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Sampling techniques for burbot in a western non-wadeable river

Z. B. KLEIN

Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, Moscow, ID, USA

M. C. QUIST

U.S. Geological Survey, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, Moscow, ID, USA

D. T. RHEA

Wyoming Game and Fish Department, Pinedale, WY, USA

A. C. SENECAL

Wyoming Game and Fish Department, Green River, WY, USA

Abstract Burbot, *Lota lota* (L.), populations are declining throughout much of their native distribution. Although numerous aspects of burbot ecology are well understood, less is known about effective sampling techniques for burbot in lotic systems. Occupancy models were used to estimate the probability of detection ($\hat{p}$) for three gears (6.4- and 19-mm bar mesh hoop nets, night electric fishing), within the context of various habitat characteristics. During the summer, night electric fishing had the highest estimated detection probability for both juvenile ($\hat{p}$, 95% C.I.; 0.35, 0.26–0.46) and adult (0.30, 0.20–0.41) burbot. However, small-mesh hoop nets (6.4-mm bar mesh) had similar detection probabilities to night electric fishing for both juvenile (0.26, 0.17–0.36) and adult (0.27, 0.18–0.39) burbot during the summer. In autumn, a similar overlap between detection probabilities was observed for juvenile and adult burbot. Small-mesh hoop nets had the highest estimated probability of detection for both juvenile and adult burbot (0.46, 0.33–0.59), whereas night electric fishing had a detection probability of 0.39 (0.28–0.52) for juvenile and adult burbot. By using detection probabilities to compare gears, the most effective sampling technique can be identified, leading to increased species detections and more effective management of burbot.

KEYWORDS: conservation, detectability, gear evaluation, Green River, non-wadeable river.

Introduction

Effective management of fish requires reliable information on their population dynamics and demographics (Allen & Hightower 2010). Obtaining this information depends on techniques that are both effective and efficient at sampling target species. However, evaluating the efficacy of a given gear is often labour intensive. Furthermore, the effectiveness of a given gear likely varies when that technique is applied to a novel species or habitat (Schloesser *et al.* 2012). Without a clear understanding of the biases associated with a particular sampling technique and how those biases vary with biotic (e.g. fish behaviour, species interaction) and abiotic (e.g. environmental characteristics) factors, inferences about the target species may result in ineffective or incorrect management decisions.

A number of metrics are available to evaluate sampling techniques (e.g. density, presence–absence). Catch

Correspondence: Zachary B. Klein, Idaho Cooperative Fish and Wildlife Research Unit, Department of Fish and Wildlife Sciences, University of Idaho, 875 Perimeter Dr. MS 1141, Moscow, ID 83844, USA (e-mail: klei7686@vandals.uidaho.edu)

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rate is commonly used to examine the sampling efficiency of gears, but requires non-zero catch rates for meaningful conclusions (Quist *et al.* 2006). When catch rates are low, counts are often unreliable and must be combined with an estimate of detection to evaluate the biases of a sampling method (MacKenzie *et al.* 2002). Because a species can be present, absent or present but undetected, detectability (the probability of being detected) must be accounted for in an analysis or inference regarding changes in the status of a species can potentially be misleading (MacKenzie *et al.* 2002, 2006). Gear comparisons are often avoided because traditional estimates of detection (e.g. mark–recapture, depletion) are effort intensive. Instead, the least selective gear or multiple gears are used to reduce the effect of gear bias (Beamesderfer & Rieman 1988). However, the use of a less selective gear or multiple gears assumes perfect detection and ignores variations in detectability. Additionally, because the effort of passive (e.g. hoop nets, trammel nets) and active (e.g. electric fishing) sampling techniques varies, direct comparisons of efficacy between passive and active gears are notoriously difficult (Peterson & Paukert 2009; Hubert *et al.* 2012). Effective management of a species is dependent on being able to reliably sample that species. Therefore, understanding variations in detectability and the factors that influence detectability is integral to the effective implementation of a given sampling technique (Jensen & Vokoun 2013).

Burbot, *Lota lota* (L.), is the only freshwater member of the family Gadidae (Howes 1991). It has a circumpolar distribution that rarely extends below 40° N and occupies a diversity of lentic and lotic habitats throughout Eurasia and North America (McPhail & Paragamian 2000). Stable burbot populations exist in Alaska, much of Canada and several Eurasian countries including Latvia, Lithuania, Switzerland and Russia (Stapanian *et al.* 2010). However, across much of its distribution, burbot is declining or has been completely extirpated, including populations in Eurasia, Canada and the contiguous USA (Tammi *et al.* 1999; McPhail & Paragamian 2000; Dillen *et al.* 2008; Stapanian *et al.* 2008, 2010). Although burbot conservation is a major focus around the world, basic questions remain regarding the most effective technique for sampling burbot.

In Wyoming, burbot is a concern not only because of declines in its native distribution but also because of its illegal introduction outside of its native distribution within the state. Burbot is native to the Wind-Bighorn River and Tongue River drainages in northern Wyoming but has been extirpated from the Tongue River basin (Hubert *et al.* 2008; Gardunio *et al.* 2011). In the Wind-Bighorn River system, burbot populations have declined due to angler exploitation and water development (Hubert *et al.* 2008). In the 1990s, burbot was illegally introduced outside of its native distribution into Big Sandy Reservoir and began dispersing throughout the Green River drainage in the early 2000s (Gardunio *et al.* 2011). Since its introduction, burbot have been sampled upstream of the confluence of the New Fork and Green rivers downstream to Dinosaur National Monument on the Utah–Colorado border (Gardunio *et al.* 2011). Although burbot conservation is a focus in the Wind–Bighorn River drainage, its introduction to the Green river system raises concern regarding management of socially and economically important trout fisheries [i.e. brown trout, *Salmo trutta* Linnaeus L., rainbow trout, *Oncorhynchus mykiss* (Walbaum) and Colorado River cutthroat trout, *Oncorhynchus clarkii pleuriticus* (Richardson)], and conservation of native non-game fishes [i.e. bluehead sucker, *Catosomus discobolus* Cope, flannelmouth sucker, *Catosomus latipinnis* Baird & Girard and roundtail chub, *Gila robusta* Baird & Girard (Gardunio *et al.* 2011)]. Additionally, burbot not being native, yet abundant, in the Green River facilitates research that would otherwise be limited by low numbers of burbot in other areas of its native distribution.

Burbot have been sampled using a variety of techniques (e.g. hoop nets, gill nets, electric fishing). However, little research has focused on comparing the effectiveness of different techniques in lotic and lentic habitats. The majority of studies focused on evaluating the effectiveness of sampling techniques for burbot have occurred in lentic systems using passive gears (Bernard *et al.* 1991; Spence 2000; Prince 2007). Although many of these studies used similar sampling techniques, drawing general conclusions is difficult due to inconsistent results. For example, Bernard *et al.* (1991) concluded that hoop nets were effective at capturing burbot in central Alaskan lakes. In contrast, Prince (2007) reported that cod traps had higher catch rates of burbot than hoop nets in Moyie Lake, British Columbia. Confounding general conclusions further, catch rates for trammel nets were nearly double those of hoop nets and cod traps in Boysen Reservoir, Wyoming (A. Dux, Idaho Department of Fish and Game, unpublished data). Results from gear comparisons in lotic systems for burbot are equally inconclusive. For example, Horton and Strainer (2008) compared slat traps, cod traps and hoop nets in the Missouri River, Montana, but found little difference in catch rates across gears. Bernard *et al.* (1991) compared large- and small-diameter hoop nets using the same mesh size (i.e. 25-mm bar mesh) as Horton and Strainer (2008) and found that larger diameter hoop nets had higher catch rates than small-diameter hoop nets. Although direct comparisons between hoop nets used in lotic systems are difficult due to myriad configurations (e.g.

diameter, mesh size), the conflicting results demonstrate the need to further evaluate hoop nets for sampling burbot. In addition, inconsistent results from studies comparing passive gears highlight the need to evaluate different sampling techniques for burbot in lotic systems.

Given conservation concerns surrounding burbot and the lack of information on the most effective techniques for sampling them, the objective of this study was to compare the detection probability ($\hat{p}$) of small-mesh hoop nets (i.e. 6.4-mm bar mesh), large-mesh hoop nets (i.e. 19-mm bar mesh) and night electric fishing for juvenile and adult burbot in the Green River drainage of Wyoming using occupancy modelling. An additional objective was to investigate the effect that habitat characteristics (i.e. dominant substrate, current velocity, depth, in-stream cover, bank type and water temperature) had on the probability of detection.

Study site

The Green River is the largest tributary of the Colorado River and drains portions of Wyoming, Utah and Colorado (Fig. 1; Wyoming Game & Fish Department 2010). The Green River originates in the Wind River Range of western Wyoming and flows for approximately 235 km before entering Fontenelle Reservoir. From Fontenelle Reservoir, the Green River flows for about 150 km until it enters Flaming Gorge Reservoir at the Wyoming–Utah border. Upstream of Fontenelle Reservoir, the Green River is characterised by high gradient runs interspersed with pool–riffle habitat and substrate characteristics of alluvial deposits (e.g. cobble, gravel, sand and silt; Kurtz 1980). From Fontenelle Dam to its confluence with the Big Sandy River, the Green River is characterised by long runs averaging 450 m (Wiley 1974). After its confluence with the Big Sandy River, the Green River is relatively low gradient and the substrate is dominated by sand and gravel substrate (Wiley 1974). Most of the Green River basin is characteristic of a high desert climate with monthly temperature averages of −9°C (January) and 17°C (July) and low annual precipitation (25.4 cm; Wyoming Game & Fish Department 2010).

Methods

Sampling was conducted in four river sections during summer (22 May–12 July 2013) and two river sections during autumn (6–25 September 2013). Sections were not a unit of interest *per se*; rather, they were simply used to allocate sampling effort. Sections were selected to ensure habitat was representative of the Green River. Each section was divided into 150-m long sites. In total,

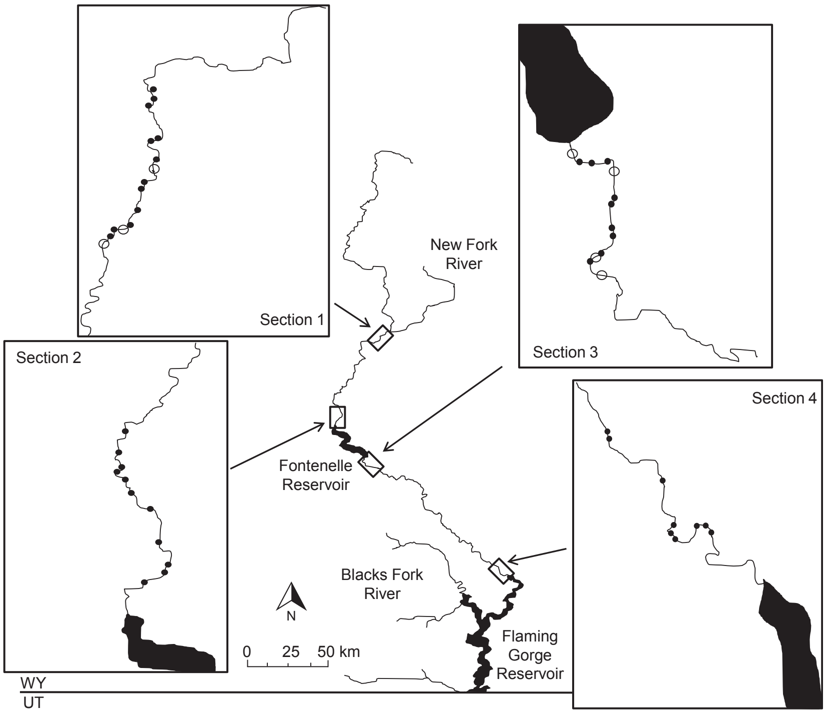

Figure 1. Sections used for juvenile and adult burbot sampling in the Green River, Wyoming, during the summer and autumn (2013). Boxes depict each section in detail with sites sampled in the summer and autumn (black circles) and sample sites added in the autumn to increase sample size (open circles).

49 sites were randomly selected for sampling (n = 41 in summer; n = 28 in autumn; Fig. 1). All sites were sampled with small- and large-mesh hoop nets (6.4-mm and 19-mm bar mesh, respectively) and night electric fishing. Sites were sampled over a 9-day period such that each site was sampled three times with each sampling gear. For example, on day one, a given site was sampled with a randomly selected small- or large-mesh hoop net. The following day, the site was sampled using the size of hoop net not used the previous day. The site was sampled using night electric fishing on the final day to avoid biasing hoop net detections. Three days of consecutive sampling events were considered a sampling occasion, and each site was sampled on three occasions per season. The summer sampling season commenced at the most downstream section and continued upstream until all sites had been sampled in triplicate. To increase sample size during autumn, 19 sites were added to the 41 sites sampled in the summer. However, due to unusually high precipitation and logistical constraints, only the most upstream sites (n = 15; section 1, Fig. 1) and the sites immediately downstream of Fontenelle Dam (n = 13; section 3) were sampled during autumn.

The small-mesh hoop nets had seven 0.6-m-diameter hoops and an overall length of 3 m. A fish ‘escape route’ was installed between the sixth and seventh hoop to allow fish to escape from of a lost net. ‘Escape routes’ consisted of a 20 × 20 cm square opening covered with 6.4-mm bar mesh secured with 3-mm untreated cotton twine. The large-mesh hoop nets had four 0.91-m-diameter hoops and an overall length of 2.9 m. Each net had a ‘Mammalian Escape Hatch’ placed between the third and fourth hoop, consisting of a 20 × 41 cm square opening covered with a 36 × 61 cm square section sewn on three sides (A. Senecal, unpublished information). Each net was positioned parallel to the current with the cod end anchored upstream. Nets were baited with white sucker, *Catostomus commersonii* (Lacepède), a non-game, non-native species in the system and fished for 24 h.

Night electric fishing was conducted using a drift boat equipped with a 5000 W generator and Smith-Root VVP-15B electrofisher (Smith-Root, Vancouver, WA). Electric fishing power output was standardised to 2750–3200 W with a duty cycle of 45% and a frequency of 45 Hz (Miranda 2009). One netter was positioned on the bow of the boat using a 2.4-m long dip net with 6-mm bar knotless mesh. Electric fishing proceeded downstream from the upstream-most point of each 150 m site. Fish were enumerated, weighed (g) and measured for total length (mm). Effort was recorded as the total number of seconds electricity was applied to the water.

Six site-specific habitat variables (i.e. current velocity, depth, wetted width, dominant substrate, in-stream cover and bank type) that were hypothesised to affect the probability of occupancy and detectability were measured once per season at each site. Channel units (i.e. riffle, run, pool) were sampled separately for habitat characteristics. Channel-unit length and width were measured to the nearest 0.5 m using a laser rangefinder. Sites with a single channel-unit type were divided in half by length, and each half was treated as a channel unit. In-stream habitat characteristics were measured along a transect that ran diagonally from the uppermost point of the channel unit to the lower-most point of the channel unit on the opposite bank. In-stream habitat characteristics (i.e. current velocity, dominant substrate, in-stream cover and depth) were measured at 20, 40, 50, 60 and 80% of the transect’s length (Flotemersch *et al.* 2006; Sindt *et al.* 2012). Depth was measured using a 2-m sounding pole or a portable electronic depth finder (Marcum LX-i, Minnetonka, MN). Current velocity was measured using a portable flow meter (Flo-Mate Model 2000; Marsh-McBirney, Loveland, CO). The dominant substrate was visually estimated as organic matter–silt (<0.0004–0.262 mm), sand and gravel (0.263–64.0 mm), cobble (64.0–256.0 mm), boulder (>256.0 mm) or bedrock (modified from Orth & Maughan 1982). Presence or absence of cover (i.e. large woody debris, overhanging vegetation, undercut banks, boulders, macrophytes and man-made structures) was recorded at each transect point (Fitzpatrick *et al.* 1998; Neebling & Quist 2010). The proportion of each bank type (i.e. alluvium and vegetation, eroded, undercut, vegetation, boulder, anthropogenic structure, woody debris and sand and gravel) was visually estimated for each site. An additional bank type called ‘alluvium bluff’ was estimated for each site and was characterised by steep, eroded banks with large substrate (≥64.0 mm; modified from Jaeger *et al.* 2005). In addition to site-specific habitat variables, sample-specific habitat variables [i.e. temperature (°C), conductivity (μS cm^{-1}), Secchi depth (m)] thought to affect occupancy and detection probabilities were measured at each sampling event following fish sampling.

Length structure was assessed for each sampling technique for burbot in the Green River for summer and autumn (2013) using length-frequency histograms (Fig. 2). Although gear-specific size selectivity was not a primary focus of this study, length-structure information provided valuable insight into the management of burbot.

Occupancy modelling was used to estimate the detection and occupancy probability of each sampling gear for juvenile and adult burbot for the summer and autumn. Single-season, single-species occupancy models

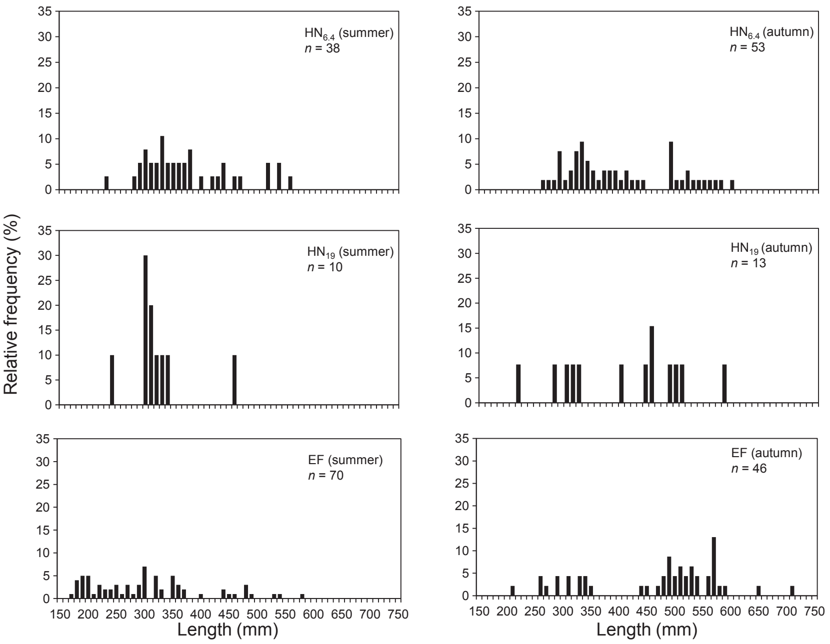

Figure 2. Length-frequency histograms for burbot collected with 6.4-mm bar measure mesh hoop nets ($HN_{6.4}$), 19-mm bar measure mesh hoop nets (HN_{19}) and night electric fishing (EF) in the Green River, Wyoming, during the summer and autumn (2013).

were constructed using programme PRESENCE (Hines 2006). Single-season, single-species models were chosen because they directly addressed the research questions and their simple form allowed for inclusion of multiple covariates. Programme PRESENCE estimates both the probability of a site being occupied (Ψ) and the probability of a species being detected at an occupied site ($\hat{p}$) using the detection history of repeatedly sampled sites. Detection histories represent the detection or non-detection of a species at a series of sampling events. A sampling occasion was then represented by the detection history of three independent sampling events, and a season was represented by three sampling occasions. Because a species can either be present, absent or present but undetected at a site, PRESENCE uses the detection history to estimate the probability of detection at a site known to be occupied during a single sampling event within a sampling occasion. Occupancy is then the probability that a species is present at a site and requires a single positive detection to estimate (MacKenzie *et al.* 2005, 2006). Although obtaining estimates of detectability was the primary focus of this study, it was necessary to model occupancy by season and habitat to acquire the best model fit (MacKenzie & Bailey 2004). The estimates of the probability of detection ($\hat{p}$) were used as an index for comparing sampling techniques (Schloesser *et al.* 2012; Haynes *et al.* 2013). Additionally, site- and sample-specific covariates were added to occupancy models to explain variability in detection by season and gear for juvenile and adult burbot.

Mean depth, current velocity, substrate composition and bank type were hypothesised to be the primary factors affecting the probability of detecting burbot (Dixon & Vokoun 2009; Eick 2013; Table 1). Flaming Gorge and Fontenelle reservoirs were hypothesised to act as thermal refugia for burbot during periods of high mean water temperature and as a ‘source’ of burbot. Therefore, a covariate of river kilometre (rkm) between sites and the downstream reservoir was included.

A Spearman’s rank-order correlation coefficient was used to assess multicollinearity amongst covariates (Higgins 2003; Sindt *et al.* 2012). If two covariates were highly correlated ($|\rho| > 0.70$), only the most ecologically important and interpretable variable was used in candidate models (Burnham & Anderson 1998). For example, mean depth was highly correlated with maximum depth ($|\rho| > 0.85$); mean depth was used because it better represented depth of the overall site.

Gear-specific detection probability (detectability) was investigated using a multistep approach. First, the probability of occupancy was estimated within the context of covariates hypothesised to influence occupancy, while the detectability parameter was held constant (Smith 2013). Each of these models was ranked using Akaike’s information criterion adjusted for small sample size (AIC_c; Burnham & Anderson 1998). The occupancy

Table 1. Mean and standard error (in parenthesis) of site- and sample-scale (indicated by asterisk) variables used to evaluate detection probabilities for juvenile and adult burbot sampled in the Green River, Wyoming, during the summer and autumn (2013). River section 1 is the most upstream section followed by sections 2, 3 and 4

		Summer				Autumn	
Variable	Description	Section 1	Section 2	Section 3	Section 4	Section 1	Section 3
Depth_{CV}	Mean coefficient of variation of depth	0.36 (0.02)	0.38 (0.02)	0.31 (0.02)	0.34 (0.04)	0.31 (0.02)	0.30 (0.02)
$\text{Depth}_{\text{Mean}}$	Mean depth	1.13 (0.05)	0.94 (0.02)	0.93 (0.03)	0.96 (0.10)	0.89 (0.06)	1.01 (0.06)
$\text{Velocity}_{\text{CV}}$	Mean coefficient of variation of water column velocity	0.31 (0.03)	0.39 (0.06)	0.36 (0.06)	0.46 (0.07)	0.46 (0.05)	0.41 (0.05)
$\text{Velocity}_{\text{Mean}}$	Mean water column velocity (m s^{-1})	0.60 (0.03)	0.55 (0.05)	0.61 (0.09)	0.39 (0.05)	0.40 (0.04)	0.45 (0.04)
Proximity	Distance from downstream reservoir (rkm)	46.86 (1.20)	8.77 (1.71)	120.70 (0.58)	8.72 (1.23)	46.28 (1.04)	120.60 (0.58)
Temp*	Temperature for each sampling event (°C)	19.94 (0.16)	18.80 (0.11)	14.65 (0.09)	16.07 (0.24)	16.19 (0.17)	17.10 (0.12)
$\text{Secchi}_{\text{Mean}}$	Mean Secchi disc depth (m)	0.63 (0.00)	0.72 (0.03)	0.83 (0.00)	0.61 (0.04)	0.46 (0.02)	0.93 (0.00)
$\text{Substrate}_{\text{Co-Bl}}$	Proportion of cobble and boulder substrate	0.53 (0.10)	0.81 (0.08)	0.83 (0.09)	0.12 (0.06)	0.63 (0.06)	0.93 (0.06)
$\text{Substrate}_{\text{Bl}}$	Proportion of boulder substrate	0.08 (0.04)	0.01 (0.01)	0.06 (0.06)	0.02 (0.02)	0.07 (0.03)	0.21 (0.03)
Bank_{Veg}	Proportion of vegetated bank	0.51 (0.07)	0.50 (0.08)	0.07 (0.04)	0.41 (0.12)	0.42 (0.06)	0.16 (0.06)
Bank_{Av}	Proportion of alluvium - vegetated bank	0.03 (0.02)	0.13 (0.05)	0.64 (0.10)	0.00 (0)	0.19 (0.06)	0.48 (0.06)
$\text{Bank}_{\text{Bluff}}$	Proportion of alluvium bluff bank	0.14 (0.06)	0.14 (0.07)	0.15 (0.07)	0.29 (0.08)	0.17 (0.06)	0.22 (0.06)
$\text{Bank}_{\text{Eroded}}$	Proportion of eroded bank	0.40 (0.05)	0.33 (0.05)	0.31 (0.09)	0.48 (0.09)	0.37 (0.07)	0.33 (0.07)
Mp	Proportion of instream macrophyte cover	0.36 (0.09)	0.85 (0.07)	0.63 (0.13)	0.72 (0.11)	0.30 (0.04)	0.57 (0.04)
Effort*	Total number of minutes for each sampling event (Min)	870.10 (6.65)	906.40 (3.19)	892.10 (4.48)	933.60 (11.25)	928.90 (2.70)	957.60 (2.70)

model with the lowest AIC_c value was then used to estimate the probability of detection for each gear. Again, estimates of occupancy were not the direct focus in this study. Rather, the best estimate of occupancy was needed, so the occupancy component of the model could be held constant while $\hat{p}$ was estimated. Detection probabilities of each sampling technique were estimated using a gear-specific model without including covariates. Gear-specific models consistently had more support (i.e. lower AIC_c value) than models with constant detection. Therefore, the influence of habitat characteristics on the detectability of each gear was evaluated using gear-specific detectability models. To address changes in detectability by gear, gear-specific models included interactions with habitat covariates. Because habitat characteristics likely influence life-history stages differently, juvenile and adult burbot detections were modelled separately for the summer samples (juvenile < 280 mm; adult ≥ 280 mm; Amundsen *et al.* 2003). Due to low catch rates and a shortened sampling period, it was necessary to pool juvenile and adult burbot detections during autumn. To facilitate a more direct comparison of detection probabilities between the summer and fall, juvenile and adult burbot detections from the summer were also pooled. Because Fontenelle Dam acts a barrier to fish movement, the upper (sections 1 and 2) and lower (sections 3 and 4) portions of the river were considered separate for modelling. An overdispersion parameter (*c*) was estimated for all models using a Pearson chi-square statistic and parametric bootstrap (1000 iterations). The $\hat{c}$ from the most parameterised model within the suite of candidate models was used to adjust estimated model variance for overdispersed models ($\hat{c} > 1$; Burnham & Anderson 1998; MacKenzie *et al.* 2006). If models were overdispersed, quasi-AIC_c (QAIC_c) was used to rank the models, and an additional parameter was added to further penalise candidate models (Burnham & Anderson 1998). All models with a ΔAIC_c or ΔQAIC_c of ≤ 2 were considered plausible (Burnham & Anderson 1998; Schloesser *et al.* 2012).

Results

Substrate, bank composition and Secchi depth varied little amongst sites within a section but were highly variable between river sections and seasons (Table 1). For example, the most downstream sites (section 4) averaged 12% (SE = 0.06) large substrate (cobble and boulder), whereas sites immediately downstream of Fontenelle Dam (section 3) averaged 83% (SE = 0.09) large substrate. Similarly, an overall difference of 64% alluvium and vegetated bank was apparent between sites directly upstream of Flaming Gorge Reservoir (section 4) and sites immediately downstream of Fontenelle Dam (section 3). Depth and current velocity were relatively homogenous throughout the Green River. A difference of only 0.24 m in mean depth was observed during the summer between the shallowest (mean depth ± SE; 0.89 ± 0.06 m) and deepest (1.13 ± 0.05 m) sections sampled. Current velocity exhibited similarly low variability. For instance, current velocity differed by only 0.22 m s^{-1} between sites directly downstream of Fontenelle Dam (highest mean current velocity) and sites immediately upstream of Flaming Gorge Reservoir (lowest mean current velocity) during the summer.

Length structure of burbot in the Green River increased for all gears from the summer to autumn. Additionally, the mean length of burbot sampled by all gears increased from the summer (mean length ± SE; 333.4 ± 8.9 mm) to autumn (435.7 ± 10.4 mm; Fig. 2). The greatest range of lengths of burbot was collected with night electric fishing for both seasons.

During the summer, 118 burbot were sampled using small-mesh hoop nets (n = 38 burbot), large-mesh hoop nets (n = 10 burbot) and night electric fishing (n = 70 burbot). Adult burbot were detected at 25 of the 41 sites, and juvenile burbot were detected at 29 of the 41 sites. Models including gear-specific effects on detection

Table 2. Candidate models used to evaluate the influence of site- and sample-scale covariates on occupancy (Ψ) and detection probability ($\hat{p}$) for juvenile and adult burbot sampled in the Green River, Wyoming, during the summer and autumn (2013). Only models with a ΔAIC < 2 and the most parameterized model (indicated in bold font) in each candidate set are included. Akaike's information criterion (AIC$_c$) or quasi-Akaike's information criterion adjusted for small sample size (QAIC$_c$) was used to rank all candidate models. Models ranked using QAIC$_c$ are indicated with an asterisk. Model weight (w_i) and total number of parameters (K) are also included. The relationship of covariates with detectability is indicated as positive (+) or negative (−)

Season and life history stage	Model name	AIC$_c$ or QAIC$_c$	ΔAIC$_c$ or QAIC$_c$	w_i	K
Summer					
Adult	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ (Gear, −Section)	160.26*	0.00	0.45	6
	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ (Gear, +Temp, −Effort, −Velocity$_{\text{Mean}}$, +Section)	168.83*	8.57	0.00	11
Juvenile	Ψ (−Proximity), $\hat{p}$ (Gear, −Section)	175.01*	0.00	0.39	6
	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ Gear, −Effort, −Velocity$_{\text{Mean}}$, +Section)	180.02*	5.01	0.03	10
Adult–Juvenile	Ψ (−Proximity), $\hat{p}$ (Gear, −Section)	176.22*	0.00	0.39	6
	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ (Gear, +Temp, −Effort, −Velocity$_{\text{Mean}}$, +Section)	185.03	8.81	0.00	11
Autumn					
Adult–Juvenile	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ (Gear, −Proximity, −Section)	248.56*	0.00	0.98	9
	Ψ (Bank$_{\text{Eroded}}$), $\hat{p}$ (Gear, −Effort, −Velocity$_{\text{Mean}}$, +Section)	261.23*	12.67	0.00	10

Table 3. Estimated detection probabilities ($\hat{p}$) and standard error [in parenthesis; adjusted by the variance inflation factor ($\hat{c}$)] for night electric fishing (EF), 6.4-mm mesh hoop net (HN$_{6.4}$) and 19-mm mesh hoop net (HN$_{19}$) for juvenile and adult burbot sampled in the Green River, Wyoming, during the summer and autumn (2013)

	Detection probability ($\hat{p}$)		
Season and life history stage	EF	HN$_{6.4}$	HN$_{19}$
Summer			
Adult	0.30 (0.20–0.41)	0.27 (0.18–0.39)	0.09 (0.05–0.18)
Juvenile	0.35 (0.26–0.46)	0.26 (0.17–0.36)	0.10 (0.05–0.17)
Adult–Juvenile	0.35 (0.26–0.46)	0.26 (0.17–0.36)	0.10 (0.05–0.17)
Autumn			
Adult–Juvenile	0.39 (0.28–0.52)	0.46 (0.33–0.59)	0.18 (0.10–0.30)

probabilities had clear support for juvenile and adult burbot (Table 2). None of the top models from each suite of candidate models from the summer (i.e. juveniles, adults and pooled) contained habitat covariates. Probability of detection of adult burbot in the summer was lowest with large-mesh hoop nets (Table 3). Night electric fishing and small-mesh hoop nets had similar probabilities of detection for adult burbot. The probability of detection for juvenile burbot using night electric fishing and small-mesh hoop nets were similar during the summer. Detectability of juvenile burbot was similar to that of adult burbot for all gears during the summer.

A total of 112 burbot was captured during autumn using small-mesh hoop nets (n = 53 burbot), large-mesh hoop nets (n = 13 burbot) and night electric fishing (n = 46 burbot). Adult burbot were detected at 20 of the 28 sites, and juvenile burbot were detected at 5 of the 28 sites. In autumn, the probability of detection of juvenile and adult burbot was similar for small-mesh hoop nets (Table 3) and night electric fishing. Although there was an overall increase in detection probability during autumn compared to summer, large-mesh hoop nets continued to have the lowest probability of detection. Along with gear type, models containing proximity to downstream reservoir had the most support (Table 2). When controlling for upper and lower sections, the proximity to downstream reservoir negatively influenced the probability of detection for all gears (Table 4).

Discussion

A gear-specific effect on detection probabilities was apparent for juvenile and adult burbot for both seasons. The effect of gear on detection probabilities was not surprising given the long-established biases associated with different fish sampling techniques (Paukert 2004; Hetrick & Bromaghin 2010). For instance, Schloesser *et al.* (2012) reported that detection probabilities for a number of Missouri River fishes were most influenced by sampling technique (i.e. gillnet, trammel net, otter trawl). Similarly, Beamesderfer and Rieman (1988) found a species-specific pattern of size selectivity for largemouth bass, *Micropterus salmoides* (Lacepède), walleye, *Sander vitreus* (Mitchill) and northern pikeminnow, *Ptychocheilus oregonensis* (Richardson), in John Day Reservoir, Oregon, using traps nets, electric fishing, angling and two sizes of monofilament gill nets. In the current study, length frequencies indicated that night electric fishing was the least size-selective gear. A number of factors have been identified as influencing gear biases including biological, technical and environmental characteristics (Reynolds & Kolz 2012). The effect of environmental factors on gear bias was accounted for by including habitat characteristics in occupancy models. However, the current study did not assess the technical and biological factors influencing gear bias. Without further research, it is difficult to assess all the constituents that influence the vulnerability of burbot to a given gear. Regardless, the findings of the current study highlight the need to better understand the biases and variations in detectability of both active and passive gears for sampling burbot.

Only one other study has evaluated both passive and active sampling techniques for burbot in lotic systems. Smith (2013) sampled burbot in the Kootenai River, Idaho and the Green River, Wyoming, using day electric fishing and 6.4-mm bar mesh hoop nets as part of a larger fish assemblage study. Smith (2013) reported that detection probabilities ($\hat{p} \pm$ SE; 0.19 ± 0.14) for juvenile burbot were identical between 6.4-mm bar mesh hoop nets and day electric fishing. However, detection probabilities for adult burbot were quite different between day electric fishing (0.12 ± 0.09) and 6.4-mm

Table 4. Beta estimates from top models used to evaluate the occupancy (Ψ) and detection probability ($\hat{p}$) for juvenile and adult burbot sampled in the Green River, Wyoming, during the summer and autumn (2013). Parameter estimates and confidence limits are presented on the odds-scale

			Confidence limits	
Season and model	Covariates	Estimate	Lower	Upper
Autumn				
Ψ (Bank$_{\text{Talus}}$), $\hat{p}$ (Gear, Proximity, Section)	Ψ	1.75^{e+00}	5.70^{e-01}	5.39^{e+00}
	Ψ (Bank$_{\text{Talus}}$)	1.24^{e+03}	1.63^{e-04}	9.44^{e+09}
	p (Section)*	1.14^{e-07}	9.65^{e-09}	1.34^{e-06}
	HN$_{6.4}$ (Proximity)*	9.12^{e-02}	6.41^{e-02}	1.30^{e-01}
	HN$_{19}$ (Proximity)*	1.12^{e-01}	7.65^{e-02}	1.65^{e-01}
	EF (Proximity)*	1.45^{e-01}	1.06^{e-01}	1.98^{e-01}
	HN$_{6.4}$*	8.69^{e+11}	1.49^{e+10}	5.06^{e+13}
	HN$_{19}$*	4.05^{e+10}	5.56^{e+08}	2.95^{e+12}
	EF*	1.55^{e+10}	3.86^{e+08}	6.25^{e+11}

*Significant parameter estimates.

bar mesh hoop nets (0.61 ± 0.14). Although the detection probabilities for juvenile and adult burbot using 6.4-mm bar mesh hoop nets from this study were different from Smith (2013), direct comparisons are tenuous given disparate sampling designs between the two studies. Specifically, Smith (2013) set four 6.4-mm bar mesh hoop nets at each sampling event for 12 h and pooled detections of burbot, whereas a single 6.4-mm bar mesh hoop net was set for 24 h per sampling event in the current study. The addition of three nets per site likely influenced the detection probabilities reported by Smith (2013). Smith (2013) reported much lower detection probabilities using day electric fishing compared with the results for night electric fishing from the current study. No studies have specifically compared day and night electric fishing for burbot. However, previous research has reported higher catch rates and greater species diversity using night electric fishing compared to day electric fishing (Sanders 1992; Pierce *et al.* 2001; McInerny & Cross 2004). The disparity in catch rates between day and night electric fishing is likely due to differences in the ecology of the study species. For example, Sanders (1992) reported collecting approximately double the number of flathead catfish, *Pylodictis olivaris* (Rafinesque), in the Ohio and Muskingum rivers using electric fishing at night compared to the day. Flathead catfish is reported as being largely inactive during the day and exhibiting greater movement and feeding during the dusk, night and dawn (Daugherty & Sutton 2005; Vokoun & Rabeni 2006). Burbot is similarly inactive during the day but begin foraging at dusk and continue until dawn (Carl 1995). Additionally, burbot often use large substrates as cover during periods of inactivity (Dixon & Vokoun 2009; Eick 2013), which may reduce their vulnerability to capture by electric fishing (Reynolds & Kolz 2012). The sedentary behaviour of burbot coupled with their affinity for large substrate habitats likely accounts for decreased detectability observed with day electric fishing.

Models including covariates for habitat were not strongly supported using data from the summer. However, proximity to downstream reservoir negatively influenced the detection probability of all gears during autumn. The relationship between gears and proximity to downstream reservoir may be related to pre-spawning movement exhibited by adfluvial burbot. Adfluvial populations of burbot occur in eastern North America, northern latitudes of the Missouri River system and the Kootenai River system of Idaho and British Columbia (Paragamian 2000; Hubert *et al.* 2008). Many adfluvial burbot populations undergo pre-spawning movement in autumn (Lawler 1963). Pre-spawning movement of burbot in the Green River likely coincided with autumn sampling as evidenced by the increased mean length of burbot and higher relative abundance. For instance, 64 juvenile and adult burbot averaging 379.3 mm were caught in section 1 during the autumn compared with 31 juvenile and adult burbot averaging 287.5 mm caught in section 1 during the summer. Because detectability is often positively related to site-specific abundance (Royle & Nichols 2003), sampling conducted at sites closer to source reservoirs likely resulted in increased detection probabilities.

Occupancy modelling provided a useful framework to compare active and passive sampling techniques while accounting for the potential influence of habitat characteristics on detectability. Night electric fishing and small-mesh hoop nets had similar detection probabilities for juvenile and adult burbot in the Green River; however, night electric fishing required substantially less effort. Although night electric fishing is possible in many western systems, safety considerations of working at night will likely preclude its use in many river systems. Additionally, electric fishing efficiency decreases with increasing water depth (Reynolds & Kolz 2012). In situations where electric fishing is not possible, small-mesh hoop nets may be a suitable alternative. Not only are hoop nets effective at sampling deep habitats, but concerns of working in a lotic system at night can be avoided in that hoop nets can be set during the day. Regardless of the sampling technique employed, it is important to note that given the prevalence of adfluvial populations of burbot in many rivers, sampling conducted in autumn will likely be more effective than in other seasons. By using night electric fishing, small-mesh hoop nets or a combination of the two, burbot can be efficiently and effectively monitored in lotic systems characteristic of the western USA.

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