

**FISH COMMUNITY SIZE SPECTRA AND THE ROLE OF VESSEL
AVOIDANCE IN HYDROACOUSTIC SURVEYS OF BOREAL LAKES AND
RESERVOIRS**

By

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ABSTRACT

Hydroacoustic data were used to quantify vessel avoidance by fishes, and derive fish community size spectra in two shallow boreal systems in eastern Manitoba, Canada. Lac du Bonnet reservoir and adjoining lakes at Nopiming Provincial Park were studied during summer 2011 and 2012. The magnitude of boat avoidance varied between these relatively similar water bodies ($p = 0.04$), but was not significantly influenced by fish depth or survey speed. Length-frequency spectra were determined from acoustic surveys at Lac du Bonnet, and acoustic data were used to map bathymetry of the reservoir. Community abundance (spectra height) was greater in 2011 than 2012 ($p < 0.05$), and decreased through the summer. Spatial variation in spectra parameters appear to be related to physical habitat characteristics. I conclude that vessel avoidance should be quantified *in situ*, and that acoustic size spectra may be used to monitor differences in fish communities over time and among habitats.

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INTRODUCTION AND OVERVIEW

The development of hydroelectric generating capacity is increasing globally as the need to move toward renewable sources of power generation becomes more widely recognized (Frey and Linke 2002; World Bank 2013). In Canada alone, there are more than 470 hydroelectric facilities operating to generate more than 60% of the country's electricity (CHA 2008). With rising concern over greenhouse gas emissions from the burning of fossil fuels, hydropower will continue to play an important role in Canada's energy market. While hydropower provides an efficient renewable energy source, the construction and operation of hydroelectric generation facilities impacts physical, chemical and biological process in the aquatic environments in which they operate (Rosenberg et al. 1997).

When a river is dammed to form a reservoir the landscape is fundamentally altered; this conversion from a lotic to a more lentic system affects physical habitat conditions within the water body (Baxter 1977). Changes in the flow regime of a natural river or lake that may occur with the construction of a hydro generating facility include a dampening of seasonal fluctuations in flow, an increase in the flooded area and water depth upstream of the dam coupled with decreased water depth and flooded area downstream, and changes in bottom substrate composition and bed mobility (Baxter 1977; Cobb et al. 1992; Kondolf 1997; Bunn and Arthington 2002). These physical changes impact the biological community in the regulated river or newly created reservoir. Changes in water level, flow, and substrate can alter the composition, abundance, and distribution of vegetation (Wilcox and Meeker 1991; Hill et al. 1998;

Nilsson and Berggren 2000) and benthic invertebrates (Voshell and Simmons 1984; Cobb et al. 1992; Hart and Finelli 1999; Dewson et al. 2007). In addition, changes in nutrient levels and water residence time brought by flow alteration can result in changes to the abundance and diversity of phyto- and zooplankton (Basu and Pick 1996; Paterson et al. 1997; Thouvenot et al. 2000).

Fish communities are a fundamental component of aquatic ecosystems, and are valued both economically and recreationally through small and large scale fisheries. It is important that changes in fish communities due to anthropogenic activities, such as hydropower generation, be monitored and assessed. The combined impacts of alterations in biotic and abiotic conditions when a river is regulated or a reservoir is formed can have significant impacts on fish communities in these aquatic systems. Flooding of riverbanks with the creation of a reservoir may lead to a loss of preferred spawning habitat, negatively impacting fish populations (Hubbs and Pigg 1976; Paragamian et al. 2001). In addition, changes in planktonic community structure and abundance may alter the diet composition and feeding success of planktivorous fishes (McQueen et al. 1986; Johnston and Mathias 1994). The construction of dams also creates a barrier to fish movement that can result in localized extinctions of migratory fish species (Beamish and Northcote 1989; Gehrke et al. 2002; Schilt 2007). These differences in biotic and abiotic conditions of a reservoir, relative to that of a natural lake, also make reservoirs more susceptible to invasions by non-native species (Havel et al. 2005). At present, hydro companies utilizing Canadian aquatic ecosystems lack a satisfactory method of assessing changes in fish

abundance, distribution, and habitat use in the ecosystems their operations impact (Smokorowski et al. 2011).

Aquatic ecosystems, such as those in which hydropower operations occur, are largely considered to be shaped by size-based processes (Boudreau et al. 1991). An organism's body size is one of its most fundamental traits, and has been linked to numerous life history traits such as reproduction (Fenchel 1974), productivity (Sheldon et al. 1972) and metabolic rate (Gillooly et al. 2001). Particularly for fishes, body size is also directly related to feeding, as predation is largely limited by an individual's gape size, resulting in predators consistently consuming prey items smaller than their own size; body size therefore plays a significant role in structuring trophic interactions (Elton 1927). Body size has also been shown to impact organisms at a population scale, with population densities generally lower for organisms with larger individual body size (Brown et al. 2004). The use of ecological indicators that consider organism body size are especially useful in aquatic environments, as the structure and function of these systems is driven largely by size-based processes (Boudreau et al. 1991).

Fish community size spectra have been widely used to monitor the impacts of commercial fishing on the size-structure and abundance of fish communities (e.g. Rice and Gislason 1996; Kantoussan et al. 2009; Sweeting et al. 2009). Size spectra are relationships of the relative abundance of organisms by size, and can be used to evaluate the status of aquatic ecosystems (Emmrich et al. 2011), the effectiveness of fisheries management (Dickie et al. 1987), and track long term changes in fish community size and structure (Rice and Gislason 1996; Daan et al. 2005). In addition, size spectra have been

used to evaluate the effectiveness of a marine protected area, and to assess differing levels of fish productivity between habitats (Macpherson et al. 2002). Historically, size spectra have been based on catch data obtained through netting surveys and fisher reports; the effectiveness of hydroacoustic methods for size spectra analysis is not yet known, but application of this method may prove advantageous.

Hydroacoustic methods allow for non-invasive, efficient surveys of fish communities in both freshwater and marine environments, providing information on fish distribution, abundance, and individual body-size (Simmonds and MacLennan 2005). Other fish sampling methods, such as gillnets, trawling, and electrofishing, are more labour intensive, and are more likely to result in fish mortality (Argent and Kimmel 2005). For these reasons, acoustic methods provide a time-efficient, cost-effective, and environmentally friendly alternative to other more widely used methods of fish assessment. In addition, acoustic surveys have been shown to provide a more representative sample of the length distribution of fish communities, when compared with netting surveys (Coll et al. 2007), as the effectiveness of the gear is not largely biased by fish size. Hydroacoustic data also provides information on physical habitat (i.e. water depth, bathymetry, substrate, vegetation) and plankton abundance, allowing for the direct comparison between fish and other environmental characteristics (Godlewski et al. 2004). The ability of hydroacoustics to efficiently quantify fish abundance, while providing reliable estimations of individual fish lengths (Love 1971; Simmonds and MacLennan 2005), suggests that acoustic methods are well suited for size-based studies of fish communities. Acoustics may be a useful tool for application with size spectra analysis to

assess relative importance of different habitats, and tracking long term changes in fish community abundance and structure. Despite this potential, to the best of the author's knowledge, there have been no studies published in the primary literature that have applied size spectra theory to acoustic survey data.

While hydroacoustics is useful for the collection of high resolution data on fish community abundance and structure, the survey method should not initiate avoidance responses by fishes for the data to be reliable. Noise transmitted from survey vessels, as well as turbulence in the water column from hull and propeller movement may trigger an anti-predator response by fishes, causing them to flee from an approaching vessel (Mitson 1995; Frid and Dill 2002). The presence of avoidance reactions by fishes may lead to bias and imprecision in acoustic surveys of fish communities (Simmonds and MacLennan 2005; DeRobertis and Handegard 2013). While the impact of large survey vessels on fishes in marine environments, particularly for schooling fish, has been well documented (Soria et al. 1996; Mitson and Knudsen 2003; Skaret et al. 2005; Hjellvik et al. 2008), responses of fishes in freshwater environments have not received the same attention, with few studies focusing on these reactions (e.g. Draštík and Kubečka 2005; Godlewska et al. 2009). Compared to the marine environment, freshwater acoustic surveys present different challenges with regards to vessel avoidance: study sites are typically shallower, and fish may perceive a greater potential threat from the survey vessel due to increased proximity, leading to a greater likelihood of avoidance (Ydenberg and Dill 1986). However, survey vessels in freshwater are generally significantly smaller than those used in marine research, which should result in comparatively less noise and turbulence introduced into the water column. Decreased noise levels associated with smaller survey

vessels may be expected to result in a lower magnitude of avoidance. The potential for expansion of hydroacoustic survey methods in freshwater ecosystems makes developing a simple method to quantify avoidance a priority for further investigation.

In this thesis, I contribute to furthering the development of hydroacoustic methods for assessment and monitoring of fish communities in lakes and reservoirs. In chapter 1, I investigated the impact of a small survey vessel on the acoustic surveys in shallow lakes and reservoirs. I develop a method to efficiently quantify vessel avoidance *in situ* through comparisons of fish densities, sizes, and depth distributions among paired experimental transects. Results are presented from two boreal systems in Manitoba, Canada, and highlight differences in avoidance that may occur among similar systems. In chapter 2, I determined if fish community size spectra could be effectively derived from hydroacoustic survey data for use in fish assessments of lakes and reservoirs. Fish abundances and individual lengths are estimated from acoustic data, and applied to size spectral methods typically used with catch data. Size spectra characteristics are found to reflect differences in fish community structure among areas of varying physical habitat characteristics. The importance of spatial and temporal variation in spectra parameters for applications of acoustic size spectra for long term monitoring in aquatic ecosystems are also discussed.

CO-AUTHORSHIP STATEMENT

All manuscripts in this thesis were co-authored with George Rose. I am the principle author for both chapters presented in this thesis, and was the principle contributor to project design, field research, data analysis and manuscript preparation.

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1. QUANTIFYING FISH AVOIDANCE OF SMALL ACOUSTIC SURVEY VESSELS IN BOREAL LAKES AND RESERVOIRS

1.1 Abstract

Mobile hydroacoustic surveys are increasingly used to assess the distribution and abundance of freshwater fish; yet fish may avoid moving vessels, potentially introducing bias in these assessments. In this paper, avoidance in boreal lakes and reservoirs was quantified by developing a simple method based on paired drift:drift (D:D) and drift:motor (D:M) transects. Two systems in eastern Manitoba, Canada were studied: Lac du Bonnet reservoir and Nopiming. Acoustic data were collected using a digital DTX echosounder (BioSonics, Inc, Seattle, USA), with a downward facing 200 kHz split-beam transducer, deployed from 5.5 m vessels (Boston Whalers) modified for acoustic research. An avoidance coefficient (A_c) was developed based on comparisons of acoustic fish densities while the vessel moved over the same transects by drifting and by motoring at survey speeds. A_c did not differ significantly from 1 (no avoidance) at Nopiming (median of 0.81, $n = 13$), but did at Lac du Bonnet (median of 0.51, $n = 31$, $p < 0.05$). Variability in A_c was as high in transect pairs, and was unrelated to fish depth (mean 6.9 m at Lac du Bonnet; 13.1 m at Nopiming) or survey speed (up to $3.70 \text{ m}\cdot\text{s}^{-1}$, 7 knots). Results indicated that fishes did not dive in the presence of the motoring vessel, nor was avoidance size-based. We did not detect any evidence of fish attraction to our drifting vessel. Our results suggest that boat avoidance during acoustic surveys of shallow boreal lakes may vary in relatively similar water bodies, but can be quantified experimentally.

1.2 Introduction

Boat avoidance can introduce uncertainty in mobile surveys of fish abundance. In hydroacoustic surveys, avoidance may lead to bias (and imprecision) in estimates of fish communities (Simmonds and MacLennan 2005; De Robertis and Handegard 2013). In theory, such uncertainties may be accounted for by quantification of avoidance; however, in practice this is not often done, in part, because no standard method of quantification is available.

Avoidance behaviour by fishes is likely a general anti-predation response, with fishes reacting to a perceived threat triggered by the approaching vessel (Fréon et al. 1993b; Frid and Dill 2002; Vabø et al. 2002). Noise transmitted from the vessel is likely responsible for most boat avoidance (Mitson 1995; De Robertis et al. 2010), but visual disturbance and turbulence in the water column may also lead to avoidance (De Robertis and Handegard 2013). Vessel noise is largely attributed to the rotations of the motor and propeller, noise transmitted from activities onboard, and interaction of the hull with the water.

Typical avoidance reactions include lateral movements away from the path of an approaching vessel (Soria et al. 1996), and vertical avoidance, wherein fish may dive deeper in the water column (Gerlotto and Fréon 1992; Jørgensen et al. 2004). Lateral avoidance may cause fewer fish to be ensonified, with or without target strength (TS) changes, whereas diving responses may alter both the depth distribution of fishes and the angle of fish relative to the acoustic beam, potentially impacting both TS measurements and resultant integrated backscatter (Henderson et al. 2007). Schooling fish may also

change their distributions when approached by a vessel, altering depth distributions, and schools may become laterally or vertically compressed (Guillard et al. 2010). These behavioural changes could result in error in estimates of fish abundance, biomass, and TS (Vabø et al. 2002; Ona et al. 2007).

Avoidance reactions are difficult to generalize. Previously reported responses are highly variable (Fréon et al. 1993a; De Robertis and Handegard 2013), differing among species (Misund 1997), size classes (Draštík and Kubečka 2005; Godlewska et al. 2009), distance from the vessel, and depth of fishes (Vabø et al. 2002; Guillard et al. 2010). Survey vessel properties such as the size of the vessel, noise characteristics, engine power, and vessel speed are all likely to impact the magnitude of observed avoidance (Mitson 1995; Mitson and Knudsen 2003).

The majority of published literature on the response of fishes to survey vessels has focused on the reactions of schooling fish in marine environments, and have shown that schools display a range of lateral and vertical movement away from an approaching vessel (Soria et al. 1996; Mitson and Knudsen 2003; Skaret et al. 2005; Hjellvik et al. 2008). Fish have also been shown to be attracted to stationary or drifting research vessels in the marine realm (Castro et al. 2002; Røstad et al. 2006), with fish abundance increasing within a drift. However, no similar studies have been reported from freshwater environments. Boat avoidance responses by non-schooling fish, especially those in freshwater environments, have not received the same attention, with only a few studies focusing on these reactions. Compared to its marine counterpart, hydroacoustic research in freshwater introduces different challenges with regards to the potential for avoidance. Freshwater sites are typically shallower than marine study areas; therefore the threat

perceived by closer ranging fishes may be heightened, resulting in greater likelihood of avoidance responses (Ydenberg and Dill 1986). However, survey vessels used in freshwater are typically smaller and less powerful than marine research vessels, which may result in comparatively less disturbance (i.e. transmitted noise and turbulence) of the water column, which may lessen avoidance.

Previous studies in lakes and reservoirs have suggested that boat avoidance reactions to small vessels may be greater with smaller fish (Drašník and Kubečka 2005; Godlewska et al. 2009) and may only be a significant issue at very short ranges (<10m) (Blanchfield et al. 2005; Drašník and Kubečka 2005). In addition, in eutrophic waters visual cues do not appear to contribute to avoidance, with motor noise being the main contributing factor (Godlewska et al. 2009). The potential for expansion of hydroacoustic survey methods in shallow freshwater ecosystems makes further study of fish reactions in these environments, and the development of a simple method to quantify avoidance a priority for further investigation.

In this study, our objective was to develop a simple method to quantify avoidance of survey vessels in shallow (< 26 m) boreal Canadian lakes. Avoidance was hypothesized to be influenced by survey speed and fish depth, and differ with fish size. We also tested for changes in depth distribution of fishes with passage of the survey vessel.

1.3. Methods

1.3.1 Study Sites

This study was conducted in July and August of 2011 and 2012 in eastern Manitoba, Canada (Figure 1.1). Study sites were located within a chain of adjoining lakes in Nopiming Provincial Park (Manigotagan Lake and Quesnel Lake, hereafter, “Nopiming”; 50° 52' N, 95° 35' W, maximum depth = 25 m), and at Lac du Bonnet (50° 22' N, 95° 53' W, maximum depth = 26 m), a hydropower reservoir along the Winnipeg River. These water bodies are located within boreal Canadian Shield, in the Hudson Bay drainage basin. Water clarity is low in these lakes, and differs between sites, with Nopiming having secchi depths generally >1.5 m, and Lac du Bonnet varying with flow, but typically <1 m across the entire reservoir, and nearing 0 m in certain areas in late summer. Lac du Bonnet is well mixed, with water temperatures around 20°C through the water column for the duration of the sampling seasons in both 2011 and 2012. At Nopiming, surface waters neared 20°C, decreasing to 7°C in the deepest parts of the lake.

Both systems contain diverse fish communities, with similar species compositions that differ slightly between the two sites (CAMP 2013; Boisclair, D., Université de Montréal-HydroNet, Montréal, Québec, Canada, unpublished data). Greater species richness occurs at Lac du Bonnet (CAMP 2013). The fish community within these systems consists largely of percids (e.g. walleye *Sander vitreus*, sauger *Sander canadensis*, yellow perch *Perca flavescens*), coregonids (cisco *Coregonus artedii*, lake whitefish *Coregonus clupeaformis*), catostomids (e.g. White Sucker *Catostomus commersonii*), burbot (*Lota lota*), small mouth bass *Micropterus dolomieu*, and northern

pike *Esox lucius*. Lake sturgeon *Acipenser fulvescens* are present within Lac du Bonnet, but not at Nopiming.

1.3.2 Experimental Procedure

The boats used in this study were designed to be as quiet as possible, given readily accessible equipment and practical constraints. Two 5.5 m Boston Whalers were purpose-modified for hydroacoustic work (Stern 2012), and powered by four-stroke outboard engines (90HP Honda at Lac du Bonnet, 50HP Mercury at Nopiming). The Boston Whalers used were stock foam-cored boats and provided a highly stable platform, limiting turbulence from hull movements, and dampening radiated noise from activities and vibrations onboard the vessel. Unfortunately, noise levels and reduction could not be quantified as equipment necessary to do so was not available at these sites. Increasing availability of portable systems for noise quantification may allow for *in situ* quantification of noise signatures of small vessels used in future surveys (Marine Technology Reporter 2013).

Hydroacoustic data were collected using a BioSonics DTX echosounder (BioSonics, Seattle, Washington, USA) with a downward facing 200 kHz split-beam transducer (6.5° beam angle, pulse length = 0.40 ms) deployed on an adjustable arm off the port side of the vessel at a depth of 0.3 – 0.5 m. The system was calibrated *in situ* using a 36 mm tungsten carbide sphere (target strength = -39.6 dB re 1 m²), following Foote et al. (1987). Ping rates were set to maximum achievable; three frequencies were transmitting sequentially (200, 430, and 1000 kHz, though only data from the 200 kHz transducer is presented in this study), resulting in an average ping rate of 6 pings•s⁻¹ for

each frequency. The far field of the 200 kHz transducer began at approximately 1.0 m from its face. At vessel speeds to $< 3.1 \text{ m}\cdot\text{s}^{-1}$ (6 knots) there was 100% overlap of measurable volumes within the half-power beam in sequential pings (discounting any effects of vessel motion). Overlap, and hence sequential hits on a single target, is expected to be greater at slower speeds and for larger targets.

Boat tracks and acoustic data were georeferenced utilizing a Garmin GPS Model 17x HVS (Garmin International, Olathe, Kansas, USA) with an approximate accuracy of 3 m. To further reduce noise potential from possible electrical interference, the echosounder and all scientific equipment were powered by rechargeable battery packs, electrically independent of the engine alternators and batteries.

Paired transects were used to determine if fish exhibited avoidance responses to the survey vessel. Each pair comprised either two drift transects (i.e. drift:drift; D:D) or one drift immediately followed by one motor transect (i.e. drift:motor; D:M). For D:D pairs, the first transect was completed by allowing the vessel to drift passively across an area with the motor off. At the end of this line, the boat was driven to the beginning of the original transect, without crossing the area that had been sampled, and was allowed to drift over the same area a second time. D:D transect paths were in all cases similar, but somewhat differing drifts were unavoidable. Drifts were primarily driven by forcing from water currents, but wind also contributed to vessel movement along the transects. Drift speeds were $< 1 \text{ m}\cdot\text{s}^{-1}$ in all cases. For D:M pairs, the first transect was drifted as described. For the second transect of a D:M pair, the motor was started, and the GPS track of the drift transect was immediately duplicated in a reverse order at a constant speed haphazardly selected from a range of $1.03 - 3.70 \text{ m}\cdot\text{s}^{-1}$.

Transects spanned a variety of depths and locations within the water bodies. The water depths of transect pairs ranged from 4.2 – 21.0 m, with a mean depth for pairs of 8.8 m (SD = 4.0 m). Transect pairs never exceeded three hours in duration from the start of the initial transect to the completion of the second transect, and in most cases was less than one hour in duration, covering a mean distance of 576 m (SD = 472 m). The precision of the GPS tracks of each pair is thought to be within 3 m, but as the acoustic beam footprint is less than this (approximately 1 m beam footprint diameter for each 10 m depth increment) at every depth within the study sites, constant mean fish density is assumed at these temporal and spatial scales. Transects were completed in relatively calm waters with winds generally $<20 \text{ km}\cdot\text{h}^{-1}$. Weather conditions for transects varied from sunny to light rain, occasionally changing over the course of a single transect line. Acoustic measures of fish communities may differ between day and night surveys (Fréon et al. 1993a, 1993c; Ye et al. 2013), though no standard appears to exist for which provide a more accurate assessment of fish stocks in freshwater (Draštík et al. 2009). Within our study, all experimental work was completed during daylight.

1.3.3 Acoustic Data Processing

Hydroacoustic data were analyzed in Echoview 5.4 software (Myriax, Hobart, Tasmania, Australia). Within our study systems, schooling was not observed during the day within the pelagic zone, allowing for the detection of single fish targets. As backscatter was comprised primarily of single fish echoes, counting techniques were employed utilizing the fish tracking module in Echoview to select individual fish within single target echograms. Fish tracking was chosen for data processing since data from

other single target processing techniques, such as echo counting (see Kieser and Mulligan 1984), may be biased towards larger bodied and deeper fishes. Large and deep fish are more likely to be pinged on multiple times due to organism size and/or beam footprint, which could potentially skew echo count data to over represent these individuals, especially when dealing with a large size range of fishes. Single target and fish tracking parameters were adjusted from default for drift and motor transects (Table 1.1); single target detection was set at a minimum of -60 dB re 1 m², however only fish tracks of a mean acoustic size (target strength; TS) of > -55 dB re 1 m² were included in analyses. Detected fish tracks were manually checked to ensure quality of detections based on characteristics of signal surrounding the fish track, such as separation distance from bottom or nearby fish targets, interference from side-lobe reflection, and the presence of dense plankton signal.

1.3.4 Avoidance Coefficient

The density of fish (F) within each transect was calculated as the average number of fish detected per cubic meter of water sampled within each transect. Sampled volume (V ; m³) was estimated as a wedge, following Kieser and Mulligan (1984), excluding the top 1.0 m of the beam from analysis to account for noise in the acoustic data from surface turbulence and bubbles:

$$(1.1) \quad V = D \cdot \partial_D \cdot L - 1.0 \cdot \partial_{1.0} \cdot L$$

Where D is the mean depth (m) of the transect, ∂ is the beam radius at depth D , and L is the distance travelled in the transect. Fish densities were compared between transects within each pair, with differences defined as the *avoidance coefficient* (Ac):

$$(1.2) \quad Ac = F_f / F_i$$

Where Ac is defined as the avoidance coefficient, F_i is the average fish density (fish•m⁻³) in the initial drift transect of a pair, and F_f is the average fish density in the second transect. F_f may be from a second drift or a motoring transect. An Ac value ≥ 1.0 indicates no avoidance, whereas a value approaching $Ac = 0$ is considered to indicate near complete avoidance. Avoidance coefficients were compared between D:D and D:M pairs to determine if there was a significant change in the magnitude of Ac between the two categories of transect pairs. Within D:M pairs, the influence of study site, initial fish depth, and motoring speed on Ac was examined. Fish depth in the initial transect of the pair was used to test for depth-dependant avoidance, as this represents the depth distribution of fishes in the absence of the motoring vessel. Speed for each D:M pair was determined as the average speed travelled by the vessel during the motoring transect.

1.3.5 Diving Index

The change in average depth of fish between drift and motor transects was also examined, by defining an index of fish diving (DI) as the difference in mean fish depth between the drift and motor transects, as a proportion of the average depth of the transect:

$$(1.3) \quad DI = (FD_i - FD_f) / Depth$$

where FD_i is the average depth (m) of fishes in the initial drift transect, FD_f is the average depth (m) of fishes in the second transect, and $Depth$ is the average depth of transects (m) within the pair, determined as:

$$(1.4) \quad Depth = (\text{mean depth}_{\text{initial}} + \text{mean depth}_{\text{final}}) / 2$$

The difference in fish depth between the initial and final tracks was normalized for overall water depth since fish in deeper waters have a wider depth range available, which may bias the result across a range of depths if not accounted for. A negative diving index indicated that fishes were, on average, deeper in the motor transect than the drift, and would be considered evidence of diving reactions to the moving vessel.

1.3.6 Change in Target Strength

Average acoustic size (\overline{TS} ; dB re 1 m²) of fish for each transect was defined as the average of TS (dB re 1 m²) of all tracked fish within that transect (averaged in the arithmetic domain). The change in \overline{TS} (ΔTS) within paired drift and motor transects was calculated as:

$$(1.5) \quad \Delta TS = (\overline{TS}_{\text{final}}) - (\overline{TS}_{\text{initial}})$$

A negative ΔTS value indicates that, on average, TS was larger in the initial drift than the second (motoring, or drift) transect, a positive value that TS was smaller.

1.3.7 Vessel Attraction

In order to test for attraction of fish to the drifting vessel, we examined the change in fish abundance along a drift transect. If the vessel was attracting fish, it would be

expected that the number of fish detected in the beam would increase along the drifted transect as fish move towards the vessel (Røstad et al. 2006). Drift transects were divided into five minute intervals, and the number of fish detected per time interval determined. Linear regressions of fish abundance per interval, versus time, were then completed for each drift transect, excluding those transects which did not contain a sufficient number of fishes to allow for a valid regression. Significant increases in fish abundance over time are considered to be evidence of attraction.

1.3.8 Statistical Analysis

Error structure for A_c did not conform to a normal distribution; therefore generalized linear models (GLM), with quasi-poisson error structure, were used to:

1. Assess the influence of study site, motoring speed, and fish depth on the magnitude of A_c
2. Determine if A_c differed between D:D and D:M pairs at Lac du Bonnet.
3. Test for differences in A_c of D:M pairs at Lac du Bonnet between 2011 and 2012.

Paired-sample Wilcoxon tests were used to determine if A_c values for different groups varied from $A_c = 1$.

Transect and initial fish depths were compared between Lac du Bonnet and Nopiming using a single-factor ANOVA. Single-sample t -tests were used to determine if DI or Δ TS differed from zero. Linear regression was applied to determine if DI values were correlated to transect depth or fish depth. Mean TS did not conform to a normal distribution, therefore a Wilcoxon test was used to determine if TS differed between study

sites. One transect pair was excluded from DI and Δ TS analysis, as no fish were detected in the motor transect and the values could therefore not be calculated. An additional Δ TS value was identified as an outlier ($> \text{mean} + 3\text{SD}$), and excluded from analyses.

All statistical analyses were completed using statistical software package R, version 2.14.2 (R Foundation for Statistical Computation, Vienna, Austria). Significance was assessed at $\alpha = 0.05$ for all analyses.

1.4. Results

1.4.1 Avoidance Coefficient

A total of 50 transect pairs were completed (Appendix); 13 at Nopiming, and 37 at Lac du Bonnet. All pairs at Nopiming were D:M pairs, while both D:M ($n = 31$) and D:D ($n = 6$) pairs were completed at Lac du Bonnet (Figure 1.2). Transects pairs at Nopiming were characterized by deeper waters (ANOVA: $F_{1,42} = 41.08$, $p < 0.01$), fish deeper in the water column (ANOVA: $F_{1,42} = 32.84$, $p < 0.01$) and fish of larger TS (Wilcoxon; $W = 42.5$, $p < 0.01$) than those at Lac du Bonnet (Table 1.2).

Study site had a significant influence on A_c of D:M pairs (Table 1.3; GLM: $\chi^2 = 4.42$, $p = 0.04$). We detected no significant influence of fish depth or survey speed on A_c (Figure 1.3). Transects at Nopiming did not show significant avoidance, with median A_c not different from one (Wilcoxon: $V = 50.5$, $p = 0.75$; $n = 13$, median = 0.81, 95% CI = 0.69, 1.48). At Nopiming, A_c values ranged from 0.52 to 2.31, with $A_c < 1$ in 61% of

transect pairs. In contrast, avoidance was apparent at Lac du Bonnet, with a median A_c value significantly less than one ($V = 64.0$, $p < 0.01$; $n = 31$, median = 0.51, 95% CI = 0.45, 0.79). A_c values at Lac du Bonnet ranged from 0 to 1.95, with $A_c < 1$ in 81% of transect pairs. There was no significant difference in A_c between years (GLM: $\text{chisq} = 0.07$, $\text{df} = 1$, 29 , $p = 0.79$) for D:M pairs at Lac du Bonnet in 2011 (median = 0.53, $Q1 = 0.47$, $Q3 = 0.96$, $n = 15$) and 2012 ($n = 16$, median = 0.37, $Q1 = 0.14$, $Q3 = 0.85$).

A comparison of D:D ($n = 6$) and D:M ($n = 31$) pairs at Lac du Bonnet revealed a significant difference in A_c values among types of transect pairs (Table 1.2, GLM: $\text{chi-sq} = 16.38$, $p < 0.01$). While D:M pairs showed significant avoidance, results of D:D pairs did not indicate avoidance reactions occurring, with a median A_c value that did not differ from one ($V = 18.0$, $p = 0.16$; median = 1.60, 95% CI = 0.52, 3.83). A_c of D:D pairs ranged from 0.52 to 3.83, and in 33% of pairs A_c was less than one.

1.4.2 Diving Index

All subsequent analyses were completed on D:M pairs only. Diving index (DI) values did not differ from zero, ($t = -1.63$, $p = 0.11$; -0.03 ± 0.13 (mean \pm SD), $n = 43$; Figure 1.4a), and do not suggest that fish moved into deeper waters during motored transects. DI did not vary significantly between the two study sites (ANOVA: $F_{1,41} = 3.40$, $p = 0.07$). There was no change in DI with decreasing transect depth (linear regression: $F_{1,41} = 0.56$, $p = 0.46$), or with differences in mean fish depth during the drift transect (linear regression: $F_{1,41} = 0.12$, $p = 0.73$), or motor transect (linear regression: $F_{1,41} = 2.10$, $p = 0.15$).

1.4.3 Change in target strength

Δ TS (n = 42) did not differ from zero (Figure 1.4b) within D:M transect pairs (t = 0.35, p = 0.72; 0.2 ± 3.7 dB re 1 m^2), indicating no change in mean target strength measured between drift and motor transects. The magnitude of Δ TS did not differ significantly between Lac du Bonnet (0.37 ± 3.9 dB re 1 m^2 , n = 29) and Nopiming (-0.18 ± 3.14 dB re 1 m^2 , n = 13) (ANOVA: $F_{1,40} = 0.20$, p = 0.65).

1.4.4 Vessel Attraction

No consistent pattern was observed in changes in fish abundance along drift transects (n = 40). A significant relationship between fish abundance per five minute interval, with increasing time along a drift, was only detected in 15% (n = 6) of drift transects, of which only three showed a positive relationship (Table 1.4). The lack of a consistent increase in abundance with time within drift transects suggests that fish were not being attracted to the vessel.

1.5 Discussion

Our results indicate that fish avoidance of small acoustic survey vessels can be quantified in small boreal lakes employing a simple method based on comparing drifts and motored runs of the same transects. Variability in the magnitude of avoidance coefficients likely resulted from temporal variation in the fish community along each transect.

A strong relationship between avoidance and depth is not supported by our results in these shallow boreal lakes. We did not observe increased avoidance when fish are shallower within the water column, which may have been expected as fish closer to the vessel might perceive a greater threat from increased vessel noise and turbulence from the propeller and hull moving through the water. Our study is consistent with previous findings of limited avoidance at distances >10 m (Blanchfield et al. 2005; Draštík and Kubečka 2005), but extends them to even shallower depths in boreal lakes. We caution, nonetheless, that in lakes that are deeper than our study lakes, or with a different survey platform, depth may be an important factor. In deeper marine systems, fish have been shown to avoid large survey vessels at distances on the order of hundreds of meters, and to move vertically in the water column away from the vessel (e.g. Vabø et al. 2002; Mitson and Knudsen 2003). Fish targets observed in our trials did not shift distribution into deeper waters in the presence of the motoring vessel, suggesting that diving reactions were not a prominent avoidance mechanism.

Large and small fish may avoid vessels in different proportions (Draštík and Kubečka 2005), due to differences in swimming ability (Bainbridge 1958), or likelihood of identifying the vessel as a potential predator. As aquatic ecosystems are typically size structured (Kerr and Dickie 2001), capturing this structure in acoustic data is important for potential use in size-based studies. TS:length relationships are well established (e.g. Love 1971; Kubečka and Duncan 1998), and comparisons of avoidance among fish of differing TS may be used to examine differing levels of vessel avoidance by fishes of varying size (Draštík and Kubečka 2005). Here, Δ TS may be interpreted as a rough

indicator of differential avoidance among fishes of various lengths. Our data do not indicate any significant change in mean TS between drift and motoring tracks, therefore avoidance in these systems is not considered to be size-selective. However, TS is also influenced by body composition of fishes (e.g. swim bladder shape and volume), and therefore varies among species (Foote 1980; Frouzova et al. 2005). The orientation of fish relative to the acoustic beam also impacts measured TS of individual fish (Hazen and Horne 2003; Simmonds and MacLennan 2005). Differences in TS due to changes in fish orientation in the presence of our vessel were likely minimal, however, as diving reactions were not found to be significant.

The cause of differences in avoidance between Lac du Bonnet and Nopiming are difficult to determine with our data. Consistent with studies in European lakes and reservoirs (e.g. Draštik and Kubečka 2005; Godlewska et al. 2009), visual cues do not appear to have played a role; although both lakes had relatively low water transparency, the clearer lake (Nopiming) showed decreased avoidance relative to Lac du Bonnet. There is also no evidence of habituation to vessels as a factor influencing avoidance, as has been observed in other studies (Vabø et al. 2002; Doksæter et al. 2012); avoidance was greater at Lac du Bonnet, where the use of motorized boats is much more prevalent due to a greater population and abundant recreational boating, relative to the remote Nopiming. The noise characteristics of the vessels and motors used at the two sites may have differed, despite their intended similarity. The more powerful motor used at Lac du Bonnet may have led to an increase in perceived threat by fishes, resulting in the greater avoidance observed at this site. However, we were not able to measure the vessel

noise signatures and are therefore limited to speculation. Whatever the cause, it points to the importance of undertaking *in situ* avoidance experiments utilizing the specific platform and equipment to be used in surveys.

Our data suggest that fish were not attracted to our drifting vessels, nor moved in and out of the drifting acoustic beam (Benneheka et al. 1995), thereby over-estimating drift densities and inflating avoidance coefficients. The number of fish within drifts did not consistently increase with time, which has been used as an indicator of attraction in marine systems (Røstad et al. 2006). In addition, if attraction were occurring, significant avoidance coefficients would have been expected at both study sites, but this was not observed. Nonetheless, the potential for a drifting vessel to attract fish, and hence inflate density estimates, should be tested for in all empirical determinations of an avoidance coefficient.

The need for methods to quantify the impact of vessel avoidance in acoustic surveys has been well recognized (Vabø et al. 2002; Hjellvik et al. 2006; DeRobertis and Handegard 2013). The application of a correction factor may be a useful method of accounting for this vessel impact. Here we computed an avoidance coefficient, which is reflective of the fish density measured by a motoring vessel, as a proportion of the undisturbed fish density. The fish density estimate obtained from an acoustic survey may be multiplied by the inverse of the avoidance coefficient, with associated uncertainty intervals, yielding an estimate of the actual fish density corrected for bias due to avoidance of the survey vessel.

Consider a hypothetical acoustic survey which yields a fish density estimate of 0.3 fish•m⁻². An estimate at a site where no significant avoidance was detected, such as Nopiming, needs no correction. In contrast, where avoidance is determined to be significant, *Ac* estimates may be applied to obtain an adjusted value, with corresponding confidence intervals:

$$(1.6) \quad \text{Density}_{(\text{Adjusted})} = \text{Density}_{(\text{measured})} \cdot (1/Ac), \pm 95\% \text{ CI}$$

The data distribution should be examined to determine if median or mean avoidance is the appropriate correction factor. At Lac du Bonnet, *Ac* values were not normally distributed, therefore the median *Ac* would be used to correct for vessel avoidance (Helsel 1987).

Following equation (1.6), and given median *Ac* = 0.51, with a lower confidence limit at *Ac* = 0.35, and an upper limit at *Ac* = 0.86, an adjusted density at Lac du Bonnet may be obtained:

At median Ac:

$$(1.7) \quad \text{Density}_{(\text{Adjusted})} = 0.3 \text{ fish/m}^2 \cdot (1/0.51) = 0.59 \text{ fish/m}^2$$

Lower & upper limits:

$$(1.8) \quad \text{Density}_{(\text{lower})} = 0.3 \text{ fish/m}^2 \cdot (1/0.35) = 0.86 \text{ fish/m}^2$$

$$(1.9) \quad \text{Density}_{(\text{upper})} = 0.3 \text{ fish/m}^2 \cdot (1/0.86) = 0.35 \text{ fish/m}^2$$

These estimates would be applied to obtain population estimates that explicitly account for avoidance of the vessel by the fish under study. In the present study, we attempted to determine avoidance over a range of survey speeds and depths, in order to inform the

optimum survey conditions. Future work might focus on the actual survey speed in an attempt to increase precision in avoidance coefficient estimates.

The paired transect method used in this study has some limitations. Firstly, the method is limited to areas in which drifting, by either wind or current forcing, is feasible without disturbance from other vessels, and that a drifting vessel can be used as a suitable control (i.e. does not initiate avoidance) for motoring transects at survey speeds. In addition, we assume here that the sudden start of the motor between the drift and motoring track does not inflate avoidance relative to typical survey conditions (Doksæter et al. 2012).

While these assumptions may contribute to uncertainty in the quantification of avoidance, there is no evidence that any of them played important roles in the present study. The paired transect method appears to be useful and cost effective for the estimation of the impact of a motoring vessel on fish densities measured in acoustic surveys, and should be applicable to a wide range of environments as long as drifting is feasible. *In situ* quantification of boat avoidance should be done under typical survey conditions.

We conclude that in acoustic surveys of boreal lakes and reservoirs, boat avoidance can be quantified using relatively simple methods developed here. Survey platform design should attempt to reduce noise, and new portable technologies may enable *in situ* measurements of boat noise (something not possible in the present study). Of primary importance, avoidance should not be assumed to be constant among different

systems or vessels, but should be quantified *in situ* allowing for the estimation of an avoidance correction factor to be applied to acoustic survey estimates of fish density.

Table 1.1: Single target and fish track detection parameters used in Echoview 5.4

| Single Target Detection | |
|---|-----|
| TS threshold (dB) | -60 |
| Pulse length determination level (dB) | 6 |
| Minimum normalized pulse length | 0.7 |
| Maximum normalized pulse length | 1.5 |
| Maximum beam compensation (dB) | 15 |
| Maximum standard deviation of minor and major axis angles (degrees) | 1.2 |

| Track Detection Properties | |
|------------------------------------|---|
| Minimum number of single targets | 1 |
| Minimum number of pings in track | 1 |
| Maximum gap between single targets | 2 |

Table 1.2: Summary of site characteristics and avoidance indicators for drift:drift (D:D) and drift:motor (D:M) transect pairs. Significant ($p < 0.05$) differences between study site characteristics, and in avoidance indicator values between D:M pairs at Lac du Bonnet (LdB) and Nopiming are marked with an asterisk (*)

| | D:D | D:M | | |
|---|-----------------|------------------|------------------|---|
| | LdB | LdB | Nopiming | |
| N | 6 | 31 | 13 | |
| <i>Depth</i> (m); mean \pm SD | 7.5 \pm 1.9 | 7.0 \pm 2.0 | 13.1 \pm 4.4 | * |
| Fish Depth, initial (m); mean \pm SD | 3.9 \pm 1.7 | 4.2 \pm 1.3 | 8.8 \pm 4.0 | * |
| \overline{TS} (dB re 1 m ²); median | -47.5 | -47.9 | -40.0 | * |
| (Q1, Q3) | (-49.9, -46.3) | (-50.45, -46.25) | (-40.9, -37.5) | |
| <i>Ac</i> ; median | 1.60 | 0.51 | 0.81 | * |
| (Q1, Q3) | (0.84, 2.36) | (0.35, 0.86) | (0.76, 1.55) | |
| DI; mean \pm SD | 0.00 \pm 0.17 | -0.01 \pm 0.14 | -0.08 \pm 0.08 | |
| ΔTS (dB); mean \pm SD | 2.0 \pm 7.3 | 0.4 \pm 3.9 | -0.2 \pm 3.1 | |

Table 1.3: Generalized linear model results of factors influencing *Ac*. The impact of fish depth, vessel speed and study site was examined for drift:motor (D:M) transect pairs (n = 46). Differences among years (2011 n = 15, 2012 n = 16), and the impact of transect pair type (drift:drift, n = 6; D:M, n = 31) was examined at Lac du Bonnet. Variables are considered as fixed effects. Interactions were not significant.

| Parameter | Variable type | p | chi-square |
|---|----------------------|----------|-------------------|
| All Drift:Motor Pairs | | | |
| Fish depth, initial transect (m) | Continuous | 0.73 | 0.12 |
| Vessel speed (m•s ⁻¹) | Continuous | 0.82 | 0.05 |
| Study site | Categorical | 0.04 | 4.42 |
| 2011 vs. 2012, Lac du Bonnet | | | |
| Year | Categorical | 0.79 | 0.07 |
| Drift:Drift vs. Drift:Motor, Lac du Bonnet | | | |
| Pair Type | Categorical | <0.01 | 16.38 |

Table 1.4: Linear regressions of the number of fish detected per five minute interval over time within drift transects did not reveal any consistent pattern in changes in abundance with time. A significant positive relationship would be considered to suggest attraction of fish to the vessel.

| Relationship | N | R² (mean ± SD) | Slope (mean ± SD) |
|--------------------------------------|----------|--|------------------------------------|
| Significant (p<0.05), positive slope | 3 | 0.67 ± 0.25 | 1.99 ± 2.41 |
| Significant (p<0.05), negative slope | 3 | 0.85 ± 0.12 | -1.72 ± 1.68 |
| Not significant | 34 | 0.23 ± .21 | -0.24 ± 0.63 |

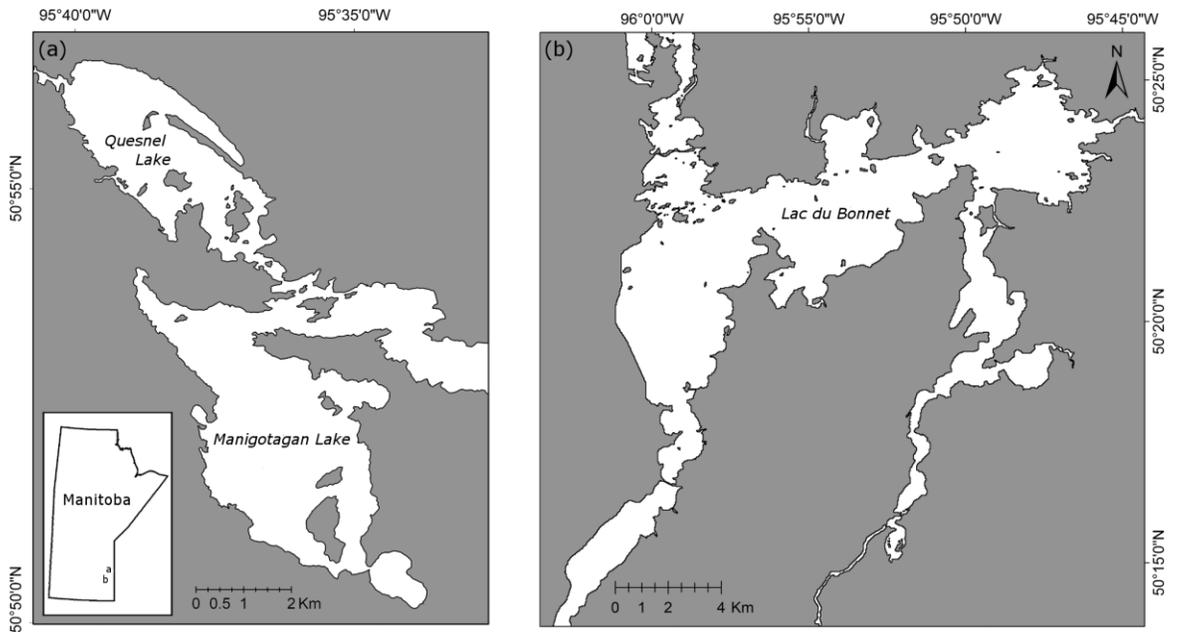


Figure 1.1: Maps of (a) the Nopiming system (Manigotagan and Quesnel Lakes), and (b) Lac du Bonnet, Manitoba, Canada. Experimental transects were completed within Nopiming in July and August of 2011, and at Lac du Bonnet in July and August of 2011 and 2012.

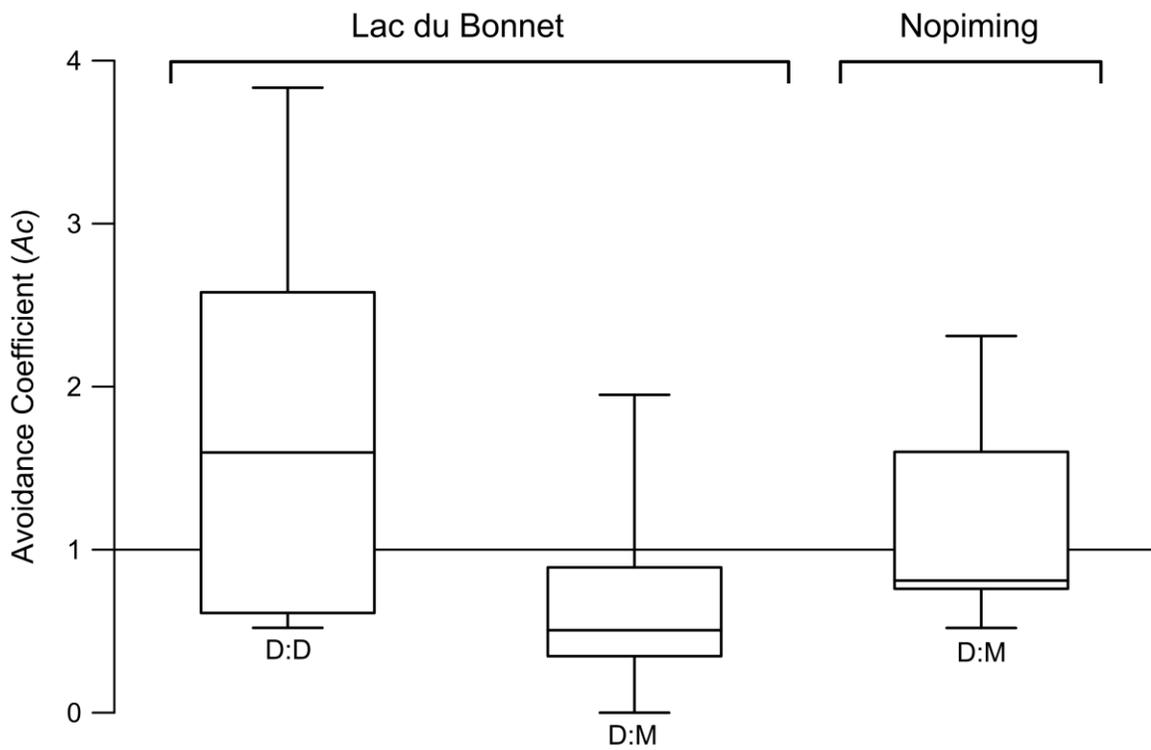


Figure 1.2: Avoidance coefficients (Ac) of paired drift:drift (D:D) and drift:motor (D:M) transects differed significantly among study sites, Lac du Bonnet and Nopiming, with significant avoidance ($p < 0.05$) occurring at Lac du Bonnet. Avoidance was not detected in drift:drift (D:D) transect pairs at Lac du Bonnet.

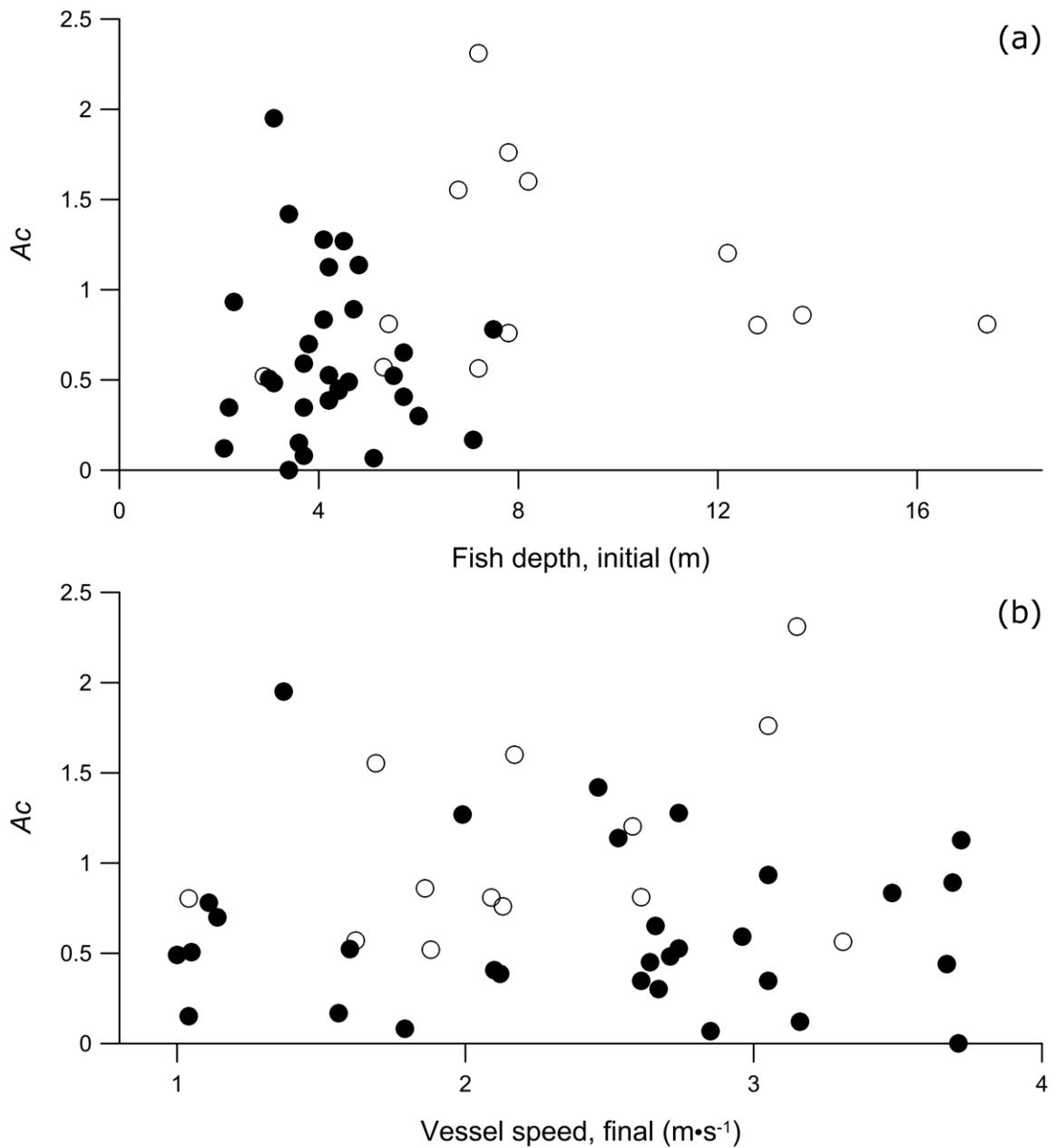


Figure 1.3: Ac values for drift:motor pairs were not found to be significantly influenced by (a) initial fish depth or (b) vessel speed, but were significantly different between the two study sites, Lac du Bonnet ($n = 31$, solid circles) and Nopiming ($n = 13$, open circles).

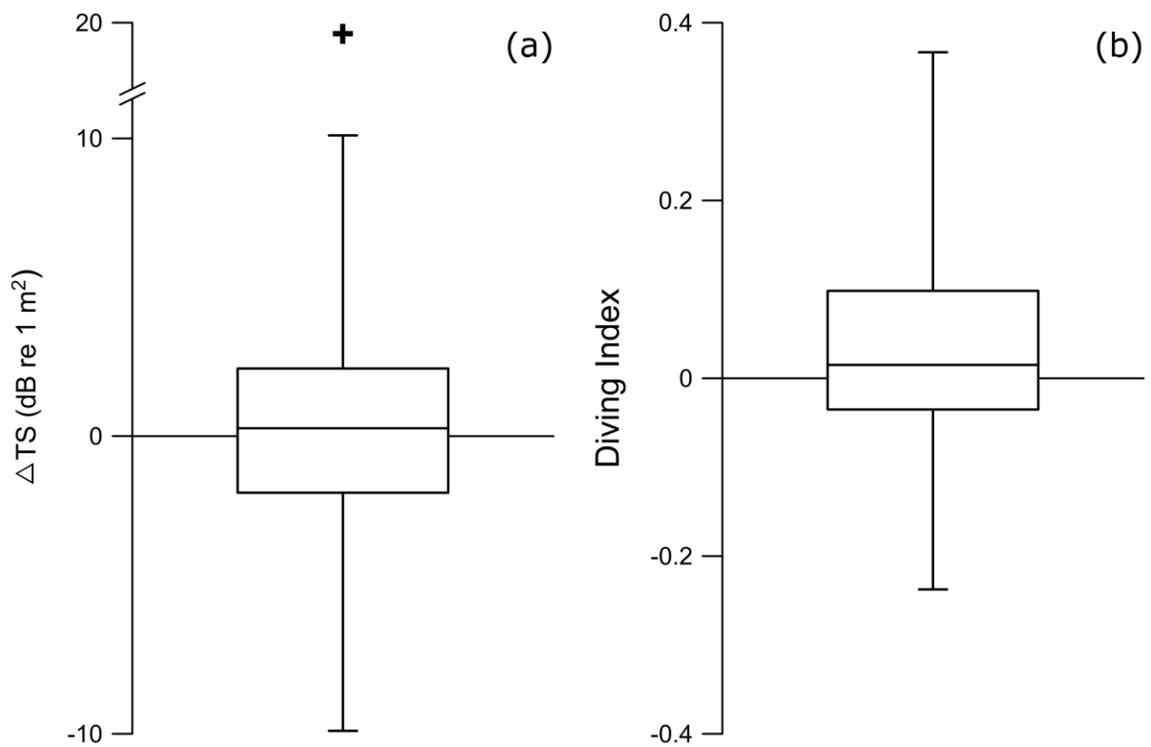


Figure 1.4: (a) Diving index (DI) within pairs of drift:motor transects did not differ significantly from zero ($p = 0.11$, $n = 43$). (b) Changes in mean target strength of fishes (ΔTS) between drift and motor transects within pairs did not differ significantly from zero ($p = 0.72$, $n = 42$). Boxplots show minimum, maximum, median, and first and third quartiles. Outliers ($\text{mean} \pm 3SD$) are indicated by a cross.

2. USING ACOUSTIC SIZE SPECTRA TO ASSESS FISH COMMUNITIES IN A BOREAL RESERVOIR

2.1 Abstract

Ecological indicators that consider organism body size are especially useful in aquatic ecosystems, as the structure and function of these systems are driven largely by size-based processes. Here, we used 11 hydroacoustic surveys in 2011 and 2012 (BioSonics DTX, split-beam 200 kHz) to derive length-frequency spectra of fish communities at Lac du Bonnet, a hydropower reservoir along the Winnipeg River, Manitoba, Canada. Acoustic data from the predominant single fish echoes, tracked and converted to fish lengths (5-50 cm), formed significant spectra in all cases ($p < 0.05$, R^2 ranged from 0.77-0.99). Fish abundance (spectra height) within the reservoir was greater in 2011 than in 2012 ($p < 0.05$), varied consistently between 3 basins of the ecosystem, and decreased through the sampling season in both years. Size composition (spectra slope) of fish communities did not differ between years, but were related to physical habitat characteristics with slopes steeper ($p < 0.05$) in areas characterized by shallow waters and large littoral zones. Spectral variations were consistent with fish catch using gillnets. Acoustically derived size spectra provide an efficient means to detect and monitor fish community dynamics over varied spatial and temporal scales in boreal systems where single acoustic targets predominate.

2.2 Introduction

Body size is a fundamental property of all organisms, being linked to life history traits and physiological properties, including production (Sheldon et al. 1972), metabolic rate (Gillooly et al. 2001), reproductive efficiency (Fenchel 1974), and trophic interactions (Elton 1927; Thiebaut and Dickie 1993; Yvon-Durocher et al. 2011; Giacomini et al. 2013). As a consequence of these links, the abundance of organisms varies predictably with body size in aquatic ecosystems (Kerr 1974; Boudreau et al. 1991; Thiebaut and Dickie 1993).

Community size spectra, describing the relative abundance of organisms by size, provide a useful method for evaluating the status of aquatic ecosystems (Petchey and Belgrano 2010; Emmrich et al. 2011; Murry and Farrell 2014). Spectra provide information on underlying size-based mechanisms in communities (Boudreau et al. 1991), and create a link between traits at the individual and population level with the structure and dynamics of the community (White et al. 2007). In general, the slope of size spectra is indicative of the relative abundance of fish by size, while spectra height indexes overall community abundance (Daan et al. 2005; Yemane et al. 2005; Sweeting et al. 2009). An examination of changes in size spectra over time can be used to investigate systematic shifts in fish communities (Emmrich et al. 2011) and assess the impact of environmental changes on aquatic communities (Rice and Gislason 1996; Duplisea and Castonguay 2006; Petchey and Belgrano, 2010). Comparisons of spectra among and within systems may be used to assess spatial differences in productivity (Sprules and Munawar 1986; Macpherson et al. 2002).

Fish size spectra have typically been derived from catch data (e.g. Rice and Gislason 1996; Emmrich et al. 2011). However, methods used to capture fish, such as gillnetting, electrofishing, and trawling, are size-selective and introduce bias in size spectra analyses (Hamley 1975; Bonvechio et al. 2008). Hydroacoustic methods, on the other hand, are theoretically less size selective if the majority of fish can be isolated within the acoustic beam. In freshwater ecosystems where this condition holds, acoustic methods can provide a more representative sample of the length distribution of a fish community than netting surveys (Coll et al. 2007; Kubecka et al. 2009). In addition, directivity-related variations in target strength (e.g., Nakken and Olsen 1977; Gauthier and Rose 2001) may be less prominent than in oceanic ecosystems because shallower water depths limit vertical movements. However, to the best of our knowledge, there have been no published studies in which size spectra from freshwater systems were derived from acoustic data.

The principal objective of this study was to determine if useful length-frequency spectra of freshwater fish communities in a boreal reservoir could be derived from hydroacoustic data. We hypothesized that spectra slope and height would index variations in fish abundance and size structure resulting from variations in physical habitat, season and year. We then discuss how acoustic size spectra might be used to monitor the status of fish communities in freshwater ecosystems.

2.3 Methods

This study was conducted in July and August of 2011 and 2012, at Lac du Bonnet (50° 22' 27" N, 95° 53' 48" W), a hydropower reservoir in southeastern Manitoba, Canada (Figure 2.1). This site is located within the boreal Canadian Shield region within the Hudson Bay drainage basin. The main channel of flow travels north from the Winnipeg River into Lac du Bonnet, with flow magnitude generally decreasing along the west-east axis of the lake. Lac du Bonnet has been dammed since the construction of MacArthur Falls generating station in 1952 (Manitoba Hydro 2013a). Flow is also regulated upstream at the Seven Sister's generating station, which began operation in 1931. Both generating stations operate as run of the river facilities (Manitoba Hydro 2013b). Additional inputs from the Bird River and Lee River flow into the eastern portion of Lac du Bonnet.

A 5.5m Boston Whaler was modified for hydroacoustic work (Stern 2012) and powered by a relatively quiet 90 HP Honda 4-stroke outboard engine. The Whaler provided a highly stable platform with a foam cored hull which limited noise transmission into the water column. This boat-motor combination was expected to limit noise in the acoustic data and decrease the potential for vessel avoidance by fishes (Wheeland and Rose 2014). Hydroacoustic data were collected with a BioSonics DTX echosounder (BioSonics, Seattle, WA, USA) with downward facing splitbeam transducers operating at 200 kHz (beam width = 6.5°, pulse width = 0.4 ms) and 430 kHz (beam width = 6.9°, pulse width = 0.4 ms). Transducers were deployed on an adjustable arm off the port side of the vessel, with the face of the transducers submerged to a depth of 0.3-0.5 m. Ping

rate was set to maximum achievable ($6 \text{ pings} \cdot \text{s}^{-1}$ for each frequency). Each frequency was calibrated *in situ* using tungsten-carbide calibration spheres, following Foote et al. (1987). The echosounder and all scientific equipment were powered by rechargeable battery packs, keeping them electrically isolated from the engine in order to reduce the potential for electrical noise interference. Acoustic data were recorded with controlling software Visual Acquisition 6 (BioSonics Inc., Seattle, WA, USA), and georeferenced with a Garmin GPS, Model 17x HVS (Garmin International, Olathe, KS, USA) with an accuracy of approximately 3 m.

Hydroacoustic surveys were completed as a series of parallel transects oriented perpendicular to the long axis of the reservoir (Figure 2.1). Spacing between transects was 1.85 km (one nautical mile). Tracks for multiple surveys were offset from one another by approximately 300 m in order to maximize coverage for mapping physical habitat, and to facilitate analysis of variation introduced through differences in survey coverage. Surveys were completed at a constant speed of $3.1 \text{ m} \cdot \text{s}^{-1}$ (6 knots), and were limited to depths >3 m. All surveys were completed during daylight hours, beginning at least 30 minutes after sunrise.

Hydroacoustic data were edited and analyzed using Echoview 5.0 software (Myriax, Hobart, Tasmania, Australia). Data from the 200 kHz transducer were used for all information on fishes. The single target detection and fish tracking modules were used to isolate and measure individual fish. Single target detection parameters were derived empirically and kept consistent for all surveys (Table 2.1). The fish tracking module grouped sequential single targets from individual fish; all tracked fish were manually

checked to ensure quality of detections based on characteristics of adjacent signal (e.g., interactions with other fish, side-lobe interference, noise). Fish tracking reduced the potential bias of non-tracked single target data (i.e., echo counting), in which larger fish or those at greater range may be represented by a greater number of pings. Lengths (total length; TL) of individual fish were estimated from mean TS of each fish track at 200 kHz, using the standard equation derived by Love (1971). Fish estimated to be <5 cm TL were excluded from data analysis due to uncertainty in separating fish signal from that of dense plankton in the low target strength range. An upper bound on fish included in size spectra was set at a maximum TL of 50 cm due to rarity of larger fishes detected in our surveys (<5% of total fish detections within surveys and similarly rare in catch data).

The derivation of fish community size spectra was adapted from Emmrich et al. (2001) for use with hydroacoustic surveys. Fish counts from acoustic data were binned in 5cm length intervals. To make spectra comparable between surveys and among basins, counts were normalized based on the volume of water sampled by the acoustic beam, so that all spectra are representative of an equal volume. Fish counts were normalized following:

$$(2.1) \quad N_L = C_L \cdot \left(\frac{V}{5.21E+04} \right)$$

where N_L is the volume-normalized count, C_L is the number of fish within length bin L . One of the surveys was chosen at random to be the representative sample volume, and fish counts from all additional surveys were normalized to represent this volume, in our surveys $5.21E+04 \text{ m}^3$. Sampled volumes (V) were estimated as a wedge, based on the

summation of acoustic beam volume along 10m intervals of survey track. This volume was calculated as:

$$(2.2) \quad V = \sum_{i \rightarrow n} [(D_i \cdot \delta_{Di} \cdot 10m) - 2.27]$$

Where n is the number of 10 m intervals within the survey, D_i is the average depth of the 10m interval, and δ_{Di} is the half-power beam radius at depth D . The constant of 2.27 represents the volume of the top 2 m of the beam in each 10 m interval; this portion of the water column is excluded to account for surface noise and the transducer near-field.

Size spectra were computed through model II major axis regression (Legendre 2013), with the natural logarithm of normalized fish counts being the dependent variable, and the natural logarithm of the midpoint of each length bin the independent variable. Height of the spectra at the midpoint of the length range, on a natural log scale, was computed from regression equations for each individual spectrum (Daan et al. 2005).

Acoustic data from the 430kHz transducer collected during the fish surveys were combined with additional transects oriented perpendicular to survey lines as well and near shore tracks circumscribing the reservoir's shoreline and the perimeter of islands within the study area. Vessel speed ranged from 1.0-3.1 $\text{m}\cdot\text{s}^{-1}$ for the collection of acoustic habitat data. The lake bottom depth was determined in Echoview, and imported into ArcGIS 10.1 (Esri, Redlands, CA, USA) where a nearest neighbour interpolation was used to interpolate depths between transects, generating a 20 m resolution raster bathymetric grid. Areas of submerged aquatic vegetation were manually identified in the acoustic data.

Bottom substrate was assessed at a number of sites within the reservoir, with locations chosen based on visual assessment of differences in acoustic bottom signal. Underwater video (Sea-Drop Camera 950, SeaViewer, Tampa, FL, USA) was used to determine if substrate was hard (i.e. rocky; consisting of boulder, cobble and/or bedrock), or soft. If substrate was determined to be soft from the video, an Ekman grab (2011: 25.2 cm x 25.2 cm; 2012: 15.2 cm x 15.2 cm) was used to take a sample of the substrate, which was then visually classified as sand or mud.

Thermal profiles of the water column were measured periodically within each sampling season by dropping a Minilogger II (VEMCO, Halifax, NS, Canada) at a constant rate from the water surface to the lake bottom.

Data on the species composition of the fish community were collected using gillnets. Nets were tied end to end forming a gang comprised of 10 m lengths of 5/8" and 1" mesh, and 20 m lengths each of 1", 2", 3", and 4 1/4" mesh. Nets were equipped with a float line and a lead line, and gangs were anchored at each end with a sand bag. Each net was approximately 2 m in height. Gillnet sampling locations were chosen to cover a range of habitat characteristics across the area of the reservoir. In 2011 all nets were set at the bottom of the water column, while in 2012, each set consisted of one gang of nets set on the bottom, and a second gang set midwater, with the exception of Set 3 where the pelagic net did not set properly and was therefore excluded. All sets were completed during daylight. Sets were generally 1 hour in duration, with the exception of Set 9, which was deployed for 2.5 hours due to boat malfunctions. Fish collected were identified to species

whenever possible and measured for total length. Fragments of individual fishes found in the nets were noted for species presence, but lengths were not taken.

ANOVAs were used to test for variation in spectra parameters. Regression analysis was applied to spectra slope and heights to assess seasonal changes. All statistical analyses were done in R statistical software, version 2.14.2 (R Foundation for Statistical Computation, Vienna, Austria).

2.3 Results

Three basins were identified within Lac du Bonnet, based on bathymetry and shoreline morphology (Figure 2.1). The maximum depth recorded was 25.2 m, located in Basin 1, with a mean depth of 7.7 m for the whole lake. Macrophytes were located only at depths shallower than 2.5 m, with growth limited by low water clarity. Hence, macrophytes were not present on the survey transects which were limited to areas >3 m deep. Video analysis and grab samples revealed that substrate composition ranged from bedrock to mud, and varied among the three basins as well as within basins (Table 2.2). In Basin 1, areas of rocky, sandy, and muddy substrate were identified. The bathymetry of Basin 1 is complex, with steep slopes along the main channel of flow, and extensive flat areas along the western shoreline in the northern portion of the basin. Basin 2 showed very little variability in bathymetry, with generally low slopes and a depth of 6.5-8 m across the majority of the basin. The substrate in this basin was mainly mud, with a few rocky outcrops. Basin 3 is bowl-shaped, with steeper slopes nearer the shoreline consisting

largely of bedrock, and depths to 15 m at the centre of the basin where substrates were mostly mud.

Thermal profiles did not indicate the presence of a thermocline during our study, with water temperatures generally $23^{\circ}\text{C} \pm 3^{\circ}$ from the surface to the deepest point of the lake (25 m) in both years of the study.

Six surveys were completed in 2011 and five in 2012, with size spectra computed for the reservoir as a whole (Table 2.3), and for each basin (Table 2.4). Size spectra regressions were significant in all cases ($p < 0.05$); Figure 2.2), with a R^2 values ranging from 0.87 - 0.99. All slopes were negative. For surveys late in the season in both 2011 and 2012 there were no fish detected in some of the larger length classes. Vacant length classes were excluded from spectra regressions to eliminate a floor effect (Rice and Gislason 1996) that may skew parameters.

Spectra height for the reservoir was significantly greater ($p = 0.02$, $F_{1,9} = 7.67$) in 2011 (mean \pm SD = 2.81 ± 0.41) than in 2012 (2.15 ± 0.36). In both years spectra height declined as the summer progressed (2011: $p < 0.01$, $F_{1,4} = 64.64$; 2012: $p = 0.02$, $F_{1,3} = 17.94$; Figure 2.3). Spectra slopes did not significantly differ between 2011 (-2.66 ± 0.35) and 2012 (-2.73 ± 0.29) ($p = 0.72$, $F_{1,9} = 0.14$). The magnitude of spectra slopes decreased as the season progressed in 2011 ($p < 0.01$, $F_{1,4} = 27.16$), but not in 2012 ($p = 0.86$, $F_{1,3} = 0.04$; Figure 2.4). Slopes were significantly steeper ($p < 0.01$, $F_{1,9} = 13.80$) in surveys in which transects spanned Hay Bay (mean \pm sd = -2.83 ± 0.21 , $n = 8$) as opposed to those that did not (-2.31 ± 0.19 , $n = 3$).

Spectra height was not significantly different when compared among the three basins (2011: $p = 0.245$, $F_{2,15} = 1.60$; 2012: $p = 0.15$, $F_{2,12} = 2.22$), however mean height was greatest, in Basin 3 in both years (Figure 2.5). Heights differed significantly between 2011 and 2012 within Basin 1 ($p < 0.01$, $F_{1,9} = 14.27$) and 2 ($p < 0.01$, $F_{1,9} = 29.30$). No difference in height was observed in Basin 3 between the two years ($p = 0.08$, $F_{1,9} = 3.93$). Within Basin 3, spectra height decreased throughout the summer in both 2011 ($p < 0.01$, $F_{1,4} = 33.82$) and 2012 ($p < 0.01$, $F_{1,3} = 142.00$). No significant change in height was observed through the sampling season within Basin 1 or 2 in either year (Table 2.5).

Spectra slope varied significantly among the three basins ($p < 0.01$, $F_{2,29} = 8.63$, Figure 2.6), with Basins 1 and 3 having similar slopes that differed from that of Basin 2 (Tukey: Basins 1:2 $p < 0.01$, Basins 1:3 $p = 0.40$, Basins 2:3 $p = 0.03$). Slope did not differ significantly between years in any of the three basins (Basin 1: $p = 0.83$, $F_{1,9} = 0.05$; Basin 2: $p = 0.36$, $F_{1,9} = 0.94$; Basin 3: $p = 0.07$, $F_{1,9} = 4.21$). Slope declined significantly over the summer in Basins 2 and 3 in 2011, but no change was evident in any basin during 2012 (Table 2.5).

Two gillnet sets were completed in 2011 in Basin 1, with 8 species captured. In 2012, a total of seven gillnet sets were completed: three in Basin 1, two in Basin 2, and two in Basin 3. Thirteen species were identified within the catch, with composition differing somewhat among basins (Table 2.6). Walleye *Sander vitreus*, White Sucker *Catostomus commersonii*, and Northern Pike *Esox lucius* were the only species caught in all three of the basins in 2012. In Basin 1 the most abundant species in the gillnet catch was Lake Sturgeon *Acipenser fulvescens*, a species caught only in this basin. Mooneye

Hidon tergisus, Cisco *Coregonus artedi*, Emerald Shiner *Notropis atherinoides*, and Shorthead Redhorse *Moxostoma macrolepidotum* were caught in Basins 1 and 2, but not in Basin 3. Spottail Shiner *Notropis hudsonius* and Yellow Perch *Perca flavescens* were caught in Basin 2 and 3, but not in 2012 in Basin 1, despite their presence in the 2011 catch; this likely reflects differing locations for gillnet sets between the two years (Figure 2.1). In addition, Lake Whitefish *Coregonus clupeaformis* were only captured in Basin 3 in 2012, despite its presence in the 2011 catch in Basin 1.

2.4 Discussion

Our data indicate that the relative abundance and size composition of freshwater fish communities in boreal lakes and reservoirs can be indexed by size spectra determined from hydroacoustic surveys. The acoustically-based size spectra were consistent with the theoretical negative relationship between abundance and body size of aquatic systems (Kerr and Dickie 2001) and spectra derived from catch data (e.g. Rice and Gislason 1996). Our results suggest that hydroacoustics may be an efficient method for constructing size spectra with which to monitor freshwater fish communities.

While there is no way of verifying the absolute parameters of the size spectra determined here, spectra varied consistently in all three basins with respect to known environmental and fish community differences. For example, spectral heights (abundance) were highest in Basin 3, consistent with expected higher water residence time due to greater depths and increased distance from the main channel of flow (Rueda

et al. 2006). These features enable the plankton community to become better developed (Basu and Pick 1996), and may allow for greater fish productivity (Rawson 1952; Mills and Schiavone 1982). This is consistent with local knowledge of productivity and fish abundance in the reservoir (D. Leroux, Conservation and Water Stewardship, Fisheries Branch, Box 4000, Lac du Bonnet, MB, personal communication), and the limited catch data presented here.

Spectral slopes also varied among basins, but not between years. Differences in morphometric characteristics among the three basins may be linked to variations in the size structure of the fish communities and hence spectra parameters, consistent with Emmrich et al. (2011). The steeper spectra slopes in Basin 2 suggest that this area contains a higher proportion of small-bodied fishes than is present in the other basins. This is especially evident at Hay Bay in the northern portion of Basin 2; an interpretation that is consistent with Hay Bay having relatively shallow waters which may limit the presence of larger predatory fishes (Harvey and Stewart 1991), and favour the abundance of smaller fishes. Areas of submerged and emergent vegetation within Hay Bay's relatively large littoral zone can also act as refuge from predation for small and juvenile fishes (Crowder and Cooper 1982; Weaver et al. 1997). In addition, the limited fetch of sheltered bays can impact community structure (Brind'Amour et al. 2005), and may have contributed to observed differences in spectral slopes. The slopes in Basins 1 and 3 were similar to each other, but lower than in Basin 2, reflecting a balance between small bays providing habitat for small-bodied fishes, and deeper areas better suited to larger fishes.

For the most part, spectra slopes within each basin were consistent with gillnet catches, with both indicating an abundance of small fish in all 3 basins (Figure 2.7). The length distribution of fish caught in Basin 1 did not show the expected decrease with increasing length; catch proportions were greatest at the smallest (<10 cm) and largest (>40 cm) size ranges. Given the vagaries of gillnet catches (Argent and Kimmel 2005; Dennerline et al. 2012) this abundance of large fishes is at least consistent with the shallower spectra slopes in Basin 1. Catch proportions in Basins 2 and 3 decreased with increasing length class, consistent with expected size spectra distributions. In Basin 2 the proportion per size class decreased more rapidly than in Basin 3, consistent with the steeper acoustic spectra slopes observed in Basin 2. In addition, large fish were much more abundant in the net catch in Basin 1, consistent with its shallower spectral slopes.

Seasonal changes in size structure are likely to have significant impacts on community dynamics (Rudolf 2012). A decrease in spectra height (i.e. community abundance) was observed over the sampling season in both years of this study, possibly a consequence of mortality over the summer season. Movement between basins may also account for some of observed decrease in spectral height, which was attributable mostly to changes in Basin 3, which has the most abundant plankton community (R. Pollom, Memorial University of Newfoundland, unpublished data). Basin 3 experienced large phytoplankton blooms in both years, possibly leading to oxygen depletion in late summer and fish movement to other basins (Kramer 1987; Vanderploeg et al. 2009). In particular, juvenile Yellow Perch, which were common in Basin 3, may actively avoid areas of low dissolved oxygen (Suthers and Gee 1986). Redistribution of fishes from Basin 3 could

have counteracted decreases in abundance in Basins 1 and 2, explaining why changes in height across the season were only observed in Basin 3. Observed changes in spectra parameters through the season highlight a need for consistency in survey timing to accurately assess changes in fish communities over multiple years.

Assessing spectral slope in conjunction with height may help in interpretations of changes in fish community structure and abundance. For example, in 2011, both height and slope decreased in Basin 3 as the season progressed. This suggests that the decrease in overall abundance within this basin did not occur equally for fish of all sizes, with a relatively greater decrease in smaller fishes. Over a number of years, a decrease in spectra height combined with shallower slopes may reflect poor recruitment or decreases in juvenile fish abundance. Steepening of spectral slope combined with decreases in height can reflect changes resulting from fishing effort that targets large-bodied fishes (Bianchi et al. 2000), whereas a steepening of slope with increases in height suggests stronger incoming recruitment. Monitoring of size spectra over longer time periods may reveal mechanisms underpinning such dynamics in fish communities.

In conclusion, we have shown that hydroacoustic methods can be used to assess the size spectra of a boreal freshwater ecosystem, where fish are broadly distributed and present for the most part as single targets and not dense aggregations. Spectra appear to have potential to capture spatial, inter-annual and seasonal dynamics. Considerations of survey route and timing appear to be vital in order to obtain spectral measurements that can be effectively compared across years. Finally, further research is needed to exploit

these methods, but our data suggest that acoustic size spectra may prove useful as a cost-effective and efficient tool in long term monitoring of lakes and reservoirs.

Table 2.1: Single target and fish track detection parameters used in Echoview 5.0

| Parameter | |
|---|------|
| Single Target Detection | |
| TS threshold (dB) | -55 |
| Pulse length determination level (dB) | 2.00 |
| Minimum normalized pulse length (dB) | 0.35 |
| Maximum normalized pulse length (dB) | 1.50 |
| Maximum beam compensation (dB) | 15.0 |
| Maximum standard deviation of axis angles | 1.20 |
| Track Detection | |
| Minimum number of single targets | 1 |
| Minimum number of pings in track | 1 |
| Maximum gap between single targets | 2 |

Table 2.2: Physical habitat characteristics of the three basins defined within Lac du

Bonnet

| | Basin 1 | Basin 2 | Basin 3 |
|--------------------------------|-----------------|-----------------|-----------------|
| Area (km ²) | 38.7 | 27.0 | 18.4 |
| Mean Depth (m) | 7.4 | 6.6 | 9.9 |
| Max Depth (m) | 25.2 | 12.9 | 15.0 |
| Slope (degrees); mean \pm SD | 1.03 \pm 1.21 | 0.46 \pm 0.94 | 0.69 \pm 1.32 |
| Slope (degrees); maximum | 17.26 | 11.59 | 13.36 |
| Substrate recorded | Mud, Sand, Rock | Mud, Rock | Mud, Rock |

Table 2.3: Length frequency spectra regressions (Model II major axis) for each survey across the whole of Lac du Bonnet ($p < 0.05$ for all regressions). n is the number of points included in the regression, after zero values at the upper end of the size range were excluded.

| Year | Survey | Day of | Volume | | R^2 | Intercept | Slope | Height |
|------|--------|--------|-------------------|---|-------|-----------|-------|--------|
| | | year | sampled (m^3) | n | | | | |
| 2011 | 1 | 207 | 1.42E+05 | 9 | 0.98 | 12.39 | -3.09 | 3.302 |
| | 2 | 214 | 1.51E+05 | 9 | 0.99 | 11.48 | -2.82 | 3.191 |
| | 3 | 215 | 1.49E+05 | 9 | 0.97 | 11.21 | -2.83 | 2.896 |
| | 4 | 229 | 1.45E+05 | 9 | 0.98 | 9.77 | -2.41 | 2.689 |
| | 5 | 230 | 1.53E+05 | 9 | 0.98 | 10.46 | -2.69 | 2.562 |
| | 6 | 239 | 1.41E+05 | 8 | 0.97 | 8.34 | -2.09 | 2.198 |
| 2012 | 7 | 210 | 1.43E+05 | 9 | 0.94 | 10.49 | -2.68 | 2.618 |
| | 8 | 215 | 1.56E+05 | 9 | 0.96 | 10.24 | -2.66 | 2.435 |
| | 9 | 221 | 1.46E+05 | 9 | 0.94 | 9.19 | -2.43 | 2.049 |
| | 10 | 224 | 1.51E+05 | 6 | 0.87 | 11.34 | -3.21 | 1.896 |
| | 11 | 237 | 1.53E+05 | 8 | 0.92 | 9.58 | -2.66 | 1.773 |

Table 2.4: Length frequency spectra regressions (Model II major axis) within each basin, for every survey completed in 2011 and 2012 ($p < 0.05$ for all spectra regressions).

| Year | Survey | Day of | | Volume | | | | | |
|------|--------|--------|-------|-------------------|---|-------|-----------|-------|--------|
| | | year | Basin | sampled (m^3) | n | R^2 | Intercept | Slope | Height |
| 2011 | 1 | 207 | 1 | 6.04E+04 | 9 | 0.92 | 9.69 | -2.44 | 2.54 |
| | | | 2 | 2.68E+04 | 5 | 0.99 | 15.16 | -4.10 | 3.12 |
| | | | 3 | 5.47E+04 | 9 | 0.95 | 13.03 | -3.18 | 3.68 |
| | 2 | 214 | 1 | 6.72E+04 | 9 | 0.91 | 10.78 | -2.72 | 2.78 |
| | | | 2 | 2.72E+04 | 7 | 0.84 | 15.41 | -4.33 | 2.70 |
| | | | 3 | 5.73E+04 | 8 | 0.96 | 11.35 | -2.65 | 3.56 |
| | 3 | 215 | 1 | 6.82E+04 | 9 | 0.96 | 9.77 | -2.41 | 2.70 |
| | | | 2 | 3.04E+04 | 8 | 0.87 | 14.73 | -4.18 | 2.45 |
| | | | 3 | 5.04E+04 | 9 | 0.94 | 12.08 | -3.07 | 3.08 |
| | 4 | 229 | 1 | 6.82E+04 | 8 | 0.92 | 10.46 | -2.72 | 2.47 |
| | | | 2 | 2.37E+04 | 6 | 0.92 | 11.15 | -3.00 | 2.35 |
| | | | 3 | 5.36E+04 | 9 | 0.85 | 10.14 | -2.46 | 2.93 |
| | 5 | 230 | 1 | 7.50E+04 | 8 | 0.97 | 10.74 | -2.81 | 2.48 |
| | | | 2 | 2.79E+04 | 5 | 0.92 | 10.37 | -2.58 | 2.80 |
| | | | 3 | 5.08E+04 | 9 | 0.88 | 10.26 | -2.62 | 2.57 |
| | 6 | 239 | 1 | 6.30E+04 | 8 | 0.94 | 9.05 | -2.27 | 2.39 |

| | | | | | | | | | |
|------|----|-----|---|----------|---|------|-------|-------|------|
| | | | 2 | 2.57E+04 | 8 | 0.97 | 7.73 | -1.82 | 2.38 |
| | | | 3 | 5.21E+04 | 5 | 0.94 | 6.45 | -1.53 | 1.96 |
| 2012 | 7 | 210 | 1 | 6.15E+04 | 9 | 0.92 | 8.57 | -2.11 | 2.38 |
| | | | 2 | 2.95E+04 | 6 | 0.86 | 14.00 | -4.05 | 2.10 |
| | | | 3 | 5.21E+04 | 9 | 0.91 | 11.78 | -3.05 | 2.83 |
| | 8 | 215 | 1 | 7.45E+04 | 9 | 0.96 | 9.42 | -2.42 | 2.31 |
| | | | 2 | 2.92E+04 | 4 | 0.94 | 12.52 | -3.76 | 1.49 |
| | | | 3 | 5.21E+04 | 8 | 0.92 | 11.28 | -2.91 | 2.73 |
| | 9 | 221 | 1 | 6.51E+04 | 9 | 0.87 | 8.16 | -2.12 | 1.95 |
| | | | 2 | 2.67E+04 | 6 | 0.81 | 12.52 | -3.78 | 1.41 |
| | | | 3 | 5.36E+04 | 6 | 0.96 | 10.60 | -2.89 | 2.13 |
| | 10 | 224 | 1 | 6.93E+04 | 6 | 0.84 | 11.55 | -3.36 | 1.67 |
| | | | 2 | 3.16E+04 | 6 | 0.89 | 11.66 | -3.36 | 1.78 |
| | | | 3 | 5.02E+04 | 6 | 0.77 | 12.38 | -3.52 | 2.04 |
| | 11 | 237 | 1 | 6.56E+04 | 8 | 0.85 | 9.30 | -2.55 | 1.82 |
| | | | 2 | 2.94E+04 | 5 | 0.78 | 13.14 | -4.04 | 1.27 |
| | | | 3 | 5.73E+04 | 5 | 0.93 | 11.93 | -3.64 | 1.23 |

Table 2.5: Regression results for change in height and slope for each basin, each year, by day of the year.

| | Year | Basin | p | F | df | R ² | Equation |
|--------|------|-------|--------|--------|-----|----------------|-----------------|
| Height | 2011 | 1 | 0.11 | 4.08 | 1,4 | 0.51 | -0.009x + 4.52 |
| | | 2 | 0.17 | 2.67 | 1,4 | 0.40 | -0.016x + 6.09 |
| | | 3 | <0.01* | 33.82 | 1,4 | 0.89 | -0.050x + 14.02 |
| | 2012 | 1 | 0.12 | 4.80 | 1,3 | 0.62 | -0.024x + 7.26 |
| | | 2 | 0.18 | 2.98 | 1,3 | 0.50 | -0.023x + 6.66 |
| | | 3 | <0.01* | 142.00 | 1,3 | 0.98 | -0.062x + 15.90 |
| Slope | 2011 | 1 | 0.97 | <0.01 | 1,4 | <0.01 | 0.0003x -2.63 |
| | | 2 | <0.01* | 40.05 | 1,4 | 0.91 | 0.081x - 21.23 |
| | | 3 | 0.02* | 13.90 | 1,4 | 0.78 | 0.043x - 12.05 |
| | 2012 | 1 | 0.52 | 0.52 | 1,3 | 0.15 | -0.019x + 1.75 |
| | | 2 | 0.99 | 1.00 | 1,3 | <0.01 | -0.000x - 3.78 |
| | | 3 | 0.11 | 0.11 | 1,3 | 0.62 | -0.027x + 2.83 |

Table 2.6: Gillnet catch for each basin; in 2011 only benthic nets were used, while in 2012 benthic and midwater nets were set, with the exception of set 3, where the pelagic net was not usable.

| Basin | Set ID | Date | Depth (m) | Species | n | Mean Length (mm) | % of set |
|-------|--------|-----------|-----------|--------------------|----|------------------|----------|
| 1 | 1 | 19-Aug-11 | 20 | Lake Sturgeon | 25 | 418 | 86% |
| | | | | Channel Catfish | 2 | 95 | 7% |
| | | | | Lake Whitefish | 1 | 315 | 3% |
| | | | | Sauger | 1 | 320 | 3% |
| 1 | 2 | 19-Aug-11 | 5 | Yellow Perch | 68 | 64 | 69% |
| | | | | Spottail Shiner | 20 | 75 | 20% |
| | | | | Emerald Shiner | 6 | 90 | 6% |
| | | | | Walleye | 2 | 173 | 2% |
| | | | | Sander Sp. | 2 | 85 | 2% |
| | | | | Sauger | 1 | 215 | 1% |
| 1 | 3 | 13-Aug-12 | 16 | Lake Sturgeon | 12 | 472 | 57% |
| | | | | Walleye | 5 | 542 | 24% |
| | | | | White Sucker | 2 | 395 | 9% |
| | | | | Emerald Shiner | 1 | - | 5% |
| | | | | Shorthead Redhorse | 1 | 270 | 5% |
| 1 | 4 | 14-Aug-12 | 15 | Sauger | 1 | 280 | 50% |
| | | | | White Sucker | 1 | 516 | 50% |
| 1 | 5 | 14-Aug-12 | 7 | Sauger | 5 | 237 | 33% |
| | | | | Mooneye | 3 | 250 | 20% |
| | | | | Cisco | 2 | 74 | 13% |
| | | | | Walleye | 2 | 288 | 13% |
| | | | | Emerald Shiner | 1 | 87 | 7% |

| | | | | | | | |
|---|---|-----------|----|--------------------|----|-----|-----|
| | | | | Northern Pike | 1 | 500 | 7% |
| | | | | White Sucker | 1 | 540 | 7% |
| 2 | 6 | 17-Aug-12 | 6 | Walleye | 15 | 180 | 45% |
| | | | | Sander sp. | 6 | 82 | 18% |
| | | | | Spottail Shiner | 5 | 77 | 15% |
| | | | | Yellow Perch | 3 | 116 | 9% |
| | | | | Northern Pike | 2 | 633 | 6% |
| | | | | Mooneye | 1 | 100 | 3% |
| | | | | White Sucker | 1 | 431 | 3% |
| 2 | 7 | 22-Aug-12 | 8 | Emerald Shiner | 23 | 87 | 49% |
| | | | | Walleye | 15 | 289 | 32% |
| | | | | Yellow Perch | 7 | 63 | 15% |
| | | | | Cisco | 1 | 81 | 2% |
| | | | | Shorthead Redhorse | 1 | 364 | 2% |
| 3 | 8 | 17-Aug-12 | 8 | Yellow Perch | 74 | 78 | 72% |
| | | | | Spottail Shiner | 13 | 77 | 13% |
| | | | | Walleye | 9 | 230 | 9% |
| | | | | Lake Whitefish | 4 | 286 | 4% |
| | | | | Northern Pike | 1 | 531 | 1% |
| | | | | White Sucker | 1 | 275 | 1% |
| 3 | 9 | 22-Aug-12 | 14 | Lake Whitefish | 5 | 391 | 31% |
| | | | | Sander sp. | 5 | 93 | 31% |
| | | | | Sauger | 3 | 297 | 19% |
| | | | | White Sucker | 1 | 480 | 6% |
| | | | | Northern Pike | 1 | 670 | 6% |
| | | | | Walleye | 1 | 190 | 6% |

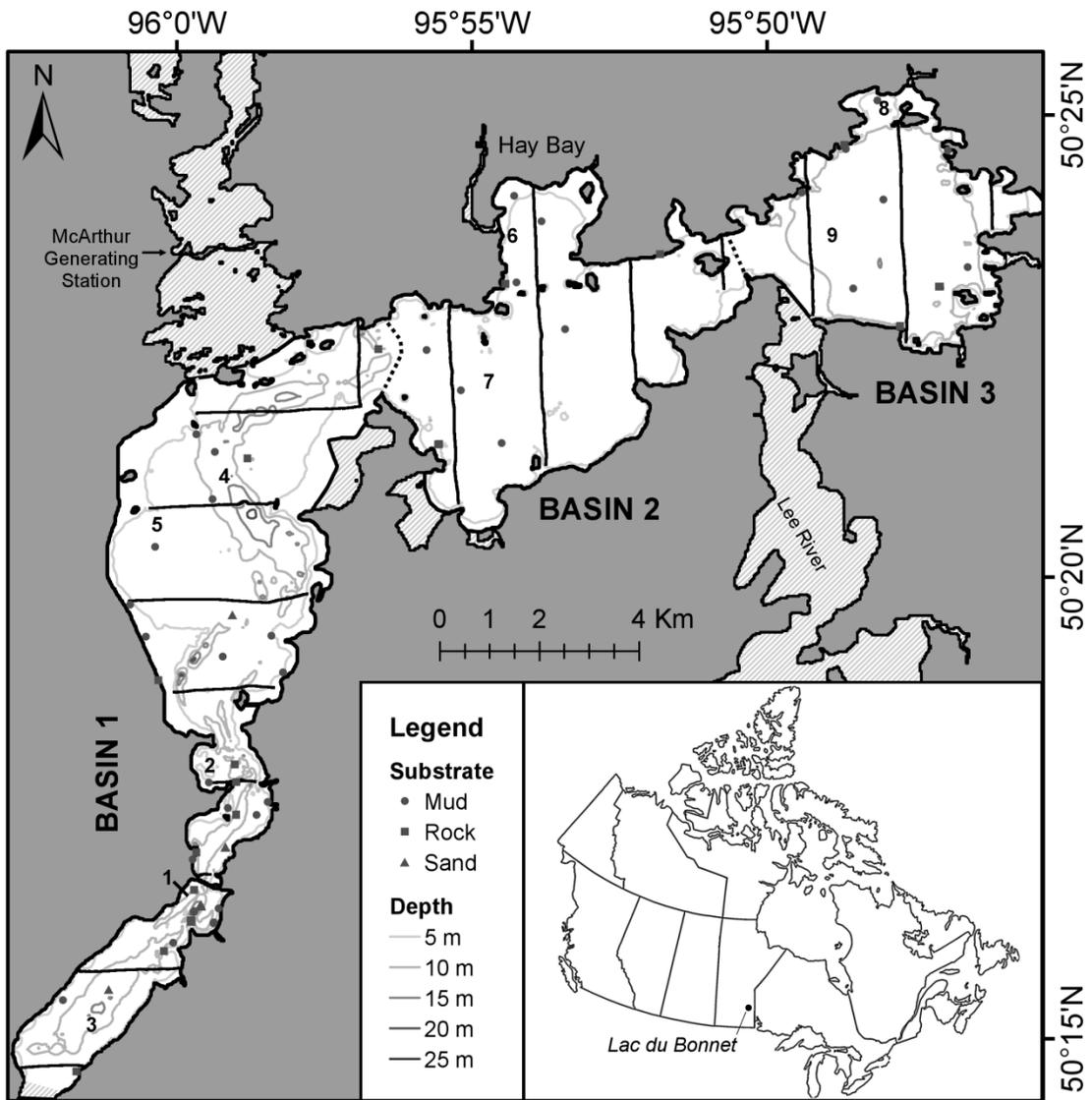


Figure 2.1: Bathymetry of Lac du Bonnet, Manitoba, Canada, as determined from hydroacoustic data. An example survey track is shown here, survey 10, indicated by parallel solid black lines. Dashed lines indicate basin boundaries. Grey-hatched areas were not sampled. Numbers correspond to gillnet set locations; catch details are provided in Table 2.6.

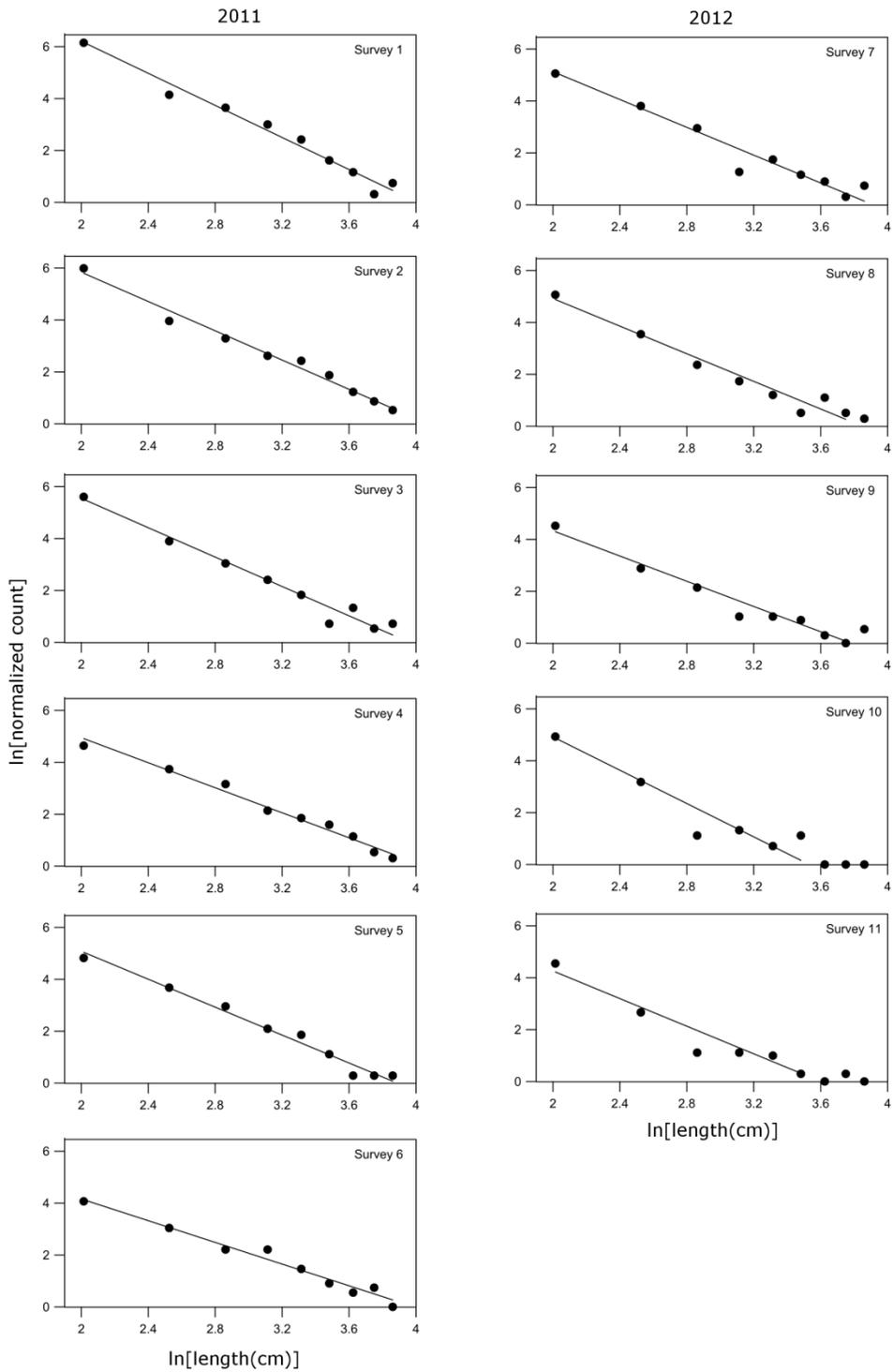


Figure 2.2: Individual spectra regressions for the whole reservoir for each survey completed; surveys 1-6 in 2011, surveys 7-11 in 2012.

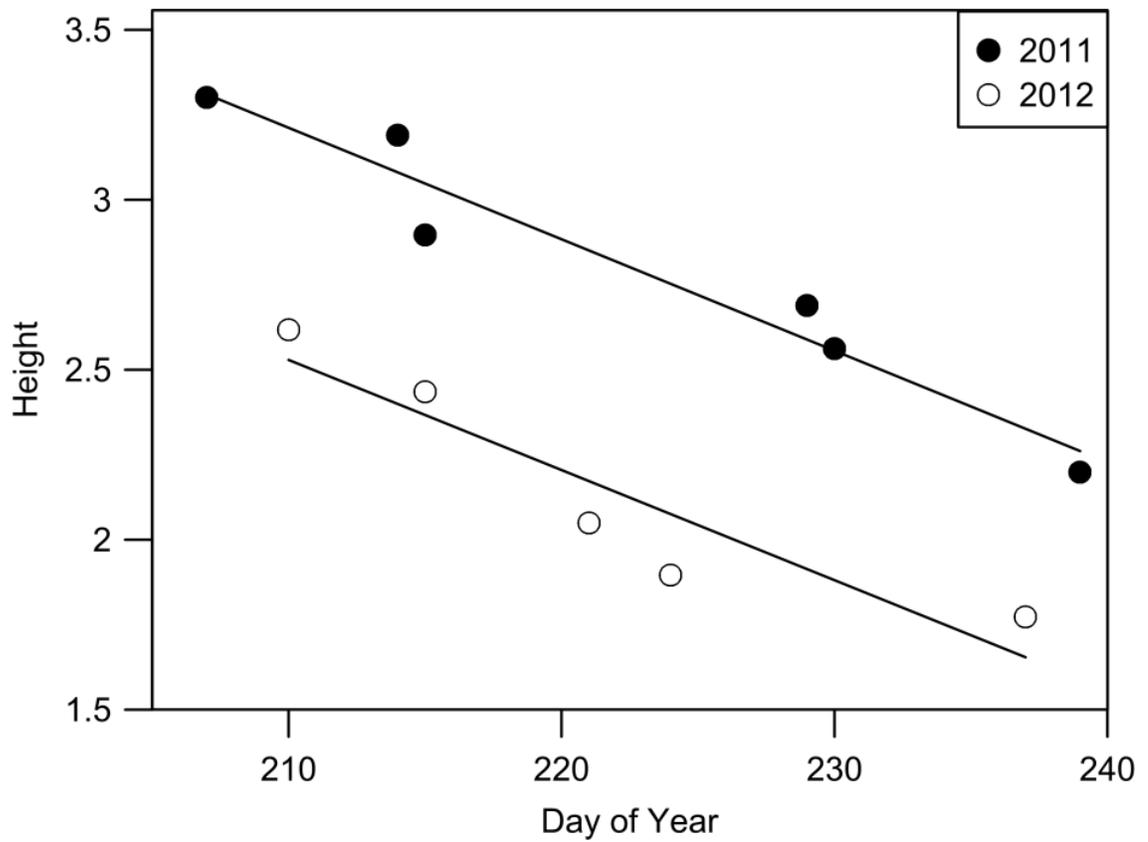


Figure 2.3: Spectra height at Lac du Bonnet was significantly greater in 2011 than 2012 ($P < 0.05$), and decreased across the sampling season in both years (2011: $y = -0.0328x + 10.10$; 2012: $y = -0.0324x + 9.33$).

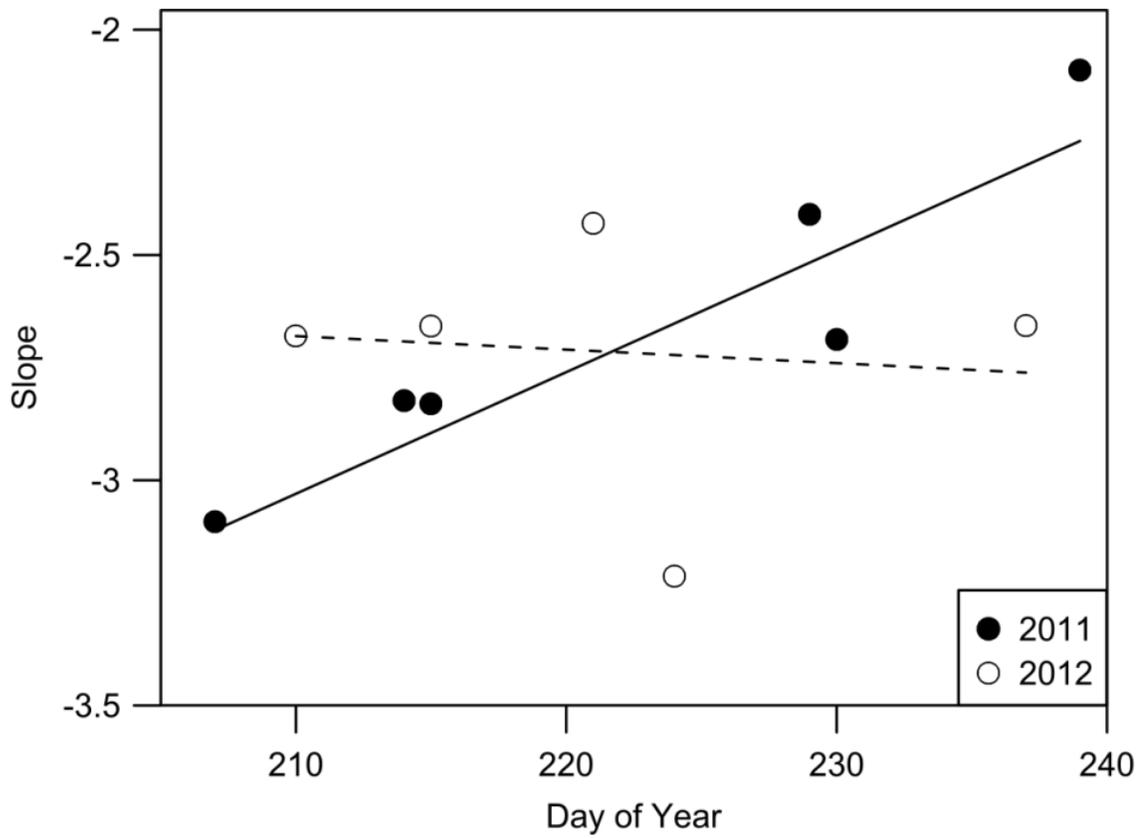


Figure 2.4: There was no significant difference in mean slope between the two years of the study. Spectra slope was only found to change significantly across the season in 2011 ($y = 0.027x - 8.70$).

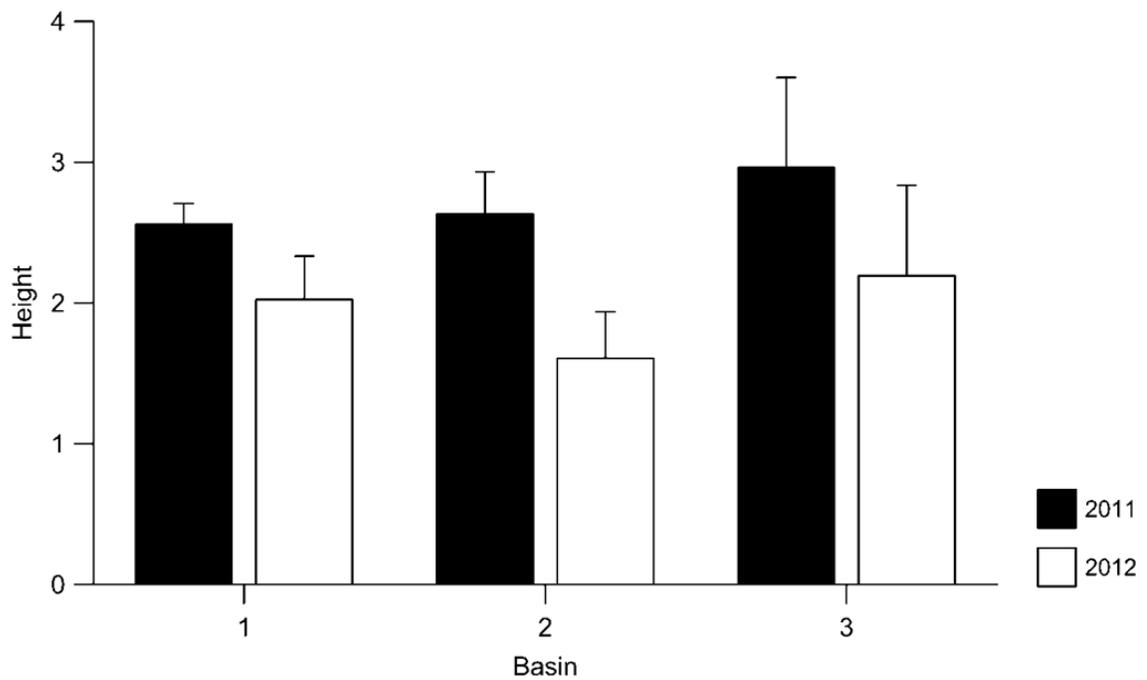


Figure 2.5: Mean spectra height of each basin in 2011 and 2012. Height was significantly greater ($P < 0.05$) in 2011 than 2012 in Basins 1 and 2, and was greatest in the third basin of the lake in both years. Error bars indicate standard deviation.

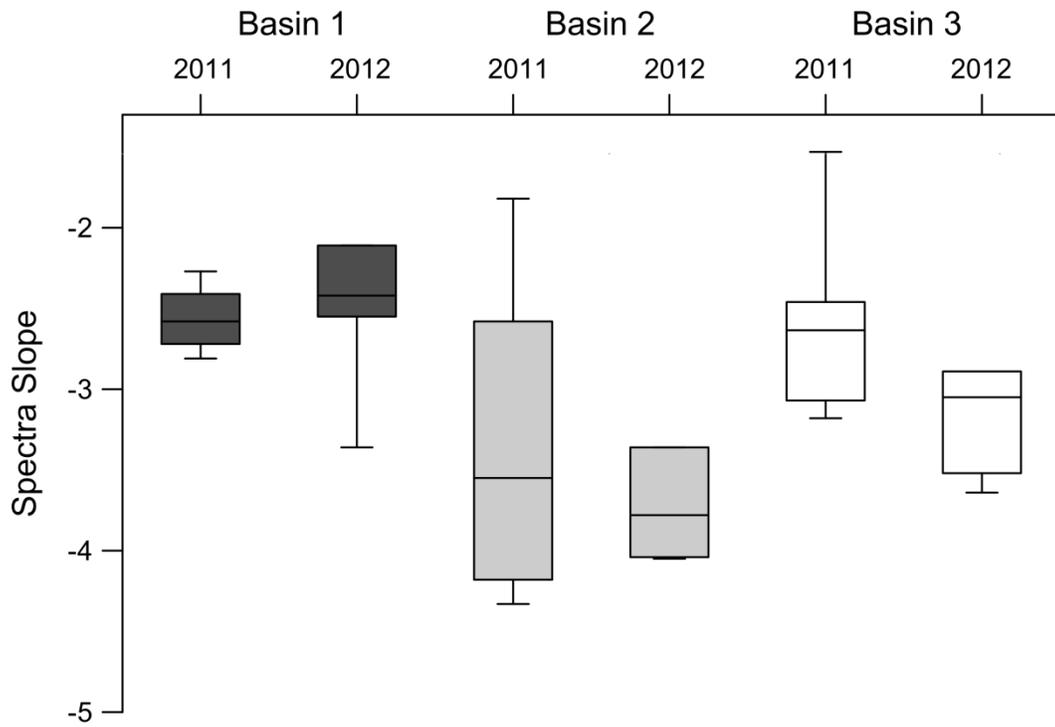


Figure 2.6: Mean spectra slope in Basin 2 was significantly steeper ($P < 0.05$) than that of the other two basins, indicating a fish community in this basin that is more heavily skewed towards smaller fishes. There was no significant difference in mean slope between years for any basin. Box plots show maximum, median, minimum, and first and third quartiles.

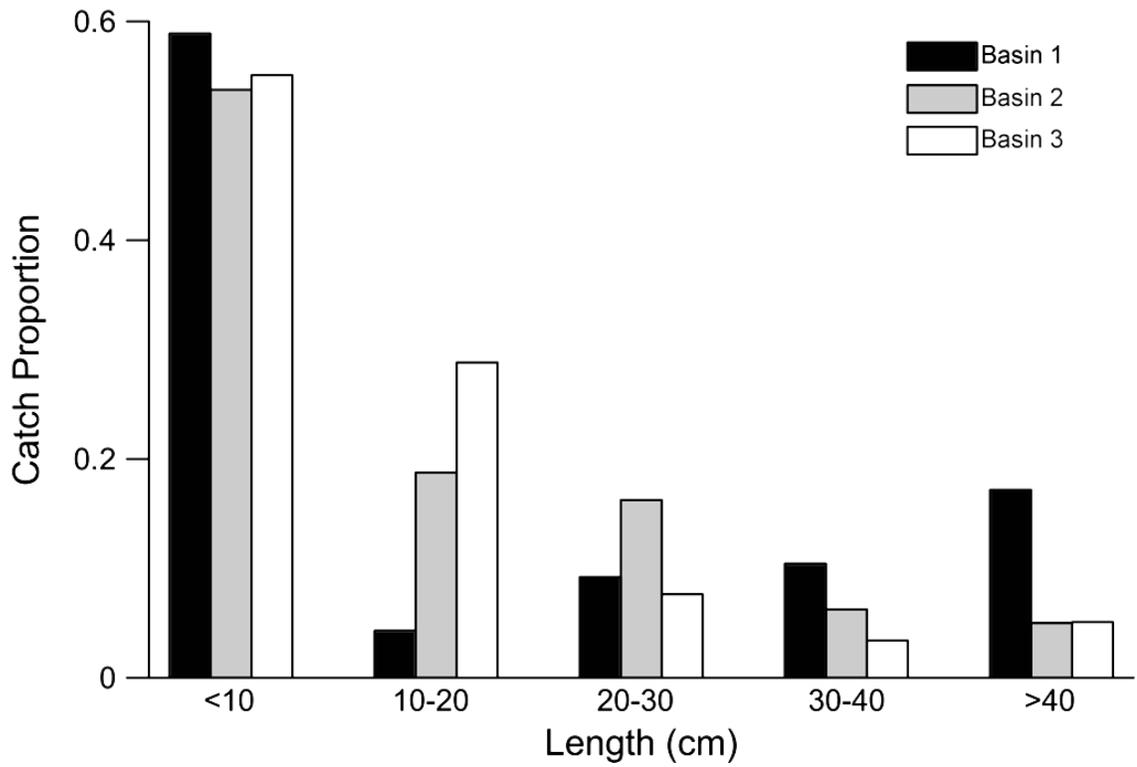


Figure 2.7: Length distribution of gillnet catch among the three basins. Catch was combined for 2011 and 2012 in Basin 1.

SUMMARY

There is an identified need to monitor aquatic ecosystems, and the organisms which inhabit them, especially in areas impacted by anthropogenic activities such as hydropower development. Hydroacoustics allow for non-invasive, efficient, and cost effective assessments of fish communities. In this thesis I developed a method to quantify the impact of vessel avoidance in acoustic surveys in shallow lakes and reservoirs, and investigated the potential for the application of acoustic methods to size-based studies of fish communities.

Potential biases associated with acoustic surveys in freshwater resulting from vessel avoidance reactions are not currently understood, and standard methods for measuring the impact of avoidance acoustic assessments of fish communities are required. The first chapter of this thesis presented results from experimental work quantifying fish avoidance in response to small acoustic survey vessels in shallow waters. Avoidance was determined to differ in magnitude between the two boreal systems studied: Lac du Bonnet, a hydropower reservoir along the Winnipeg River, and Manigotagan and Quesnel lakes, two adjoining lakes in Nopiming Provincial Park, Manitoba. These systems have a similar maximum depth (25-26m), and contain fish communities with similar species compositions. In both systems, the magnitude of avoidance was not significantly influenced by fish depth or survey vessel motoring speed. Our results do not support visual stimuli as a significant source of avoidance, as avoidance was greater in the system with lesser water clarity. Avoidance was not found to differ among fish of different lengths, and fish were not observed moving into deeper

waters in the presence of the motoring vessel. Differences in the magnitude of avoidance between the two systems studied here highlight a need to quantify avoidance *in situ* when beginning acoustic work in any new location or with a different survey platform. The careful consideration of survey vessel choice, combined with *in situ* measures of fish avoidance, should lessen uncertainty associated with acoustic estimates of fish abundance and distribution.

In the second chapter of this thesis, acoustic methods were applied to size spectral theory of aquatic ecosystems. Aquatic ecosystems are shaped by size-based processes, with communities that may be characterized by relationships between abundance and body size. While size spectra have often been derived from catch data, no published studies existed in which community size spectra were based from acoustic data. Here, hydroacoustic surveys of the pelagic zone of Lac du Bonnet were used to assess fish community structure and abundance based on parameters of fish length-frequency spectra. This approach takes advantage of the lesser size-bias of acoustic data as compared to net data, and eliminates species identification problems that occur with acoustic data. The acoustic size spectra were consistent with expected ln-linear relationships of abundance and fish length, as typically derived from both theory and empirically-based catch data. Height of size spectra were used as a measure of fish community abundance, while spectral slopes were representative of the relative abundance of fishes by size. Differences in spectral height indicated that fish community abundance was greater in 2011 than 2012, and that abundance decreased through the summer season. Variation in spectra parameters within the reservoir reflected differences

in physical habitat characteristics which influence fish community structure. Spectra slopes were steepest in areas with shallow waters and large littoral zones (i.e. Basin 2), which provide habitat for small bodied and juvenile fishes. In contrast, basins within the reservoir with deeper waters tended to have shallower spectra slopes.

The acoustic methods for avoidance quantification and size-spectra analysis developed within this thesis may provide useful tools for industry and management agencies involved in routine monitoring of fish communities in areas impacted by hydropower (or more generally, in freshwater ecosystems). However, it is worth noting some limitations associated with these methods. Current relationships between target strength and length have been developed largely based on studies from marine species, with few studies looking at freshwater species (e.g., Frouzova et al. 2005; Kubečka and Duncan 1998). Research aimed at furthering the development of standard equations for freshwater fishes is warranted, and would be useful in attempts to limit potential bias introduced into size spectra from conversions of acoustic size to fish length. In addition, current acoustic technologies and analyses are limited in their ability to resolve individual species in diverse systems such as the ones studied within this thesis. A combination of acoustics, coupled with traditional catch methods such as gill netting, is recommended when attempting to describe the structure and composition of fish communities.

Acoustic surveys enable cost and time-efficient assessment of pelagic communities within lakes or reservoirs, and have minimal impact on fish mortality as compared to widely used netting surveys. In addition, the equipment and protocols for these surveys are easily transferred among different systems, and can therefore be

implemented on a large scale. Parameters of acoustic size spectra may reveal changes in fish community abundance and structure over time, informing managers of potential impacts associated with hydropower or other human interventions. While further research is suggested to better understand the sources of variability associated with these methods, our data suggest that avoidance by fishes can be effectively quantified, and acoustic size spectra may prove useful in the long term ecological monitoring of fish communities in lakes and reservoirs.

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APPENDICES

Appendix 1. Survey vessel, a 5.5m Boston Whaler, and transducer deployment arm. The arm was lowered into the water, with transducer faces submerged to a depth of 0.3-0.5 m.



Appendix 2. Transect characteristics and avoidance indicator values of both drift:drift (DD) and drift:motor (DM) transect pairs at both study sites, Lac du Bonnet (LdB), and Nopiming (Nop).

| Pair Type | Site | Year | Transect length | | Vessel speed | | <i>Depth</i> (m) | \overline{TS} | ΔTS (dB) | FD _i (m) | FD _f (m) | DI | F_i (fish•m ³) | <i>Ac</i> |
|-----------|------|------|-----------------|-------|----------------------|-------|---------------------|-----------------|---------------------|------------------------|------------------------|-------|---------------------------------|-----------|
| | | | (m) | | (m•s ⁻¹) | | | | | | | | | |
| | | | Initial | Final | Initial | Final | | | | | | | | |
| DD | LdB | 2012 | 345 | 404 | 0.17 | 0.17 | 5.8 | -51.7 | -0.7 | 2.2 | 2.1 | 0.02 | 1.56E-02 | 1.68 |
| DD | LdB | 2012 | 465 | 475 | 0.25 | 0.24 | 6.0 | -47.7 | 5.7 | 3.7 | 2.3 | 0.23 | 8.66E-03 | 0.61 |
| DD | LdB | 2012 | 579 | 545 | 0.21 | 0.19 | 6.1 | -47.2 | 0.0 | 3.8 | 4.7 | -0.15 | 6.72E-03 | 1.51 |
| DD | LdB | 2012 | 115 | 80 | 0.06 | 0.04 | 7.1 | -50.6 | 0.5 | 3.6 | 5.1 | -0.21 | 1.08E-02 | 3.83 |
| DD | LdB | 2012 | 155 | 221 | 0.06 | 0.08 | 9.5 | -46.0 | -7.7 | 7.1 | 6.0 | 0.12 | 1.15E-02 | 0.52 |
| DD | LdB | 2012 | 196 | 190 | 0.09 | 0.11 | 10.2 | -39.6 | 14.1 | 2.7 | 3.4 | -0.07 | 1.74E-03 | 2.58 |
| DM | LdB | 2011 | 3217 | 3175 | 0.45 | 1.11 | 10.7 | -34.6 | -7.3 | 7.5 | 7.5 | 0.00 | 2.32E-03 | 0.78 |
| DM | LdB | 2011 | 615 | 621 | 0.28 | 1.14 | 6.8 | -44.7 | -0.8 | 3.8 | 4.3 | -0.07 | 1.66E-02 | 0.70 |
| DM | LdB | 2011 | 1542 | 1660 | 0.22 | 2.10 | 8.8 | -41.3 | 0.9 | 5.7 | 5.8 | -0.01 | 7.34E-03 | 0.41 |
| DM | LdB | 2011 | 744 | 778 | 0.19 | 1.00 | 8.3 | -39.4 | 0.4 | 4.6 | 4.4 | 0.02 | 6.28E-03 | 0.49 |

| | | | | | | | | | | | | | | |
|----|-----|------|------|------|------|------|------|-------|------|------|------|-------|----------|------|
| DM | LdB | 2011 | 833 | 820 | 0.23 | 1.37 | 4.3 | -47.0 | 4.6 | 3.1 | 2.7 | 0.09 | 7.34E-03 | 1.95 |
| DM | LdB | 2011 | 639 | 650 | 0.22 | 1.05 | 4.3 | -51.6 | -1.3 | 3.0 | 3.1 | -0.02 | 2.25E-02 | 0.51 |
| DM | LdB | 2011 | 401 | 394 | 0.13 | 2.74 | 8.5 | -50.4 | 0.4 | 4.2 | 4.1 | 0.01 | 9.19E-02 | 0.53 |
| DM | LdB | 2011 | 165 | 329 | 0.11 | 2.74 | 10.2 | -47.9 | 0.7 | 4.1 | 4.8 | -0.07 | 2.28E-02 | 1.28 |
| DM | LdB | 2011 | 326 | 264 | 0.17 | 2.64 | 6.8 | -47.2 | -0.2 | 4.4 | 3.4 | 0.15 | 1.39E-02 | 0.45 |
| DM | LdB | 2011 | 371 | 377 | 0.11 | 2.61 | 5.5 | -50.2 | 1.6 | 3.7 | 4.4 | -0.13 | 3.54E-02 | 0.35 |
| DM | LdB | 2011 | 550 | 574 | 0.17 | 1.60 | 8.9 | -49.9 | 0.2 | 5.5 | 5.1 | 0.05 | 1.23E-02 | 0.52 |
| DM | LdB | 2011 | 636 | 636 | 0.16 | 3.72 | 8.6 | -47.7 | -0.7 | 4.2 | 5.9 | -0.20 | 5.07E-03 | 1.13 |
| DM | LdB | 2011 | 809 | 782 | 0.36 | 2.46 | 4.6 | -47.8 | 1.7 | 3.4 | 3.9 | -0.11 | 1.38E-02 | 1.42 |
| DM | LdB | 2011 | 425 | 412 | 0.21 | 3.67 | 6.4 | -46.9 | 4.9 | 4.4 | 3.8 | 0.09 | 4.17E-02 | 0.44 |
| DM | LdB | 2011 | 1246 | 1251 | 0.36 | 2.66 | 9.4 | -42.1 | 10.1 | 5.7 | 5.8 | -0.01 | 3.88E-03 | 0.65 |
| DM | Nop | 2011 | 619 | 629 | 0.24 | 1.88 | 7.4 | -46.9 | -1.4 | 2.9 | 4.4 | -0.20 | 6.18E-03 | 0.52 |
| DM | Nop | 2011 | 371 | 382 | 0.23 | 2.13 | 11.9 | -40.9 | -3.6 | 7.8 | 7.8 | 0.00 | 1.23E-02 | 0.76 |
| DM | Nop | 2011 | 538 | 534 | 0.21 | 1.86 | 21.0 | -42.8 | -2.1 | 13.7 | 14.6 | -0.04 | 3.34E-03 | 0.86 |
| DM | Nop | 2011 | 612 | 600 | 0.17 | 1.04 | 19.4 | -39.9 | 0.3 | 12.8 | 13.5 | -0.04 | 6.58E-03 | 0.80 |

| | | | | | | | | | | | | | | |
|----|-----|------|------|------|------|------|------|-------|------|------|------|-------|----------|------|
| DM | Nop | 2011 | 589 | 630 | 0.18 | 1.62 | 11.4 | -42.4 | 6.1 | 5.3 | 8.3 | -0.26 | 9.28E-03 | 0.57 |
| DM | Nop | 2011 | 1229 | 1221 | 0.30 | 2.17 | 11.6 | -36.9 | -0.1 | 8.2 | 8.0 | 0.02 | 1.72E-03 | 1.60 |
| DM | Nop | 2011 | 424 | 435 | 0.24 | 3.15 | 12.7 | -37.5 | 0.5 | 7.2 | 8.7 | -0.12 | 7.78E-04 | 2.31 |
| DM | Nop | 2011 | 565 | 586 | 0.14 | 2.58 | 14.9 | -40.1 | -2.3 | 12.2 | 12.3 | -0.01 | 2.81E-03 | 1.20 |
| DM | Nop | 2011 | 471 | 472 | 0.16 | 3.05 | 9.8 | -40.1 | -1.0 | 7.8 | 9.3 | -0.15 | 6.05E-03 | 1.76 |
| DM | Nop | 2011 | 726 | 734 | 0.29 | 1.69 | 11.0 | -39.5 | -4.7 | 6.8 | 7.6 | -0.07 | 3.09E-03 | 1.55 |
| DM | Nop | 2011 | 778 | 843 | 0.08 | 3.31 | 11.6 | -37.1 | 3.7 | 7.2 | 8.6 | -0.12 | 1.70E-03 | 0.56 |
| DM | Nop | 2011 | 262 | 315 | 0.14 | 2.61 | 8.0 | -34.1 | 4.1 | 5.4 | 5.8 | -0.05 | 4.38E-03 | 0.81 |
| DM | Nop | 2011 | 602 | 588 | 0.36 | 2.09 | 19.9 | -40.0 | -1.9 | 17.4 | 18.6 | -0.06 | 6.10E-03 | 0.81 |
| DM | LdB | 2012 | 562 | 606 | 0.16 | 2.96 | 4.8 | -49.4 | 4.6 | 3.7 | 3.3 | 0.08 | 1.34E-02 | 0.59 |
| DM | LdB | 2012 | 585 | 693 | 0.17 | 2.71 | 4.5 | -51.4 | 1.3 | 3.1 | 3.4 | -0.07 | 3.28E-02 | 0.48 |
| DM | LdB | 2012 | 452 | 625 | 0.15 | 2.12 | 4.8 | -50.9 | 2.7 | 4.2 | 3.9 | 0.06 | 2.65E-02 | 0.39 |
| DM | LdB | 2012 | 542 | 540 | 0.20 | 1.79 | 6.7 | -49.6 | 1.9 | 3.7 | 3.8 | -0.02 | 8.40E-03 | 0.08 |
| DM | LdB | 2012 | 427 | 505 | 0.20 | 1.99 | 5.2 | -49.3 | 0.4 | 4.5 | 3.3 | 0.23 | 9.51E-03 | 1.27 |
| DM | LdB | 2012 | 404 | 528 | 0.17 | 3.16 | 5.7 | -51.6 | 2.3 | 2.1 | 1.9 | 0.04 | 2.63E-02 | 0.12 |

06

| | | | | | | | | | | | | | | |
|----|-----|------|-----|-----|------|------|------|-------|-------|-----|-----|-------|----------|------|
| DM | LdB | 2012 | 385 | 476 | 0.21 | 3.05 | 6.1 | -50.5 | -2.3 | 2.2 | 2.8 | -0.10 | 8.85E-03 | 0.35 |
| DM | LdB | 2012 | 433 | 459 | 0.23 | 3.48 | 5.1 | -51.6 | -5.3 | 4.1 | 4.1 | 0.00 | 1.77E-03 | 0.83 |
| DM | LdB | 2012 | 373 | 380 | 0.13 | 2.53 | 5.6 | -54.3 | -0.1 | 4.8 | 4.2 | 0.11 | 1.45E-03 | 1.14 |
| DM | LdB | 2012 | 475 | 509 | 0.24 | 3.05 | 6.0 | -45.6 | -2.7 | 2.3 | 4.5 | -0.37 | 5.30E-03 | 0.93 |
| DM | LdB | 2012 | 545 | 667 | 0.19 | 3.69 | 6.0 | -47.3 | -0.2 | 4.7 | 4.8 | -0.02 | 1.02E-02 | 0.89 |
| DM | LdB | 2012 | 190 | 193 | 0.11 | 3.71 | 10.3 | -35.6 | NA | 3.4 | NA | NA | 4.50E-03 | 0.00 |
| DM | LdB | 2012 | 115 | 174 | 0.06 | 2.85 | 8.0 | -50.7 | -2.9 | 5.1 | 3.2 | 0.24 | 2.16E-02 | 0.07 |
| DM | LdB | 2012 | 80 | 128 | 0.04 | 1.04 | 6.7 | -47.0 | 19.6* | 3.6 | 6.0 | -0.36 | 2.07E-02 | 0.15 |
| DM | LdB | 2012 | 155 | 179 | 0.06 | 2.67 | 9.9 | -49.4 | 5.8 | 6.0 | 5.9 | 0.01 | 6.89E-03 | 0.30 |
| DM | LdB | 2012 | 221 | 262 | 0.08 | 1.56 | 9.0 | -44.8 | -9.9 | 7.1 | 6.0 | 0.12 | 9.96E-03 | 0.17 |

Appendix 3. GPS tracks for each survey completed at Lac du Bonnet. Surveys 1-6 occurred in 2011, 7-11 in 2012.

