Developing Monitoring Protocols For Aquatic Ecosystems in Forested Regions of Alberta

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EXECUTIVE SUMMARY

This report is part of a larger endeavor to develop a monitoring program capable of detecting changes in the biological diversity of Alberta's forested region. It describes: 1) protocols to monitor changes in aspects of the biological diversity of standing water bodies, and 2) a process to integrate standing and flowing water protocols within the larger monitoring design. Additional information on the monitoring protocols for flowing water bodies related to the: i) development of a functional stream hydrography layer, ii) use of Indian Remote Sensing Satellite (IRSS) imagery to quantify linear disturbances and iii) statistical analyses of stream monitoring data is provided as an appendix to the report.

We recommend that the Alberta Forest Biodiversity Monitoring Program monitor the abundance and diversity of phytoplankton, zooplankton, benthic invertebrate, amphibian, aquatic bird and fish communities within standing water bodies in Alberta's forested region. All six elements perform important ecosystem functions within standing water bodies, are moderately or highly responsive to watershed disturbances and climate change, and the majority comprise diverse assemblages. In order to categorize standing water body types and to develop empirical models that can explain changes in the abundance and diversity of these communities, we identified a suite of environmental variables that should be measured as part of the monitoring program.

A hierarchical decision process was developed to identify the type of habitat identified by the systematic grid. We recommend that the AFBMP use a single line hydrography layer to determine whether the sampling point is located in flowing water habitats. If the sampling point is located within a stream channel, then as many as possible of the flowing water protocols should be applied. In contrast, if the sampling site does not fall within a stream channel, then the site should be identified as non-stream and as many as possible of the terrestrial and standing water protocols would be applied.

While the protocols to monitor the biodiversity of stream ecosystems was completed in 1999, developments over the last 2 years warranted additional discussions

of the: i) development of a functional stream hydrography layer, ii) utility of the Indian Remote Sensing Satellite (IRSS) data and iii) statistical analyses of monitoring data which are appended. We recommend that the AFBMP recognize additional costs required to develop a functional stream hydrography layer from the Provincial Government's Base Features Geographical Information System and that these expenditures are included the process of estimating overall program costs. Second, while data obtained from the IRSS is the best and most up to date information available, its 5.8 m spatial resolution will underestimate: i) some types of watershed disturbances and ii) the overall level of industrial development unless the imagery is updated regularly. Lastly, a diversity of statistical approaches are used to detect changes in the abundance, composition and diversity of aquatic organisms. While there is a level of consistency in how data are analyzed, a number of important issues remain unanswered including how results from some analyses compare with alternative statistical approaches. We recommend that the AFBMP consider creating a working group to examine how data collected by the AFBMP will be analyzed.

SCOPE

The content of this report was identified after discussions between the client (Alberta Sustainable Resource Development, Mr. Harry Stelfox, Contract administrator) and the research team (Alberta Research Council and the University of Alberta). These discussions resulted in the development of a Contract Terms of Reference that were agreed to by the client and the research team.

CHAPTER 1 - INTRODUCTION

1.1 APPROACH AND RATIONALE

The Alberta Forest Biodiversity Monitoring program was established in 1997 with the objective to develop, test, and evaluate a long-term monitoring program capable of detecting changes in the biodiversity of Alberta's forests (Schneider et al. 1999). When implemented, the program would monitor a suite of aquatic and terrestrial plant and animal species to test for changes in biological diversity at multiple scales from sub-basins to landscapes.

The program is guided by eight principles. The program should: 1) support existing commitments (local, national, and provincial) for biodiversity monitoring, 2) use a common, standardized methodology applied across all of Alberta's forested natural regions, 3) monitor both aquatic and terrestrial ecosystems, 4) include elements that represent life forms from diverse taxonomic groups and trophic levels, 5) operate based on a hierarchy of spatial scales, 6) occur in locations having a wide range of land use histories, including those with limited human influence, 7) include descriptors that may explain changes in the abundance and distribution of species (i.e., environmental variables), 8) be transparent in development and implementation, and 9) where possible, use standard monitoring techniques (Schneider et al. 1999, Scrimgeour and Kendall 1999).

In identifying elements, sampling protocols and related environmental variables that should be measured to detect changes in the biological diversity of standing water bodies, we assumed that sampling would be: i) completed by a field crew of 2-3 people, ii) conducted at a given site within a maximum of 2-3 days and, if possible be iii) integrated with sampling of terrestrial elements.

1.2 OBJECTIVES

The objectives of this report were three-fold. First, we describe protocols required to monitor changes in the biological diversity of standing water bodies in

forested regions of Alberta by monitoring six aquatic elements of phytoplankton, zooplankton, benthic invertebrate, amphibian, aquatic bird and fish communities. We also: i) identify a suite of environmental variables (i.e., water body and watershed descriptors) that should be measured in addition to the six elements and ii) discuss some of the logistical difficulties that will likely arise when sampling standing water bodies. Based on the Contract terms of reference agreed upon by the client, identification of the monitored elements is not accompanied with an exhaustive discussion of the selection process through which elements were selected compared with those that were not. Second, we describe an overall sampling approach that can be used to discriminate between: 1) flowing water systems and 2) standing water bodies and terrestrial habitats. Third, we provide an appendix containing additional information on the monitoring protocols developed for flowing water bodies (Scrimgeour and Kendall 1999). Specifically, we address issues related to: i) development of a functional stream hydrography layer, ii) the use of Indian Remote Sensing Satellite (IRSS) imagery to quantify linear disturbances and iii) statistical considerations in the analysis of stream monitoring data.

CHAPTER 2 – MONITORING PROTOCOLS FOR STANDING WATER BODIES

2.1 INTRODUCTION AND SELECTION OF AQUATIC ELEMENTS

Based on our review of all potential monitoring elements, we identified six element communities (i.e., phytoplankton, zooplankton, benthic macroinvertebrate, amphibians, fish and aquatic birds) that should be monitored to describe aspects of the biological diversity of standing water bodies in forested regions of Alberta. All six elements fulfill the criteria identified by the AFBMP technical committee.

In addition to describing sampling methods for each element, we present: i) a suite of environmental variables that could be used to develop predictive models explaining changes in the abundance, composition and diversity of the six selected elements and ii) a hypothetical layout to assist with the selection of sampling points within individual water bodies.

2.2 DEFINING THE SAMPLING PLOT

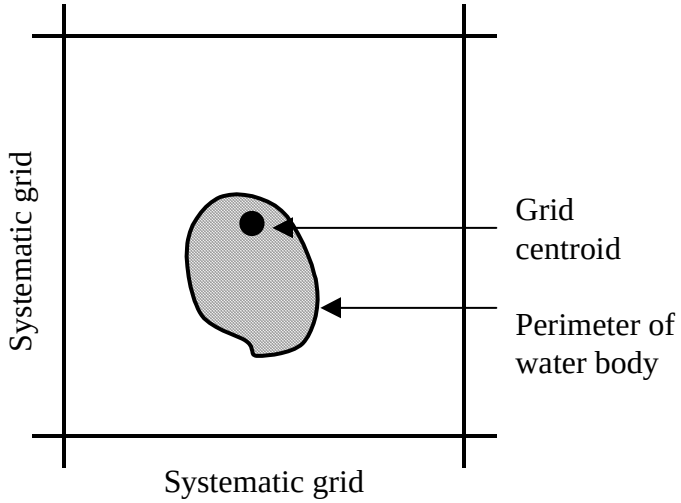
Quantifying lentic communities is typically completed using the entire water body as the sampling unit (e.g., Schindler 1990, Planas et al. 2000, Prepas et al. 2001) and in many cases requires substantial amounts of effort to quantify communities inhabiting large water bodies. We recommend that the AFBMP adopt a fixed plot size of 75 ha or less when sampling standing water bodies in forested regions of Alberta. A 75 ha maximum plot size is beneficial because: i) all six aquatic elements can be sampled within such an area over a three day period, ii) it does not depart from a systematic approach to select sampling sites and iii) it does not require a substantial financial investment as would be required when sampling large (e.g., > 250 ha) water bodies, such as the large wetland complexes in the boreal plains, and to a lesser extent, large lakes (Mitchell and Prepas 1990).

We recommend that the AFBMP adopt a maximum plot size of 75 has when sampling standing water bodies and describe how such a plot size could be applied in the

three scenarios where the water body size is: i) less than 75 ha, ii) equals 75 ha, and iii) exceeds 75 ha (Fig. 1). Where the water body is less than, or equal to 75 ha, we suggest that the water body area defines the sampling plot and that the entire water body is sampled (Fig 1A). In contrast, when the watershed body exceeds 75 ha, a 75 ha area is established within the water body to delimit the sample plot. In the latter case, the 75 ha plot could be identified by using existing Geographic Information Systems land cover data bases to identify the perimeter of the 75 ha plot.

Use of a maximum 75 ha plot requires the development of a set of decision rules to determine where the plot should be established when the water body size exceeds 75 ha. While several alternatives exist, the 75 ha plot could be delineated by dividing the water body using a line that extends in a west-east direction and using a GIS query to create the 75 ha plot by moving the line in a north-south direction until it encompasses 75 ha that is relatively well distributed around the sampling centroid (Fig. 2). Lastly, in some cases, the entire 75 ha plot will fall within large lakes or wetland complexes. In these cases, the sample plot will not encompass margins of standing water bodies. Protocols to deal with this case are discussed in sections 2.3.1 to 2.3.6.

A) Small standing water body (= 75 ha)



B) Large standing water body (>> 75 ha)

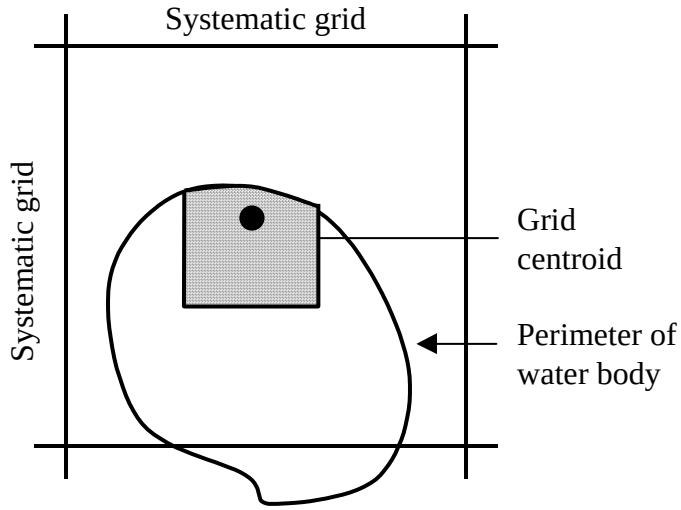
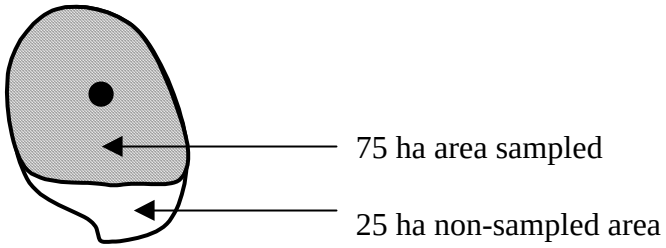


Figure 1 Comparison of hypothetical sample plots in a small (A) and a large (B) standing water body. Where the water body is 75 ha or less, we recommend that the entire water body should comprise the sampling plot, whereas a 75 ha plot within the water body should be used when the water body exceeds 75 ha. Depending upon habitat availability, as many as possible aquatic elements should be sampled within, or adjacent to the plot.

A) Small standing water body (100 ha)



B) Larger standing water body (200 ha)

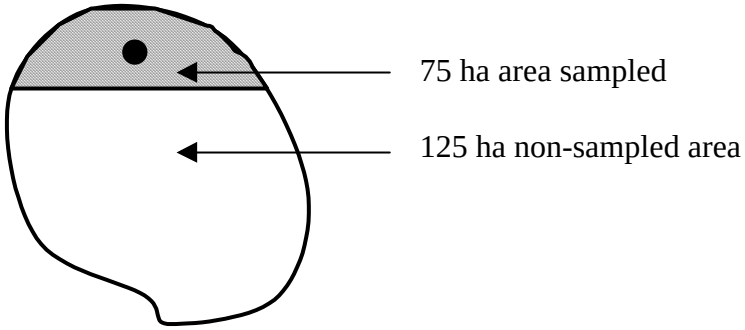


Figure 2 Comparison of hypothetical 75 ha sample plots in a small (100 ha) and a large (200 ha) standing water body. Filled circle represents the centroid of the systematic grid.

2.3 SELECTED ELEMENTS TO MONITOR ASPECTS OF THE BIOLOGICAL DIVERSITY OF STANDING WATER BODIES

2.3.1 Phytoplankton

Community Structure

Phytoplankton, the algae that live in open water, is perhaps the most well-studied component of lentic systems. Phytoplankton communities typically include cyanobacteria, a group of photosynthetic bacteria (previously known as blue-green algae), green algae, golden-brown algae, yellow-green algae, diatoms, cryptomonads, dinoflagellates, and euglenoids. Although phytoplankton communities are influenced by many factors such as light, temperature, pH, turbulence, competition, and selective grazing, certain recurrent patterns in biomass, species diversity and community composition are correlated with nutrient concentrations (LaZerte and Watson 1985, Smith 1990, Duarte et al. 1992, Watson et al. 1997, Fig. 3). For instance, nutrient-poor systems have low phytoplankton biomass and are typically dominated by green algae or a combination of golden-brown algae and cryptomonads (Schindler 1987, 1990, Duarte et al. 1992). In contrast, phytoplankton communities in eutrophic, or nutrient-rich systems, often shift toward dominance by very few taxa, usually cyanobacteria (Fig. 3), and sustain very high algal biomass (e.g., Prepas et al. 2001). Moderate enrichment tends to result in increased taxonomic diversity and more variable levels of biomass (Sommer et al. 1986). Changes in the nutrient status of a water body due to disturbances such as forestry and fire can alter phytoplankton community structure (Planas et al. 2000).

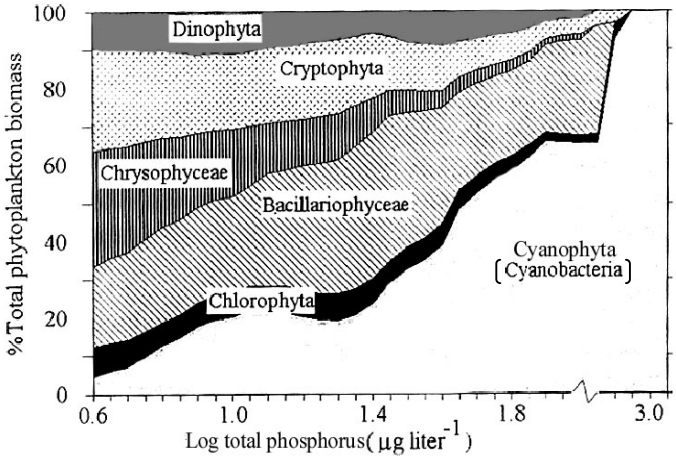


Figure 3 Changes in temperate zone phytoplankton community structure, excluding the picophytoplankton, with changes in water column total phosphorus in primarily stratified lakes (modified from Watson et al. 1997 with permission).

Generally, standing water bodies within Alberta's boreal forest are naturally high in phosphorus with a correspondingly high proportion of cyanobacteria species (e.g., Trimbee and Prepas 1987). Even in naturally eutrophic systems, shifts in nutrient status can lead to further proliferation of cyanobacteria (Prepas et al. 2001). Thus, species composition of phytoplankton will likely be an indicator of disturbance if such disturbances result in elevated nutrient loadings (e.g., Harig and Bain 1998).

Sampling Alternatives

Typical sampling devices for phytoplankton consist of bottles, tubes and pumps; nets are not recommended because even fine mesh nets are not sufficient for capturing small phytoplankton (American Public Health Association 1989). Bottles provide point samples from the water column, whereas tubes provide integrated samples from the surface to a preselected depth. Bottles are best for examining small-scale spatial variation of organisms within lakes, but less so for documenting changes through time and differences among regions. Integrated samples provide a more thorough view of phytoplankton communities than do single depth samples taken using bottles.

Recommended Protocols

Sampling Gear

We recommend that the AFBMP monitor phytoplankton communities by taking vertically integrated samples using tubes. With the exception of very shallow (water depth < 50 cm) water (Discussion to follow), tubing can be used in a wide range of depths, and thus the same method can be used in the majority of water bodies encountered. We recommend the standard technique consisting of clear polyethylene tubing with a 2.54-cm inner diameter, one-way valve at the bottom, and an attached lead weight (e.g., Campbell et al. 1998, Prepas et al. 2001). The tube is extended from the water surface to the bottom of the euphotic zone (>1% of ambient surface light) and retrieved back to the water surface. In Alberta, the bottom of the euphotic zone can be estimated as twice the secchi depth, although we suggest that it be determined using a light meter. In Alberta, the deepest euphotic zone will likely be ~12 m (Mitchell and Prepas 1990), and we recommend the construction of both a short (4 m) and a long

(15 m) tube for sampling. Our experience (e.g., Scrimgeour et al. 2000) is that the shorter tube is considerably easier to use than the long tube and should be appropriate for the majority of standing water bodies.

Many fen and bog complexes contain substantial areas of shallow water (e.g., 5 to 50 cm) interspersed with emergent vegetation and areas that extend above the water surface. In these situations, the use of the sampling tube may not be overly useful and in the situations we recommend that phytoplankton be collected using the small tube sampler, held horizontally, as described for zooplankton (See Section 2.3.2).

Sample Design

Design considerations - As is typical of aquatic organisms, phytoplankton exhibit considerable spatial and temporal variation in biomass and species composition. Communities follow general patterns of seasonal succession, and spatial variation is a function of many interacting factors. Typically, samples for phytoplankton are taken in the pelagic zone of lakes. Often, only one sample is collected from the point of maximum depth to the water surface. However, because phytoplankton distribution is patchy, samples should be collected from more than one location.

Recommended sampling design - We recommend taking a total of six samples with three phytoplankton samples from the euphotic zone from pelagic, or "deep water" sites. One of these stations is located at the deepest point of the basin, and the remaining two are randomly selected within the pelagic zone, or, in water bodies <3m deep, within a "deep" water zone (See section 2.3.2). Limiting sampling to one habitat type or area within the water body will improve precision by removing some of the variance associated with the different habitat types, but in many cases, shallow water bodies will not have an extensive pelagic zone. We also recommend taking an additional 3 samples from the littoral zone. While two pelagic samples appear to be adequate for characterizing phytoplankton community composition in Alberta lakes (e.g., Prepas et al. 2001) the degree of inter-sample variance is not well documented. Thus, we recommend that the AFBMP collect three samples from the euphotic zone and three from the littoral zone. These samples should be processed to better describe relations between sampling

effort and variance of density, community composition and richness estimates and that the amount of sampling effort should be increased or decreased based on the results of these comparisons. Because all samples from a water body are taken from the euphotic zone, issues of depth stratification and weighting of samples inherent with sampling zooplankton (Section 2.3.2) and fish (See 2.3.5) are not as critical.

Table 1 Overview of sampling protocols to monitor phytoplankton communities from standing water bodies.

Attribute	**Sampling Details**
1. Sampler	1 short (4 m) and 1 long (15 m) tube sampler
2. Habitats	Pelagic (or "deep" water) and littoral zones
3. Number of stations	Six stations: 3 in the pelagic zone and 3 from the littoral zone
4. Selection of stations:	
a) Water bodies >3m deep	Pelagic zone: 2 randomly selected, 1 at maximum water depth Littoral zone: three randomly selected
b) Water bodies 1-3 m	Pelagic zone: 1 randomly selected at depth >2 m, 1 at maximum water depth Littoral zone: three randomly selected
c) Water bodies <1m deep	5 randomly selected, 1 at maximum water depth
5. Samples	100 mL of water collected at each station

Sample Preservation

Phytoplankton samples should be preserved in Lugol's solution and stored in 100 mL amber glass or opaque polyethylene bottles in the dark. The final concentration of Lugol's solution in the sample should be 1% (1:100) (Wetzel and Likens 1991).

Sample Processing

Considerations - As with zooplankton, it is necessary to take subsamples of the phytoplankton for identification and counting. The subsamples are generally further "subsampled" by identifying and counting organisms in only a predetermined number of "fields" within the counting chamber. There are a variety of considerations involved in subsampling and estimating numbers and biomass of phytoplankton. Determining the number of fields to count, algal cells to count, and cell volume are a few of the decisions that need to be made. The number of fields examined and the number of algal cells counted will vary depending on the density of organisms in the chamber. Taxa occurring at higher densities are often counted over fewer fields than rarer taxa (Venrick 1978). As long as data are expressed as the number of organisms per field, all counts can be standardized to the original sample volume. Counting 100 organisms for each important taxa is recommended for achieving a 95% confidence interval within ± 20% of the mean (Lund et al. 1958). We suggest that phytoplankton samples taken by the AFBMP be processed initially to describe relationships between mean and variance estimates of density, biomass and community composition resulting from different sub-sampling volumes and numbers of fields counted. Prepas et al. (2001) used a subsampling protocol for samples from boreal Alberta lakes that may prove appropriate for the AFBMP. For detailed instructions on subsampling, counting and biomass estimates, see Sournia (1978), Venrick (1978), Wetzel and Likens (1991) and references therein. Taxonomic keys for the identification of phytoplankton and biovolume estimates are provided by Lewis (1976), Prescott (1978), Wetzel (1983), and Dillard (1999).

Recommended protocols - Phytoplankton samples can be counted using several different devices. Typical methods include the sedimentation technique and use of an inverted microscope (i.e., the Utermöhl method, Utermöhl 1958), modified sedimentation chambers for use with a compound microscope, a variety of counting cells such as the Sedwick-Rafter and Palmer-Maloney cell, filtration onto membrane filters and centrifugation (Wetzel and Likens 1991). The standard method for counting phytoplankton, and one that is widely considered the best technique (Wetzel and Likens

1991) is the Utermöhl method (Utermöhl 1958) and its use is recommended for the AFBMP.

The Utermöhl method uses a chamber into which a subsample of 1-100 ml is placed and left to sediment out. Phytoplankton are then counted and identified under an inverted microscope. The variance to mean ratio of several subsample counts should be tested for conformation to a random distribution with the chi-square test (see Prepas 1984).

The basics of the Utermöhl method are as follows (Utermöhl 1958, Lund et al. 1958, Wetzel and Likens 1991):

1. Mix the sample by inverting the sample bottle very gently, and then pour a portion into the chamber so that the water from the sample beads upward above the top edge of the chamber.
2. Slide the chamber cap across the top, removing any excess water and enclosing a sample of exact volume without air bubbles. Chambers are available in 1, 5, 10, 25, 50 and 100 ml sizes. Use smaller volume chambers for samples with higher plankton densities.
3. Allow the organisms to sediment out according to the following guidelines: time in hours = height of sedimentation chamber in cm x 3.
4. Identify and enumerate the phytoplankton using an inverted microscope. Each microscope must be calibrated against a known scale using a Whipple ocular micrometer. Detailed instructions can be found in American Public Health Association standard methods (1989).

We propose that the target measures (i.e.,dependent variables) group for phytoplankton be richness, diversity, relative abundance or biomass of dominant taxonomic groups, and that identification be made to the genus level. It is clear from the literature that examining both taxonomic composition and dominance (e.g., % biomass) in combination is the important measure, not taxonomic richness alone (LaZerte and Watson 1985, Eloranta 1986, Hecky and Kilham 1988). Because we recommend focusing on shifts in community structure rather than on a direct measure of species

richness in relation to stressors, identification to the genus level should provide an appropriate level of resolution while reducing laboratory processing time. In addition, important aggregate groups (e.g., edible and non-edible algae, nitrogen-fixing algae) can be assessed from this level of taxonomic identification.

Costs

Table 2 Cost estimates for equipment to monitor phytoplankton communities in standing water bodies. Costs are in Canadian dollars.

Sampling equipment	Cost
Short tube sampler	45[a]
Long tube sampler	95[b]
100 mL glass bottle (cost per bottle)	0.6
Preservative (cost per sample)	0.2
Sample identification	
a) High taxonomic (predominantly to genus and species) resolution (cost per sample)	95
b) Low taxonomic (predominantly family and genus) resolution (cost per sample)	65

[a] 4 m length, [b] 15 m length.

Sampling platform		Cost
1) Road accessible		
Small water body		
a) Shallow	14.5' canoe[a]	1120
	2 horsepower, 4 stroke motor	1000
b) Deep	10' aluminum boat	1000
	2 horsepower, 4 stroke motor	1700
Large water body	12' aluminum boat	1300
	5 horsepower, 4 stroke motor	1700
2) Remote access		

a) Small water body	10' inflatable boat	2000
	2 horsepower, 4 stroke motor	1700
b) Large water body	12' inflatable boat	2500
	5 horsepower, 4 stroke motor	1700

Safety equipment	Sampling Platform	Cost
Life jackets	3	200
Flares	2	100
First aid safety kit	1	300
Flashlight	3	90

2.3.2 Zooplankton

Community Structure

Organisms that comprise zooplankton range in size from a few microns up to about 15 mm. Typically rotifers and crustaceans (primarily copepods and cladocerans) dominate the zooplankton, along with some insect larvae (Chaoboridae) and aquatic mites. In temperate lakes, zooplankton communities can consist of 50-100 species (Morgan 1980, Kobayashi et al. 1998) and as many as 200 genera may be represented (excluding protists) across waterbodies of Alberta (Clifford 1991). Assemblages in undisturbed systems are often species-rich, and dominated by large-bodied taxa (e.g., Locke and Sprules 1994, Harig and Bain 1998). However, zooplankton communities are highly sensitive to pH, pollutants, nutrient levels and changes in competition and predation that typically reduce richness and promote dominance by small-bodied taxa (See Harig and Bain (1998) and Stemberger and Miller (1998) and references therein). Thus, collection of zooplankton samples can provide several metrics including species richness, dominance of large-bodied zooplankton, and zooplankton biomass (Harig and Bain 1998) that can be used to evaluate the effects of anthropogenic disturbances on the biological diversity in forested regions of Alberta.

Sampling Alternatives

Traditional methods of sampling zooplankton include use of vertical and horizontal tow plankton nets and volumetric samplers (e.g., bottles, traps, tubes and pumps; de Bernardi 1984). Several advanced techniques exist including use of acoustic devices, optical plankton counters, video systems, and laser fluorosensors. However these techniques either do not provide taxonomic information or are not suitable for surveys (reviewed in Pinel-Alloul 1995). Bottles and traps sample a relatively small volume of water and thus require large numbers of samples to measure community composition. They are most effective when taking point samples (i.e., depth specific samples), but less effective for estimating overall species richness. Nets are by far the most commonly employed sampling equipment. They filter organisms directly from the water column, and can be used to take integrated samples across a variety of depths.

Each piece of equipment differs in its ability to capture particular species and taxa differing in size and behaviour. For example, the smallest organisms captured will be a function of the mesh size used, and motile animals may sense and avoid a slow moving apparatus. Thus, to produce comparable samples among water bodies, it is important to use consistent methods and equipment. Further, unless sampling effort is extremely high, richness is typically underestimated and results should be considered an index, rather than an absolute measure of total richness, as is the case for all community elements measured.

Traps and bottles can be used to construct an integrated sample by sampling a range of depths, and then creating a composite sample weighted by volume of depth strata. This is a well-proven method, however, since only a small portion of the water column is sampled at a time, constructing an integrated sample is time consuming. This method may be impractical when sampling designs include multiple sites within a lake. Although vertical net hauls are the most common method of collecting samples, they suffer from net clogging, variable filtering efficiencies, and bow-wave avoidance by mobile species, which can result in their under representation. Tubes sample a known volume of water, do not clog, and produce less of a hydrodynamic wave. Further, tubes are superior to nets when sampling littoral areas because they can be used in shallow waters containing dense vegetation. Studies have rated tube samplers to be as much as

50% more efficient than nets (Graves and Morrow 1988, DeVries and Stein 1991, Campbell et al. 1998) and similar in efficiency to Schindler-Patalas traps (DeVries and Stein 1991). A detailed account of the strengths and weaknesses of various sampling equipment is provided by de Bernardi (1984).

Recommended Protocols

Sampling Gear

We recommend that the AFBMP monitor zooplankton communities using tube samplers (DeVries and Stein 1991, Knoechel and Campbell 1992). These integrated samplers are appropriate for measures of whole-lake richness, as opposed to point-samplers, which are used when detailed information on vertical distribution within a water body is desired.

Two different tube samplers will be necessary to sample deep and shallow water habitats. For deep, macrophyte-free water, we recommend the simple and inexpensive tube sampler described by Knoechel and Campbell (1992). It consists of a collapsible 10.2 cm diameter tube, 15 m in length, fitted with a self-actuating flap mechanism and plankton net. In shallow and vegetated areas, where the above sampler may not work, a clear plastic tube will be sufficient. DeVries and Stein (1991) use a tube 0.16 cm thick with a 7.3 cm internal diameter. A cap with a hole (0.18 cm inner diameter) is fitted to the top, and a stopper inserted into the hole once the tube is fully submersed. This tube need only be as long as the water being sampled is deep, thus a 3 m length should be appropriate in most cases. Mesh sizes used to sample zooplankton can range from as small as 35 μm for studies that include microzooplankton (rotifers; e.g., Basu et al. 2000), and up to 153 μm (e.g., Harig and Bain 1998) for larger scale projects. As a practical compromise, we recommend a mesh size of 64 μm for both samplers (Carter et al. 1986, Field and Prepas 1997). The two sampling devices can be calibrated by taking samples using each device at the same location and comparing capture efficiencies. Smiley and Tessier (1998) used the Knoechel & Campbell (1992) tube sampler in combination with a smaller tube for vegetated areas, and found no difference in their capture efficiencies.

Sample Design

Design considerations - One challenge when sampling zooplankton is dealing with high spatial and temporal variability within water bodies. Zooplankton assemblages display seasonal shifts in population abundance, inter-annual changes in species composition, and differing species turnover rates. Therefore, a consistent time frame for sampling and taking an appropriate number of samples from each water body will be important in minimizing these sources of variance.

Arnott et al. (1998) provide useful information on how the number of zooplankton samples taken within a lake affects cumulative species richness measures using data from lakes in eastern Canada. Many studies and monitoring programs base richness estimates on single samples from individual lakes. Arnott et al. (1998) found that for crustaceans, single samples detected only 50% of the annual species pool and 33% of the total (i.e., over several years) estimated species pool. Increasing the temporal or spatial extent of sampling improves richness estimates (Fig. 4), but the species detected will differ in each case. In their study, the deepest point in the lake yielded the highest richness, likely because it sampled a greater volume of water and hypolimnetic habitat.

Recommended sampling design - If the AFBMP allows only one sampling episode, as is the current scenario, we recommend sampling at five stations within a water body to improve richness estimates. To achieve the best precision in estimating characteristics of the community, sampling should be stratified by habitat. In lakes, the simplest division is into littoral and pelagic habitats. Taking multiple samples from each habitat type will help to overcome the problem of sampling heterogeneously distributed organisms. By definition, the littoral zone extends from shore to deeper areas of the water body where light levels cannot support rooted macrophytes. The pelagic zone of a

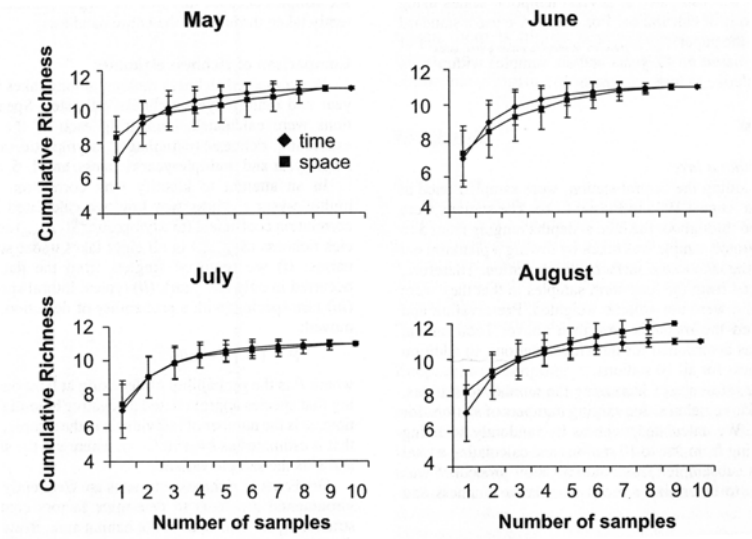


Figure 4 Mean cumulative richness of crustacean zooplankton for two sampling regimes in Plastic Lake. Spatial samples (squares) were taken in May, June, July and August at 9 or 10 locations. Both richness estimates were obtained by repeatedly subsampling the database for different combinations of the same numbers of samples (After Arnott et al. 1998; with permission).

lake is the open water area beyond the littoral zone and in Alberta may comprise areas >3 m depth. Many water bodies in boreal Alberta are quite shallow (e.g., Mitchell and Prepas 1990, Prepas et al. 2001) and thus may consist only of littoral habitat.

Deeper water bodies (>3 m) - Ideally, samples should be taken across depth strata in proportion to the volume that each contributes to the total volume of the lake. Given the number of sites to be sampled and the lack of a priori information, however, we recommend the collection of three samples from the pelagic zone and three samples from the littoral zone of each water body (Table 3). Whenever possible, samples should be collected using the larger tube sampler. Samples should extend from the water surface to ~ 0.25 m off the bottom in pelagic waters. Littoral samples should be taken at three of the 12 benthic invertebrate stations (see section 2.2.3). These stations are randomly selected along the 1-2.0 m depth zone of the lake. The deepest point of the basin should be included as one of the pelagic stations. The other pelagic station should be chosen

Table 3 Overview of sampling protocols to monitor zooplankton communities from standing water bodies.

Attribute	Sampling Details
1. Sampler	Large (10.2 cm diameter) and small (7.3 cm) tubes
2. Habitats	Pelagic (or "deep") zone and littoral zones
3. Number of stations	6 stations per water body: 3 pelagic, 3 littoral
4. Selection of stations:	
a) Water bodies >3m deep	Littoral: 3 randomly selected along 1 to 2m depth zone[a] Pelagic: 2 randomly selected, 1 at maximum water depth
b) Water bodies 1-3 m	Littoral: 3 randomly selected along 1 to 2m depth zone[a] Deep: 2 randomly selected at depth greater than 1m, 1 at maximum water depth
c) Water bodies <1m deep	Littoral: 6 randomly selected along transect where depth =1/2 maximum depth [a] Deep: 1 randomly selected at depth greater than 1/2 maximum depth, 1 at maximum water depth
5. Samples	Routinely 1 sample per station. Three replicate samples at each station for 10 water bodies to assess variance.

[a]see Benthic Invertebrate protocol for details (Section 2.3.3).

randomly by dividing the pelagic zone into squares, and randomly selecting one square for sampling.

Shallow water bodies (<3 m) - If the water body is 1-3 m deep, three samples should be taken along the 1-2 m strip as for littoral benthic invertebrate stations in deep lakes. For water bodies shallower than 1 m, the three shallow samples should be taken along a transect where depth equals one-half of the maximum depth. Also, the deepest location should be sampled as well as a second, randomly selected "deep" water station (i.e., >1 m in water bodies 1-3 m deep; >one-half the maximum depth for water bodies <1 m deep). In shallow fen and bog complexes (e.g., water depths of 5 to 50 cm), we recommend that zooplankton be collected using the small tube sampler that should be

held horizontally within the middle portion of the water column and that at least 5 liters of water is drawn.

We further recommend taking three replicate samples at each station for a subset of water bodies. Taking replicate samples at each station for 10 water bodies should be adequate to permit an assessment of inter-sample variance. Samples taken during the first 1-5 years of monitoring should be used to better understand relationships among sampling effort, community attributes and variance. A simplified sampling protocol that may be less expensive could be implemented if analysis determines that fewer than six stations yields reliable information.

Composite samples - It is not uncommon for researchers to collect several zooplankton samples from a site and pool the samples to create a composite for easier processing. Before making composite samples from the different zones for processing, careful consideration should be given to the depths at which samples are collected and hence the proportion of the lake that each stratum sampled represents. We suggest that individual zooplankton samples are initially processed and that a statistical exercise be completed to assess the effects of composite sample mixing on density, composition and richness rather than mixing of samples in the field. Results from this exercise should be used to to determine future sampling effort (i.e., number and location of sampling stations). As a result, water depths of all sampling stations and the tube size used should be recorded. For the statistical exercise, the hypothetical composite sample should be mixed in proportion to the area of lake each zone represents. For example, if the littoral and pelagic zones represent 80% and 20% of the lake respectively, then 80% of the composite sample should be from the littoral samples and 20% from the pelagic samples (Prepas 1984). Note that integrated samplers weight all strata occurring at a station as if they contribute equally to the lake volume, that is, the epilimnion is underrepresented with a bias towards organisms in the lower depths.

Sample Preservation

To prevent distortion of the body structure and thus facilitate taxonomic identification, zooplankton should first be narcotized by placing them in carbonated water (e.g., soda water, Alkaseltzer, Bromoseltzer). After zooplanktoners have stopped moving (about 1 minute), samples should be preserved in a 4% buffered formalin-sucrose solution. The solution is made by adding 2 g borax (buffer) to every 98 ml of 40% formaldehyde. The mixture is diluted by adding 1 part water to 9 parts buffered 40% formaldehyde and sugar is added at 40 g/L.

Sample Processing

Considerations - The large numbers of organisms typically collected in zooplankton samples makes it impractical to count every individual and subsampling protocols are normally used. For both richness and abundance measures, a high degree of error can be introduced if subsamples contain small numbers of animals. Bottrell et al. (1976) illustrated the relationship between variance and mean density of individuals within subsamples and showed that the coefficient of variation declined and stabilized at approximately 8% when subsamples contained 60 or more individuals per taxon of interest. We recommend that the AFBMP estimate subsampling error for richness and abundance estimates by taking multiple subsamples from several source samples. The variance to mean ratio of several subsample counts should be tested for conformation to a random distribution with the chi-square test (see Prepas 1984).

Recommended protocols - Computer processing techniques exist for semi-automated sorting and counting of samples and measuring of body sizes (e.g., Mills and Confer 1986) and if resources permit, we recommend this approach. Alternatively, subsamples should be generated using a plankton splitter (e.g., Folsom or Motodo plankton splitters) or by pipetting out a known volume of the sample for counting. Although several methods exist, the following is a useful technique (McCauley 1984, Wetzel and Likens 1991):

1. Dilute the zooplankton sample with water so that pipetted subsamples will contain the desired number of individuals. Record all manipulations of sample volume in order to calculate the original concentration of animals.
2. Mix the sample in a graduated cylinder or similar vessel.
3. Immediately obtain a subsample of a fixed volume with a wide-mouthed (i.e., >4 mm) automatic, volumetric pipet or a Hansen-Stempel pipet.
4. Add the pipetted subsample to a counting device such as a Sedgwick-Rafter cell or sedimentation chamber for counting with an inverted and/or stereomicroscope.

Given the large number of sites to be sampled by the AFBMP each year, identifying zooplankton to species may be impractical. There are a large number of species, many of which require dissection of appendages for species level identification. We recommend that zooplankton be identified to the lowest feasible taxonomic group (e.g., genus or species) when possible. A coarser taxonomic level (e.g., family) should be considered only when finer-scale identification involves time consuming dissection.

A species list, taxonomic keys and pertinent references for each group in Alberta are provided by Clifford (1991). In addition to detailed identifications, aggregate variables such as taxonomic association, body size, life-history stage, and feeding guilds should also be considered (Stemberger and Lazorchak 1994). This approach is consistent with the AFBMP's emphasis on monitoring target groups, and would reduce costs. Stemberger and Lazorchak (1994) found that aggregate variables were easily generated, required a low level of taxonomic training, and explained more of the variance than did species-level data.

The AFBMP has stated an emphasis on richness measures and while zooplankton richness has been shown to be an effective measure of change, it should be noted that other measures can be calculated using the quantitative methods recommended in this protocol. Specifically, density, relative abundance, and biomass are commonly calculated, and if the AFMBP chooses to invest in semi-automated computer processing, little extra effort and cost would be involved in obtaining these measures after initial equipment purchases.

Statistical Power

Urquhart et al. (1998) developed power curves to detect regional-scale trends for several lake variables, including number of zooplankton taxa, using field data from the Environmental Monitoring and Assessment Program (EMAP; Messer et al. 1991). EMAP has a revisit sampling design similar to that currently proposed for the AFBMP, with resampling of sites every 4 years (with extra sampling in a subset of lakes). Using EMAP protocols, zooplankton was sampled from the deepest point in each lake using a coarse (202 μm) and a fine (48 μm) mesh plankton net. EMAP found that concordant variation (i.e., the variation of all sites together around an annual average or trend) strongly affected the ability to detect trends over time. Statistical power increased primarily as a function of increasing the number of years of monitoring, rather than increased numbers of sites or samples within a site. Models showed that with a year component of variance of ~0.15, a 3% annual change was detected with 90% certainty in approximately 15 years (Fig. 5). Covariates can be used to remove some of the yearly concordant variation so that the same power can be achieved in fewer years.

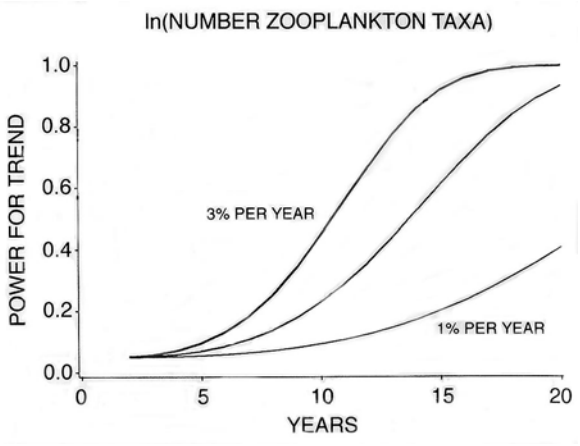


Figure 5 Power to detect trends of 1, 2, and 3% of ln number of zooplankton taxa using the observed components of variance from EMAP field data (After Urquhart et al. 1998: with permission).

Costs

Table 4 Cost estimates for equipment to monitor zooplankton in standing water bodies. Costs are in Canadian dollars.

Sampling equipment	Cost
Small tube sampler	40
Large tube sampler	100
Anaesthetic (cost per sample)	0.1
Preservative (cost per sample)	0.2
Sampling jars (per jar and lid)	1
Miscellaneous writing and note supplies	0.1
Sample Identification [a]	
a) High taxonomic (predominantly genus and species) resolution (cost per sample)	65
b) Low taxonomic (predominantly family and genus) resolution (cost per sample).	45

[a] Assuming subsampling procedures. Costs are cost per sample and include cost of entry of species counts into an Excel spreadsheet for release to the AFBMP by the consultant.

Sampling platform	Platform type	Cost
1) Road accessible		
Small water body		
a) Shallow	14.5' canoe[a]	1120
	2 horsepower, 4 stroke motor	1000
b) Deep	10' aluminum boat	1000
	2 horsepower, 4 stroke motor	1700
Large water body	12' aluminum boat	1300
	5 horsepower, 4 stroke motor	1700

2) Remote access		
a) Small water body	10' inflatable boat	2000
	2 horsepower, 4 stroke motor	1700
b) Large water body	12' inflatable boat	2500
	5 horsepower, 4 stroke motor	1700
Safety equipment		Cost
Life jackets	3	200
Flares	2	100
First aid safety kit	1	300
Flashlight	3	90

2.3.3 Benthic Invertebrates

Community Structure

Benthic invertebrates are animals that live on and in sediments and vegetation of water bodies. Benthic communities are extremely diverse. Major representatives in lentic systems usually include insects, segmented worms (oligochaetes and leeches), gastropods, microcrustacea (ostracods), and macrocrustacea (mysids, isopods, decapods and amphipods). Macroinvertebrate assemblages of streams and rivers (i.e., lotic invertebrates) have been used extensively as indicators of ecosystem health (Rosenberg and Resh 1993, Karr and Chu 1999). Indices such as the Benthic Index of Biotic Integrity (B-IBI), Invertebrate Community Index (ICI) and rapid assessment approaches are currently used in biological monitoring of streams (see Rosenberg and Resh 1993, Davis and Simon 1995). Although these measures have been used successfully for lotic waters, they have not been tested extensively in lentic habitats. Nonetheless, a huge amount of information exists on the use of benthic macroinvertebrates as indicators in running waters (e.g., Rosenberg and Resh 1993) and they hold great potential as an indicator group for standing water bodies.

In general, a variety of stressors (e.g., eutrophication, Tucker 1958; pollution, Pearson 1975; mining impacts, Osborne et al. 1979; acidification, McNicol et al. 1995a) can lead to reduced taxa richness, and dominance by a few opportunistic species. Because of the lack of information on monitoring in lentic systems, it is difficult to recommend a specific target group. As a result we recommend a community-based design, combined with statistical analyses that attempt to identify elements that show strong responses to changes in environmental predictor variables compared to those that do not. If statistical analyses indicate that particular elements, compared to others, are more responsive to watershed disturbance, then samples could be processed selectively for the responsive elements. We do, however, recommend that the AFBMP focus largely on macrobenthic fauna and that sampling nets are fitted with 250 μm mesh. Although this mesh size will underestimate some meiobenthos (< ~250-100 μm, rotifers, copepods, young chironomids, small oligochaetes, nematodes) and the microbenthos (< 100 μm, protozoans, juveniles of larger forms), both of which contribute substantially to density and species estimates, biological diversity of larger bodied animals is still substantial and has been shown to be responsive to watershed disturbances (e.g., 250 μm and 1000 μm, Voelz and Ward 1991; 908 μm and 1000 μm, Kerans and Karr 1994). We recommend focusing on macrobenthos because: 1) the AFBMP's goal is not to monitor total species numbers, but rather numbers within target groups, 2) the ecology of the macrobenthos is reasonably well understood, and 3) sample processing time increases dramatically when sampling for meio- or microbenthos.

Given that additional work is needed to select individual elements, we suggest that initial sampling should focus on the entire benthic invertebrate community and that this information be used to refine sample processing in the future. Pilot studies or small-scale research would be an excellent step to define and evaluate the utility of individual target groups within the macrobenthos and we strongly recommend doing so. In general, richness measures are often useful (Kerans and Karr 1994, Rosenberg et al. 1999), such as the number of mayfly, caddisfly, diptera or Chironomidae taxa. Other measures have also proven useful such as species richness, community composition and dominance measures and will likely be strong indicators of watershed disturbances.

Alternatively, metrics such as percent amphipods and percent insects have been shown to be good indices for discriminating between littoral benthic communities (Somers et al. 1998).

Sampling Alternatives

A variety of quantitative samplers are used to collect lentic benthos with grab samplers and corers most prevalent. Grab samplers (e.g., Ekman, Petersen, PONAR, van Veen, Smith-McIntyre) use strong closing metal jaws to collect benthos from lake substrata whereas core samplers (e.g., Kajak-Brinkhurst, multiple corers) need to be pushed down into sediments or use gravity to isolate an area of known size. Other samplers include hydraulic (or airlift) samplers, emergence traps for collecting emerging adult invertebrates, and suction devices for hard substrates. Qualitative samples are often collected using sweep nets. We recommend the use of a quantitative sampler, when possible, because the increase in quality of data obtained outweighs the greater effort involved in the actual sampling process.

The Ekman grab is the most frequently used sampler, followed by sediment corers, and within this sampler type, the Kajak-Brinkhurst corer (Downing 1984). Each device is limited in the type of substrate it can sample. For example, light-weight Ekman or core samplers perform poorly in highly compacted sediments or substrata dominated by large cobbles or boulders, whereas a heavy grab like a PONAR often fails to close properly in soft sediments (Downing 1984). Unless specifically designed to sample aquatic plants, the majority of samplers can not be used effectively in dense vegetation. Both Ekman and corer devices are inefficient samplers in extremely soft sediments (Wetzel and Likens 1991). Ekman grabs can sample a somewhat greater range of sediment types, and are easier to use in low water visibility conditions. Comparative studies have generally found corers to have higher accuracy than Ekman grabs and small area samplers such as corers are also most efficient (see references in Downing 1984). Macroinvertebrates can be quantitatively sampled from macrophytes (e.g., Cheruvelil et al. 2000), however, separating the animals from the plants can be tedious (see Downing (1984) for a treatment of common quantitative samplers).

Recommended Protocols

Sampling Gear

We recommend that the AFBMP use a corer to sample benthic invertebrates from standing water bodies. We recommend the use of a 7.6 cm diameter (i.e., 46 cm^2) corer made of PVC pipe fitted with an extension handle (France et al. 1991). The bottom edge of the corer should be sharpened to facilitate sampling in vegetation or resistant substrates (e.g., shallow wetland habitats where bryophytes are dominant). Core samples may not be able to taken in some habitat types and thus we also recommend taking sweep net samples in each water body. Sweep samples will provide semi-quantitative samples if effort is standardized in terms of area and time sampled and will provide measures for comparison from these habitats. A D-frame net with 250 μm mesh is appropriate. To standardize effort for sweep samples, we recommend taking two 1-m long sweeps over the same location, one in each direction, for a total of 10 seconds.

Sample Design

Design considerations - Maximum densities and diversity of most freshwater benthos occur in shallow water, peaking between 1-2 m, and decrease substantially with increasing depth (e.g., Diggins and Thorp 1985). Their distribution within the littoral zone is very heterogeneous, reflecting the variable nature of the habitat. European researchers have successfully characterized lakes based on profundal (i.e., the area below the deepest extent of all plant growth) macroinvertebrates, which show predictable changes with lake nutrient status (Wiederholm 1980, Dinsmore et al. 1999). Because the number of species within a target group is the measurement for the AFBMP, and not total richness, we do not recommend attempting to sample all habitats within a water body. Further, many standing water bodies in Alberta do not stratify and the majority of Alberta lakes are shallow. As a result reporting regional based trends in biological diversity of profundal benthos will be limited because of low sample sizes. We do not advocate sampling of profundal benthos as part of the AFBMP.

Recommended sampling design - We recommend a minimum of 12 core samples be collected per water body. This is based on the number of replicate samples required for macrobenthos densities to produce standard errors within 20% of the mean density when using a sampler of 50 cm^2 in cases where density of taxa of interest is 300 m^2 (Downing 1984). Densities of several major Alberta taxa are found in this range (G. Scrimgeour, unpublished data). We do not have information on this type of relation for estimates of taxa richness, but presumably data sufficient to yield density estimates will also provide reliable richness measures. In reviewing the efficiency of all sampling protocols, the AFBMP should determine whether 12 within lake samples is sufficient or whether more or less samples should be collected.

We also recommend that the AFBMP collect net sweeps at three of the randomly selected core sites and that these should be analyzed to determine the extent to which data from the three net sweeps compliments that collected using the corer. If the two techniques provide similar data, then collections using the net sweeps should be discontinued.

We recommend restricting sampling to soft sediments in the littoral zone (defined here as water <3 m in depth), stratified along the 1-2 m depth zone. By sampling from a restricted depth zone, diversity and density will be less variable, and thus the precision estimates of the samples will also be improved. To select sampling locations, we recommend that a GIS exercise be completed as part of the presampling efforts to identify 12 randomly selected sampling locations. At each station one core sample should be taken at a depth of 1-2 m where the sediment and vegetation permit, and one sweep net sample should be taken at three stations. It is very important that a description of the substrate, water depth and light availability be recorded at each sampling station.

For water bodies shallower than 1 m, follow similar procedures, only use one-half the maximum depth as the target depth zone. In highly vegetated water bodies or compacted sediments, it may not be possible to collect samples using the corer and six sweep net samples should be taken at these sites.

Table 5 Overview of recommended protocols to sample macroinvertebrate communities from standing water bodies.

Attribute	**Sampling Details**
1. Sampler	Corer sampler D-frame sweep net
2. Habitats	Littoral zone
3. Number of stations	Corer: 12 stations per water body Sweeps: 3 to 6 stations depending on substratum type (i.e., where vegetation permits)
4. Selection of stations:	
a) Water bodies >1 m deep	Randomly selected along 1-2 m depth zone
b) Water bodies <1 m deep	Randomly selected along transect where depth =1/2 maximum depth
5. Number of samples	One replicate sample per station

Sample Preservation and Processing

Grab contents should be sieved in the field through a 250-μm wash bag and preserved in 4 % formaldehyde. Invertebrate samples should be identified to genus or species depending on the availability of taxonomic keys (e.g., Clifford 1991) during initial surveys. These data, perhaps from the first 3 to 5 years, should be analyzed to determine whether lower levels of taxonomic resolution identify similar relationships to predictor variables compared to results based on genus and species level identification.

Rosenberg et al. (1999) compared the performance of a family-level model to that of models using lower taxonomic levels, and to derive biological measures (metrics) for discriminating between impaired and unimpaired sites in the Fraser River catchment. Their analysis found the family-level model to be superior to the others. Several books provide in depth detail on using macroinvertebrates in biomonitoring (Rosenberg and Resh 1993, Davis and Simon 1995, Karr and Chu 1999), and should be consulted when determining the target group. Density is a commonly used measure in benthic

community studies, and, if desired, could be obtained from the quantitative methods described in the protocol.

A diversity of laboratory procedures including the sugar flotation method and sub-sampling can be used to reduce sample processing costs (e.g., Cromar and Williams 1991, Wrona et al. 1982). Considerable debate exists as to whether laboratory subsampling is appropriate for benthic samples. Some argue that subsampling compromises data quality and that entire samples should be counted (e.g., Doberstein et al. 2000). Others suggest that counting as few as 100 individuals yields the same results as doubled or tripled effort (Somers et al. 1998). While the average length of time required to handpick a sample from a grab sampler (Ekman, Ponar and Peterson) is 2.65 h, the overall range is highly variable (0.1-10.9 h) (Resh et al. 1985). We recommend that efforts should be made to count entire samples and that the AFBMP initiate a pilot study to evaluate the consequences of subsampling on various population and community descriptors (e.g., density and relative abundance).

Costs

Table 6 Cost estimates for equipment to monitor benthic invertebrates in standing water bodies. Costs are in Canadian dollars.

Sampling Equipment	Cost
Corer sampler	75
D-frame sweep net	150
250 μm wash bag	50
Preservative (cost per sample)	0.2
Sampling jars and lid	1
Sample Identification[a]	
High taxonomic resolution	55
Low taxonomic resolution	35

[a] Costs are per sample and include cost of entering species counts into an Excel spreadsheet.

Road accessible	Sampling Platform	Cost
Small water body		
a) Shallow	14.5' canoe[a]	1120
	2 horsepower, 4 stroke motor	1000
b) Deep	10' aluminum boat	1000
	2 horsepower, 4 stroke motor	1700
Large water body	12' aluminum boat	1300
	5 horsepower, 4 stroke motor	1700
Remote access		
a) Small water body	10' inflatable boat	2000
	2 horsepower, 4 stroke motor	1700
b) Large water body	12' inflatable boat	2500
	5 horsepower, 4 stroke motor	1700

[a] 10' inflatable boat may also be effective in this habitat

Safety equipment		Cost
Life jackets	3	200
Flares	2	100
First aid safety kit	1	300
Flashlight	3	90

2.3.4 Amphibians

Community Structure

Only ten species of amphibians occur in Alberta, four of which (western toad, *Bufo boreas*; Canadian toad, *Bufo hemiophrys*; wood frog, *Rana sylvatica*; boreal chorus frog, *Pseudacris triseriata*) are found quite extensively in the boreal region (Russell and Bauer 1993). Although present in forested regions, tiger salamanders (*Ambystoma tigrinum*) often comprise scattered populations in boreal Alberta. The northern leopard

frog (*Rana pipiens*) has been historically recorded in the boreal region, but, in addition to dramatic decreases in their abundance within Alberta over the last 20 years, their distribution in northern Alberta is poorly understood. Amphibians are typically considered sensitive to environmental disturbance (Blaustein and Wake 1990, Wake 1991) and declines in both richness and abundance have been associated with human encroachment and disturbance such as agriculture (Kolozsvary and Swihart 1999) and the loss of forests (e.g., Hecnar and M'Closkey 1996, Mensing et al. 1998, Knutson et al. 1999). Indeed, Alberta populations of the northern leopard frog and Canadian toad have been listed as "at risk" and as "may be at risk" because of apparent declines in their distribution. All of the amphibian species of boreal Alberta are associated with wetlands, ponds, lakes or running water for at least part of their lifecycle.

With only a maximum of 6 species occurring in boreal Alberta, species richness may not be an effective indicator of changes in watershed condition. However, changes in amphibian presence or abundance and increases or decreases in the extent of their distributional range could be measured. Thus, we recommend that the AFBMP monitor amphibians around standing water bodies.

Sampling Alternatives

Standard techniques for inventory and monitoring of amphibians in terrestrial habitats include visual searches (e.g., visual encounter surveys, quadrat, transect, and patch sampling), the use of callback tapes to elicit a response (e.g., audio strip transects), and drift fences and traps (e.g., pitfall or funnel traps) that capture moving animals (Heyer et al. 1994, Gingras et al. 1999). In aquatic habitats, eggs, larvae or metamorphs can be sampled. However, not all methods are suitable for all habitats, species or seasons. For example, call surveys to detect frogs and toads are only effective during the spring breeding season, and can not be used to detect salamanders. Gingras et al. (1999) compared four survey techniques potentially suitable to detect amphibians in riparian zones of two streams in Alberta's boreal forest. They identified a visual survey technique as the most cost- and information- efficient protocol. The survey comprised one or two people slowly walking a fixed distance along a transect searching for amphibians on the ground surface. Once encountered, all amphibians were identified

and counted. Because none of the Alberta species are considered stream-dependant (i.e., stream habitat specialists), the visual search method described by Gingras et al. (1999) should also apply to standing water bodies. If desired, additional measurements of length and weight can be taken. Readers should refer to Gingras et al. (1999) for a full discussion on comparisons of the four sampling methods.

Recommended Protocols

Sampling Design

Based on Gingras et al (1999), we recommend the use of a low intensity visual search (Crump and Scott 1994) to monitor amphibians near standing water bodies. Visual surveys should be conducted along two 200 x 1 m transects (400 m^2). The location of these transects should be selected randomly by dividing the perimeter of the water body into 20 evenly spaced points, with two points representing the beginning of each transect selected using a random number table. If the start point for the second transect overlaps with that of transect 1, an additional point would be selected. If the perimeter of the water body is less than 400 m, the entire perimeter of the water body should be surveyed. Transects should be located between 3-5 m from the edge of the water body, beyond the point where the ground is still wet, and emergent vegetation (e.g., cattails). Surveys take approximately 30 to 60 minutes per transect (see Gingras et al. 1999 for additional details). Transects should only be established if the sample plot encompasses a length of shoreline or if the shoreline is located within 100 m of the plot perimeter.

Table 7 Overview of sampling protocols to monitor amphibian communities adjacent to standing water bodies.

Attribute	Sampling Details
1. Sampling method	Low intensity visual surveys along transects
2. Habitats	Riparian zones 3 to 5 m from the water body
3. Number of stations	Two - 200 m x 1 m transects per water body (i.e., search area = 400 m^2)
4. Transect locations	Selected randomly

Costs

Table 8 Cost estimates for equipment to monitor amphibians adjacent to standing water bodies. Costs are in Canadian dollars.

Sampling equipment	Cost
Hip chains	200
Flagging tape (cost per site)	1.5
Writing supplies (cost per transect)	0.5
Optional equipment (if weighing/measuring):	
Caliper	15
pesola scale (or use electronic balance if available)[a]	85
Latex gloves (cost per site)	1
Plastic bags (cost per site)	1

[a] Electronic balance = $450.00

2.3.5 Fish

Community Composition

Alberta's fish fauna is composed of 51 native species belonging to 13 families plus an additional eight introduced species (Nelson and Paetz 1992). A diversity of habitats exist in northern Alberta that support cold- (e.g., trout, white fish and grayling) and cool- (e.g., pike, walleye, yellow perch) water fish in addition to the more

ubiquitously distributed warm-water species, such as some of the minnows (Nelson and Paetz 1992). As a consequence of the filtering of post-glacial colonization processes, however, most fishes in Alberta are thought to be tolerant of a relatively wide range of habitat conditions (Nelson and Paetz 1992).

Within individual bodies of water, species richness of fish generally increases as a function of lake size (Tonn and Magnuson 1982, Griffiths 1997) and declines with increasing isolation (Magnuson et al. 1998) and disturbance (Minns et al. 1994). Exceptions can occur due to species introductions (Karr and Chu 1999, Whittier 1999), nutrient enrichment (Schulz et al. 1999) or when normally depauperate coldwater habitats become degraded and are invaded by species with wider environmental tolerances (Lyons et al. 1996).

The widely used index of biological integrity was originally developed for fish assemblages of running waters in the United States (Karr 1981) but has also been applied to evaluate effects of watershed disturbances on some lake assemblages (Schulz et al. 1999, Whittier 1999). Some examples of index metrics are the number of native fish species, number of intolerant species, and the relative abundance of top carnivores (Karr and Chu 1999). Many potential metrics, however, may not be successful for a region like Alberta due to the naturally depauperate assemblages (Whittier 1999). Furthermore, tracking only a subset of species (e.g., minnows, intolerant species) is not likely to be effective and thus we recommend that the AFBMP focus on the entire fish community, at least during the initial stages of the program. In addition, because the methods for estimating relative abundance (i.e., catch-per-unit-effort CPUE) are essentially the same as for determining species presence/absence, we recommend that the former be determined.

Sampling Alternatives

Fish sampling generally involves passive (e.g., entanglement, entrapment or angling gear) or active (e.g., trawls, dredges, encircling nets, electrofishing) capture methods (Murphy and Willis 1996). Acoustic devices are also sometimes used for measuring the abundance, size and distribution of pelagic fish, but in the majority of

situations cannot identify species unless combined with capture techniques (e.g., gill nets) where comparisons of acoustic data can accurately discriminate among species.

Active methods are advantageous in that most sample a defined space (and thus unit of effort) and should be less influenced by fish behavior than passive gear (Murphy and Willis 1996). Although some encircling nets can be used from shore (e.g., beach seine), most active methods require a larger boat and more people than do passive gear and their effectiveness is typically habitat (and often species) specific. For example, electrofishing in lakes typically requires a large boat and motor, which would be unsuitable for remote sites.

In general, passive techniques are simpler and easier to use than active gear and have been used extensively to gather data on species composition, fish abundance, and size distribution (e.g., Robinson and Tonn 1989, St-Onge and Magnan 2000, Paszkowski and Tonn 2000). Gill nets and trammel nets are the two primary entanglement gears and have similar biases. Trammel nets tend to be less lethal than gill nets, but fish removal from trammel nets is considerably more difficult and time-consuming than from gill nets. Trammel nets are rarely used in fisheries research, whereas gill nets are widely used to monitor fish populations (e.g., Fisheries Techniques Standardization Committee 1992, Appelberg et al. 1995, Murphy and Willis 1996, Morgan 1998). Entrapment gear, such as minnow traps and fyke nets, are also commonly used to provide information on fish abundance and species composition. Minnow traps are significantly smaller and easier to set than many other entrapment gears, and are effective for several small species in Alberta (Robinson and Tonn 1989).

Recommended Protocols

Sampling Equipment

The ability to detect fish species is strongly affected by capture technique (Jackson and Harvey 1997), thus we suggest that sampling of fish communities in standing water bodies include several gear types. Specifically, we recommend that the AFBMP inventory lentic fish using multi-mesh (experimental) gillnets, minnow traps, and beach seines. Gillnets have proven effective for giving comparable structure and abundance estimates in multi-species assemblages and are used in other aquatic

monitoring programs (e.g., EMAP, Messer et al. 1991; U.S. and Canada, Fisheries Techniques Standardization Committee 1992; Finland, Norway and Sweden, Appelberg et al. 1995) and research studies that quantify effects of watershed disturbances on fish communities (e.g., St-Onge and Magnan 2000). Small-bodied fishes can occur in high numbers in Alberta water bodies and can be more efficiently sampled with minnow traps, especially in very shallow and densely vegetated areas, which dominate many water bodies in the forested region of Alberta. Bait can be used in the minnow traps to attract fish, and we recommend a small pilot study to compare the abundance, composition, and species richness of catches using baited and non-baited traps.

Beach seines are useful for catching species that are not well represented in passive gear (e.g., Iowa darter), but may be of limited utility in habitats dominated by macrophytes or other structure (e.g., woody debris). Thus, during initial stages of the project, we recommend use of a seine and a comparison between the species caught by this gear with those obtained from gill nets and minnow traps. If these comparisons indicate that gill nets and minnow traps capture the same species as with a seine, the use of seine nets should be discontinued.

The recommended benthic gill net consists of 12 panels of different mesh sizes that increase in a geometric series (Table 9, Appelberg 2000), which reduces overall size selectivity. This distribution of mesh sizes is used extensively in northern Europe (Appelberg et al. 1995), and is now being used in Ontario (E. Snucins, Cooperative Freshwater Ecology Unit, Laurentian University, pers. comm.). Each gill net is 30 m long (12 panels of 2.5 m per panel) and 1.5 m deep, made of monofilament nylon and is attached to a buoyancy line and a lead line. Nets are typically set at the sediment surface, but can be set mid-water, as is sometimes done in pelagic areas of large lakes to target pelagic species (e.g., cisco, spottail shiner) that would otherwise be underrepresented using bottom sets (Baker et al. 1997). Minnow traps are wire traps with funnel-shaped openings on both ends through which fish enter and then are held within the trap. We recommend the standard Gee minnow traps (42 cm long by 23 cm at widest point) with 6.35 mm mesh size. Finally, beach seines should be used where appropriate conditions allow (i.e., shallow inshore areas free of obstructions with relatively smooth, firm substrate). A beach seine 8.7 m long and 1.2 m high with 5 mm

mesh size and a bag is appropriate and is operated by two people holding opposite ends of the net and hauling it through the water.

Table 9 Recommended mesh-size distribution of gill nets for sampling fish communities of standing water bodies in Alberta. Note that mesh size is listed as the knot to knot (i.e., square) measurement.

Mesh panel number	Mesh size (mm)	Thread diameter (mm)
1	43	0.20
2	19.5	0.15
3	6.25	0.10
4	10	0.13
5	55	0.23
6	8	0.10
7	12.5	0.13
8	24	0.16
9	15.5	0.15
10	5	0.10
11	35	0.20
12	29	0.16

Sample Design

Design considerations - When using gill nets, sampling effort should increase with the size of the water body to capture all netable species (Degerman et al. 1988, Morgan 1998). Further, Jackson and Harvey (1997) demonstrate how detection probabilities for different fish species increase with sampling effort for several gear types, including gill nets and minnow traps. Our (W.M. Tonn, unpublished data) previous work in small lakes in the aspen parkland - boreal forest transitional zone of Alberta illustrate how species accrual curves are affected by sampling effort, lake size, and the number of species. Although the data were not collected specifically to test for fishing effort relations and the total number of species used assumes that all netable species were caught, these data show that the asymptotic relationship between the number of species captured and sampling effort increased with lake size (Fig. 6) and the number of species per lake (Fig. 7).

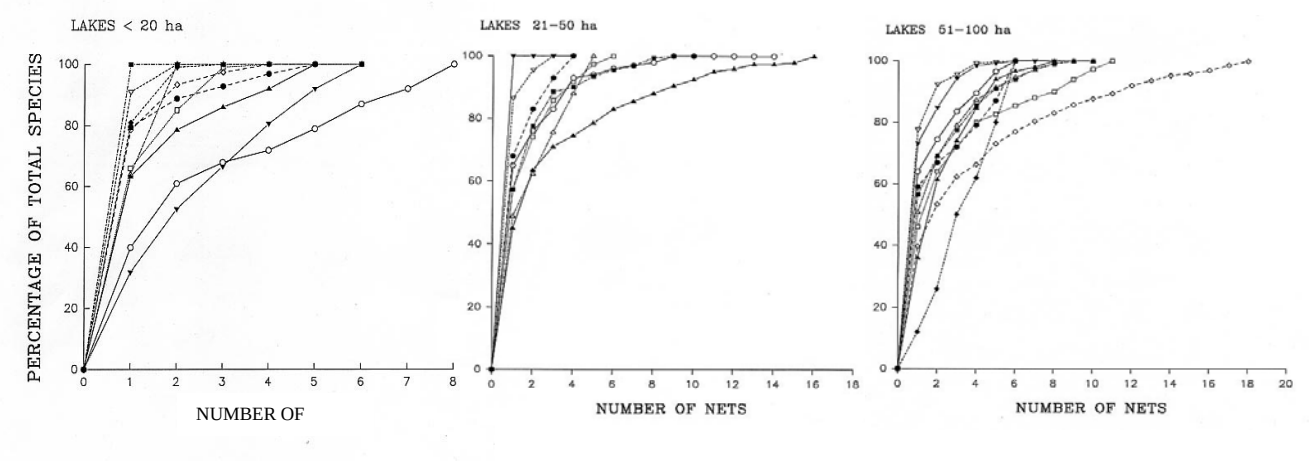


Figure 6 Relationships between cumulative percentage of fish species caught versus sampling effort (using gill nets) in three sizes of lakes in Alberta. From W.M. Tonn, unpublished data.

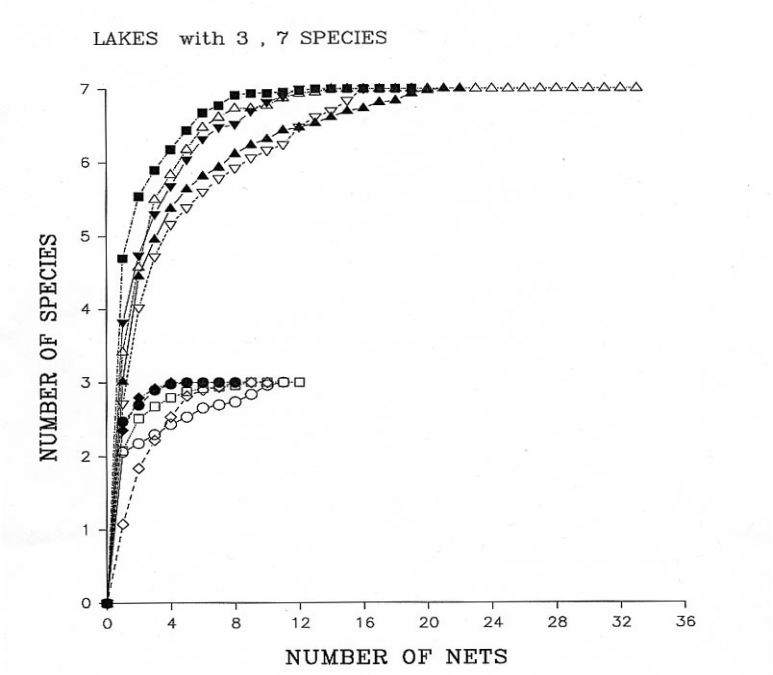


Figure 7 Cumulative number of fish species caught in Alberta lakes with either 3 or 7 species under various

Recommended protocol - Gill nets - Our protocol for gill netting is based largely on Swedish standard methods (Appelberg 2000), with appropriate modifications for Alberta (W.M. Tonn, unpublished manual). We recommend that sampling effort of gill nets be adjusted for area and depth of the water body (Table 10) as adopted for using fall

walleye index netting (FWIN) protocols (Morgan 1998). Using this design, larger, deeper water bodies will require two nights/three days to sample and smaller, shallower sites, one night/two days to sample. Sampling should be stratified by depth to increase precision of CPUE estimates (Degerman et al. 1988). We suggest depth zones for Alberta lakes of: 1) <3 m, 2) 3-6 m, and 3) >6 m. Nets should not be set in water where dissolved oxygen levels < 2 mg/L, as few fish are likely to occur in these hypoxic conditions. In very shallow (< 1.0 m) or heavily vegetated water bodies it may only be possible to set minnow traps; our experience suggests that most fish in such systems will be suitably small.

Stations for benthic gill nets should be randomly selected within each depth zone. This can be accomplished by dividing the perimeter of the water body or the length of shoreline included in the sample area into 20 equally-distanced points and then using a random number table to identify locations where nets are to be set. If sampling requires greater than one night, nets should be moved to new locations for the second night.

Study sites within a 75 ha offshore plot could be established by overlying a 100 m x 100 m grid to produce 1 ha cells. Dissolved oxygen profiles should be taken in 20 of the central cells to determine if and where levels fall below 2 mg/L. Once the depth at which dissolved oxygen equals 2 mg/L is determined, that depth is divided vertically into three depth zones. Nets are set in the water column with the net bottom on the lower boundary of each zone (e.g., for a depth zone of 4 to 7 m, the net would extend from 5.5 to 7 m).

Differences in sampling effort in each depth zone is designed to approximate water volumes of each depth stratum (Table 10) and the fact that the majority of fish species and biomass tend to occur in shallower zones of lakes. The orientation of each net should be determined randomly by selecting one of eight compass directions (i.e., N, NE, NW, E, W, S, SE, SW). In the > 6 m depth zone, one of the two nets should be set in the pelagic zone and extend from 1.5 m below the water surface to 3 m deep and an additional net set on the bottom of the depth profile at which dissolved oxygen is $\geq$ 2 mg/L. Nets should be set in the evening 2 to 3 hours prior to sunset and left out overnight to catch both day and night active species. Because species richness estimates

appear to be positively related to duration of the set, nets should be left for 12 hours (Minns and Hurley 1988). This relation has not been tested rigorously, however, and data from initial surveys completed during the first 3 to 5 years of the program should be used to evaluate relations between duration of net sets and total catch, species diversity estimates, and mortality rates for different standing water body types.

As with all sampling methods, gill nets are not free of bias. Fish movements and behaviour, weather conditions, water temperature, water transparency, and location of the gill nets affect capture rates. Thus, efforts must be made to sample under generally similar conditions and within in a common time frame (e.g., mid-to late-summer), which minimizes environmental extremes and the spawning seasons of most Alberta species. Gill nets also tend to over-represent large fish and under-represent small fish, but this can be compensated for by the geometric distribution of mesh sizes and, if necessary, by using correction factors specific to each species. As with any trapping method, data from gill-net surveys will only provide a relative index of fish abundance. For comparative studies, the important assumption is that catch probability does not vary appreciably among water bodies or when sites are sampling repeatedly through time.

Table 10 Summary of suggested sampling effort (number of net sets) and locations of gill nets to monitor fish communities in standing water bodies.

A. Lake <50 ha			
Maximum depth (m)	<3	3-6	>6
Total net-nights	4	6	10
Depth zone (m)			
<3	4	4	6
3-6	0	2	2
>6	0	0	2[a]

B. Lake 50-75 ha or 75 ha plots with shoreline			
Maximum depth (m)	<3	3-6	>6
Total net-nights	6	8	12
Depth zone (m)			
<3	6	6	8
3-6	0	2	2
>6	0	0	2[a]
C. 75 ha plots without shore line			
Upper zone	0	0	6
Mid zone	0	0	4
Lower zone	0	0	2

[a] one net set from 1.5 m to 3 m from surface, one set at the depth at which dissolved oxygen exceeds >2 mg/L.

Interpreting the table:

1. Identify the portion of the table (A, B, C) that corresponds to the surface area of the standing water body.
2. Find the column that corresponds to the maximum depth.
3. Locate the total number of net-nights for the water body and number of nets to be set within each depth zone.

Note: Effort should be divided equally between nights, for example, for 12 net nights, 6 and 6 nets should be set on each of two nights.

The large number of standing water bodies that are projected to be sampled annually precludes the adoption of FWIN protocols (Morgan 1998). These protocols are largely based on gill netting during the fall when surface water temperatures range between 10 and 15^{o}C (Morgan 1998) and are based on the lake as the sampling unit. However, we suggest that the AFBMP consider applying protocols described here in addition to those described by FWIN protocols for Alberta's largest lakes. Because large lakes are relatively scarce in Alberta, FWIN protocols could be applied to 5-10 lakes on an annual basis without greatly increasing overall project costs. The resulting data would allow for large-scale comparisons.

Minnow traps - Minnow traps should only be set in water <3 m. Minnow traps should be randomly distributed within the <3 m zone and moved daily. For water bodies <50 ha, set 10 minnow traps for one night. For water bodies 50-75 ha, set 10 minnow traps on each of two nights, i.e., 20 trap-nights. Minnow traps should be set and retrieved at approximately the same time as the gill nets. Crewmembers should ensure that dense macrophytes or other materials do not block the funnel-shaped openings. The effects of bait and chemical light sticks on capture rates should also be evaluated.

Beach seine –Seining can only be performed in certain habitats and we recommend that the AFBMP evaluate the extent to which data from beach seines adds to that provided from gill nets and minnow traps before it is incorporated as a standard collection technique. An initial evaluation of beach seines should be completed in shallow shore areas, less than 1 m in depth, with a reasonably smooth, firm substrate free of snags, dense macrophytes and other obstructions. Efforts should be made to make two seine hauls of at least 100 m^2 per water body. Because suitable sampling sites are often scarce, it may not be possible to select seining sites randomly or to seine the full 100 m^2 and the area seined at each location should be determined on each sampling occasion. The area sampled should be recorded by multiplying the length of seine net that was effectively sampling the lakeshore times the distance between the two field

members holding each end of the net. If there are many potential seining sites, two sites should be randomly sampled in a manner similar to gill net and minnow trap locations.

Catches using seine nets are also strongly affected by the time of day when sampling is completed. We recommend that, if possible, seining be completed during the dusk period to increase catch rates of nocturnal and crepuscular species. If sampling is completed during the dusk period additional safety precautions will need to be developed and communicated to field crews.

For estimates of relative abundance (as catch or biomass per unit effort), the total catch (number or biomass of fish) is standardized as the number (or biomass) of fish per gear (net or trap) per hour (nets, trap) or per m^2 (seine).

Set-lines – The use of set lines comprising baited hooks provides the opportunity to detect species (e.g., burbot and suckers) that are not often detected in gill nets. We recommend that the AFBMP complete a pilot study to evaluate the extent to which catches from set-lines augment, rather than duplicate, data on species presence-absence. Pilot studies could include the setting of two 50 m lines each containing 10 baited hooks (hooks size = # 6) that extend from the lake margin to the center of the lake. Lines should be set overnight and retrieved in the morning to reduce fish mortality.

Table 11 Overview of sampling protocols to monitor fish communities from standing water bodies.

Attribute	**Sampling Details**
1. Sampler	Multi-mesh gill nets
	Minnow traps
	Beach seine (evaluation process required)
2. Habitats	
a) gill nets[a]	With shoreline: 3 depth zones (<3 m, 3-6 m, >6 m)
	Without shoreline: 3 vertically stacked depth zones, divided equally within water with >2 mg/L dissolved oxygen
b) minnow traps[a]	Littoral zone (<3 m depth)

c) beach seine[a]	Shallow shore areas < 1 m deep
3. Number of stations	Gill nets: 4-12 net-nights Minnow traps: 10-20 trap-nights Beach seine: 0-2 seine hauls
4. Selection of stations	Gill nets: random within each depth zone Minnow traps: random within shallowest depth zone Beach seine: where appropriate habitat exists
5. Number of samples	Each trap, net or seine equals one sample (or unit of fishing effort), therefore one sample per station

[a] Field personnel should receive training on how to repair nets.

Fish Processing

Gill nets and minnow traps should be retrieved in the order in which they were set. At a minimum, all fish should be identified and counted based and the mesh panel in which they were captured recorded. Though not necessary for species assemblage information, measuring length (fork length (FL)) and mass would allow for length frequency and biomass calculations and provide information on fish condition factors and growth rates. Measuring the first 100 individuals of each species would likely be sufficient for these purposes. Age structures could be collected from the first 30 (if needed only for archiving) or 100 (for population studies) fish of each important species (see below). Live fish should be resuscitated and released; dead fish should have their air bladders punctured and then be returned to the water body. Fish caught by minnow traps or seining should be transferred to containers of fresh lake water, anaesthetized if necessary, processed and released. Where the identification of fish species in the field is difficult perhaps due to small body size or poor condition of animals, we recommend retaining voucher specimens for identification in the laboratory.

Aging structures - Changes in the growth and metabolism of fish can result in pattern formation of hard structures of fish. These patterns can be used to estimate fish age and growth. If possible, aging structures should be retained and archived. We recommend collecting aging structures from some fish collected during the AFBMP.

Mackay et al. (1990) provide information on collecting structures and aging for fish in Alberta. Generally, two structures are collected specific to each species and commonly include scales, fin rays, cleithra, opercular bones, and otoliths. In the case of small fish, the whole animal may be preserved.

Genetics and stomach content analyses – Genetic relations within and between fish stocks in Alberta is poorly known. We recommend that the AFBMP consider the collection of tissue samples for genetic analyses or at the very least inform other agencies (e.g., Alberta Conservation Association, Department of Fisheries and Oceans) that such samples could be collected by the AFBMP on a cost-recovery basis.

Analysis of stomach contents of large predatory fish provides a further opportunity to determine the presence of species that be underrepresented or absent from collections using other methods. We recommend that the AFBMP complete a pilot study to evaluate the extent to which information based on the presence of fish species in stomachs of piscivores augments, rather than duplicates, information obtained from other capture techniques.

Costs

Table 12 Cost estimates for equipment to monitor fish communities from standing water bodies. Costs are in Canadian dollars.

Sampling equipment	Cost
Gill nets[a]	700 per net x 6 = 4200
Beach seine net	200
Minnow traps	17 each x 8 = 136
Miscellaneous equipment (e.g., compass, buckets, scissors, writing supplies)	100
Small (30 cm) and large (100 cm) measuring boards	220
Portable electronic balance (Ohaus advanced series) [b]	
200 g (0.01g accuracy)	950
1200 g (0.1 g accuracy)	850
Storage and ageing of structures (per structure)	5 to 8

[a] the nylon mesh of gillnets is easily damaged by woody debris, beaver, large fish or during removal of fish. As a result, an individual gill net may need to be replaced after 15-30 net nights depending on the abundance of large fish, small fish (e.g., minnows), beaver, woody debris and the level of care taken by the field crew when removing fish from the nets.
[b] Weigh scales should be checked and routinely calibrated.

Sampling platform		Cost
1) Road accessible		
Small water body		
a) Shallow	14.5' canoe[a]	1120
	2 horsepower, 4 stroke motor	1000
b) Deep	10' aluminum boat	1000
	5 horsepower, 4 stroke motor	1700
Large water body	12' aluminum boat	1300
	5 horsepower, 4 stroke motor	1700
2) Remote access		
a) Small water body	10' inflatable boat	2000
	5 horsepower, 4 stroke motor	1700
b) Large water body	12' inflatable boat	2500
	5 horsepower, 4 stroke motor	1700

[a] 10' inflatable boat may also be effective in this habitat

2.3.6 Aquatic birds

Community Structure

Aquatic birds have been the subjects of extensive monitoring in large part because of the social importance of ducks and geese as game species. However, monitoring programs for aquatic birds or waterfowl tend to measure population abundance, reproductive success or contaminant burdens rather than focusing on community richness (e.g., U.S. Dept. of the Interior, Fish and Wildlife Service and Environment Canada, Canadian Wildlife Service 1987, Fox et al. 1991, Beauchamp et al. 1996).

In general, species richness of aquatic bird communities increases with water body size (e.g., Paszkowski and Tonn 2000), trophic status (e.g., Hoyer and Canfield 1994), macrophytes (e.g., Fairbairn and Dinsmore 2001) and food (e.g., Hanson and Butler 1994). Other influences include geographic isolation, lake morphometry, climactic conditions and water quality (see references within Kerekes 1994). Aquatic birds fit into several feeding guilds such as piscivores, herbivores, and insectivores that should result in species assemblages that reflect aspects of the ecosystem. Another important grouping is cavity nesters, because forest harvesting may reduce nesting sites for this group and change patterns of nest predation (Vander Haegen and DeGraaf 1996). The most commonly documented disturbances affecting assemblages of aquatic birds are acidification (McNicol et al. 1995c) and eutrophication (Kauppinen and Väisänen 1993) that, respectively, typically decrease and increase bird abundance, biomass and richness. Evidence also indicates that species richness may increase in Boreal Alberta on water bodies where the surrounding forest has been harvested, possibly due to invasion of species more typical of prairie habitats (Pierre 2001).

The term "aquatic birds" can encompass a broad spectrum of groups such as waterfowl, riparian birds, and shorebirds. Our recommended target group of aquatic birds is non-passerine birds that feed at or beneath the surface of the water (sensu Paszkowski and Tonn 2000). This will include waterfowl (e.g., ducks, geese), diving (e.g., loon), wading (e.g., herons) and aerially foraging piscivores (e.g., eagle, kingfisher, tern), along with others such as rails, coots, cranes and pelicans. These

species are most strongly linked to aquatic systems, depending on these habitats for at least some life history attribute. Most are also relatively conspicuous, can be identified by sight alone and monitored outside of the breeding season. We do not recommend tracking riparian birds because the majority of birds found in the riparian zone will already be monitored under the terrestrial bird protocol (Moses et al. 2001). Our recommended target group also excludes shorebirds. Although shorebirds appear to be sensitive to anthropogenic disturbance (Fitzpatrick and Bouchez 1998, Page et al. 1999), they can be sporadic in occurrence and inconspicuous which make them difficult to monitor. Some current monitoring programs for aquatic birds in Canada are the Canadian Lakes Loon Survey (McNicol et al. 1995b), Prairie Shorebird Survey, Spring Waterfowl Breeding Population Survey and Waterfowl Brood Survey (U.S. Dept. of the Interior, Fish and Wildl. Service and Environment Canada, Canadian Wildl. Service 1987).

We recommend that the AFMBP focus on species richness of aquatic birds. However, counting birds in addition to identifying birds adds little extra effort to surveys and thus we suggest recording numbers of individuals of each species. Indeed, the relative abundance of certain species can provide useful information, such as cavity nesting buffleheads (Pierre 2001).

Sampling Alternatives

Typical methods of measuring aquatic birds include aerial or ground surveys (either on foot or by boat) (Bibby et al. 2000). Aerial surveys involve identifying and counting birds from a plane, often flown along transects over lakes. This method is used for the Spring Waterfowl Breeding Population Survey and the Waterfowl Brood Survey and is calibrated by ground crews (U.S. Dept. of the Interior, Fish and Wildl. Service and Environment Canada, Canadian Wildl. Service 1987). Ground surveys are often conducted on foot, either by walking along the perimeter of the water body or surveying from one or more vantage points along the shore and counting all visible aquatic birds (Wishart 1983). Some ground surveys involve flushing birds from shoreline vegetation. Boat surveys usually follow close to the shore of the water body in a boat using a small, quiet boat motor or paddling/rowing. The latter two types of surveys may be

supplemented by callback surveys to improve detection of secretive species, where a tape recording of birdcalls is used to solicit vocalizations (Gibbs and Melvin 1993). In addition to the floating platform (e.g., boat or canoe), binoculars are the main piece of equipment needed for surveys.

Besides the obvious costs associated with aerial surveys, species identifications can be difficult from the air and aerial surveys may be more effective in general on large water bodies. Ground surveys on foot are useful for small water bodies where launching a boat would disturb and flush the birds before they could be identified, but can be hampered by tall emergent vegetation or a lack of a good vantage point (Wishart 1983), as is our experience in surveying aquatic birds in forested regions of northern Alberta. Boat surveys are generally more efficient for larger water bodies. For surveys done by boat or on foot, crew members need to ensure that birds that flush and land elsewhere are not counted twice. Surveys are affected by time of day, weather and visibility.

Recommended Protocols

Considerations

Surveying only once or twice per year may not provide reliable density estimates. The number of birds on any given water body will be related to where birds are in the breeding cycle. Once pair bonds dissolve, male waterfowl form all-male groups and migrate to large lakes and marshes (Salomonsen 1968). This results in large density changes from spring to late June and early July. The presence of broods later in summer will also need to be accounted for. Thus, the timing of surveys needs to be considered if calculating bird density.

Sampling Method

We recommend conducting visual surveys aided with binoculars, and supplemented with callback tapes for the AFBMP's aquatic bird monitoring protocol. Crew members will need to be able to identify aquatic birds primarily by sight. Training for waterbird identification is considerably easier than for songbirds. Two different surveys will be necessary to accommodate very small ($\leq$5 ha) and larger (>5 ha) water bodies. On the smaller sites, a boat crew might prematurely flush birds, and thus

surveys should be conducted on foot from the shore, whereas a boat should be used for the larger sites. The ≤5 ha cutoff size is only a guideline - some water bodies larger than 5 ha with an appropriately shaped basin and good vantage points may be more efficiently surveyed from shore than by boat, and crew members should use their judgement in determining the best method. Secretive species in Alberta that should respond to callback tapes are the pied-billed grebe (*Podilymbus podiceps*), American bittern (*Botaurus lentiginosus*), Virginia rail (*Rallus limicola*), yellow rail (*Rallus limicola*), and sora (*Porzana carolina*). Because of the difficulty in counting hidden birds, we recommend classifying these secretive species as present or absent only.

For the small water bodies, each survey should begin approximately 100 m from the water body edge. Surveying from this distance will minimize flushing. Binoculars should be used to aid in identification. Once all possible identifications have been made at this distance, crew members should approach the water body as quietly as possible. They should proceed until a suitable vantage point is reached, which may include standing in the emergent vegetation. Here, another survey of the water body should be made. If some portions of the site are not visible, another vantage point for a second survey should be selected. Birds seen flying over the water body, rather than seen on or flying within the basin, should be specifically noted as such. The start and end time of the survey should be recorded. Once the visual survey is completed, callback tapes of the selected species should be played to attempt to elicit a response. Callback tapes should consist of 30 seconds of vocalizations interspersed with 10 seconds of silence.

For larger water bodies, surveys should be conducted by boat. The boat should be rowed along the perimeter of the water body or 75 ha plot, approximately 3-8 m from shore or emergent vegetation (e.g., Paszkowski and Tonn 2000). The distance each bird was sighted along the survey should be marked on a map of the water body, so that species richness can be calibrated per length of shoreline. We recommend this procedure because aquatic birds are typically associated with the shoreline and thus more appropriately indexed against length of shoreline than surface area. This will be relevant for the 75 ha plots where none, or only one side of the plot may consist of shoreline. Using a boat motor may be necessary in wavy, windy conditions. Both duration and distance traveled should be recorded, and binoculars used to aid in

identification. Callback tapes should be played every 400 m until presence is determined.

We recommend that surveys be conducted between 0800h and 1200h and if possible under favorable weather conditions, with temperatures in the range of 5-23°C, winds less than 24 km/h, and no precipitation or ground fog (Wishart 1983). Surveying the perimeter of a 75 ha water body will take approximately 1-1.5 hr (C.A. Paszkowski, unpublished data). There are a number of field guides suitable for identifying birds in Alberta, including Birds of Alberta (Fisher and Acorn 1998).

Table 13 Overview of sampling protocols to monitor aquatic bird communities from standing water bodies.

Attribute	**Sampling Details**
1. Sampling Method	Visual ground surveys Callback recordings for secretive species
2. Habitats	Entire water body or 75 ha area
3. Survey type	On foot for water bodies ≤5 ha By boat for water bodies >5 ha
4. Selection of stations	Visual surveys: entire water body Callback surveys: once on water bodies ≤5 ha every 400 m on water bodies >5 ha or until species detection
5. Number of surveys	One per visit to water body

Costs

Table 14 Cost estimates for equipment to monitor aquatic birds in standing water bodies. Costs are in Canadian dollars.

Sampling equipment	Cost
Binoculars (7 x 35 power)	200
Tape player, cassette of bird calls	50
Stopwatch	15
Bird field guide (two books)	60

Writing supplies (cost per site)		1
Sampling platform		
1) Road accessible	Sampling Platform	Cost
Small water body		
a) Shallow	14.5' canoe[a]	1120
	2 horsepower, 4 stroke motor	1000
b) Deep	10' aluminum boat	1000
	5 horsepower, 4 stroke motor	1700
Large water body	12' aluminum boat	1300
	5 horsepower, 4 stroke motor	1700
2) Remote access		
a) Small water body	10' inflatable boat	2000
	5 horsepower, 4 stroke motor	1700
b) Large water body	12' inflatable boat	2500
	5 horsepower, 4 stroke motor	1700

[a] 10' inflatable boat may also be effective in this habitat

2.4 ENVIRONMENTAL VARIABLES

2.4.1 Introduction

We recommend that the following environmental variables be measured in conjunction with measurement of biotic elements to: i) develop empirical models explaining variance in population and community characteristics and ii) assist with the characterization of water body types (See 3.2.1 to 3.2.3). The vast majority of variables described below (Table 15) can be determined using standard techniques and reflect the minimum number of variables that should be quantified.

Table 15 Environmental variables that should be measured along with biotic elements from standing water bodies.

Watershed descriptors	
a) Watershed area (km)	- Calculate from digital elevation model
b) Watershed slope	- Calculate from digital elevation model
c) Percent forest	- Derive from AVI or Phase III
d) Percent wetland	- Derive from 1:20,000 scale topographic maps, existing wetland maps or remote sensing data sources
e) Percent vegetated	- Derive from AVI or Phase III and/or remote sensing
f) Disturbance attributes	- Percent of watershed as linear disturbances (e.g., seismic, roads, rights-of way), % as patch disturbances (e.g., well sites, harvest blocks) - Derive from Base features (i.e., IRSS-based updates), 1:20,000 scale topographic maps, or remote sensing data sources

Site variables – habitat	
a) Water body shape and area	
- Large water bodies (e.g., >30ha)	- Derive from 1:20,000 scale topographic maps, existing wetland maps or remote sensing data sources
- Small or ephemeral water bodies	- Low intensity visual estimate based on field survey using laser range finder
b) Depth	
- Lakes: depending on size, bathymetric map of 75 ha plot or entire lake	- Depth finder along multiple transects
- Wetlands: water depth at sampling sites	- Sample site depths

c) Water volume	- Derive from bathymetric map and water body area and shape maps
e) Secchi depth	- Three measurements within open water habitats.
f) Shoreline complexity and number of inflowing and outflowing streams	- Depending on water body size, shoreline complexity should be calculated as the ratio of water body perimeter compared to that of the total area of the lake expressed as a circle. - Derive from AVI, 1:20 000 scale maps or remote sensing data
g) Macrophyte cover	- The type and extent to which macrophytes cover of the water surface based on visual estimates
Site variables- physical-chemical	
a) Total phosphorus b) Total nitrogen	- Two vertically integrated water samples from two deep locations in the water body
c) Dissolved organic carbon	- As described above
d) Colour	- As described above
e) Conductivity	- Measured *in-situ* using integrated samples and conductivity meter
f) Dissolved oxygen (DO), pH, light attenuation and water temperature profiles	- DO, pH, light, temperature meters attached to a graduated depth line. Estimates based on two vertical profiles within deep water area from the water surface to standing water body bottom

2.4.2 Recommended Protocols

Watershed Descriptors

We recommend that the AFBMP measure several watershed and water body descriptors in addition to monitoring biotic elements. The surface areas of the water body can be estimated from bathymetric maps whereas shoreline complexity and can be expressed as the ratio of the water body perimeter to the perimeter of a circle of the same area as the water body. Other morphological or landscape variables that have been calculated for other studies include the distance to next nearest water body, drainage basin area, percent coverage of the drainage basin by wetlands compared with upland deciduous or coniferous forest, and drainage basin slope. These attributes can be obtained from topographic maps or aerial photographs.

Site Descriptors: Habitat and Physical and Chemical Attributes

Weather conditions. Field protocols should also include descriptions of sampling time and periods and weather conditions including wind direction and relative strength, precipitation, cloud cover, air temperature and wave height. Instantaneous measures of barometric pressure during and immediately prior to sampling (up to 7 days) should be obtained from the nearest Provincial or Federal meteorological weather station.

Depth. In the absence of existing reliable data, we recommend developing depth profiles and measuring mean and maximum depth of each water body. Bathymetric maps can be developed by taking depth measurements using a depth sounder along predetermined transects prior to sampling. The number of transects strongly affects the accuracy of bathymetric maps but three to five transects in most cases should provide sufficient data. However, decisions on the number, location and orientation of transects should be based on an initial screening of the water body prior to sampling and decisions

made on site by the field crew. If the water body contains a long axis, the suggested method would be to establish one transect down the center of the water body, and at least two, equally spaced transects perpendicular to central axis. For rounder water bodies, transects laid out in a w-pattern are effective. Depth measurements should be recorded at consistent distances along each transect on pre-printed figures outlining the water body. This map can then be used when determining habitat and depth zones, and later digitized to create bathymetric contour lines using computer programs (See Prepas et al. 2001).

Secchi depth. Secchi depth provides a measure of water transparency and should be estimated using a standard Secchi disc consisting of a weighted, black and white disc with a 20-cm diameter that is deployed off the shady side of the boat. The secchi disc depth is the average of the two depths at which the disc disappears and reappears from view of the naked eye and should be measured at the deepest station in the water body. Secchi depth should be measured in addition to completing light attenuation profiles using a light meter.

Macrophytes. Macrophytes are an important habitat feature of aquatic systems and the type and cover provided by aquatic vegetation can strongly influence the structure of invertebrate, fish and waterfowl communities (e.g., aquatic birds, Fairbairn and Dinsmore 2001). We recommend that the AFBMP visually estimate the extent to which submerged, emergent and floating macrophytes cover the water surface within six 10 by 15 m plots located adjacent to the shoreline (Baker et al. 1997). The location of each plot should be randomly selected and these surveys should be conducted from the shoreline to 10-m offshore. Record water depth at the 10-m point. Cover of each macrophyte class within each of the 10 x 15 m plots should be recorded using the following semi-quantitative rankings:

a. Very heavy (>75% coverage)
b. Heavy (40 to 75% coverage)
c. Moderate (10 to 40% coverage)

d. Sparse (< 10% coverage)
e. Absent

Lastly, the extent to which macrophytes extend beyond the outer perimeter of the 10-m plot from the shoreline should also be recorded.

Water temperature, pH, dissolved oxygen and light profiles. Vertical profiles should be taken to describe changes in water temperature, pH, dissolved oxygen and light attenuation with water depth. Many multiprobe meters include probes to measure water temperature, pH and dissolved oxygen concentrations whereas light attenuation should be measured with a Licor meter or an equivalent product that is capable of measuring light availability simultaneously above and below the water surface (i.e., two light cells). Depending on water depth, measurement of water temperature, pH, dissolved oxygen and light should be measured at 0.25 to 1 m depth intervals depending on water depth (i.e., larger depth intervals in deep habitats [e.g., 1 m where depth <10 m], smaller depth intervals in shallow areas [e.g., 0.25 to 0.5 m intervals where depth <5 m]).

Water chemistry. The collection of water for chemical analyses is routinely completed as part of limnological studies and often plays an important role in interpreting biological data. We recommend that the AFBMP collect water samples for total nitrogen, total phosphorus, colour, conductivity and dissolved organic carbon determinations. Water samples should be collected at each of the 2 pelagic, or "deep" water stations. Samples should be taken using the same methods and equipment described for phytoplankton sampling (i.e., integrated water samples from the euphotic zone collected using a clear polyethylene tube with a 2.54-cm inner diameter, one-way foot valve and an attached lead weight). The tube is extended from the water surface to the bottom of the euphotic zone and a 250 mL volume of water is collected and stored in dark plastic bottles. Samples should be stored in a dark cooler on ice and to avoid contaminating samples, powder-free latex gloves should be worn during sample collection and workers should not place hands inside or on the lip of the water sampler.

Water samples should be transported to a laboratory and analyzed within three days of collection. Because samples need to reach a laboratory as soon as possible, they should be collected towards the end of the shift. We recommend that water samples be analyzed by a recognized laboratory using standard laboratory methods (e.g., American Public Health Association 1989, Wetzel and Likens 1991).

Table 16 Overview of sampling protocols to describe selected physical and chemical variables from standing water bodies.

Chemical Variables	Sampler	Locations
Total nitrogen	Integrated tube sampler	2 pelagic (or "deep") stations
Total phosphorus	Integrated tube sampler	2 pelagic (or "deep") stations
Dissolved organic carbon	Integrated tube sampler	2 pelagic (or "deep") stations
pH	Multiprobe meter	deepest station
Dissolved oxygen	Multiprobe meter	deepest station
Conductivity	Multiprobe meter	deepest station

Costs

Table 17 Costs to measure physical and chemical environmental variables from standing water bodies. Costs are in Canadian dollars.

Sampling equipment or analysis costs	Cost
Laboratory analysis: Total nitrogen[a] (cost per sample)	11
Laboratory analysis: Total phosphorus[a] (cost per sample)	10
Laboratory analysis: Dissolved organic carbon[a] (cost per sample)	10
250 mL sample bottle	3

Short tube sampler	45[b]
Long tube sampler	95[c]
Cooler and ice	50
Multiprobe meter	4000
Depth sounder (i.e., medium quality fish finder)	400
Secchi disc with calibrated line	90
Writing supplies	0.1
Powder-free latex gloves (cost per sample)	0.8

[a] Costs represent the Government of Alberta fee schedule when samples are processed at the Limnology Laboratory, Department of Biological Sciences, University of Alberta. Some laboratories may offer a small reduction in costs to analyze pre-filtered water samples.
[b] 4 m length, [c] 15 m length

2.5 A HYPOTHETICAL SAMPLING DESIGN FOR A SMALL WATER BODY

If the AFBMP monitors all six elements, sample points could be hypothetically resemble that shown in Fig. 8.

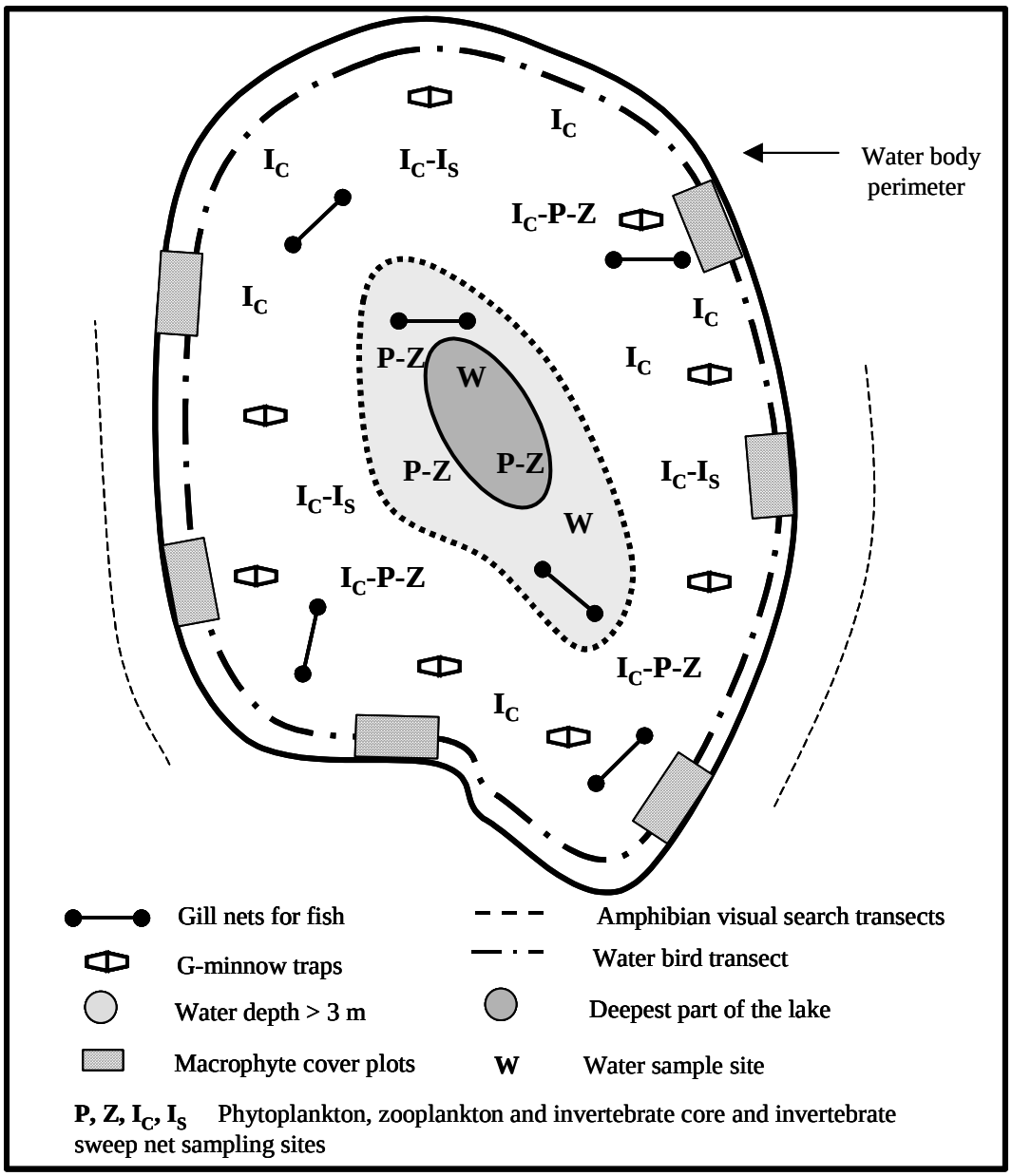


Figure 8 A hypothetical layout of sampling points to monitor phytoplankton, zooplankton, littoral invertebrates, fish, amphibians and aquatic birds within a 75 ha standing water body.

2.6 LOGISTICAL CONSIDERATIONS WHEN SAMPLING WETLANDS AND SHALLOW LAKES

2.6.1 Introduction

Standing water bodies within Alberta's forested region are dominated by lakes and bog and fen wetlands and small lakes. Wetlands comprise a diverse array of typically shallow water bodies that are saturated for sufficient lengths time to promote wetland or aquatic processes as indicated by poorly drained soils and hydrophybic vegetation (National Wetlands Working Group 1988). In general, wetlands can be divided into five classes and further separated into forms (Table 18) and physiognomy and vegetation types (i.e., Treed [coniferous, hardwood], Shrub [tall, low, mixed], Forb, Graminoid [grass, reed, tall rush, low rush, sedge], moss, lichen aquatic [floating, submerged, non-vegetated]. In general, wetlands differ based on: i) whether they are composed largely of organic or inorganic soils, ii) the location of the water table in terms of the wetland surface, iii) interaction of surface water with nutrient rich groundwater, and iv) the extent to which they are inundated by standing or slowly moving water. Wetlands are structurally variable and impose some logistical constraints because of their shallow but often variable water depths, and the presence of patches of open water interspersed with vegetation. Boreal plains wetlands often comprise peatland (bogs, poor fens, rich fens) and non-peatland swamps and marshes (Prepas et al. 2001).

The objective of this section was to identify and discuss a number of logistical considerations when sampling shallow lakes and wetlands. While surmountable, logistical constraints associated with sampling shallow lakes and wetlands lakes will increase program costs.

Table 18 Wetland classes and forms developed by the National Wetlands Working Group (1988).

Wetland Class	**Wetland Form**
Bog	Atlantic plateau bog, basin bog, blanket bog, collapse scar bog, domed bog, flat bog, floating bog, lowland polygon bog, mound bog, northern plateau bog, palsa bog, peat mound bog, peat plateau bog, polygonal peat plateau bog, shore bog, slope bog, string bog, veneer bog.
Fen wetlands	Atlantic ribbed fen, basin fen, channel fen, collapse scar fen, feather fen, floating fen, horizontal fen, ladder fen, lowland polygon fen, net fen, Northern ribbed fen, pals fen, shore fen,slope fen, snowpatch fen, spring fen, stream fen
Marsh wetlands	Active delta marsh, channel marsh, coastal high marsh, coastal low marsh, estuarine high marsh, estuarine low marsh, floodplain marsh, inactive delta marsh, kettle marsh, seepage track marsh, shallow basin marsh, shore marsh, stream marsh, terminal basin marsh, tidal freshwater marsh
Swamp	Basin swamp, flat swamp, floodplain swamp, peat margin swamp, shore swamp, spring swamp, spring swamp, stream swamp
Shallow water wetlands	Channel water, delta water, estuarine water, kettle water, non-tidal water, oxbow water, shallow basin water, shore water, stream water, terminal basin water, thermokarst water, tidal water, tundra water.

2.6.2 Logistical Considerations When Sampling Wetlands and Small Lakes

Safety

The AFBMP should consider that the safety of its field crew as the highest priority when designing and implementing a monitoring program. Sampling in and adjacent to flowing and standing water bodies raises concerns related to drowning, hypothermia, encounters with wildlife (e.g., moose, bears) and safety concerns related to

the use of quads and Argo's. We suggest that the AFBMP require all field staff to receive advanced training in: i) first aid, ii) cardio-pulmonary resuscitation (CPR), iii) emergency first aid, iv) water safety, v) use of quads and Argo's, vi) orientation and vii) dealing with problem wildlife.

Access to Sample Plots

Establishing all-season and summer roads in wetlands is expensive because large amounts of soils and bedrock need to be removed and replaced with stable material on which the road surface is developed. In addition, constructing roads within wetlands in many cases is socially unacceptable because of the their high ecological value. Unless there is no option, all-season public roads are often located in upland areas. As a result, access to wetlands in general, and the presence of the plot within large wetland complexes, creates logistical challenges associated with: i) transporting the field crew and sampling equipment to the plot and; ii) moving within the plot and describing prescriptive means of travel.

Because most of Alberta's large lakes are readily accessible by road, boats can be used to travel from the boat launch to the sample plot, whereas small, isolated lakes can be accessed using a Beaver, Cessna 185 or 206 fitted with floats. If sufficiently large and stable, beaver lodges can provide the field crew with a dry place from which to unpack equipment prior to sampling.

Depending on distance, wetland plots and small lakes could be accessed using all terrain vehicles (i.e., quads, Argo) and/or helicopters. In the majority of cases, large amounts of sampling equipment precludes walking from roads to sampling plots and the absence of a continuous area of deep (e.g., 1 m) unobstructed water (1 km) will likely preclude access using a floatplane. Depending on terrain, we recommend that the AFBMP consider accessing sites using quads and Argo's when distances from adjacent roads to the plot are relatively short (e.g, 1 to 10 km) but consider using helicopters when distances are substantially longer. In many cases, Argo's, and to a lesser extent, quads, towing light-weight trailers should provide an effective, means of transportation to wetland plots. However, crossing deeply incised streams can be problematic, imposes safety concerns and in these situations, access using helicopters may be required.

While effective in most settings, helicopters are expensive (e.g., $700 to $900 hour including fuel costs) and sampling costs can increase dramatically when: i) distances from the helicopter base to the plot are high, ii) large amounts of equipment require multiple slinging events and iii) field crews are deposited and retrieved from sample plots over several days.

Our experience in northern Alberta over the last 8 years indicate that access to wetlands will be strongly affected by the physical structure of the wetland including: 1) adjacency of upland habitats, 2) vegetation cover, 3) presence of domes, 4) size and position of treed plateaus within bogs and fens, and 5) the extent to which substratum consists of loosely consolidated peat or marl deposits. We strongly suggest that the AFBMP develop an access plan for each wetland prior to sampling using aerial photography, AVI, and data bases and local knowledge held by staff of forestry companies and organizations such as Alberta Conservation Association and Ducks Unlimited.

Moving Within the Water Body

Under most weather conditions, moving within small and large lakes in motorized aluminum boats or small inflatable rafts is not difficult and requires only standard pre-sampling considerations including an evaluation of safety considerations. Moving within many wetlands and small, shallow lakes is also not overly difficult but heterogeneous patches of dense vegetation and large amounts of field equipment increases the amount of time required to move between sampling stations. In these cases, canoes or small rafts can provide an effective platform but because they are less stable than larger boats, field crews will need to be a trained to increase their boating skills prior to the field season.

Delineating the Sampling Plot and Identifying Sample Points

We recommend that the AFBMP establish a square 75 ha plot around the systematic grid centroid as part of the pre-sampling screening exercise. Depending on the attributes of the plot, a set of randomly selected sampling points should also be selected. In deep water bodies, corners of the sample plot should be marked with temporary floating marker buoys. Plot corners within shallow water bodies, including the majority of wetlands, could be marked with conspicuously coloured 2 to 3 m stakes, and like floating buoys, removed after sampling.

Heterogeneous Habitats

Many wetland bogs, fens and swamps are highly heterogeneous in terms of water depth and often consist of patches of open water area interspersed with emergent vegetation. Sampling within physically heterogeneous habitats requires an initial planning exercise where the overall sampling plot is delineated, mapped, to some extent, followed by a process where sampling sites are randomly selected.

Reducing the Sampling Footprint

Wetlands are considered to be highly susceptible to physical damage such as that which could arise from access using quads and Argo's. We recommend that the AFBMP minimize the physical disturbance of wetlands when designing the sampling program.

CHAPTER 3 – DEVELOPING AN OVERALL SAMPLING DESIGN FOR AQUATIC SYSTEMS

3.1 INTRODUCTION AND APPROACH

Developing an integrated sampling design for aquatic systems requires:1) development of a process to distinguish between flowing water bodies and standing water bodies, 2) an understanding of the effects of separating flowing water bodies from standing water bodies affects the overall sampling deign, 3) the extent to which flowing water bodies are paired with non-flowing water body sites (i.e., standing water bodies and terrestrial sites), and 4) whether a process of post site stratification will be applied to the set of sites identified by the systematic grid. The related issue of how existing protocols can be applied to mixed sites (i.e., where both terrestrial and standing water bodies occur at the plot scale) is also an important consideration.

The objectives of this section were to: i) develop a process through which flowing water bodies could be distinguished from standing water bodies and terrestrial sites, ii) evaluate the extent to which the distinction between flowing and non-flowing streams affects the overall sampling deign, iii) the extent to which flowing water sites are paired with non-flowing water sites, iv) whether a process of post site stratification would be applied after the sampling grid established sampling sites and v) provide an update on the integration of aquatic and terrestrial monitoring protocols that includes discussions of sampling mixed habitats. Based on the terms of reference, our contractual obligations were to initiate and report on progress on discussions each of the above issues. To this end, members of our research group (Scrimgeour, Paszkowski, Tonn) met on 17 November 2001 with Chris Shank (Government of Alberta), Jim Schieck (Alberta research Council), Rich Moses (University of Alberta), Dan Farr (Biota Research) and Brad Stelfox (Forem Consulting) to discuss these issues.

3.2 ISSUE DISCUSSIONS AND STATUS

3.2.1. Distinguishing Between Flowing and Standing Water Bodies

After establishing the overall sampling grid throughout forested regions of Alberta, we recommend that the AFBMP discriminate among flowing water bodies (i.e., streams) from non-flowing water body sites (i.e., standing water bodies and terrestrial sites) using a hydrologically corrected single line hydrography layer. This can be accomplished by overlaying the stream hydrography layer on of the sampling grid to identify where points fall within stream channels. Conventional single line hydrography layers for small streams and rivers comprise a single line that reflects the stream channel. Because the line thickness is an artifact of the software package, it does not reflect the width of the stream channel. Selection of stream sites requires that the line be converted to an actual width that resembles the stream channel. This process is routinely completed using GIS tools where lengths of the stream between noches are attributed with stream widths predetermined by the researcher. In GIS terminology, this process is called buffering. Preliminary analyses of data for the Notikewin watershed (Paul Hvenegaard and John Tchir, Alberta Conservation association, NW Boreal Region, Peace River, Alberta, unpublished data) indicate that stream widths of 3, 4, 5, 9, and 15 m may be reasonable estimates for first to fifth order streams, although relations between watershed area and stream channel width may provide improved predictive equations. Queries of GIS databases should be completed to add these widths to the single line hydrography layer.

If this process identifies the sampling point as a stream channel, protocols developed for streams (Scrimgeour and Kendall 2000) should be applied. In contrast, if the site is identified as being a non-stream, a variety of both terrestrial and standing water protocols will be applied depending on the physical attributes of that site (i.e., the plot). The objective will be to apply as many of the non-lotic (i.e., both terrestrial and standing water body protocols) protocols as are feasibly possible within the sampling plot. With the exception of situations where the paucity of standing water totally precludes applying standing water protocols, both terrestrial and standing water body protocols will likely be applied at many sites, especially in the Boreal Plains where low gradients result in an abundance of standing water.

3.2.2 Effects of Separating Flowing Water Bodies From Standing Water Bodies on the Overall Sampling Design

Because stream features are linear and generally narrow compared to other habitat types (i.e., both standing and terrestrial sites) it is unlikely that the systematic grid will identify a large number of points within stream channels (i.e., stream sampling sites). Thus, the approach described above (Section 3.2.1) will unlikely affect the overall sampling design by identifying high numbers of stream sites that could potentially reduce the ability to report on changes in the biological diversity of non-stream habitats within Alberta's forested regions.

3.2.3 Pairing of Flowing Water Bodies With Non-Flowing Water Body Sites

Given their ecological and social importance, and responsiveness to watershed disturbances, members of the AFBMP Technical Committee identified the need to sample streams as part of the AFBMP and sampling protocols for streams were developed (Gingras et al. 1999, Scrimgeour and Kendall 2000). However, because the systematic grid approach currently endorsed by the AFBMP will identify few points within stream channels (i.e., stream sampling sites), the overall sampling grid would identify insufficient stream sites to allow it to report trends in biological diversity in stream ecosystems.

Based on discussions by the AFBMP Technical Committee, streams would be selected by pairing each of the non-stream points identified by the systematic grid with the closest stream site. The extent to which stream sites are paired (e.g., 1:1) or partially matched (< 1: 1 pairing) with non-stream sites has been debated in the past, and consensus on this issue has not been reached. This issue was discussed further on 17 October where the majority agreed that non-stream sites would be paired 1:1 with stream sites. Based on the existing systematic grid design, about 1250 stream sites would be sampled as part of the AFBMP and matched 1:1 with non-stream sites. Financial projections currently being used to determine the financial resources to support the AFBMP on an annual basis assume the 1:1 pairing of non-stream and stream sites.

An issue related to the integration of lotic and non-lotic sampling is how sampling teams will travel between the non-lotic and lotic sites. Costs to travel between these sites would be expected to be lowest if sites are adjacent so that sampling teams can walk between sites. Travel costs between lotic and non-lotic sites would be considerably higher if travel required the use of helicopters. The mode of travel between the non-lotic and lotic sites is an important driver of overall program costs.

Dan Farr and colleagues at the Foothills Model Forest completed an initial GIS query in 1999 to quantify distances between stream and non-stream sites. Using the 1:1,000,000 scale typography, they found that distances between non-stream and stream sites varied with natural region. While data collected at the 1: 1,000,000 scale maps would overestimate travel distances, compared with queries completed at higher resolutions, their data indicated average travel distances of up to 1-2 km between sites. Given difficulties in transporting equipment it is unlikely that sampling crews will be able to walk effectively between stream and non-stream sites. The question of how they move between sites is yet to be determined and should be considered as part of the overall description of site attributes when planning field operations.

3.2.4 Post Stratification of Sampling Sites

Inherent in the above, is that the current sampling design does not include a process of stratification of sampling points after they have been identified by the systematic grid, other than if the grid location is identified as a stream, an adjacent non-stream site will be identified and sampled as described above. Thus, non-stream sites identified by the systematic grid would be sampled irrespective of their physical attributes. However, after implementation data from the AFBMP would be used to report regional trends based on stratification of habitat type (e.g., ecoregion comparisons).

3.2.5 Integrating Terrestrial and Aquatic Protocols

Based on the terms of reference, the research team was asked to facilitate and support efforts to integrate aquatic and terrestrial protocols into the overall sampling design. The research team supported these tasks by : i) allocating project funding to Jim Schieck to organize the 17 November, and subsequently provide a summary of these discussions to Harry Stelfox, ii) reviewing meeting notes created by Jim Schieck and iii) providing Jim Schieck with draft copies of the Standing water protocols (Chapter 2), iv) reviewing sections of the report created by Jim Schieck that integrated aquatic and terrestrial protocols and v) discussing outstanding scientific issues related to the AFBMP.

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5. APPENDIX 1 – ADDITIONAL CONSIDERATIONS IN DEVELOPING MONITORING PROTOCOLS FOR FLOWING WATER BODIES

5.1 INTRODUCTION AND RATIONALE

Protocols to monitor the biological diversity of flowing water ecosystems have been previously described (Scrimgeour and Kendall 2000). However, developments over the last 2 years suggest that additional discussions of: i) the development of a functional stream hydrography layer, ii) utility of the Indian Remote Sensing Satellite (IRSS) data and iii) statistical analyses of monitoring data would assist with the development of the AFBMP.

The objectives of this chapter were three-fold. First we describe some of the difficulties associated with the developing of a functional stream hydrography layer from the Provincial Government's Base Features Geographical Information System. Second, we comment on the utility of the IRSS imagery to define linear disturbances. Lastly, we comment on several challenges associated with the sampling period, statistical analyses and data storage.

5.2 CONSIDERATIONS

5.2.1 Stream Hydrography

Development of a biomonitoring program for flowing water bodies needs to be completed in conjunction with the development of a single line hydrography layer and a digital elevation model. These databases are required to identify and describe stream networks, delineate watershed boundaries and attribute some of their physical properties. The following material describes some of the problems and solutions that the authors have experienced in developing functional stream hydrography layers and delineating watersheds in three study watersheds in northwestern Alberta.

The ability to identify and describe stream networks and delineate and describe physical attributes of their watershed boundaries are fundamental tasks required when developing a monitoring program for stream ecosystems. In Alberta, these tasks are accomplished using the Provincial Governments 1:20,000-scale single line network (SL-NET) and the Base Features digital elevation model (BF-DEM). While these data can be used to create a functional stream network layer, they require additional work to remedy a suite of errors and inconsistencies. Without such remedial actions, the resulting watersheds and their streams cannot be grouped, classified and attributed with an appropriate level of accuracy.

Creating a functional stream hydrography layer for over large areas requires merging multiple map sheets and the applying GIS process to:

1) Resolve errors in terrain and hydrography
2) Identify areas where the DEM and the SL-NET do not match well
3) Resolve errors in stream directionality and connectivity

Further, because political boundaries between Alberta and British Columbia were not based on watershed boundaries, headwaters of many of watersheds in western Alberta extend into British Columbia (e.g., Kakwa watershed). Watersheds that include areas in both provinces require additional work to: 1) delineate and quantify watershed areas and 2) attribute stream size (i.e., Strahler system). Depending on who is requesting the information, obtaining digital information from British Columbia can cost about $250.00- $300.00 for each 1:250,000 scale map sheet. Lastly, integrating stream hydrography layers in Alberta with those in other provinces can be complicated when such layers were created using different spatial scales. For example, single line hydrography layers in British Columbia are derived at 1:50,000 scale compared with the 1:20,000 scale used in Alberta's Base Features database. Differences in scale can produce different stream ordering and, over or underestimate, the occurrence of small streams.

5.2.2 Quantifying Linear Disturbances Using IRSS Imagery

Aquatic communities are strongly affected by watershed attributes including anthropogenic activities arising from forest harvesting and oil and gas activities. One of the primary objectives of the stream monitoring program is to quantify the extent of these and other industrial activities, and explain spatial and temporal variation in aquatic biota. Establishing these linkages requires information on the structure of aquatic communities as well as a suite of attributes that describe the physical structure of the watershed including anthropogenic disturbances. While a diversity of image types and sources are available, the Provincial Government, and a number of independent research studies, use geographic information systems (GIS) layers created primarily between 1996-1999 by the Government of Indian's Remote Sensing Satellite (IRSS imagery). These data were incorporated into the Alberta Government's Base features GIS database and can provide relatively up to date information on linear disturbance features, including roads, seismic lines, cut lines, railway lines and transmission rights-of-way. Information on harvest blocks (e.g., date, harvest method, areal extent) and silvicultural prescriptions may be obtained from the Forest companies after an assignment agreement has been established.

While the Provincial Governments Base features database, including the linear disturbance layers provided by IRSS imagery, is the best data set currently available, a number of challenges exist when using these data related to the spatial scale and age of the imagery.

IRSS imagery is 1:50,000 scale, black and white imagery that provides a 5.8 m spatial resolution. This resolution is less that the 1:20,000 imagery that forms much of the Base features database and may result in two main errors related to:
1) underestimation of small-scale disturbances and 2) misclassifications of small-scale features. Underestimation of small-scale features includes narrow cut-lines and low-impact seismic lines that do not result in detectable changes in overstory or ground vegetation. Because these disturbances are difficult to detect, identifying them as a specific feature (e.g., cut line versus seismic line) is difficult. In these cases, such features are typically identified as cut lines rather than seismic lines and results in

underestimates of the density and areal extent of seismic lines and an overestimation of cut lines. The extent to which these incorrect estimations influence the ability to explain variation in aquatic communities is not known.

The second concern when using Base features information is that it does not provide an up-to-date representation of existing landuse practices. In forested regions of Alberta, IRSS imagery was collected primarily between 1996-1999. Thus, while an accurate representation at that time, continued activities of forest harvesting, conversion of forests to agricultural lands and the rapid expansion of oil and gas activities has rapidly altered landscape patterns, at least at moderately small scales. In some watersheds located in foothills regions, the expansion of oil and gas exploration and extraction, related with high oil and gas commodity prices, has resulted in rapid increases in road networks, density of well sites and related infrastructure. Our preliminary comparisons of road networks quantified using IRSS compared with recent ground-truthing in July-August 2001 for the Simonette watershed located south east of Grande Prairie indicate a 2-12% increase in road density between 1998 and 2001, depending on the spatial scale at which analyses were performed and the location. In addition, many road networks have been established in sub-basins that were previously inaccessible by road. The extent to which the age of the IRSS imagery reduces our ability to account for variation in aquatic communities is poorly understood but is likely to be a substantial error in terms of watershed disturbance classifications.

5.2.3 Sampling and Statistical Considerations

The first tier of tasks completed by the AFBMP include: i) development of an overall sampling deign, ii) identification of criteria through which biological elements should be selected, iii) selection of potential predictor environmental variables, iv) development of sampling protocols, and v) completion of plot-scale test of sampling protocols. A second tier of issues that needs to be addressed by the AFBMP includes a suite of issues related to the statistical analyses other than deriving empirical relationships between sampling effort and mean and variance estimates of selected elements.

The objective of this section was to briefly discuss five issues associated with sample timing and the statistical analysis and storage of data collected by the AFBMP. Further evaluations of these, and other issues, would benefit the AFBMP by identifying the need to consider statistical analyses when determining the: i) number of plots sampled annually (i.e., spatial intensity of the program), ii) length of time between repeated sampling (i.e., the re-visitation period), iii) number of biological elements measured at each site, iv) number of within-site samples required to achieve a predetermined level of variance, and vi) extent to which stream sites are paired with non-stream sites (i.e., wetland and upland sites). These issues need to be addressed to ensure that the AFBMP is capable of detecting changes in the biological diversity of Alberta's forested regions given a certain level of financial resources.

Issue 1. When should benthic invertebrate samples be collected?

Many aquatic invertebrates have both an aquatic (larval) and terrestrial (adult) stage where larvae emerge from aquatic habitats, moult to a winged stage, mate and then die within days to weeks following emergence (Merritt and Cummins 1996). Females deposit fertilized eggs back into aquatic habitats where survivorship is thought to resemble a type III curve with extensive early mortality, but improved survival by those that survive early stages. Because the timing of the aerial adult stage is often species-specific and affected by local conditions, including water temperature and food availability (e.g., Waringer 1986, Zwick 1990, Petersen et al 1999) samples of benthic invertebrates collected from the same water body at different times of the year will differ to some extent (e.g., Reid et al. 1995).

The timing of sample collections has important implications to: i) identifying changes in biological diversity of aquatic invertebrates and ii) whether sites are identified as being impaired because of reduced statistical the power. In some cases, these errors could lead erroneous conclusions about whether a site has been impacted by industrial activities (e.g., Linke et al. 1999, Reece et al. 2001). To minimize temporal variance in population density, diversity and community composition, we suggest collecting benthic invertebrates should be collected over a relatively short period (e.g., a 6-week period) in the early fall (August-September) (David et al. 1998).

Issue 2. Multivariate versus metric approaches to identify impaired sites

A central tenet of biological monitoring is to identify sites that are impaired by anthropogenic activities and to develop, implement and monitor restoration actions that aim to repair such damage. Analytic techniques to identify impaired versus non-impaired sites using a reference condition approach (Bailey et al. 1998, Galatowitsch et al. 1999) can be classified into those that use a multimetric (i.e., i) multivariate assessment using an index that is the sum of several metrics or ii) a multimetric assessment using an index that is developed from a multivariate discriminant model; Kerans and Karr 1994, Barbour et al. 1996) compared with those that use multivariate tests (e.g., Ormerod and Edwards 1987, Rosenberg et al. 1999, Reynoldson et al 2001). The extent to which these approaches is most suitable continues to be a contentious issue in aquatic ecology (e.g., Norris 1995, Fore et al. 1996, Reynoldson et al. 1997). At some point, the AFBMP needs to consider which of the two approaches it will use to identify impaired sites. We recommend that the AFBMP use some of the data collected during the first 5 years of the program to compare and contrast predictive models based on a reference-condition approach derived using results from multimetric versus multivariate approaches.

Issue 3. Identifying the reference condition

An inherent challenge of both the multi-metric and multivariate approaches is the identification of the reference condition, that is, sites with none or minimal anthropogenic impacts. Once these conditions are established, statistical analyses are completed to quantify the probability that a site of unknown status (i.e., a test site) belongs to the population of reference sites (e.g., probability ellipses; Reynoldson et al. 2001). However, because biological conditions change at a variety of scales (e.g., ecoregion and sub-basin scales), reference conditions need to be established for a variety of biological settings. For example, to account for regional differences, the United States Environmental Protection Agency recommends that states classify lakes into categories and that reference conditions are established for each lake category. Our previous work (Prepas et al. 2001) has established within-ecoregion variance in lake

types in Alberta's boreal forest. These data indicate that lakes whose watersheds are dominated by wetlands differ markedly in water chemistry, and likely biotic communities, than those dominated by upland habitats (Prepas et al. 2001). Because these lakes have outflow streams, it is likely that stream chemistry and biotic communities in streams will also differ between wetland and upland dominated watersheds. The extent to which reference conditions need to be established for different stream types has important implications to statistical analyses because it affects sample sizes and thus statistical power and should be further evaluated by the AFBMP as part of its initial program costing evaluation.

Issue 4. Level of taxonomic resolution

Identification of benthic invertebrates is a time consuming and thus costly task and several studies have evaluated the effects of taxonomic resolution (e.g., Family versus Genus level), on the classification or ordination of benthic invertebrate community data. These studies have shown that the level of taxonomic identification does not necessarily affect resulting patterns in community types or their relationships with environmental variables, suggesting that cost efficiencies can be realized by identifying individuals to Family or Genus without adversely affecting information yield.

The level of taxonomic identification, however, is strongly affected by the overall objectives of the study, and in many cases, the level of taxonomy is based predominantly on classification of sites rather than quantifying biological diversity. In fact, the vast majority of sampling programs are designed to describe total faunal density or biomass or the density and biomass of the most abundant taxonomic groups with predetermined levels of variance. Relatively few studies are designed to quantify the overall diversity of the entire benthic invertebrate community that would typically require considerably large sample sizes (e.g., 20 to 50 replicate samples within each sampling unit) and species level identification of taxonomic groups that are known to be time consuming (e.g., Chironomidae, Oligochaeta). The extent to which the AFBMP is designed to: i) describe overall biological diversity, ii) track changes in the abundance of non-rare taxa (e.g., those comprising > 5% of all the entire community), or iii) to

quantify the extent to which anthropogenic activities affect community structure needs further discussion.

Issue 5. Data management

Data derived from a monitoring program typically includes biological, physical, geographical and chemical data types that need to be incorporated into a data base where the different data sources are relationally linked and several data management options have been discussed by the AFBMP Technical Committee. Here we provide an overview of the Ecological Data Application System (EDAS) developed by Tetra Tech, Maryland, USA (Tetra Tech Inc. 1999). While we are not necessarily recommending this software over other products, we do recommend that the AFBMP evaluate data management systems as part of the budgeting process.

EDAS is a relational database designed to manage data, calculate metrics and export of data in a variety of formats. EDAS will also complete basic descriptive statistics and some math functions and can be linked to ArcView® and Oracle®. Version 2 operates off Microsoft Access 97 and is capable of handling periphyton, benthic invertebrate, and fish community data and associated metrics. Version 3 is currently being designed and will likely be capable of linking to the Unites States Environmental Protection Agency data storage software (STORET).

EDAS Features

- Relational database designed to manage data, calculate metrics and export of data in a variety of formats
- Linkable to ArcView® and Oracle® and image files
- Microsoft Access® supports Open Database Connectivity, Structured Query Language
- Contains multiple database objects (tables, Queries, Forms, Modules)
- Performs basic descriptive statistics and math functions
- Handles multiple biological communities (i.e., periphyton, benthic invertebrate, fish)
- Flexible data manipulation through linked tables and fields

- User-friendly editing and data additions
- Based on Menu-driven interfaces
- Contains data tables capable of storing reference information on species (e.g., phylogenetic levels, functional feeding groups, pollution tolerance values)
- Capable of linking community data with environmental data
- Includes multiple data entry options (action query, direct input into a data table, data entry forms)
- Digital images can be added to data forms
- Multiple data retrieval options
- Calculates multiple metrics including measures related to taxonomy, best candidate, composition, feeding, habitat, richness, and pollution tolerance
- Includes data base management options of database compacting and repair
- Multiple field export options

5.2.4 References

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