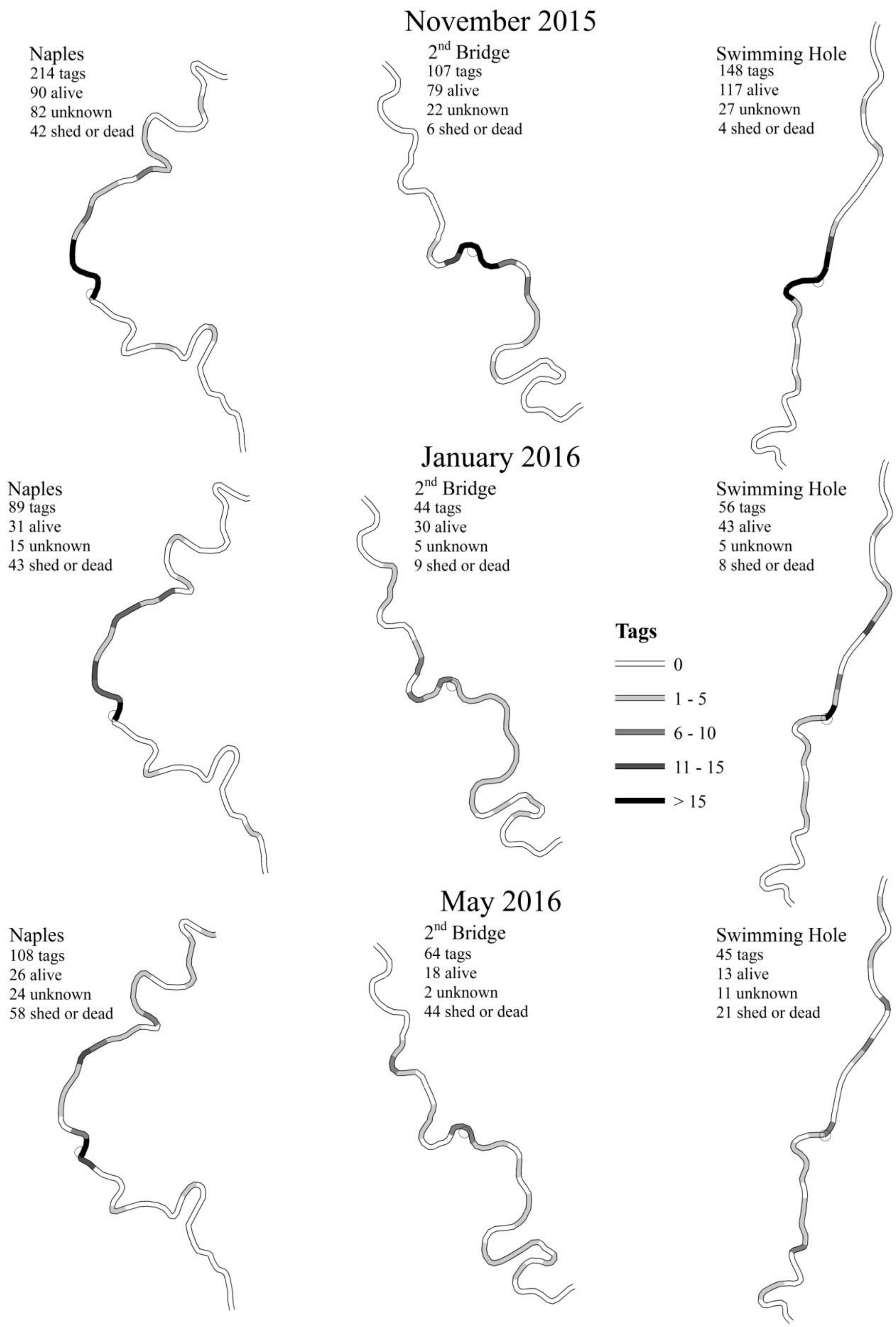


Figure 3.5. Maps of the 2 km segments centered on the three high-quality release locations in 2015 showing the locations and numbers of tags relocated for the mobile surveys conducted in November 2015, and January and May 2016. Open circles represent the stocking locations.

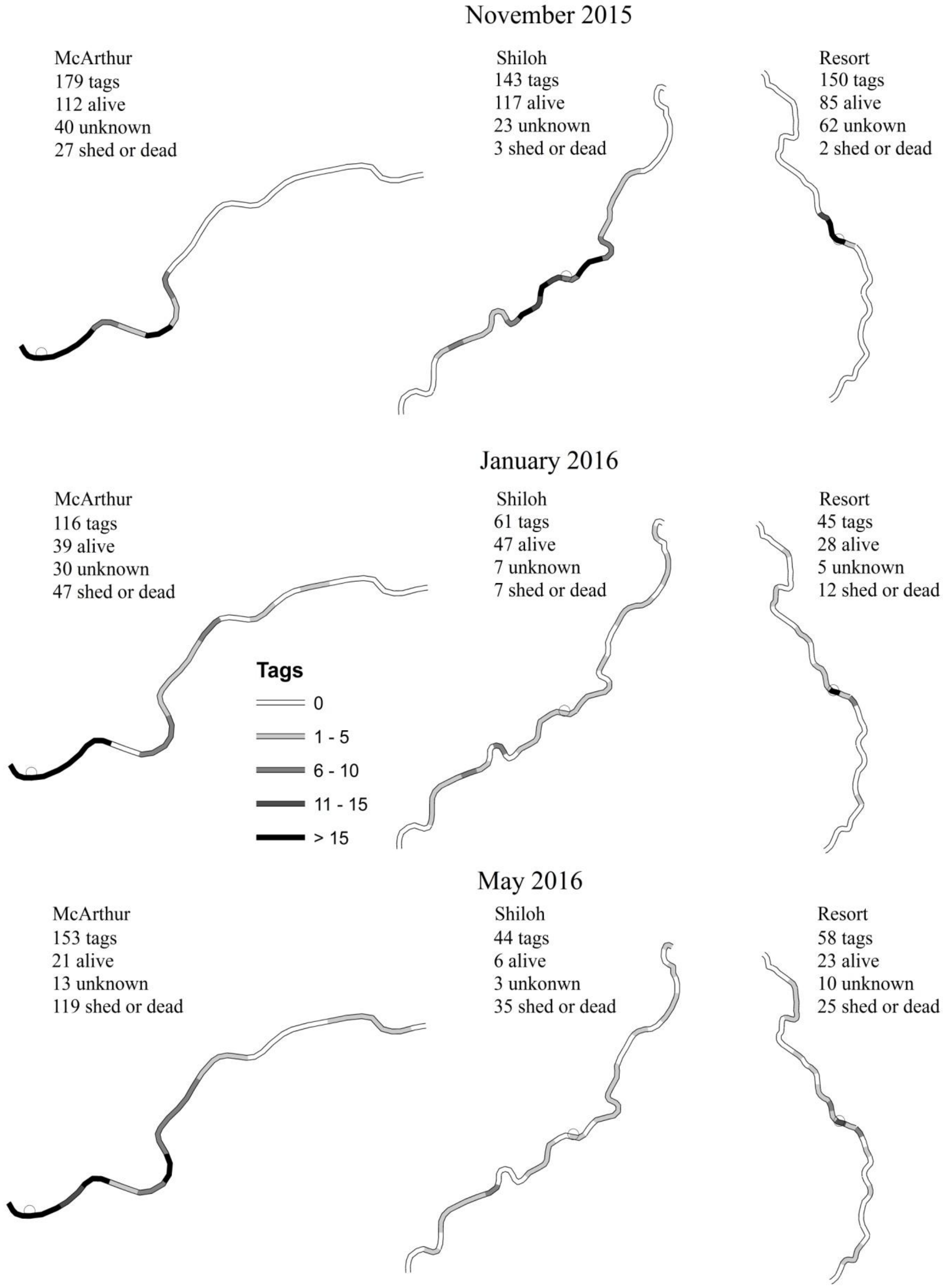
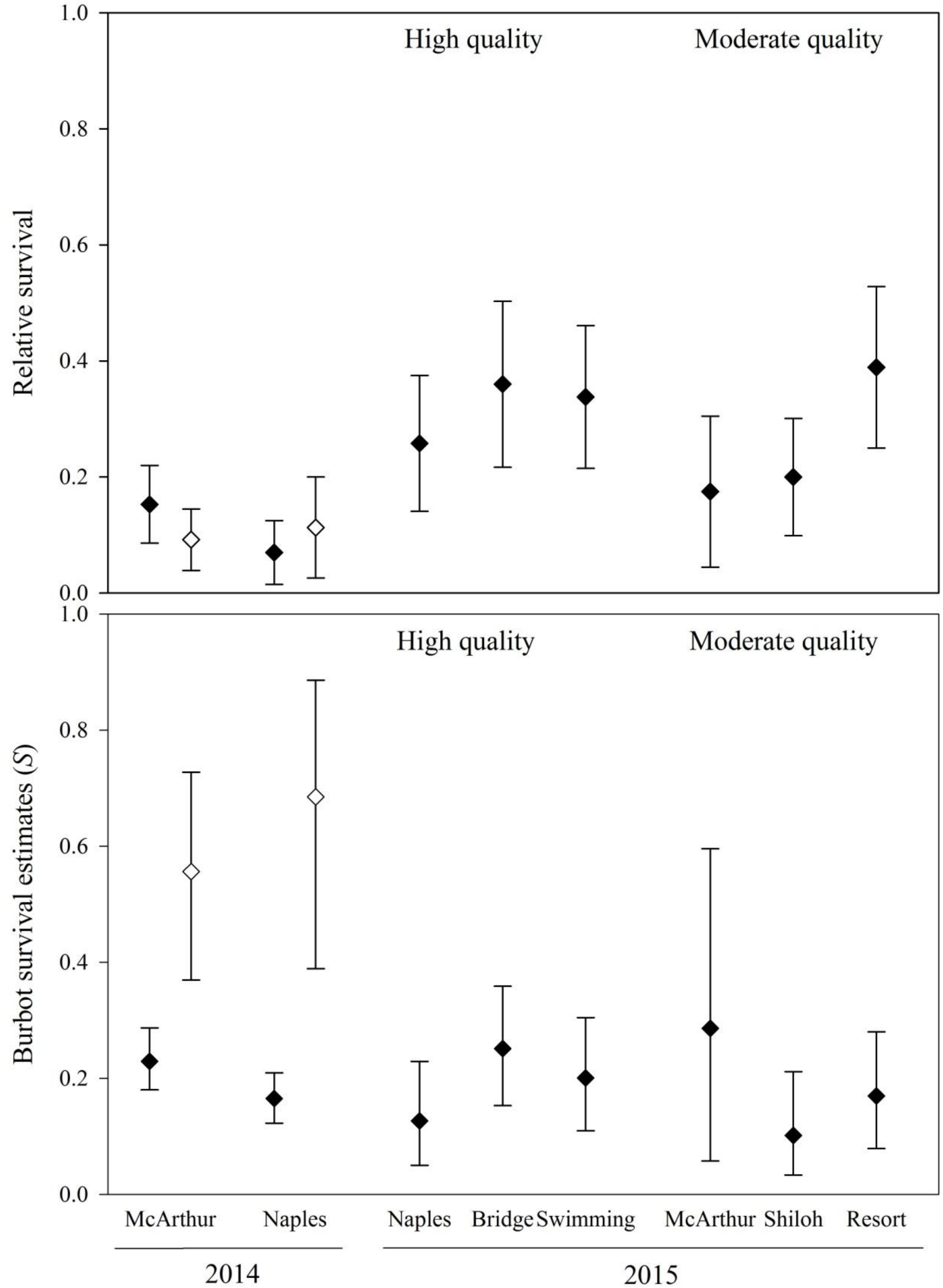


Figure 3.6. Maps of the 2 km segments centered on the three moderate-quality release locations in 2015 showing the locations and numbers of tags relocated for the mobile surveys conducted in November 2015, and January and May 2016. Open circles represent the stocking locations.



1.0
0.8
0.6
0.4
0.2
0.0
Relative survival
High quality
Moderate quality
1.0
0.8
0.6
0.4
0.2
0.0
Burbot survival estimates (S)
High quality
Moderate quality
McArthur
Naples
Naples
Bridge
Swimming
McArthur
Shiloh
Resort
2014
2015

Figure 3.7. Estimates of relative survival calculated from 2015 and 2016 longitudinal distribution surveys of Deep Creek (± 95% confidence interval) and estimates of survival (± 95% credible interval) calculated from mark-recapture analyses of Burbot stocked into Deep Creek, Idaho in 2014 and 2015. Relative survival estimates are displayed in the top panel, and survival estimates from mark-recapture analyses are displayed in the bottom panel. Black diamonds represent initial seven-month survival rates. Open diamonds represent survival rates after the first seven months. High-quality release locations in 2015 are Naples, 2nd Bridge, and Swimming Hole. Moderate-quality release locations in 2015 are McArthur, Shiloh, and Resort. Bridge is the estimate for the 2nd Bridge release location, and swimming is the estimate for the Swimming Hole release location.

Chapter 4: General Conclusions

Burbot are one of the iconic species in the lower Kootenai River and were important to local economies through commercial and recreational fisheries, and to the Kootenai Tribe of Idaho in the form of subsistence and cultural harvests. As such, restoration of Burbot in the lower Kootenai River is an important goal for managers and other stakeholders in the basin. The broad goal of this thesis was to aid in restoration efforts by providing insight into one of the release strategies currently being used for Burbot in the lower Kootenai River. The specific objectives were to provide information on the survival, movement, distribution, and habitat and species associations of juvenile Burbot released into Deep Creek. Each chapter of this thesis addressed specifics of the aforementioned goals to provide insight on the fate of Burbot after they are released into a small tributary of the Kootenai River.

The findings of chapter two identified that distance to the nearest stocking location was the most important factor associated with Burbot occurrence and relative abundance. Additionally, Burbot were closely associated with deep habitats, low current velocity, and large substrate. Deep habitats and large substrates are commonly reported as important characteristics for Burbot (Dixon and Vokoun 2009; Eick 2013; Klein et al. 2015). Chapter three sought to evaluate survival, movement, and distribution of juvenile burbot in Deep Creek. Stationary passive integrated transponder (PIT) tag antenna detections and mobile PIT tag antenna surveys of Deep Creek indicated that Burbot had poor dispersal. Patterns in dispersal were similar between years and release locations. Estimates of survival from the relative return of live and dead or shed tags (0.07-0.39), and from mark-recapture analyses were low (0.10-0.29) and similar between years and release locations. Once Burbot survived the first 7 months, survival improved to over 50%. Survival estimates from this study are some of the first for age-0 hatchery-reared Burbot released in lotic environments.

Results from this research may be used to guide management strategies. One management strategy would be to continue releasing juvenile Burbot into small tributaries of the Kootenai River. When implementing this strategy, managers might consider selecting multiple release locations and releasing Burbot at densities similar to those used in 2015 in Deep Creek. When selecting release locations, managers may consider selecting sites with deep habitats and large substrate within 1 km of the release location. Increasing the number of release locations and decreasing stocking density did not result in increased survival. However, this release strategy is more resilient to localized events that might increase mortality. Furthermore, releasing Burbot at lower densities may reduce competition for prey resources. An alternative management strategy would be to stop releasing juvenile Burbot into small tributaries. Survival of Burbot released in Deep Creek was low regardless of changes in release strategies. Mechanisms driving low survival are unclear. However, the amount of predation, and the diversity and abundance of prey resources in Deep Creek have yet to be quantified. Without a clear understanding of the factors affecting low survival, it may be difficult to ensure successful releases of Burbot in small tributaries. Regardless of the management strategy, further research is needed to ensure efficient and effective stocking methods.

Priorities for future research include identifying the mechanisms driving low survival. This includes quantifying the amount of predation on Burbot in Deep Creek and the diversity and relative abundance of available prey resources. Additional concerns include the effect of release strategies on the habitat and species associations of Burbot released in Deep

Creek. Investigating the habitat associations of Burbot after the change in release strategies from 2014 to 2015 may confirm that Burbot are moving to areas of deep habitat with large substrate close to release locations thus confirming that the observed patterns were not influenced by release location. Attempting similar studies in other small tributaries of the Kootenai River would provide additional insight into whether the observed patterns are similar to those for other small tributaries or are unique to Deep Creek. As such, managers could learn which small tributaries may be more suitable for use in restoration efforts.

References

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