RESEARCH ARTICLE

Extirpation for Conservation: Applying Predictors of Extinction Risk to Eradicate Introduced Trout Populations for Lake Restoration

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ABSTRACT

The restoration of mountain lakes affected by non-native fishes requires the removal of the introduced species. While extensive research has illuminated the physical habitats and complex food webs of mountain lake ecosystems, little is known about the self-sustaining fish populations, particularly in our study area, Waterton Lakes National Park (WLNP), Canada. We used generalized linear models to examine four population characteristics associated with the vulnerability of populations to depletion by gillnetting: 1) catch per unit effort (CPUE), 2) proportion of females, 3) proportion of mature individuals and 4) length of mature trout, as a proxy for age at maturity. There were significant differences between populations in CPUE and length of mature trout, but not in the proportion of females or mature individuals. We thus incorporated the former characteristics to rank 11 trout populations by their susceptibility to eradication. Two lakes in the Lineham basin were the best candidates for trout eradication due to their low population density and large size at maturity. The application of demographic characteristics to select introduced populations for eradication is a simple yet meaningful step in restoration commonly constrained by a lack of biological knowledge.

Keywords: aquatic restoration, exotics, introduced species, *Oncorhynchus clarkii*, protected areas, *Salvelinus fontinalis*

The stocking of salmonid fishes into historically fishless mountain lakes of western North America was so widespread in the twentieth century that it created a landscape in which almost all lakes have been affected (Bahls 1992). When stocking stopped, salmonids continued to affect food webs because many introduced populations became self-sustaining. In this case, the native, fishless ecosystems have been replaced with novel systems dominated by predatory fishes. Introduced trout are known to affect zooplankton, benthic invertebrates and amphibians in mountain lake ecosystems (Carlisle and Hawkins 1998, Donald et al. 2001, Knapp et al. 2001). To reverse these effects, resource managers may decide to restore the native ecosystems by eliminating introduced salmonid populations.

Gillnetting is a viable method of eradicating trout populations in mountain lakes, though success requires multiple years of netting, substantial resources and is constrained by lake morphology (Knapp and Matthews 1998). Gillnets function by lethally entangling fish at the gills as they attempt to swim through the undetectable mesh. The successful removal of brook trout (*Salvelinus fontinalis*) from a small Sierra Nevada lake required 3679 net days over three years (Knapp and Matthews 1998). Trout from five additional lakes in Sierra Nevada were eliminated by gillnet from 1996 to 2003 (Vredenburg 2004). Brook trout were eventually eliminated from Bighorn Lake in Banff, requiring over 10,000 net nights over three years (Parker et al. 2001). Other trout-eradication projects occurred in the Devon Lakes system in Banff National Park and an additional six lakes in Sierra Nevada, (C. Pacas, Parks Canada, pers. comm., Knapp et al. 2007); both spanned multiple years. In instances where trout populations cannot be driven to extirpation, intense netting regimes have effectively suppressed population densities in several mountain lakes (Gresswell 2009, Rosenthal et al. 2012). Despite the sheer effort required, gillnetting is typically preferred over the application of piscicides such as rotenone, which have lethal affects on invertebrates and can prolong time to recovery (Anderson 1970). Gillnetting, in contrast, has little to no impact on non-target species.

Given the substantial effort required to reduce fish populations and the number of mountain lakes affected,

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a simple method of prioritizing lakes for management would facilitate restoration decisions. Indeed, ranking systems are widely used for invasive non-native plants for which numerous infestations of the same species exist (*e.g.* Pheloung et al. 1999, Skurka Darin et al. 2011). Such systems are especially practical where management priorities are difficult to assign due to limited resources. In fishes, decision-making tools for conservation or extirpation are habitat-based (Levin and Stunz, 2005), genetics-based (Allendorf et al. 1997), or species-based (Britton et al. 2011). These tools do not meet the needs of resource managers in protected areas where habitat quality is similar, genetic studies are unaffordable, and multiple populations of a species are at issue. Knapp and Matthews (1998) reported that success of gillnetting in mountain lakes is dependent on physical morphometric factors such as lake depth, surface area, outlet width, and area of stream spawning habitat. These physical morphometric characteristics are usually known, even in remote mountain lakes, whereas the biota remains relatively unstudied. Hence, we are interested in quantifying trout biology characteristics that influence the susceptibility of populations to over-exploitation by gill net.

Conservation theory suggests that certain demographic characteristics increase a population's extinction risk. Our approach is to examine general predictors of extinction risk and test their utility for predicting eradication potential of introduced species under active depletion. Firstly, populations which are small or at low density are commonly thought to be more likely to go extinct. Genetic factors such as inbreeding and loss of genetic variability result in reduced fitness and adaptability, while demographic considerations such as the Allee effect and stochasticity can push declining populations beyond recovery (Lande 1988). Secondly, a high proportion of females in a population can also have a negative association with extinction risk, especially where reproduction rates are limited by sexually-reproducing females, as in some salmonids (Blanchfield and Ridgway 1997). Thirdly, age-at-maturity has been used to predict extinction risk across multiple taxa (Hutchings et al. 2012), and is positively correlated with extinction risk in freshwater fishes (Anderson et al. 2011). Furthermore, Marschall and Crowder (1996) found that brook trout populations reacted most negatively to factors that decreased the survival of large juveniles and small adults, and that removing large mature individuals was not necessarily detrimental to the persistence of the population because brook trout can reproduce at a small size. Body size is also a major factor in the efficiency of gillnets, since catchability generally increases with fish size (Jensen 1995, Finstad et al. 2000, Borgstrøm et al. 2010). Hence, populations in which individuals mature at a large body size are easier to remove by gillnet than individuals that mature at a smaller body size. Therefore, size at maturity and the proportion of mature individuals might influence extinction risk and the success of active depletion by gillnet.

Contemporary fisheries research indicates that population characteristics are often density-dependent. Populations at low densities can compensate for overexploitation by altering life-history traits such as survival, growth and reproduction (Johnston and Post 2009), which is why population removal can be difficult to achieve (*e.g.* Meyer et al. 2003). Opposite to the compensatory response is depensatory density dependence, where a population's density is driven to a point too low for the population to recover (Rose et al. 2001). Across global fisheries, 21% of collapses are related to depensatory mechanisms (Mullon et al. 2005). In freshwater fishes, concentrations of fish in spatially limited habitat can increase their catchability, thereby reducing population densities towards an "invisible collapse" (Lewin et al. 2006).

Our approach generates knowledge of the introduced populations and synthesizes it to pinpoint populations closer to this "cliff edge" of depensation. We investigate variability in selected population characteristics across introduced salmonid populations in Waterton Lakes National Park (WLNP), Alberta, Canada, and apply our results to restoration decisions. The goal of our study is to rank the lakes in WLNP according to their suitability for restoration based on trout demographic characteristics that may render them more susceptible to depletion and ultimately extinction. We therefore examined variation in population density (Catch per Unit Effort: CPUE), the proportions of females and mature individuals in the population, and the size at maturity (fork length of mature individuals), across 11 previously stocked lakes in WLNP. We assumed that trout populations that are characterized by low density, few females, few mature individuals, and a large body size at maturation, would be more amenable to eradication by gillnet. This research will provide insight into the characteristics of introduced salmonid populations and facilitate restoration by offering a science-based system of prioritizing impacted mountain lake ecosystems.

Methods

Study Site

WLNP (49.0458°N, 113.9153°W) protects 505 km² of the southern Canadian Rockies. The weather is characteristic of mountain environments; the average snowfall is 481.5cm per year and an average of 192 days per year have a minimum temperature above 0°C. WLNP contains 22 high elevation lakes that range from 1524 to 2195m above sea level. Previously fishless, stocking by the park commenced in the 1920s and ended in the 1980s, during which brook trout, cutthroat trout (*Oncorhynchus clarkii*), and rainbow trout (*O. mykiss*), were introduced. Thirteen lakes presently

Table 1. Characteristics of study lakes in Waterton Lakes National Park, Alberta, CA. Physical characteristic data from Anderson 1975. Chemical characteristics are SC = Specific Conductivity, TDS = Total Dissolved Solids, and pH. Watershed acronyms are CL = Cameron Lake, CC = Cameron Creek, UWL = Upper Waterton Lake, BB = Blakiston Brook, BC = Bauerman Creek WBC = West Boundary Creek.

Lake	Code	Latitude (degrees north)	Longitude (degrees west)	Elev. (m asl)	Mean Depth (m)	Max Depth (m)	Area (ha)	Water-shed	SC (µS/cm^3)	TDS (g/L)	DO%	pH
Akamina	AK	49°01'00″	114°02′00″	1655	—	5.0	4.65	CL	0.086	0.056	93.6	7.2
Alderson	AL	49°02′00″	113°02′00″	1811	21.5	60.0	10.19	CC	0.126	0.082	88.9	7.6
Lower Carthew	CL	49°02′00″	113°59′00″	2159	4.8	11.0	7.33	CC	0.089	0.061	87.3	7.9
Crandell	CR	49°05′00″	113°58′00″	1524	7.9	15.5	4.53	BB	0.225	0.145	91.8	8.3
Crypt	CT	49°00′00″	113°50′00″	1963	16.9	44.0	13.44	UWL	0.049	0.032	84.2	5.9
Goat	GO	49°10′00″	114°05′00″	1982	3.4	9.3	2.35	BC	0.092	0.068	91.3	8.0
Lineham Hourglass	LH	49°05′00″	114°04′00″	2111	10.4	23.0	12.64	CC	0.094	0.063	61.2	7.4
Lineham North	LN	49°05′00″	114°04′00″	2170	11.6	29.0	18.96	CC	0.081	0.053	85.6	8.0
Lone	LO	49°05′00″	114°07′00″	2027	5.4	13.0	2.53	BB	0.029	0.019	93.5	7.6
Lower Twin	TL	49°08′00″	114°09′00″	1927	3.6	8.0	2.72	BB	0.090	0.058	98.7	7.8
Upper Twin	TU	49°08′00″	114°09′00″	1963	5.1	13.0	6.44	BB	0.021	0.013	91.6	6.9

retain trout populations that have become self-reproducing, 11 of which were included in this study (Table 1). Two lakes were excluded because they could not be accurately sampled using the same methods due to their large size.

Data Collection

Trout populations in twelve mountain lakes were sampled, but two lakes were confirmed fishless. All lakes were sampled twice in the ice-free season, between July and September 2011, except for Crypt Lake (CT), which was sampled in July and August 2012. Spring sampling occurred between June and July, summer sampling occurred in July and August. Multiple visits were made to quantify seasonal variation.

Between one and five monofilament gillnets were set in each sampling period in each lake depending on lake size (manufactured by Lundgrens Fiskredskapsfabrik AB, Stockholm, Sweden). In the more remote lakes, logistical limitations constrained the number of nets we could use and in one severe case only one net was deployed. The overnight bottom sets optimized periods of high trout activity and we aimed for a consistent net duration of 14 hours. Nets were 30 m long, with five 6 m panels of different mesh gauge (18.5, 25.0, 38.0, 43.0 and 55.0 mm) arranged sequentially. Each net was set perpendicular to the shoreline, with one end secured to a fixed feature on shore and the deep end anchored to the substrate. The orientation of the smallest mesh was alternated equally between lake-end and shore-end. The nets were spaced evenly around the perimeter of the lake and all shoreline types were covered as best as possible. Different gillnet locations were changed for each sampling period and all locations were marked using GPS.

In the morning, nets were collected from an inflatable raft. Fish were measured to the nearest millimeter, weighed to the nearest gram, identified to species, and assessed for sex and maturity. Sex and maturity were determined by dissecting each fish and observing gonads. Each fish was assigned a unique number; the corresponding mesh size and net was recorded. A subsample of the catch (n = 115) representing the range of sizes caught was sampled for stomach contents and age determination.

Statistical Analyses

We used generalized linear models (GLMs) to assess variation in the four demographic characteristics (density, length of mature individuals, proportion of females and proportion of mature individuals) across populations. Since the lakes were isolated and freshwater fish populations generally display high levels of population variation (DeWoody and Avise 2000), it is likely that each lake is a different population of trout. The characteristics were calculated for each gillnet, so each lake (population) was represented by multiple data points. CPUE was used as a proxy for density and calculated as the number of individuals caught divided by the duration of net set in hours. Fork length of mature trout, a proxy for age-at-maturity, was the mean fork length of only the mature fish in the catch. Proportions were calculated as the number of females and mature individuals divided by the total catch per net. CPUE and length data were normally distributed and hence were modeled with a Gaussian distribution. Data for the proportion metrics were not normally distributed, so a binomial distribution was applied, weighted by the number of fish caught. Explanatory variables other than population included in the model were fish species, sampling period, and their interaction. The Akaike Information Criterion (AIC, Akaike 1973) was used to select among the ten models for each demographic characteristic; the lowest AIC value represents the most parsimonious model and models within 2 ΔAIC were ordered by the number

Table 2. Results of two factor generalized linear models (GLMs) to assess the importance of population, season, species and the interaction of season and species on the variability of four demographic characteristics (CPUE, fork length of mature individuals, proportion of females and proportion of immature individuals). X indicates the factor was included in the model. Adjusted R^2 values are reported; *R^2 calculated from deviances for proportion variables.

Variable	Rank	Population	Season	Species	Interaction	AIC	R^2
CPUE	1	X	X			43.24	0.79
	2	X	X	X		43.24	0.79
	3	X	X	X	X	44.87	0.78
	4	X		X		58.59	0.72
	5	X				58.59	0.72
	6		X	X		109.17	0.24
	7		X	X	X	110.81	0.23
	8			X		110.89	0.20
	9		X			121.82	0.04
	10					122.95	0.00
Fork Length at Maturity	1	X				533.19	0.76
	2	X		X		533.19	0.76
	3	X	X			535.13	0.76
	4	X	X	X		535.13	0.76
	5	X	X	X	X	536.45	0.76
	6			X		590.21	0.28
	7		X	X		592.02	0.27
	8		X	X	X	593.60	0.26
	9					608.27	0.00
	10		X			610.18	-0.16
Proportion Female	1					77.04	0.00*
	2		X			77.56	0.09*
	3			X		77.99	0.03*
	4		X	X		78.55	0.11*
	5		X	X	X	80.71	0.12*
	6	X		X		88.41	0.35*
	7	X				88.41	0.35*
	8	X	X			89.24	0.42*
	9	X	X	X		89.24	0.42*
	10	X	X	X	X	91.40	0.42*
Proportion Mature	1					19.17	0.00*
	2			X		21.19	0.01*
	3		X			21.22	0.03*
	4		X	X		23.23	0.04*
	5		X	X	X	25.37	0.13*
	6	X		X		39.83	0.44*
	7	X				39.83	0.44*
	8	X	X			41.93	0.48*
	9	X	X	X		41.93	0.48*
	10	X	X	X	X	44.14	0.58*

of variables (Burnham and Anderson 2002). All statistical analyses were performed in R Statistical Software (R Statistical Software, R Development Core Team).

Ranking Lakes for Trout Eradication

We used the above analysis of demographic characteristics to indicate suitable factors on which to rank populations for removal. Lakes were ranked from one to eleven for each characteristic applied, where one represented the condition of the population that is most amenable to eradication. Scores for each characteristic were summed for each lake, yielding a final ranking of lakes by their suitability for restoration.

Results

Fifty nets were set in 12 lakes in 2011, plus an additional eight nets in the remaining lake (CT) in 2012. A total of 1369 trout were caught in ten of the lakes sampled. Two lakes yielded no fish. Three species were represented: brook trout, cutthroat trout, and rainbow trout. All lakes contained exclusively either brook trout or cutthroat trout, except Little Akamina Lake, in which a small number of rainbow trout were caught. No further analysis was done on this species. A combined total of 706 brook trout were caught in CR, TU, TL, and AK, while 649 cutthroat trout were caught in LN, LH, CL, AL, GO, LO, and CT (lake acronyms defined in Table 1). The finding that lakes contained only one species supported our definition of "population" as the fish biomass of one lake. Gillnetting confirmed an absence of fish in LS and CU.

Demographic Characteristics of Trout Populations

We found that population was the most important variable explaining the variation in two demographic characteristics: CPUE and fork length of mature fish. The models were close in terms of best-fit so parsimony rules (Akaike 1973) were upheld to select the best models. For the GLMs based on CPUE data, the best-fit model included population and season, which explained 83% of the variability in the dataset (Table 2; linear regression, r^2 = 0.79, $F_{11,46}$ = 19.97, p < 0.0001). Removing the season variable revealed that the variation across populations was far more important than that across seasons (linear regression, r^2 = 0.72, $F_{10,47}$ = 15.43, p < 0.0001). CPUE was consistently higher in the spring than in the summer (Figure 1a). When averaged over season, CPUE ranged from 0.63 (LH) to 1.71 (CL), except in two lakes where CPUE was much higher (TU = 2.49, TL = 2.59; Figure 1a). Correlation analysis showed that CPUE was not affected by variable set durations (r(52) = 0.060, t = 0.45, p = 0.7).

The model that best explained fork length of mature trout included only population as an explanatory variable (Table 2; linear regression, $r^2 = 0.76$, $F_{10,47} = 19.51$, $p < 0.0001$). Average values were distributed evenly across a range of 195.2 mm (TU) to 292.2 mm (LN), but mature trout were far larger in CT (327.4 mm; Figure 1b).

Population was not a main factor explaining the variance in the remaining two demographic characteristics: the proportion of females and the proportion of mature trout (Table 2 and Figures 1c and d). GLMs revealed that the best model for both characteristics was the null model, indicating that variation was also not evident across season or species (Table 2).

Ranking Lakes for Trout Eradication

Our ranking system identified two lakes, Lineham Hourglass (LH) and North Lineham (LN), as the most suitable for trout eradication by gillnet due to their combined low population density and large size at maturity (Table 3). The evaluation employed CPUE and length of mature individuals at equal weights, as our GLMs suggested that population had a similarly strong influence on both. The R^2 values for the two characteristics fell within a range of 0.03 (Table 3). We excluded the proportion of females and mature individuals from the assessment because we found no evidence of significant variation across populations (Table 2). AL, LO, CT, CR, and AK were the next highest-ranked lakes, followed by CL and GO. TU and TL were by far, the least appropriate lake for restoration by trout removal because they contained low-density populations that mature at a small body size (Table 3).

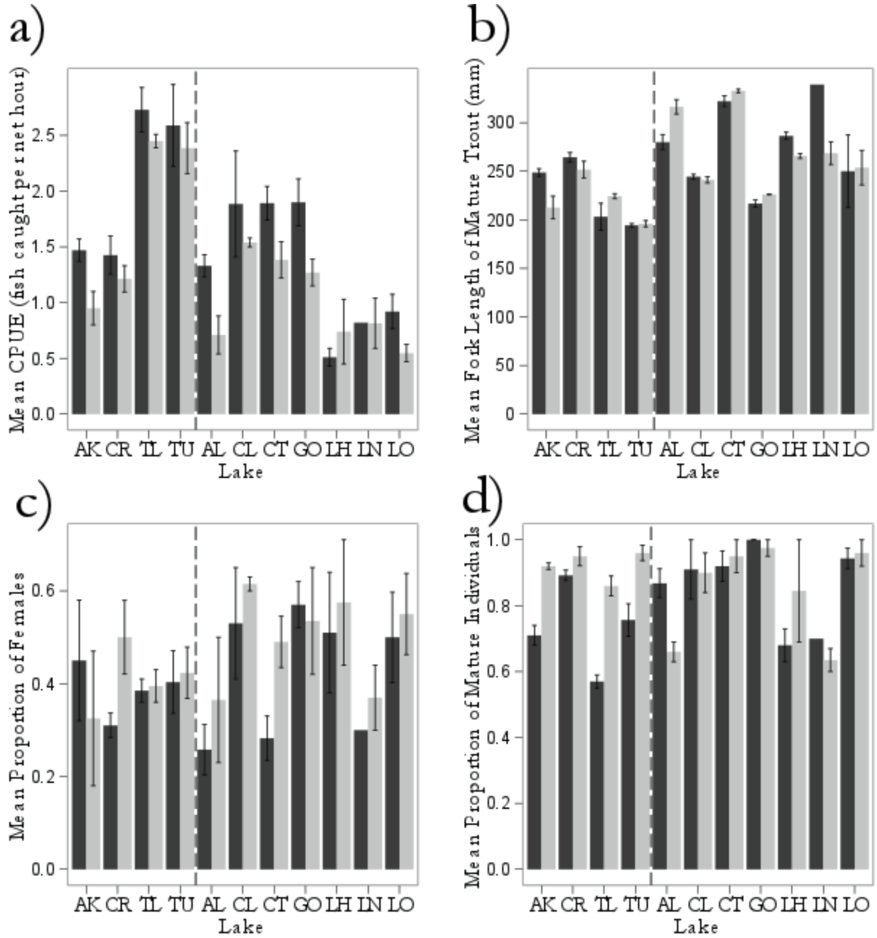

Figure 1. Mean values (± SE) of four demographic characteristics [a) mean CPUE, b) mean fork length of mature trout, c) mean proportion of females, d) mean proportion of mature individuals] for eleven trout populations in Waterton Lakes National Park, Alberta, CA. Lake codes as per Table 1. Dark grey bars represent spring data, light grey bars represent summer data. Dashed line divides brook trout (*Salvelinus fontinalis*; left) from cutthroat trout (*Oncorhynchus clarkia*; right). Only one net was cast in the spring in LN, so standard error could not be calculated.

Table 3. Ranking of 11 previously stocked Waterton Lakes National Park lakes by susceptibility, based on demographic characteristics CPUE and fork length of mature trout (FLM). Lake codes as per Table 1.

Code	CPUE	FLM	Total	Rank
AK	5	8	13	6
AL	4	3	7	2
CL	9	7	16	7
CR	6	5	11	5
CT	8	1	9	4
GO	7	9	16	7
LH	**1**	**4**	**5**	**1**
LN	**3**	**2**	**5**	**1**
LO	2	6	8	3
TL	11	10	21	8
TU	10	11	21	8

Discussion

We were able to quantify variation in demographic characteristics across introduced trout populations in WLNP to identify those populations that would be most susceptible to eradication. Our cursory ranking system specifically identified Lineham Hourglass Lake (LH) and North Lineham Lake (LN) as top candidates for trout depletion, based on demographic characteristics that had not previously been measured in WLNP.

Demographic Characteristics of Trout Populations

Our results indicated that variation in two demographic characteristics (CPUE and fork length of mature individuals) of WLNP trout populations were explained by population-level differences. These characteristics are also prevalent in the literature as predictors of extinction risk. Low-density populations are more prone to genetic and demographic factors leading to extinction (Lande 1988, Willi et al. 2006). In Canadian freshwater fishes, delayed maturity (often accompanied by larger body size) is the best predictor of extinction risk (Anderson et al. 2011). Density and size at maturity are therefore pertinent to the selection of introduced populations for removal in the event of lake restoration.

The finding that CPUE and fork length at maturity varied across populations was not unexpected given the substantial variation in lake morphometry, chemistry, and food web composition. Stocking histories and fishing use also differed from lake to lake and are further sources of variation. Similar ranges in density (< 0.1–6.8 fish hr^{-1} per 30.5 m net) and mean lengths (11–56 cm) were reported for brook trout in 183 Rocky Mountain lakes in Wyoming, USA (Chamberlain and Hubert 1996). Lacustrine populations in the eastern, native range also demonstrated impressive variation in CPUE and mean length (Lachance and Magnan 1990, Quinn et al. 1994, Magnan et al. 2005).

The reported variation in these stocked populations was generally attributed to lake morphometrics (size and elevation of lake) and the density of other fishes (Chamberlain and Hubert 1996), while fishing intensity, the density of competitors, and community complexity explained variation in density in the native range (Lachance and Magnan 1990, Quinn et al. 1994, Magnan et al. 2005). Growth rate of Alberta populations was related to amphipod abundance, productivity, and water temperature and negatively related to elevation (Donald et al. 1980). Variation in the eastern distribution was explained by competitor biomass, community complexity, salmonid diversity, and fishing intensity (Lachance and Magnan 1990, Quinn et al. 1994, Magnan et al. 2005).

Cutthroat trout populations display similar variation in CPUE and body length, but perhaps for different reasons. In the Bighorn Mountains of Wyoming, mean total length ranged from 220–425 mm and density from 0.4–2.4 fish net^{-1} $hour^{-1}$, across 19 lakes (Bailey and Hubert 2003). Unlike brook trout, cutthroat trout mean length was not associated with environmental factors but with density and lake accessibility (Bailey and Hubert 2003). Meanwhile, accessibility was the only factor associated with CPUE. In the absence of further studies on lake populations, spatial variability in size at maturity was observed cutthroat trout in Montana streams (110 mm to 180 mm; Downs et al. 1997). Overall, the demographic characteristics of density and length, of both brook trout (in novel and native habitats) and cutthroat trout, are spatially variable due to physical and chemical lake attributes, food web composition, fishing intensity, and the density of other fishes.

Although population explained most of the variation in WLNP trout density, season also had an effect. Densities were consistently lower in the summer, which could be due to decreased activity in the littoral zone during the later sampling period. During the period of summer stratification, cutthroat trout avoid near-surface waters but are nearer to the surface when lakes are mixed (spring and fall) (Nowak and Quinn, 2002, Baldwin et al. 2002). A similar trend in WNLP's dimictic lakes could be expected to reduce the efficiency of shoreline gillnet sets in the summer. Densities could also have been reduced by efficient gillnetting in the spring sampling period, leaving reduced numbers of trout vulnerable to gillnets in the summer.

The proportions of females or of mature individuals did not vary across populations. Rather, our results suggested a high degree of variability within each lake, particularly for the proportion of females (Figure 1c). Though the mean values for both species were comparable to reports by Downs et al. (1997) and Meyer et al. (2003), they do not describe the variation within each lake. Despite literature support for these factors weighing heavily on extinction risk due to female-limited reproductive strategies (Blanchfield and Ridgway 1997), we did not find the proportion of females or of mature individuals to be useful in prioritization because there was little variation across populations. These factors were thus omitted from our ranking of populations for depletion.

Management Implications

For aforementioned reasons, only trout density and fork length of mature trout were used in a ranking system to distinguish populations with higher susceptibility to population depletion by gillnet. Similar assessment tools have been principally developed for invasive land plants, where ecological gains can be optimized by prioritizing populations for management action (*e.g.* Pheloung et al. 1999, Skurka Darin et al. 2011). The management of freshwater fishes has also recently benefited from modifications of such tools to aquatic invaders. For example, Copp et al. (2009) developed the Fish Invasiveness Scoring Kit (FISK) to distinguish potentially invasive and non-invasive species, which was used as a pre-assessment for the more instructive modular assessment tool by Britton et al. (2011). The latter incorporates species prioritization, population-level risk to receiving waters, management action impacts and costs of management actions, to assess introduced fish populations for management priority. Such systems are effective because they can be molded to fit the values of a particular region while retaining the structure needed to maintain transparent decision-making in governmental organizations. Even within the umbrella of national mandates, regions may value resources differently (*e.g.* angling value) and managers can assign higher weight to the criteria that have greater importance in their particular jurisdiction. A downfall to the majority of ranking systems is that they are impact-based, which is impractical in situations where impacts are equal across the landscape, such as WLNP. Fine-tuning existing assessment tools to hone in on demographic differences that affect management action success will improve their practicality in these landscapes.

The two lakes that were ranked highest by our assessment system, LN and LH, have a similar combination of high trout population density and low size of mature trout. They also share similar abiotic factors such as elevation, lake depth, lake area, and accessibility. They are two of the highest lakes in WLNP, but are within 60m in elevation. The difference in depth between the two lakes is less than 1.5 m and the difference in area is under 7 ha. Though these lakes may contain the best populations to deplete based on biological characteristics, they are remote and difficult to access by foot and helicopter. The safest route to access LH and LN is via an 8 km trail over a ridge, followed by a few kilometers of steep off-trail terrain. Unsurprisingly, these lakes receive low visitor use and fishing pressure, but are highly valued as representations of undisturbed ecosystems. Thus, further manipulation of the Lineham Lakes basin may be opposed by conservationists and backcountry users. Nevertheless, human and physical considerations could supersede biological factors when selecting lakes for restoration by trout eradication. The responsibility of resource managers to uphold regional values when making management decisions is facilitated by ranking systems such as that presented in this study, and by the provision of hard-to-measure biological data.

Ironically, this study concurrently instructs trout eradication and conservation. That is, we have found support of intraspecific diversity in exotic populations, which can be considered as biodiversity (Fraser and Bernatchez, 2001), particularly in western North American freshwater habitats depauperate of native fish fauna (Keeley et al. 2005). If restoration is not pursued, the population characteristics investigated in this study are still valuable for the management and continued monitoring of high mountain lakes.

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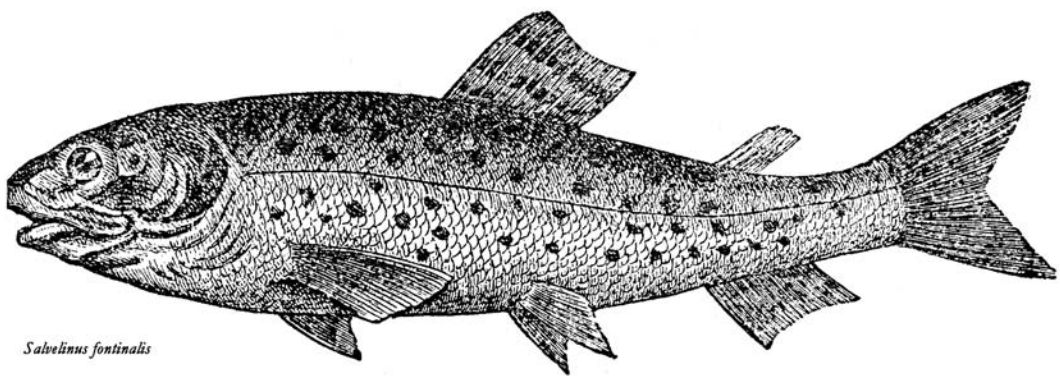

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