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FISH-3737; No. of Pages 9

ARTICLE IN PRESS

Fisheries Research xxx (2013) xxx–xxx

Contents lists available at ScienceDirect

Fisheries Research

journal homepage: www.elsevier.com/locate/fishres

Evaluating the relationship between mean catch per unit effort and abundance for littoral cyprinids in small boreal shield lakes

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ARTICLE INFO

Article history:
Received 30 April 2013
Received in revised form 23 October 2013
Accepted 25 October 2013

Keywords:
Fathead minnow
Pearl dace
Minnow trap
Trap net
Experimental Lakes Area
Mark-recapture

ABSTRACT

Catch per unit effort (CPUE) is commonly used as a relative measure of littoral fish abundance; however, few studies have examined this relationship for boreal shield lakes. We used non-linear regression to generate relationships between mark-recapture abundance estimates and mean CPUE derived from 7 years of standardized fishing using baited minnow traps for two common cyprinid species; pearl dace (*Margariscus margarita*) and fathead minnows (*Pimephales promelas*), in littoral areas of two small boreal lakes. We produced significant, positive CPUE-abundance relationships for pearl dace and fathead minnows. Pearl dace were less variable in daily CPUE during the course of the study, suggesting they may require less sampling effort than fathead minnows to precisely estimate their population size. Density estimates derived from our estimates of abundance were consistent with those from similar boreal shield lakes, providing confidence in our method to estimate abundance. Finally, we developed relationships to estimate population size from long-term monitoring data collected on these same cyprinid species using two types of small mesh trap nets. Non-linear relationships were developed between mean trap net CPUE and abundance estimates derived from minnow traps for fathead minnows, but not pearl dace. These relationships should permit population estimates from mean CPUE data collected using similar capture methods in similar lakes.

1. Introduction

Data collection to form accurate and sufficiently precise population estimates is inherently labour-intensive and can require multiple sampling of the same site for mark and recapture events. As a result, mean catch per unit effort (CPUE) has become an established metric of relative abundance for terrestrial and aquatic animals, particularly for single-event sampling programmes (Seber, 1982; Maunder, 2001; Maunder and Langley, 2004). This is especially true in freshwater systems, where mean CPUE data are often collected by single-event monitoring programmes that employ lethal sampling. Depending on what other information is collected, this may permit the estimation of population sizes and form the basis of annual harvest quotas for valuable commercial and recreational game fish species (Ontario Ministry of Natural Resources, 1991; Forage Task Group, 2011; Walleye Task Group, 2011). Further, CPUE has been used by regulators to monitor for potential changes in fish populations related to industrial activities (Gahcho Kue Project, 2010), to study effects of invasive species (Hoyle et al., 2008), human exploitation (Post et al., 2008) and anthropogenic substances (Kidd et al., 2007).

A common criticism of CPUE is that it does not provide a quantitative measure of actual fish abundance (Beverton and Holt, 1957), because it used in this way assumes strict proportionality between these two parameters (Harley et al., 2001; Maunder, 2001). However, in some cases, CPUE has been shown to be "density-invariant", whereby decreases in CPUE are not evident until fish density has been dramatically reduced (Post et al., 2002). Non-proportional relationships between CPUE and abundance, including hyperstability—where CPUE remains high while abundance drops—and hyperdepletion—where CPUE declines faster than abundance (Hilborn and Walters, 1992)—present a challenge for understanding the efficacy of CPUE as an indicator of abundance (Hubert et al., 2012). For example, commercial and recreational fishing data, which are inherently spatially biased, are prone to result in a hyperstable relationship between CPUE and abundance (Matsuishi et al., 1993; Harley et al., 2001; Gaertner and Dreyfus-Leon, 2004). Deviation from the assumption of direct proportionality may result in the over- or under-estimation of abundance using CPUE data (Harley et al., 2001; Tsuboi and Endou, 2008) and presents a serious obstacle in estimating population sizes accurately. Presently, passive sampling gear such as minnow traps and trap nets are commonly used by industry and government

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http://dx.doi.org/10.1016/j.fishres.2013.10.019

monitoring programmes to capture forage fish for environmental assessment and in long-term monitoring of aquatic systems, although no studies to date have examined how CPUE of these gear types relate to estimates of abundance.

Boreal freshwater lakes represent the most abundant lake-type globally (Schindler et al., 1996) and are affected by climate change, food web alteration and increased exploitation (Schindler and Lee, 2010). Boreal lakes are home to highly prized commercial, recreational and subsistence fish species such as lake trout (*Salvelinus namaycush*), walleye (*Sander vitreus*) and northern pike (*Esox lucius*); however, the longevity of these piscivorous fish species often makes it difficult to identify the effect of stressors on their populations. Alternatively, the short-lived nature of cyprinid species has made them model organisms for indicating effects of environmental changes on fish populations (Elser et al., 1998; Kidd et al., 2007; State of The Great Lakes, 2009: 82–88; EKATI Diamond Mine, 2012). Fisheries managers should use estimates of productivity when assessing impacts on fish communities (Minns et al., 2011), but because few agencies have the resources to initiate full-scale population studies on littoral fish communities in boreal lakes, they often rely on mean CPUE to infer changes in abundance of littoral cyprinids. Due to issues surrounding hyperstability and hyperdepletion, it has remained unknown whether cyprinid CPUE provides an accurate representation of cyprinid abundance in boreal freshwater lakes.

For a period of 7 years, we conducted an annual mark-recapture study of adult size classes of two littoral minnow species common to Canadian boreal lakes to achieve the following objectives: (1) quantify the relationships between mean CPUE and mark-recapture abundance estimates obtained from minnow traps for cyprinid species with different life-histories; (2) evaluate sampling effort and methodology to better design future monitoring programmes using minnow traps; and (3) examine the relationship between mark-recapture abundance estimates (derived from minnow traps) and mean CPUE data from a second type of trapping method (two types of trap nets) to provide estimates of population size for a long-term monitoring programme.

2. Methods

We conducted annual mark-recapture and trap net sampling of pearl dace (*Margariscus margarita*) and fathead minnows (*Pimephales promelas*) during the fall (September–October) of 1999–2005 at Lakes 260 (L260) and 442 (L442) located within the Experimental Lakes Area (ELA), northwestern Ontario, Canada (Fig. 1). Both lakes are considered oligotrophic and are similar size; L260 is 34 ha in area, with a maximum depth of 14.4 m, while L442 has an area of 16 ha, maximum depth of 17.8 m. The littoral region (≤3 m depth) of L260 is 40% of lake area and for L442 is 16%. The study lakes have similar fish communities, including lake trout as a top predator, white sucker (*Catostomus commersonii*), slimy sculpin (*Cottus cognatus*) and a number of cyprinid species such as northern red-belly dace (*Phoxinus eos*) and finescale dace (*P. neogaeus*) in addition to pearl dace and fathead minnow. From 2001 to 2003, L260 was manipulated by addition of an endocrine disrupting chemical (EDC) that subsequently resulted in a substantial decline of the fathead minnow CPUE and a moderate decline of the pearl dace mean CPUE (for details, see Palace et al., 2006, 2009; Kidd et al., 2007). L442 is a long-term reference lake at the ELA that was not altered for the duration of the study.

2.1. Mark-recapture study

Fall mark-recapture studies were designed for use with the Schnabel estimator and were comprised of 10–11 days (d) trapping periods. During these events, 30 minnow traps baited with a pasta/flour mixture were evenly set around the perimeter of the lake in the littoral zone (<3 m in depth). Minnow traps were standard Gee^{TM} traps made of square galvanized mesh (6.4 mm bar measure), but modified to have a single opening at one end. Minnow traps were collected and re-baited daily. After removal from traps, fish were placed into holding containers with lake water and transported to shore where all fish were sorted by species. The first day of trapping each year consistent of only marking fish, and subsequent days consisted of counting the number of marked and unmarked fish captured, as well as adding new marks to previously unmarked fish. Fish marking involved clipping the tip of the caudal fin using scissors (Palace et al., 2006). All previous clips and mortalities were noted and live fish were then returned to the lake. All fish were released from a single point on the lake. On the last day of mark-recapture trapping, marked fish and unmarked fish were counted and no new marks were added.

We used the Schumacher–Eschmeyer version of the Schnabel estimator to produce abundance estimates from our mark-recapture study (Schnabel, 1938; Schumacher and Eschmeyer, 1943; Ricker, 1975; Schneider, 1998). This method is based on minimizing the weighted sum-of-squares between the proportion of marked fish in a random sample and the total population using the following equation:

$$N_{\text{est}} = \frac{\sum_{d=1}^{n} C_d M_d^2}{\sum_{d=1}^{n} R_d M_d} \tag{1}$$

where:

N_{est} = estimate of population size in numbers of individual fish;
C_d = total number of fish capture on day d;
R_d = the number of recaptures (fish previously marked) caught on day d;
M_d = the number of previously marked fish available for recapture at the start of day d;
n = the total number of sampling days;
d = individual sampling day, ranging from the first (d_1) to last (d_n).

As this method uses a sample to estimate population size, there is uncertainty or error associated with the population estimate, which we calculated using the following equations:

$$s^2 = \frac{\sum_{d=1}^{n} (R_d^2/C_d) - \left[\left(\sum_{d=1}^{n} R_d M_d \right)^2 / \left(\sum_{d=1}^{n} C_d M_d^2 \right) \right]}{m-1} \tag{2}$$

where:

s^2 = sample variance;
m = number of sampling days on which fish were captured.

$$\text{SE}_{N_{\text{est}}} = \sqrt{N_{\text{est}}^2 \times \left[\frac{N_{\text{est}} s^2}{\sum_{d=1}^{n} R_d M_d} \right]} \tag{3}$$

where:

$\text{SE}_{N_{\text{est}}}$ = the standard error of the population estimate (N_{est}).

The Schnabel method assumes that (1) the population is closed (births, deaths and migration do not occur); (2) all fish are equally catchable; and (3) marked fish are not misidentified (Schnabel, 1938; Schumacher and Eschmeyer, 1943; Ricker, 1975). Both study lakes have shallow inflow and outflow streams which often run dry by mid-summer. We assume that migration among these lakes is negligible, and our systems were closed during the study (M. Docker, unpublished data). We believe that our study methods met the second and third assumptions of the Schnabel method. Traps

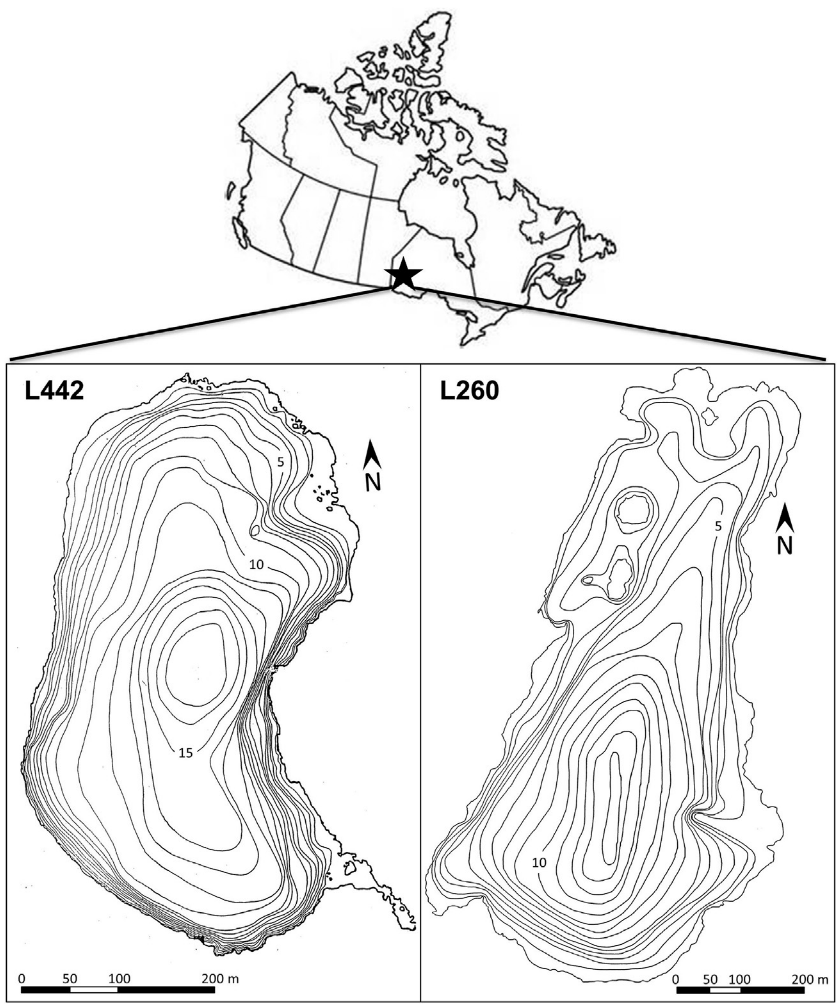

Fig. 1. Location and bathymetry of Lakes 260 and 442 at The Experimental Lakes Area, Ontario, Canada (49°34′–49°47′ N, 93°36′–93°52′ W). Contour lines represent 1 m depth intervals, starting from shore.

were set randomly in littoral areas, to evenly cover the perimeter of lakes. Although all captured fish were released from a single point in each lake over the duration of the study, the small size of our study lakes, along with evidence that fish quickly moved throughout the lake following capture and release (P. Blanchfield and M. Rennie, unpublished data), provides confidence that marked individuals were fully mixed in the population. Further, because of the short duration of the mark-recapture study (10–11 d), we assume births and deaths are negligible. Lastly, marks were clearly distinguished on fish by the presence of a clipped caudal fin. Clips from fish surviving from previous years had exhibited regrowth, and these scars were easily distinguished from fish marked during the current study.

2.2. Trap net sampling

Fall trap net collections occurred annually for 7 years in both L260 and L442 to monitor changes in minnow population size and structure. Each September–October, 2–5 small diameter (2–5 mm bar measure) mesh trap nets were set in each lake for 16–33 consecutive days; nets were emptied approximately every 2–7 d. Trap nets were of two types that either contained a central lead set perpendicular to shore (see Beamish, 1973) or without a central lead, where one wing was extended and tied to shore, set parallel to the shoreline in order to capture fish travelling in a single direction (hereafter called "wing nets"; Mills et al., 1987). Nets of either design had holding pots framed in wood and anchored in place which ranged in volume from 2.2 to 6.1 m^2, and were set in ~3–4 m of water (Beamish, 1973; Mills et al., 1987). The length of leads varied depending on the steepness of the lake bottom and the distance from shore to ~3–4 m of water, as traps were targeting the littoral area (≤3 m depth) of the lakes. The height of the pots and leads were 90 cm. Trap nets were set at the same locations each fall; however, these locations are assumed to be random with respect to cyprinids as traps are passive (un-baited), and sites were selected to target spawning lake trout, not pearl dace or fathead minnows. Following collection, all fish were transported to shore in tubs to be counted, and measured (fork length). When catches were high, only a random subset of fish was measured.

2.3. CPUE calculations

We calculated mean CPUE for pearl dace and fathead minnows for each lake and year individually by dividing the total number of fish caught by the number of traps set and collected on that sampling day. These daily CPUE values were averaged to obtain a grand mean (±SD) for each sampling period (year). For trap nets, mean CPUE values for each species were calculated by dividing total daily catches by "net days" (the number of traps fishing multiplied by the number of days traps were set) to produce a CPUE representing

Please cite this article in press as: Guzzo, M.M., et al., Evaluating the relationship between mean catch per unit effort and abundance for littoral cyprinids in small boreal shield lakes. Fish. Res. (2013), http://dx.doi.org/10.1016/j.fishres.2013.10.019

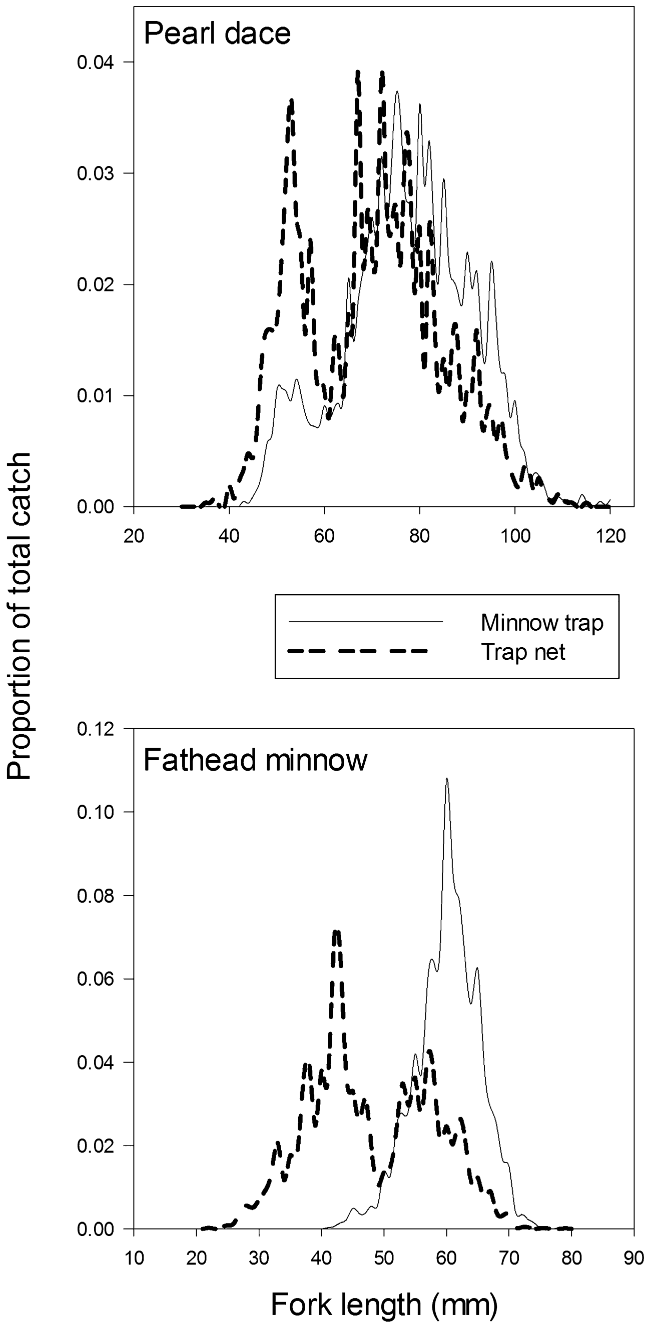

Fig. 2. Fork length frequency distributions for catches of pearl dace and fathead minnows captured by minnow traps and trap nets each spring for 1999–2005 from L442 and L260 combined.

number of fish caught per trap per day. Daily CPUE values were used to produce a grand mean (±SD) for each sampling period (year) (Mills et al., 2000; Palace et al., 2006).

2.4. Comparing selectivity of minnow traps and trap nets

Trap nets employed in this study had a smaller mesh diameter (2–5 mm) compared to minnow traps (6.4 mm), and therefore were able to catch smaller fish, many of which were YOY minnows (Fig. 2). To investigate the influence of differences in size-selectivity between gears on the relationship between trap net CPUE and mark-recapture abundance estimates from minnow traps, we also investigated the application of a minimum cut-off size—determined by fork length—applied to the lower end of the size-frequency distributions of trap nets catches. To compare the fishing abilities of the two capture methods we calculated minimum size cut-offs for each species using spring monitoring data on our study lakes. Although these data are not used in this study, they provided a means to compare the size-selectivity of minnow traps and trap nets in our study lakes, and for our study species. We estimated the minimum size of each fish species that could be captured by minnow traps by calculating the fork length that corresponded to the lower 2.5th quartile of all combined spring minnow trap catches (1999–2006) for L442 and L260. We then used the size distribution data from each haul of the trap nets to estimate the proportion of fish below the cut-off size, and re-calculated CPUE accordingly so that it excluded fish below the cut-off size. This allowed for direct comparisons of trap net CPUE and mark-recapture abundance (from minnow traps).

2.5. Data analyses

We computed an annual grand mean CPUE (±SD) from daily values independently for each species and lake from minnow trap and trap net catch data. Prior to regression analysis, we standardized variance among our variables by taking the $\log_{10}$ of each CPUE and abundance value plus one. We then used a 2nd order power model (Eq. (4)) to relate these log-transformed yearly mean CPUE values to our log-transformed mark-recapture abundance estimates (Paloheimo and Dickie, 1964). Because the goal of our study was to develop a model to convert CPUE to abundance, we treated CPUE as our independent variable and abundance as our dependent variable. While many studies are interested in how CPUE as a dependent variable varies with abundance, our approach meets the goal of our study without having to resort to inverse prediction models.

$$N_{est} = a\mathrm{CPUE}^{b} \quad (4)$$

We used the *R* statistical computing package (Version 2.14.1; R Development Core Team, 2011) to determine size-selectivity for minnow traps and for non-linear regression using the *nls* function in the *R* base package. Figures were produced using the programme SigmaPlot (Version 12.0; Systat Software Inc., Chicago, IL, USA).

3. Results

3.1. Relating minnow trap CPUE to abundance estimates

Throughout the study, the annual number of individual pearl dace marked during mark-recapture studies averaged ~3000 and ~4600 in L442 and L260, respectively (Table 1). The number of fathead minnows marked each year in L442 and L260 both averaged ~400, roughly 10-fold lower than for pearl dace. Low catches of fathead minnows in L260 in later years of the study are attributed to the collapse of this population from experimental manipulation (see Kidd et al., 2007).

We observed that the proportion of fish recaptured typically increased as the mark-recapture study progressed. The proportion of daily recaptures were high for pearl dace, which were 18.8% (±SE: 2.97%) and 23.2% (±4.54%) by the second day of the marking period and plateaued at approximately ~65% in both lakes (Fig. 3). For fathead minnows, the number of daily recaptures during the sampling period were lower than those of pearl dace, starting at 1.2% (±0.94%) and 0.2% (±0.15%) on the second day of mark-recapture sampling and plateauing after 7 d at approximately 6% and 15% in L260 and L442, respectively. The proportion of daily recaptures was much less variable for pearl dace relative to fathead minnow (Fig. 3).

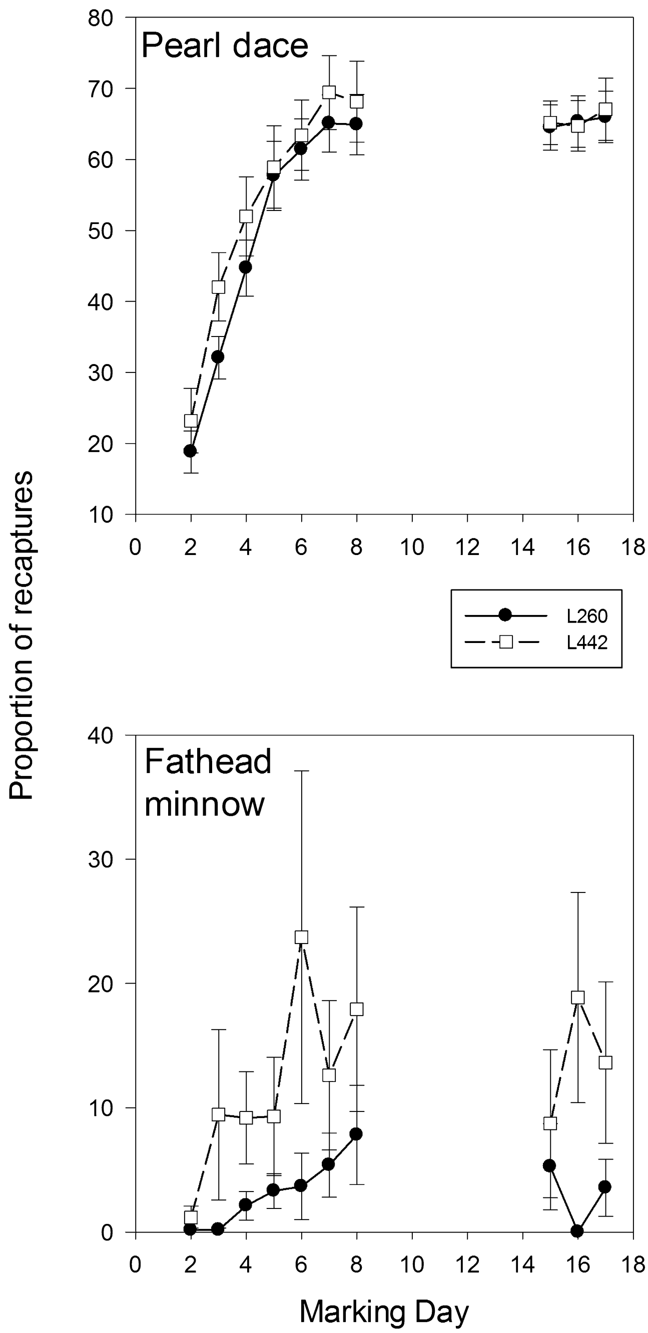

Fig. 3. Mean proportion of recaptured pearl dace and fathead minnow during fall marking periods in L260 and L442 for 1999–2005.

Abundance estimates of adult pearl dace and fathead minnows in L442 exhibited natural variation, where estimates were highest at the beginning and end of the study and lowest during the middle (Table 1). In L442, pearl dace abundances (±SE) ranged from 2023 ± 129.2 to 6680 ± 331.8, individuals, while fathead minnow populations ranged from 449 ± 1220.0 to 4525 ± 1753.3 (Table 1). In L260, pearl dace and fathead minnow populations steadily declined over the course of the study, although, this decline was more substantial in fathead minnows (Table 1).

Our regression analysis produced significant positive correlations between mean minnow trap CPUE and mark-recapture abundance estimates for both pearl dace and fathead minnows (Fig. 4a and Table 2).

3.2. Trap net CPUE versus abundance estimates

The smallest fish size captured by minnow traps was 41 mm fork length. In contrast, trap nets captured fish as small as 19 mm fork length. This resulted in a bi-modal distribution of fish lengths for spring trap net catches that included the abundant YOY size class not captured by minnow traps (Fig. 2). The estimated minimum fork length cut-offs were similar for pearl dace (51 mm) and fathead minnows (50 mm). The proportion of trap net catch data for pearl dace removed by minimum size cut-offs ranged from 0% to 53%, and from 0% to 40%, in L442 and L260, respectively. The proportion of fathead minnow catch removed by the minimum size cut-offs ranged from 23% to 85%, and from 0% to 93%, in L442 and L260, respectively.

Our regression analysis produced significant positive correlations between mean trap net CPUE and mark-recapture abundance estimates for fathead minnows, but not for pearl dace (Fig. 4b and Table 2).

4. Discussion

In this study, we found significant non-linear relationships between minnow trap CPUE and mark-recapture abundance estimates for two common littoral cyprinid species in boreal shield lakes. Assuming estimates from our mark-recapture methods accurately reflect population size of cyprinids larger than YOY, these relationships could be used to generate abundance estimates from mean CPUE values obtained using similar sampling methods in lakes with similar characteristics. The ability to derive abundance estimates from CPUE data opens up a window of opportunity for fisheries researchers to convert relative measures of abundances to actual numbers of individual fish, allowing for estimates of biomass. Our findings suggest that mean CPUE data, commonly collected by government agencies, consulting companies, and fisheries researchers to evaluate impacts of industrial activities, can accurately reflect population trends for minnow species in boreal lakes. Our results also provide evidence that mean cyprinid CPUE from small-mesh trap nets can be representative of abundance for fathead minnows, and this should allow for analysis of long-term data collected using other methods. The ability to convert cyprinid CPUE obtained from various trapping methods to abundance will allow researchers to test and evaluate theoretical concepts and ecosystem models describing the effects of stressors such as harvest (Abrams et al., 2012), predation (Abrams and Fung, 2010), mortality (Abrams, 2009), and contaminants (Gledhill and Van Kirk, 2011) on fish population dynamics.

Our study may provide a measure of confidence in the ability of mean CPUE, obtained using similar methods, to be used as an indicator of population abundance of littoral fish communities in similar boreal shield lakes. Our confirmation of CPUE as an effective measure of abundance is analogous to results of Haggarty and King (2006), who found significant correlations between the CPUE from standardized research angling and density estimates obtained from visual observation for near shore marine reef fish, and Zimmerman and Palo (2011), who identified significant correlations between CPUE and mark-recapture estimates for noble crayfish (*Astacus astacus*). In contrast to the strict proportionality between CPUE and abundance found in these studies, our results indicated the relationship between mean CPUE and abundance to be non-linear, suggesting our sampling methods were subject to hyperdepletion—where CPUE declines faster than abundance. Hyperdepletion is a function of changes in catchability, which is the proportionality between abundance and an index of abundance, such as CPUE (Arreguin-Sanchez, 1996). Catchability has been shown to be time-variant in both fishery and non-fishery data

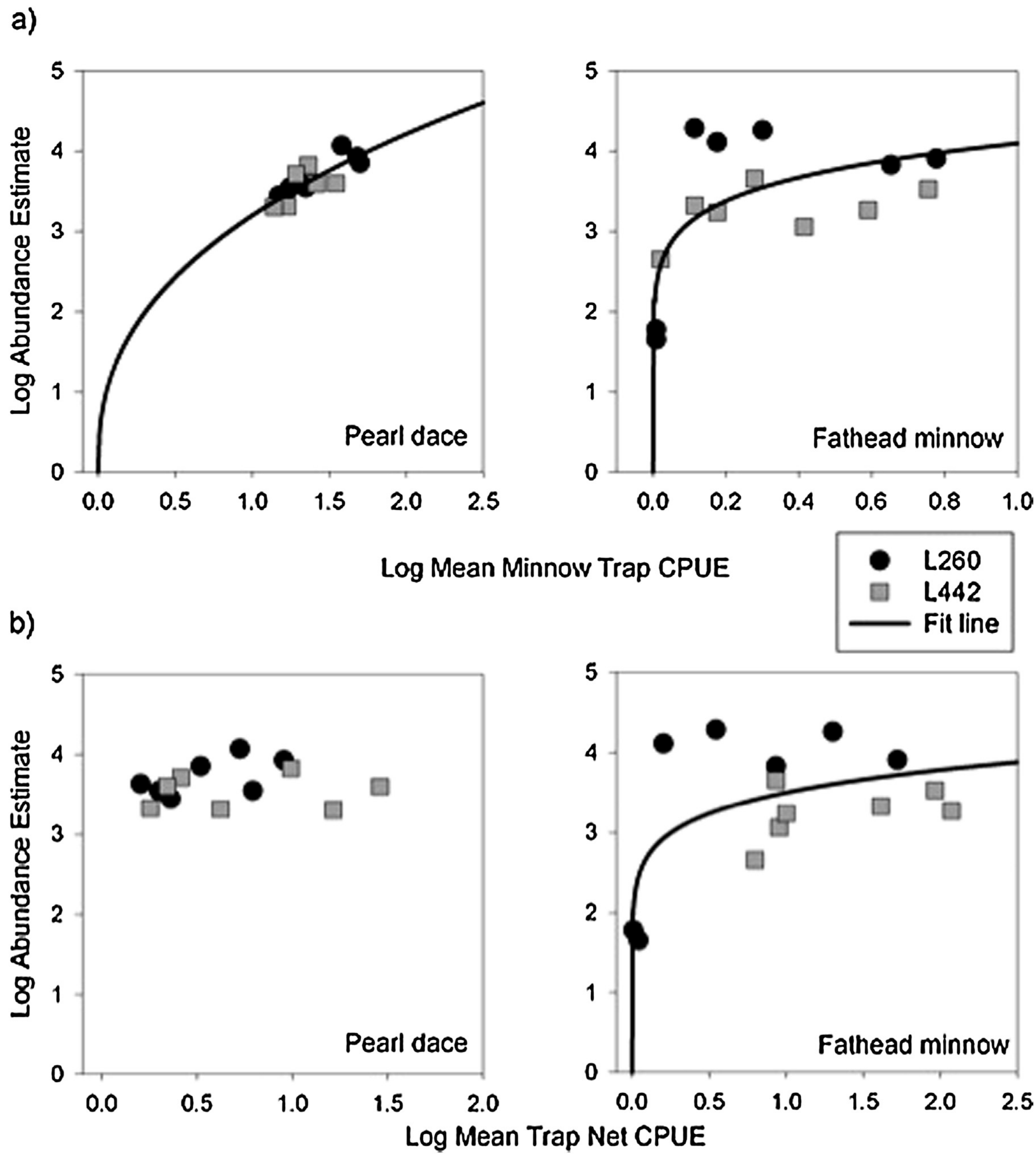

Fig. 4. Power model regressions between log-transformed abundance estimates and mean (a) minnow trap and (b) trap net CPUE for pearl dace and fathead minnows from 1999 to 2005 (data from L442 to L260 combined). Note: for trap net—abundance regressions, trap net CPUE values have been recalculated to include size-appropriate fish only. Where regression lines are not present, no significant fit between CPUE and *N* was determined.

and can be affected by numerous factors, such as density, environment, and fish behaviour (Wilberg et al., 2010). In our study, changes in the density of cyprinids over time, particularly in manipulated L260, could have resulted in density-induced variations in habitat use. These variations in habitat use could then have altered the amount of overlap with our trapping areas, thus affecting catchability (Walters, 2003). Furthermore, variations in weather and in water chemical concentrations (e.g. addition of EDC to L260) in study lakes between sampling years could also have altered the behaviour, and thus the dispersion of fish affecting catchability over the duration of the study (Wilberg et al., 2010).

We are confident that our estimates of population size based on fall mark-recapture studies are representative and unbiased. Density estimates based on our calculated abundances (mean of 7 years for L442 and pre-manipulation data (1999–2000) for L260), equated to littoral-region cyprinid density estimates of 2292 fish ha^{-1} (pearl dace: 1445 fish ha^{-1}; fathead minnow: 847 fish ha^{-1}) for L442, and 1710 fish ha^{-1} (pearl dace: 743 fish ha^{-1}; fathead minnow: 967 fish ha^{-1}) for L260. Our estimates were comparable to those of Eddy (2000) and Totsche (unpublished), who estimated the density of finescale dace in smaller, shallower ELA L115 and L632 to be 4191 (±95 CI: 382) and 3311 (±260) fish ha^{-1}.

Our results showed species-specific differences in catchability. During mark-recapture studies pearl dace exhibited a high proportion of recaptures consistently after the first 7 d of trapping.

Table 1
Summary data for fall mark-recapture studies, including the number of marked fish, the number of recaptured fish, and abundance estimates (±SE) for each year, and mean (±SD) CPUE from minnow traps and trap nets from 1999 to 2005 in L442 and L260.

Species	Year	*M*		*R*		N_{est}		Mean MT CPUE		Mean TN CPUE	
		L442	L260	L442	L260	L442	L260	L442	L260	L442	L260
Pearl dace	1999	4328	7276	2738	4406	6680 (331.8)	11,738 (368.6)	22.2 (5.7)	37.1 (11.0)	8.8 (4.3)	4.3 (2.5)
	2000	3490	7121	2364	8113	5133 (213.9)	8451 (225.9)	18.3 (4.7)	47 (10.3)	1.6 (0.9)	8 (6.3)
	2001	3860	6430	7184	9524	3959 (201.3)	7129 (210.5)	34 (3.7)	49.1 (11.0)	1.2 (1.4)	2.3 (1.8)
	2002	1961	3212	2938	2835	2108 (168.9)	4273 (224.9)	15.1 (2.9)	18.9 (4.2)	0.8 (0.5)	0.6 (0.7)
	2003	1871	2942	2805	3297	2055 (151.6)	3511 (217.5)	16 (4.3)	21.3 (3.8)	3.2 (3.8)	5.2 (5.3)
	2004	1805	2269	2385	2224	2023 (129.2)	2783 (175.6)	12.9 (4.5)	13.9 (3.1)	15.4 (25.2)	1.3 (1.1)
	2005	3490	2729	4661	2608	3945 (228.2)	3501 (202.3)	25.2 (7.8)	16.6 (2.6)	27.8 (23.2)	1 (1.7)
Fathead minnow	1999	1185	1373	282	145	3374 (135.0)	8090 (1022.8)	4.7 (2.3)	5 (1.9)	91.5 (25.6)	51.7 (33.6)
	2000	262	318	8	3	4525 (1753.3)	18,220 (7954.4)	0.9 (0.6)	1 (1.0)	7.6 (7.2)	19.1 (21.9)
	2001	414	985	100	81	1139 (249.1)	6748 (868.0)	1.6 (0.6)	3.5 (1.8)	8 (6.3)	7.6 (3.2)
	2002	15	140	1	1	449 (1220.0)	12,977 (8383.8)	0.05 (0.03)	0.5 (0.4)	5.3 (2.9)	0.6 (0.4)
	2003	143	77	5	2	1719 (732.7)	19,284 (13,209.9)	0.5 (0.5)	0.3 (0.2)	9 (5.1)	2.5 (2.5)
	2004	78	6	1	1	2108 (189.6)	44 (51.4)	0.3 (0.2)	0.02 (0.03)	40.2 (44.4)	0.1 (0.1)
	2005	756	6	191	1	1859 (140.2)	59 (99.6)	2.9 (1.5)	0.02 (0.03)	117.3 (14.4)	0.02 (0.02)

M = total number of marked fish for an individual year, *R* = total number of recaptured fish in an individual year, N_{est} = abundance estimate for each year, mean MT CPUE = mean minnow trap CPUE for an individual year, mean TN CPUE = mean minnow trap CPUE for an individual year, L442 = Lake 442, L260 = Lake 260.

This phenomenon was not evident in fathead minnows, which had highly variable proportion of recaptures and showed no increases in daily CPUE during fall trapping periods. This could suggest that we had marked a larger proportion of the pearl dace population, relative to that of fathead minnows. These differences in catchability of pearl dace relative to fathead minnows could be due to behavioural differences between species. Pearl dace primarily forage on aquatic insect larvae while fathead minnows are known to feed on detritus and plant matter (Scott and Crossman, 1973; Stewart and Watkinson, 2004). Active foraging on live prey by pearl dace would increase their likelihood of encountering passive sampling gear compared to fathead minnows, which are considered sedentary feeders. Lower mobility of fathead minnows could also provide some explanation for the large variation in fathead minnow catches experienced throughout the study. Alternatively, differences in habitat use may explain asymmetry in catches between our study species, where traps may have been set in habitats that are more frequently visited by pearl dace than by fathead minnows.

Within species variations in catch rate observed for pearl dace, may have been the result of differences in personality, where bolder individuals are more likely to be trapped than shy individuals (Biro and Dingemanse, 2008). Personality effects of catchability have been found in pumpkinseed sunfish (*Lepomis gibbosus*), where individuals captured in unbaited passive traps were bolder than those captured by active trapping methods (Wilson et al., 1993). In rainbow trout (*Oncorhynchus mykiss*), active-bold genotype fish have been shown to occupy more risky habitat and be more susceptible to gill net capture, relative to inactive-shy genotype fish (Biro et al., 2006; Biro and Post, 2008). Further, differences in body size, or reaction to fish already captured could explain differences in pearl dace and fathead minnows. Tsuboi and Endou (2008) suggested that differences in catchability of white-spotted char (*Salvelinus leucomaenis*) and red-spotted masu salmon (*O. masou ishikawae*) using a standardized fishing species-specific could be related to differences in behaviour, such as body-size dominance hierarchies, and reactions to the capture of nearby fish. While we acknowledge individual variation in capture probabilities may have existed during our study, the batch-mark method we employed did not allow for these behaviours to be quantified.

Our results, specifically the error values around our abundance estimates, indicated that our sampling design was sufficient for tracking abundance of pearl dace, but that more effort may be required for more precise estimation of fathead minnows. Catchability is typically a function of fish abundance, and as abundance varies over time, so will catchability (Maunder et al., 2006). For example, if abundance of a fish species were to decrease, the catchability or chance of that species encountering a trap would also be expected to decrease. In mark-recapture studies, this reduction in sample size would increase the uncertainty around abundance and related CPUE estimates. The abundance and mean CPUE of fathead minnows were generally lower and much more variable than pearl dace during our study. The lower abundance of fathead minnows, relative to pearl dace, combined with their short-lived nature (Scott and Crossman, 1973), support the idea that increased effort may be needed to study this species. Because both CPUE and abundance estimates were calculated using the same number of traps, this should not have affected the generated relationship between these two parameters for this species; however, the greater variation among fathead minnow catches, provides some explanation for the larger confidence intervals around our generated abundance estimates for this species. Thus, increased effort may be needed to obtain a more precise abundance estimates for fathead minnows relative to pearl dace.

Table 2
Results of power model ($N_{est} = a\text{CPUE}^b$) relating log transformed mean minnow trap and trap net CPUE to log transformed mark-recapture abundance estimates from derived from wire minnow traps.

CPUE source	Species	*df*	RSE	*a*			*b*		
				Estimate	SE	*P*	Estimate	SE	*P*
Minnow trap	Pearl dace	12	0.14	3.20	0.09	<0.001	0.40	0.08	<0.001
	Fathead minnow	12	0.59	4.10	0.29	<0.001	0.12	0.04	0.008
Trap net	Pearl dace	12	0.24	3.67	0.09	<0.001	0.02	0.03	0.49
	Fathead minnow	12	0.65	3.50	0.18	<0.001	0.11	0.05	0.03

$P < 0.05$ was used as our critical significance value. *P* = denotes the probability that a parameter estimate is significantly different than zero, SE = standard error of a parameter estimate, *df* = degrees of freedom, RSE = residual squared error.

Please cite this article in press as: Guzzo, M.M., et al., Evaluating the relationship between mean catch per unit effort and abundance for littoral cyprinids in small boreal shield lakes. Fish. Res. (2013), http://dx.doi.org/10.1016/j.fishres.2013.10.019

In addition to the effects of behaviour and population size, the size of lakes, and more specifically, the relative size of their littoral zones could affect whether sampling effort is adequate. Because we used the same number of traps in each lake, our sampling effort covered a greater proportion of total the total littoral area of L442, relative to L260. Based on lake morphology, the littoral area of L442 and L260 are 2.56 ha and 13.6 ha, respectively. If 30 traps were sufficient for trapping pearl dace in both of these systems, this equates to 11.7 and 2.21 traps ha^{-1} of littoral area in L442 and L260, respectively. However, reduced density estimates for pearl dace in L260 relative to L442 suggests that using 30 traps may not have sufficiently covered the larger littoral area of L260. Thus, to achieve a density of 11.7 traps ha^{-1}, as used in L442, 160 traps would need to be distributed around the littoral zone of L260.

We produced significant, non-linear relationships between mean trap net CPUE and mark-recapture abundance from minnow traps for fathead minnows. Trap nets like those used in this study have been used to track CPUE in lakes at the Experimental Lakes Area in Northwestern Ontario for decades. The ability to convert CPUE generated from these methods to estimates of abundance are critical for developing mechanistic ecosystem models that can be used to better understand ecosystem function in these freshwater boreal lakes. Because a large proportion of YOY fish captured in trap nets were not represented in wire minnow trap catches and in turn the abundance estimates, we used a minimum size cut-off to truncate the size distribution of trap net catches and alter daily CPUE values. We suggest that cut-offs combining data from lakes and years provide the best method for comparing how different trapping methods capture fish and are necessary to generate accurate relationships between abundance measures and CPUE data.

Acknowledgments

We gratefully appreciate the assistance with fish collections by Sandy Chalanchuk, Ken Mills, Lori Tate and many ELA staff and students. Lee Hrenchuk is thanked for her help in preparing the map. This research was supported by Fisheries and Oceans Canada (DFO), the American Chemistry Council Long Range Initiative Program and the Canadian Federal Toxic Substances Research Initiative. M. Guzzo was funded by The University of Toronto, The University of Manitoba Graduate Fellowship, and The H.E. Welch Award. We thank two anonymous reviewers whose comments improved the manuscript.

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