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Maritimes Region

A Framework for the Assessment of the Status of River Herring Populations and Fisheries in DFO's Maritimes Region

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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TABLE OF CONTENTS

ABSTRACT

Alewife (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*) are diadromous species of fish that are collectively referred to as river herring. River herring return to many of the river systems in Nova Scotia and Southwest New Brunswick and are fished together as “gaspereau”. The fishery is geographically widespread, with fishing practices and gear types that differ among rivers, and is managed primarily through effort controls. Within the Maritimes Region, the status of river herring stocks has not been regularly assessed.

Towards the goal of developing an on-going monitoring and assessment program, this framework was developed to provide an overview of the spatial scale for assessment and identification of stock units, reference points against which status could evaluated, monitoring methods, analytic methods and research recommendations, taking into account DFO’s precautionary framework for fisheries management.

River herring have a high degree of fidelity to natal rivers, the populations of each species in individual rivers are considered to be discrete. Reference points are well developed for alewife and are defined on two axes: one that identifies whether overfishing is occurring, and one that identifies whether abundance is in the critical, cautious or healthy zones. Reference points for blueback herring have not been developed. Monitoring and assessment approaches differ depending whether data are fishery dependent or independent. With fishery dependent data only, in the short term, status can only be assessed relative to fishing mortality reference levels, whereas in the longer term, statistical catch-at-age models, which are well developed for at least alewife, can be used to estimate both abundance and mortality rates enabling status evaluations on both axes. In situations where fishery independent data (escapement counts) are available, status can be evaluated on both axes in the short term, whereas in the longer term, application of statistical catch-at-age models would be expected to lead to an improved understanding of population dynamics leading to improved advice. Research that helps to apportion landings from mixed-stock fisheries to specific stocks, as well as research leading to an improved understanding of how other human activities affect stocks (e.g. efficiency of fish passage facilities, survival at dams, increased habitat in reservoirs) is anticipated to significantly improve advice within the precautionary framework.

Cadre pour l’évaluation de l’état des populations de gaspareau et des pêches dans la Région des Maritimes du MPO

RÉSUMÉ

Le gaspareau (*Alosa psuedoharengus*) et l'alose d'été (*Alosa aestevalis*) sont des espèces de poissons diadromes qui sont appelées de façon collective « gaspareau ». Les gaspareaux reviennent dans plusieurs réseaux hydrographiques de la Nouvelle-Écosse et du sud-ouest du Nouveau-Brunswick et ils font l'objet de la même pêche. La pêche est répandue sur le plan géographique; les pratiques de pêche et les types d'engins varient selon les rivières et la pêche est principalement gérée par l'entremise de contrôles de l'effort. Dans la Région des Maritimes, l’état des stocks de gaspareau n’a pas été évalué régulièrement.

Dans le but d’établir un programme de surveillance et d’évaluation continue, ce cadre a été élaboré afin d’offrir un aperçu de l’échelle spatiale pour l’évaluation et l’identification d’unités de stock, de points de référence par rapport auxquels le statut pourrait être évalué, de méthodes de surveillance, de méthodes analytiques et de recommandations de recherche qui prennent en compte le cadre de précaution du MPO pour la gestion des pêches.

Le gaspareau présente un degré de fidélité élevé pour les rivières natales. La population de chaque espèce est considérée comme discrète dans des rivières distinctes. Les points de référence sont bien mis en place pour le gaspareau et sont définis sur deux axes : un qui détermine s’il y a surpêche et l’autre qui détermine si l’abondance se trouve dans les zones critiques, zones de prudence ou zones saines. Aucun point de référence n’a été élaboré pour l’alose d’été. Les approches de surveillance et d’évaluation varient si les données sont dépendantes ou indépendantes de la pêche. À l’aide des données dépendantes des pêches seulement, à court terme, le statut ne peut être évalué qu'en fonction des niveaux de référence de la mortalité par pêche; alors qu’à long terme, les modèles statistiques de prises selon l'âge, qui sont bien élaborés au moins pour le gaspareau, peuvent être utilisés pour estimer l'abondance et les taux de mortalité, ce qui permet d'effectuer des évaluations de statut sur les deux axes. Dans les situations où les données indépendantes de la pêche (dénombrements d’échappées) sont accessibles, le statut peut être évalué sur les deux axes à court terme. À long terme, l’utilisation de modèles statistiques de prises selon l'âge devrait permettre d'obtenir une meilleure compréhension des dynamiques des populations et de fournir de meilleurs avis. Les recherches qui aident à répartir les débarquements de pêches allant des stocks mélangés aux stocks précis, ainsi que les recherches entraînant une amélioration de la compréhension des répercussions des autres activités humaines sur les stocks (p. ex. l’efficacité d’installations de passe à poisson, la survie aux barrages, l’augmentation des habitats dans les réservoirs) devraient améliorer de façon significative les avis en utilisant le cadre de précaution.

1.0 INTRODUCTION

Alewife (*Alosa pseudoharengus*) and blueback herring (*Alosa aestivalis*) are diadromous species of fish that are collectively referred to as river herring. They are indigenous to the Maritime Provinces and the eastern United States.

River herring return to many of the river systems in Nova Scotia and Southwest New Brunswick and are fished together as “gaspereau”. The fishery is geographically widespread, with fishing practices and gear types that differ among rivers, and is managed primarily through effort controls.

Fisheries for river herring are of local economic value. Reported landings in the Maritime Provinces (since 1960) peaked in 1980 at just less than 11,600 t, and averaged 6,231 t between 1997 and 1999 (DFO 2001). The Saint John and Miramichi rivers in New Brunswick produce the largest river herring yields in North America (Schmidt et al. 2003).

In addition to their importance economically, river herring are important species ecologically. They are prey species at sea and in fresh water, and are important predators that can alter zooplankton community composition within lakes (Mills et al. 1992, Gibson and Daborn 1998). They can also serve as a vector for nutrient transport from the oceans to inland waters (Durbin et al. 1979, Garman 1992, Garman and Macko 1998). As a result, human activities such as fishing and the construction of dams that impact upon river herring population size may indirectly alter the productivity and community structure within their natal watersheds (Freeman et al. 2003).

Both the species-specific distribution and status of river herring populations in the Maritimes Region are not well known. Rulifson (1994) surveyed fisheries biologists to determine the distribution and status of anadromous *Alosa* in eastern North America. Within the Maritimes Region, 129 of 131 rivers identified in the survey were reported to contain alewife and 105 rivers were reported to contain blueback herring. The majority of the alewife populations were reported to be “in decline”, whereas the majority of blueback herring populations were reported to be “stable” or “status unknown”. The basis for these evaluations is unclear, given that relatively little data exists for most of these populations.

River herring populations and fisheries have not been regularly assessed in the Maritimes Region. The last river herring assessment in the Maritimes Region was for alewife in the Gaspereau River (Kings County, NS) in 2007, and there has not been a regional assessment since 2001. In 2001, some stocks in the Maritime Provinces exhibited characteristics of over-exploited stocks (Robichaud-LeBlanc and Amiro 2001). Data collection for the assessment of Maritimes Region river herring stocks since that time has been sporadic.

Towards the goal of developing an ongoing river herring assessment program, this document was prepared to provide an outline of a general framework for collecting and interpreting data pertaining to river herring populations and their fisheries. It begins with information about the spatial scale on which populations are structured and upon which assessments should take place. It is well established that stock assessments should produce estimates of quantities such as fishing mortality rates and abundance or biomass that are indicative of status, which can then be compared to reference levels (or reference points) to evaluate the status of the stock and its fishery. There has been considerable research on reference points for alewife in the Maritimes Region, and reference points consistent with DFO’s precautionary framework (DFO 2006), are proposed in Section 3 of this document. Options for data collection to estimate abundance and biological characteristics of river herring populations are discussed in Section 4, as are analytical methods to estimate abundance and biological characteristics. Data requirements and research recommendations to implement the framework are provided in Section 5.

Specifically, the following four objectives for DFO's "Maritimes Region River Herring Framework and Case Study Application to the Tusket River Fishery" advisory process are addressed in this document:

- Characterize the appropriate spatial scales for assessment and advice that takes into consideration population differentiation, variability among the various fisheries, as well as other factors that may affect abundance. Evaluate the potential to develop Index Rivers for regional assessment and advice.
- Evaluate the options for data collection and the associated assessment methods that are possible in different regions or rivers; specifically, as related to fishery-independent abundance metrics (total abundance estimates and/or relative indices, characterization of the fishery, and the development of reference points).
    - Consider the effect of other human activities on rivers (e.g., dams) and how these would impact the above.
- Provide advice on how each run should be sampled in order to estimate abundance, quantify the commercial landings, and determine species composition, age and size distributions and other characteristics of the run.
    - Focus on sampling design within a year (i.e. sampling constant numbers (daily, weekly) vs. sampling proportional to abundance).
    - Focus on sampling frequency among years (e.g. every year, every other year and so on) required in order to detect trends in these characteristics or to appropriately evaluate status.
- Provide research recommendations to address uncertainties and gaps in the assessment framework.

The final objective:

- Evaluate the data collection and assessment methods as applied to the alewife and blueback herring populations in the Tusket River and the resulting determination of their status.

is addressed in a separate document (Bowlby and Gibson 2016). The majority of information presented in this document pertains to alewife, reflecting the greater amount of information available for this species. Many of the knowledge gaps for blueback herring identified here could be expected to be addressed if the data collections described in this document were implemented.

1.1. BIOLOGICAL OVERVIEW

Alewives and blueback herring are sympatric throughout much of their range (Loesch 1987), although blueback herring have a larger and more southerly range (Nova Scotia to Florida) than alewife (Labrador to South Carolina). The species have similar life cycles (Figure 1.1). Adult river herring migrate up coastal rivers in the spring (late-March to late-June) for spawning, with the majority of the combined runs returning in May to rivers in Nova Scotia and southwest New Brunswick. Adults spawn in fresh water during the spring, after which they return to the ocean. Young-of-the-year river herring move downstream in the late summer and early fall to winter at sea. The fish mature at two to seven years of age at which time they return to the rivers to spawn. Alewife and blueback herring are iteroparous (multiple reproductive cycles) and in non-impacted populations may spawn as many as four to six times throughout their lives.

Although their spawning periods overlap, alewives may begin their spawning run 2 to 4 weeks earlier than blueback herring. Spawning runs are thought to be highly structured, with older and larger individuals returning first, and smaller first-time spawners coming later in the run. In rivers in the Maritimes Region, alewives typically begin spawning in May, and may continue to do so over two months (Scott and Scott 1988), utilizing ponds, lakes or slow-flowing portions of streams and rivers as spawning habitat. In areas were spawning seasons overlap, the two species are isolated by the use of different spawning sites (Loesch and Lund 1977), with blueback herring spawning in areas with faster moving water (Loesch 1987). Less is known about the habitat preferences or distribution patterns of adults in the marine environment, but it is thought that there is broad mixing among species as well as populations originating in Canada and those in the United States.

River herring juvenile ecology appears highly variable. When spawned in moving water, the eggs and larvae of both species are transported downstream. In Lake Ainslie, NS, a non-tidal freshwater lake, juvenile alewives and small numbers of blueback herring are present in the lake until late August when they begin to move downstream to the estuary (O'Neill 1980). Young-of-the-year alewives and blueback herring are present in the non-tidal freshwater headpond above the Mactaquac Dam during July through September and at least to late October (Jessop and Anderson 1989). Migration from this headpond probably begins in late August, and increases rapidly through early September (Jessop 1990). In the Chesapeake Bay area, juveniles are distributed throughout tidal fresh water during spring and early summer, and may move upstream in the summer with the encroachment of saline water (Warinner et al., in Loesch 1987). In the Annapolis River, NS, a river without lakes or impoundments, alewife are present in the estuary at salinities of about 30 in July (Gibson 1996).

1.2 DESCRIPTION OF THE FISHERIES

River herring fisheries are relatively unique in terms of the number of participants, the diversity of licensed gears and their geographic extent. Where sympatric populations exist, they are harvested together in rivers or estuarine fisheries. Licensed fishing gears include square nets (currently Gaspereau River only), trap nets, and dip nets that are primarily fished above head-of-tide, as well as set and drift gill nets in estuaries, coastal waters or large lakes. Licenses are typically issued for more than one gear type within a specific geographic region and since 2009 have been subject to mandatory reporting through logbooks.

Information from the commercial fishery has historically been partitioned by Fishery Statistical District (FSD – Figure 1.2.1) rather than by river, and there are instances where information from a single river may come from multiple FSDs. There are currently four established regional Advisory Committees (Southwest New Brunswick, Yarmouth/Shelburne County, Lunenburg/Queens County and the Gaspereau River), although landings are reported from other FSDs in Nova Scotia and New Brunswick.

Data from the “older” logbook program (see Section 5) from 2000 to 2007 were used to characterize the spatial distribution of the fishery. For a high level visualization, the region was subdivided into seven areas, and the average annual reported landings (uncorrected for non-reporting) were calculated for each area (Figure 1.2.2). For the 2000-2007 time period, reported landings in Southwest New Brunswick exceeded those in the Maritimes Region portion of Nova Scotia. Within the Nova Scotia portion of the Maritimes Region, reported landings were highest in the Yarmouth-Shelburne area. The total number of unique individuals reporting commercial landings during this time period was 521, the greatest number of which (Figure 1.2.3) were in the Yarmouth-Shelburne area. These numbers do not indicate the number of licenses available or the total number of individuals fishing in a specific year, largely because specific licenses change hands. With the exception of non-reporting, they are indicative of the number of people participating in the fishery during this time period.

Within each area, catches differ from river to river (Figure 1.2.4). In eastern Cape Breton, the largest reported landings came from around the Bras d’Or Lakes (Figure 1.2.4), which also had the greatest number of participants reporting having fished (Figure 1.2.5). Along the eastern shore of Nova Scotia, the Musquodoboit River had the highest reported landings (Figure 1.2.4). The greatest number of participants in this area was in Ship Harbour/Lake Charlotte, although the participants are distributed throughout the geographic area (Figure 1.2.5). With Lunenburg and Queens County, both the greatest number of participants and the largest reported average landings are from the Medway River. In Yarmouth and Shelburne Counties, the majority of the reported landings for the 2000 to 2007 time period were from the Tusket River (Figure 1.2.4), although there are many participants reporting from other locations, such as Eel Lake/Kiack Brook (also called Belleville) and Annis River (Figure 1.2.5). For other areas in southwest Nova Scotia, both the highest number of participants and the highest landings were from Salmon River (Digby Co.). Within the inner Bay of Fundy, the highest reported landings come from the Shubenacadie and Gaspereau rivers (Figure 1.2.4), with the highest number of reporting participants fishing in the Shubenacadie River (Figure 1.2.5). In Southwest New Brunswick both the highest reported landings and the highest number of reporting participants are in the Saint John River and its tributaries.

2.0 SPATIAL CONSIDERATIONS AND STOCK UNITS FOR ASSESSMENT

For the purposes of assessment and management, there are several ways that fish stocks can be defined. Here, we define a stock as: a population or subpopulation of a particular species of fish, for which intrinsic factors, such as growth, reproductive rates, carrying capacity, natural mortality and mortality caused by human activities, can be the significant factors determining the stock's population dynamics, while extrinsic factors such as immigration and emigration are minimal and can be ignored. This definition aligns closely with the goals of assessment, which are to determine whether mortality rates and abundances (i.e. intrinsic factors) are within appropriate limits.

2.1 STOCK UNITS

There is strong evidence that anadromous *Alosa* home to natal rivers, including tagging studies that show homing to rivers of previous spawning in both American shad and river herring, as well as genetic studies that show substantial differentiation among samples collected from different rivers (McBride et al. 2014, Palkovacs et al. 2013). For smaller river systems, monitoring of river herring populations at the spatial scale of the river system would be expected to produce data that could be considered representative of an individual stock, although in some larger systems, such as the Saint John River in New Brunswick, monitoring programs at the tributary scale might be more likely to result in data representative of a stock.

Thus, the commercial river herring fishery contains multiple components: targeted fisheries on individual populations, targeted fisheries on rivers with both species (mixed-species), as well as targeted fisheries on rivers that would likely contain multiple populations (mixed-stock fisheries that may also be mixed-species). As such, there would be areas where fisheries could be monitored, but interpretation of that data would be problematic because it would not be specific to a single stock. However, there are other locations, that if monitored, one could be much more certain that the data being collected is representative of abundances and mortality rates of a specific stock.

2.2 POTENTIAL FOR INDEX RIVERS

Developing an assessment and management approach for river herring based on index rivers for monitoring presupposes that there are broader scales at which assessment results from individual populations could be considered representative of other populations at the broader scale. There are two aspects to this:

1. identification areas where all populations would be expected to be similar to each other; and
2. identification of areas where the magnitude of human impacts on river herring populations could be considered similar for all populations in the area.

Relative to regional variation, research using genetic tools has shown distinct population structuring. McBride et al. (2014) studied the genetic diversity, differentiation and population structure of alewife and blueback herring in the northeast USA and eastern Canada. They examined the 34 alewife populations and four blueback herring populations (Figure 2.1.1) using 14 microsatellite loci. They found significant differentiation among most rivers, and identified 8 alewife population clusters, several of which are represented in the Maritimes Region (Figure 2.1.2). Of the Bay of Fundy populations, the Saint John River population grouped more closely with the Gulf of St. Lawrence and Nova Scotia Atlantic coast populations, suggesting it differs from the other Bay of Fundy populations included in the study. Two genetic groups were identified in the other Bay of Fundy populations, one of which was predominant in the Gaspereau River, while the other was predominant in the Shubenacadie River and Petitcodiac River. Similarly, within the Nova Scotia Atlantic coast populations, the southernmost populations (Tusket, Argyle and Kiack Brook) appear differentiated from other populations on this coast. The eastern Cape Breton (Bras d'Or Lakes) populations appear differentiated from the Nova Scotia Atlantic coast rivers. Although delineating clusters is somewhat subjective, these results demonstrate a high level of variability among populations, regions and rivers, supporting the idea that river-specific populations are relatively discrete and should be assessed separately.

With respect to the extent that human activities within a region might be expected to be similar, as shown in Section 1.2, the magnitude of fishing activities (both the number of participants and the landings) varies among rivers that are in close proximity. Additionally, other activities, such as hydroelectric development, are present in some rivers and not others. As such, it is presently unclear the extent to which an assessment on a single river could be indicative of status of populations in other rivers, although it may be possible to extrapolate using catch-per-unit effort and watershed size to make these kinds of inferences. This possibility could be explored once data from several populations become available.

For the purposes of this framework, subdivision of the Maritimes Region into smaller geographic units and selection of one or more populations in each unit for monitoring is recommended. Based on the magnitude of landings and the genetics information above, geographic units in Nova Scotia could consist of eastern Cape Breton, the Eastern Shore, Lunenburg and Queens County, Yarmouth and Shelburne Counties, and the Bay of Fundy. In southwest New Brunswick, geographical units could include the Saint John River and its tributaries, and Bay of Fundy rivers outside the Saint John River watershed. Rivers in which monitoring can occur are discussed in Section 6, where the key consideration is the extent to which the sampling would be expected to produce results indicative of a stock.

3.0 REFERENCE POINTS CONSISTENT WITH DFO'S PRECAUTIONARY FRAMEWORK

There is much more information and data upon which to derive reference points for alewife fisheries in the Maritimes Region than there is for blueback herring. In this section, reference points for alewife, consistent with DFO's precautionary framework, are developed. Reference points for blueback herring are deferred until more population specific data are available. While the developed reference point framework closely follows DFO's precautionary framework (described below), it is extended to include a second fishing mortality (or removal) reference point to better characterize when overfishing is occurring, versus when a population is fully exploited or under exploited, as discussed in Section 3.2.

3.1 BACKGROUND

Biological reference points (RPs) are reference levels, based on the biological characteristics of a fish stock and the characteristics of its fishery, against which a metric estimated for the stock or fishery can be compared in order to determine the status of the stock and its fishery. They are used to gauge whether specific management objectives are being achieved and provide both the link between stock assessment and management objectives (Caddy and Mahon 1995), and a basis for risk analysis of management actions (Punt and Hilborn 1997).

Fisheries and Oceans Canada (DFO) has adopted "A fishery decision-making framework incorporating the Precautionary Approach (PA)" that is to be used where decisions regarding commercial, recreational, or subsistence controls on harvest and all other removals are required (Figure 3.1.1). The primary components of the PA (DFO 2006) are:

1. Reference Points (RPs) and stock status zones (healthy, cautious and critical).
2. Harvest strategies and harvest decision rules.
3. The need to take into account uncertainty and risk when developing reference points and developing and implementing decision rules.

The stock status zones are created by the Limit Reference Point (LRP) defining the critical to cautious boundary, an Upper Stock Reference (USR) defining the cautious to healthy boundary, and the removal reference for each of the three zones (DFO 2006).

The LRP ideally represents the stock status below which serious harm occurs to the stock and should be well above the level where the risk of extinction or extirpation is likely. Negative impacts to the ecosystem and long-term loss of fishing opportunities also influence the selection of the LRP. Serious harm in this context can result from either human-induced mortality or changes in population dynamics not related to human activities (DFO 2006).

The USR defines the point at which removals must begin to be reduced in order to avoid reaching the LRP. To achieve this objective the USR must be high enough in comparison to the LRP to provide sufficient time for management actions to be implemented and to have the biological effect of promoting stock increases (DFO 2006).

A Target Reference Point (TRP) may also be defined in the PA framework. It is defined to be equal to or greater than the USR and represents a stock status goal that the management system promotes (DFO 2006).

The adoption of the LRP, USR, and TRP for any stock involves a combination of biological, social, and economic considerations. A completely non-arbitrary method of determining the specific abundance where serious harm will occur to a given stock does not exist. The arbitrariness arises from uncertainty in the biological data and changes in society's perception of acceptable risk. Thus, biological, social, and economic considerations all carry weight in the definition of all these RPs. However, there is generally greater emphasis on biological considerations at the LRP and greater emphasis on the social and economic considerations at the TRP, and considerations for the USR fall somewhere in between depending on the particular fishery and the biological dynamics of the species (DFO 2006). Within the Maritimes Region, 40% of the biomass at maximum sustainable yield (B_{msy} is sometimes proposed as the LRP, whereas 80% of B_{msy} is sometimes used as the upper stock reference level (DFO 2012).

A removal reference is also defined relative to the stock status zones as the maximum acceptable removal rate from all types of fishing and other human activities. This rate in each of the three zones (critical, cautious and healthy) should not exceed the removal reference in the healthy zone. The removal reference will vary depending on the stock's location in each of the zones. It may also be influenced by factors other than those associated with stock status such as ecosystem effects, recruitment expectations, and other indicators of harvest pressures on a stock. As a result, adjustments of the removal reference in any zone do not have to be linear, and the precise shape under any given circumstances will be determined during the establishment of Harvest Control Rules (HCRs) (DFO 2006). The fishing mortality rate that produces maximum sustainable yield (F_{msy}) is often used in the Maritimes Region to establish removal reference levels (DFO 2012).

Within the USA, removal reference levels were calculated during the first coast-wide Atlantic States Marine Fisheries Commission (ASMFC) assessment of Atlantic coastal river herring stocks (Crecco and Gibson 1990). The authors recommended that in-river fishing rates should be kept 20 - 30% below μ_{msy} levels for all stocks because ocean losses to a given stock are usually unknown. Reference removal levels were provided in terms of an exploitation rate (μ), which equates to the instantaneous fishing mortality rate, F, as $\mu = 1 - e^{-F}$. Stocks were considered fully exploited if the exploitation rate was within 75% of $\mu_{msy}$and partially exploited if μ was less than 75% of μ_{msy}. Reference levels during the second coast-wide ASMFC assessment of Atlantic coastal river herring stocks (ASMFC 2012a, ASMFC 2012b) were established analogously using a reference total mortality rate rather than fishing mortality, in part because fisheries were closed in some states, and removals were not well quantified for some others. The approach was modified slightly in some cases based on the available data and specific requirements of individual states.

Here, biomass reference points that can be used as LRPs and USRs to assess the status of alewife (whether abundance is in the critical, cautious or healthy zone) are developed, as well as the fishing mortality reference points that can be used determine whether over fishing is occurring. The emphasis is on alewife because the productivity of this species is better understood, in part due to data availability. A similar framework could be developed for blueback herring after data collection for this species are initiated.

3.2 FISHING MORTALITY REFERENCE POINTS

Gibson and Myers (2003a) derived reference points for four alewife populations in the Maritimes Provinces using several methods, including: yield per recruit analyses (Beverton and Holt 1957), spawner biomass per recruit (SPR) analyses (Shepherd 1982, Mace and Sissenwine 1993), replacement-based methods (Sissenwine and Shepherd 1987, Quinn and Deriso 1999), a life cycle-based production model (Gibson and Myers 2003a, 2004), decision-theoretic methods (Ianelli and Heifetz 1995, Gibson and Myers 2004), and simulation approaches (Gibson and Myers 2003a, 2004). In all analyses, the fisheries are assumed to occur only on mature fish as the fish are returning to spawn, consistent with the majority of river herring fisheries in the Maritime Provinces (the reference points would not be appropriate for fisheries in the marine environment that include both mature and immature fish). Parameter estimates required for the reference point calculations (e.g. natural mortality, maturation schedules) were obtained by fitting a statistical catch-at-age model to the data available for the four alewife populations, as described in Gibson and Myers (2003a). For the decision-theoretic methods, Gibson and Myers (2003a, 2004) compared several methods of deriving the joint probability distribution for the SR parameters. Only the results of the method that uses both the population-specific likelihood surface and the species-level probability distribution for the SR parameters to derive their posterior probability density are reproduced here. Definitions of the reference points are provided in Table 3.2.1, values from their analyses are reproduced here in Table 3.2.2.

Although nearly all of the reference points listed in Table 3.2.2 have either been used or proposed for use in stock assessments, each has its strengths and weaknesses. A major disadvantage of yield per recruit analyses is that it does not account for the effects of exploitation on the fish stock, as evidenced by the near infinite value obtained for F_{max} for the four alewife populations (Table 3.2.2). Spawner-biomass per recruit (SPR) reference points are widely used for stocks without an informative stock-recruitment relationship, but selecting the appropriate percentage of SPR is problematic if the productivity of the stock is not known (Mace and Sissenwine 1993). Although simulation-based reference points could be used, their value is dependent on assumed values for process variability and autocorrelation, which are typically not known. The fishing mortality rate that produces maximum sustainable yield, F_{msy}, does integrate the stock-recruitment relationship, yield per recruit analyses and spawner biomass per recruit analyses, thereby accounting for population productivity, the growth of fish and the effects of fishing on the spawner biomass. For this reason, F_{msy}, calculated using the maximum likelihood estimates (MLEs) of the stock recruitment parameters, is commonly used when it can be calculated. However, when the stock recruitment relationship is poorly determined, the maximum likelihood estimate of F_{msy} can lead to over- or under-exploitation of the stock (Figure 3.2.1). In order to address the issue of poorly determined stock-recruitment relationships, Gibson and Myers (2004) proposed a decision theoretic reference fishing mortality rate, $F_{max.E[C]}$, found by maximizing the expectation of the yield. Via simulation, they showed that fishing at $F_{max.E[C]}$ provided higher equilibrium yields (on average across populations) than did fishing at the MLE of F_{msy}, while at the same time reducing the risk of over-exploitation of the stock (Figure 3.2.1). Because this reference point:

1. does address the limitations of using references points based only on yield per recruit or spawner biomass per recruit analyses,
2. does better account for uncertainty in the stock recruitment relationship when estimating F_{msy}, and
3. does, across populations, produce higher equilibrium yields than the MLE of F_{msy}, it is used as the upper removal reference level in this framework.

Estimated values of $F_{max.E[C]}$ were 0.75 for 3 of the analyzed populations, and 0.82 for the fourth population (Gibson and Myers 2003a). This higher value was from the Gaspereau River, a population that is estimated to have a higher level of natural mortality and is influenced by hydroelectric development (see Section 3.5). The use of *F*=0.75, equating to an annual exploitation rate of 0.53, is proposed as the upper removal reference level in this framework, above which populations would be considered to be overexploited.

The simulation-based reference point derivation of Gibson and Myers (2003a) highlight two important aspects of alewife fisheries:

1. when random variability is included in the model, the resulting reference points are lower (e.g. compare F_{msy} and $F_{max.mean[C]}$ for the Gaspereau and Margaree River alewife fisheries in Table 3.2.2), and
2. the yield curves are relatively flat (e.g. Figure 3.2.1).

The effect of the relatively flat yield curve is that reference exploitation rates can be reduced somewhat with only a small change in the yield in the fishery. For example, a comparison of the exploitation rates associated with $F_{max.mean[C]}$ and $F_{90\%.mean[C]}$ for the 4 populations indicates that, on average, reference exploitation rates can be reduced by 33.3% with only a 10% reduction in yield. Together, these aspects suggest that lower fishing mortality reference points are precautionary relative to the high recruitment variability exhibited by alewife without a large sacrifice in yield to the fishery. Consistent with the recommendation of Crecco and Gibson (1990) that in-river fishing rates should be kept 20 - 30% below μ_{msy} levels for all stocks because oceanic and other losses to a given stock are usually unknown, alewife populations in this region could be considered fully exploited if their exploitation rate is between 35% and 53%. Below 0.35, populations would be considered underexploited.

3.3 BIOMASS REFERENCE POINTS

Establishment of biomass reference levels for a river herring population requires to things:

1. some estimate of the productive capacity of the system, and
2. some criteria for establishing reference levels.

As criteria, the use of SSB_{msy} is proposed as the USR, and 10% the unfished equilibrium biomass ($SSB_{10\%}$) is proposed as the LRP. Information must be available, or assumptions must be made, about the underlying dynamics of the populations in order to calculate values for these reference values. Within the Maritimes Region, the Gaspereau River alewife population is currently the only population with sufficient data to estimate biomass reference points using population-specific data. However, there is information that can be used to derive reference values for other systems.

Information about the productive capacity of alewife habitat is available in the literature. Gibson and Myers (2001, 2003a, 2003c) and Gibson (2004) conducted a meta-analysis of maximum lifetime reproductive rate and carrying capacity of alewife at the species level based on the simultaneous analysis of data from eight alewife populations. Each variant of the analysis includes a further refinement or inclusion of more data. This analysis, the final version of which is available in Gibson (2004), provides probability distributions for these two parameters that can be used as proxies to estimate the unfished equilibrium biomass (SSB_0) for systems without sufficient population-specific data.

The random effects distribution for the log of the maximum lifetime reproductive rate has a mean of 2.96 and a standard deviation of 0.13, suggesting a median maximum lifetime reproductive rate of 19.2 replacement spawners per spawner. The random effects distribution for log of carrying capacity has a mean of 3.94 and standard deviation of 0.42 (Figure 3.3.1). These estimates suggest a median carrying capacity of about 51 t/km^2 of nursery area (Gibson 2004).

The carrying capacity estimates of Gibson (2004) are estimates of the recruitment asymptote defined as the limit of lifetime recruitment as the spawner biomass approaches infinity (Gibson 2004). This value is an important input in a population dynamics model; however, in the context of fisheries reference points, a more directly applicable value is the equilibrium biomass in the absence of fishing mortality (or other anthropogenic mortality), a value that can be calculated from the results of the meta-analysis. Gibson and Myers (2003a) provide the derivation of the equilibrium spawning stock biomass in the absence of fishing mortality (SSB_0) given:

1. a Beverton-Holt stock-recruitment relationship parameterized in terms of the maximum reproductive rate (the maximum of the ratio of the number of recruits to the spawner biomass, α) and the asymptotic recruitment level (in terms of the number of recruits in the cohort, R_{asy}), and
2. the spawner biomass per recruit in the absence of fishing mortality ($SPR_{F=0}$):

$$B_0 = \frac{(\alpha SPR_{F=0} - 1)R_{asy}}{\alpha}$$

Multiplying both the numerator and denominator of that equation by $SPR_{F=0}$, yields:

$$B_0 = \frac{(\alpha SPR_{F=0} - 1)R_{asy}SPR_{F=0}}{\alpha SPR_{F=0}}$$

Here, the terms $\alpha SPR_{F=0}$ and $R_{asy}SPR_{F=0}$ are the maximum lifetime reproductive rate and the lifetime recruitment asymptote (carrying capacity) provided in Gibson (2004). Using the median values for these parameters, it follows that, for a typical alewife population, the unfished equilibrium spawning stock biomass is 94.7% that of the carrying capacity. Although the use of the median estimates of carrying capacity from Gibson (2004) are proposed as the basis for deriving biomass reference points here, the analyses of Gibson (2004) do show that carrying capacity can be highly variable from population to population. Estimates of the unfished median equilibrium spawning stock biomass, and its 25th and 75th percentiles on a per unit area basis are provided in Table 3.3.1.

Population specific values for SSB_0 can be calculated by multiplying the values in Table 3.3.1 by the amount of habitat within the watershed utilized by the population. A lower limit reference point can be calculated by taking 10% of the resulting value. Obtaining SSB_{msy} is a bit more problematic because it depends not only on the carrying capacity of the population, but its productivity, which incorporates its maximum reproductive rate, growth, maturity and natural mortality, as well. Of the four populations for which reference points were analysed by Gibson and Myers (2003a), two (the Gaspereau and the Margaree) could be considered suitable for calculating the ratio of $SSB_{msy}$to SSB_0. For these populations, the ratio averaged 0.1485. In the absence of population-specific data, a proxy for the upper stock reference can be obtained by multiplying the median SSB_0 by this ratio (Table 3.3.1).

3.4 THE REFERENCE POINT FRAMEWORK

Values for the reference point framework are provided in Table 3.4.1 and an example of the framework for evaluating status is shown in Figure 3.4.1. Optimally, a population and its fishery would be near the intersection of the limit lines drawn at $\frac{\mu}{\mu_{RLL}} = 1$ and $\frac{Esc}{ESC_{USR}} = 1$, or just to the right of and just below this point. Populations and their fisheries with their values in the upper left part of the graph indicate that the population is in the critical zone with respect to abundance, and that overfishing is occurring. If a population and its fishery remain in the lower left part of the graph for several years, it would indicate that something other than fishing in the river and its estuary is affecting abundance. If a populations and its fishery are in the lower right part of the graph, there is the potential to increase the exploitation rate. For illustrative purposes, the results of the 2007 Gaspereau River alewife assessment are shown on the graph. In both the early 1980s and in the late 1990s, the exploitation rate was very high relative to the reference levels and the abundance was low. During the early 2000s the exploitation rate decreased and the spawning escapement increased. This occurred after the construction of a new fishway at the White Rock Generating Station and a reduction in fishing effort from five days per week to four days per week.

Interpretation of status evaluations within this framework is not dependent only on the status evaluation in an individual year, but also on longer changes that are occurring within a watershed or with regard to the management of the fishery, as well as annual variability. For example, if, for an existing population, access is provided to new habitat, abundance would appear low relative to the total amount of habitat prior to the population increasing in size. Similarly, annual variability would be expected to be high. If management changes are implemented, the status figure provides a good mechanism for evaluating the response of the population and the fishery, as shown above.

3.5 EFFECTS OF DAMS AND HYDROELECTRIC DEVELOPMENT ON FISHERIES YIELDS AND REFERENCE POINTS

Construction of dams and development for hydroelectric generation are two human activities which are prevalent that have the potential to impact on river herring populations, to affect fisheries yields and, importantly, reference points for the assessment and management of these fisheries. Fundamental to assessing the effects of human activities other than fishing are the timing of activities within the life cycle (e.g. does mortality occur before or after spawning) and relative to other human activities (e.g. does fishing occur upstream or downstream of dams), or both. Here, we consider the effect of dams on carrying capacity of freshwater habitat, of downstream passage mortality at hydroelectric generation stations, and of upstream fish passage efficiency.

Gibson (2004) developed a population dynamics model for alewife that can be used to evaluate downstream mortality by incorporating an instantaneous rate of turbine mortality for juvenile fish (T^{juv}) or adult fish (T^{adult}). The first part of this model (shown below), gives the contribution of first-time spawning fish (top part of the equation; if *p=0*) and repeat spawning fish (lower part of the equation; if *p>0*) to the number of fish returning to the river in year *t*, of sex *s,* of age *a*, that have spawned *p* times previously ($N_{t,s,a,p}$). The parameters describing the life cycle include the instantaneous fishing mortality rate in a given year (F_t), the sex ratio (υ_s), the instantaneous natural mortality rate for immature fish at sea (M^{juv}), the maturity schedule ($m_{t-a,s,j}$), the sex and age specific instantaneous rate of natural mortality for adults ($M^{adult}_{s,a}$), the spawner biomass (SSB_{t-a}), and a two-parameter spawner-recruit model where α is maximum number of recruits per unit biomass of spawners (at low abundance in the absence of density dependence) and R_{asy} is the asymptotic recruitment level. The full dynamical model, slightly adapted from Gibson (2004) is:

$$N_{t,s,a,p} = \left\{ \begin{array}{c} \frac{\alpha SSB_{t-a}}{\left(1+\frac{\alpha SSB_{t-a}}{R_{asy}}\right)} e^{-T^{juv}} \upsilon_s m_{t-a,s,a} e^{-M_a^{juv}} \prod_{j=0}^{j=a-1}\left(1-m_{t-a,s,j}\right) \;\; if\ p=0 \\ N_{t-p,s,a-p,0} e^{-\left(\sum_{k=t-p+1}^{t} F_k + T^{adult}p + M_{s,a}^{adult}p\right)} \; if\ p>0 \end{array} \right\}.$$

Assuming a non-selective fishery, the life cycle can be closed as:

$$SSB_t = \sum_{s,a,p} (1-u_t)\, N_{t,s,a,p} w_{s,a,p}$$

where $w_{s,a,p}$ are the weights of fish in each sex, age and previous spawning category.

From this model, a few things are evident. First, as discussed above with respect to biomass reference levels, if the amount of accessible habitat for alewife is increased as the result of construction of a dam, all other things being equal, the equilibrium population size would be expected to increase in proportion to that increase. Based on the meta-analysis used to derive biomass reference levels, the increase of a “typical” alewife population would be expected to be 51 mt/km^2, although the variance associated with this value is high. Second, because juvenile turbine mortality is occurring after density dependence but before either fishing or maturation and spawning, it would be expected to reduce the returns to the river by $1 - e^{-T^{juv}}$. Third, where adult turbine mortality occurs after spawning, its effect on population size and fishery yields would be less than that of juvenile turbine mortality even if the rates were the same.

Gibson (2004) used this model to examine the relationship between turbine mortality, fisheries yields and fishery reference points using the Gaspereau River population as an example. In this system, all fishing occurs downstream of the hydroelectric generating stations, and all spawning habitat is located upstream. He used the population dynamics parameters provided in Gibson and Myers (2003a) for this population, and used grid searches to find the fishing mortality and yields at MSY for levels of juvenile and adult turbine mortality from zero to 100% (Figure 3.5.1). His results show how equilibrium catches and spawning escapements at MSY decrease with increasing mortality. In the case of juvenile turbine mortality, F_{msy} decreases as juvenile turbine mortality increases. Because adult turbine mortality occurs after reproduction, sustainable fisheries may exist even if all adult fish die during turbine passage. In the case of 100% adult turbine mortality, equilibrium yield at MSY is reduced to 75.3% from its level in the absence of turbine mortality, whereas 100% turbine mortality reduces the spawner biomass at MSY to 49.0% its level without turbine mortality. In contrast with juvenile turbine mortality, F_{msy} increases slightly with increasing adult turbine mortality. Although the pattern would be expected to be similar for other populations, the equilibrium yields, spawner biomass and reference points would be expected to differ if different life history parameters were used.

Although not explored here, Gibson’s approach can be adapted to other watersheds and other questions. For example, if spawning habitat was available downstream of the generating station, the effects of turbine mortality would also be a function of the proportion of the total available habitat that is upstream and downstream of the generating station, and density dependent effects in both locations would need to be modeled separately. The effects of upstream fish passage efficiency, which can be highly variable, can be modelled similarly. In the case of the Gaspereau River population, where very little successful reproduction is thought to occur downstream of the generating station (Gibson and Daborn 1998), a reduction in fish passage efficiency would be expected to reduce the spawner biomass proportionately. The population-level effects can then be modelled as described above. In a system with habitat downstream of the dam, all other things being equal, the effect of changes in fish passage efficiency would be dependent on the proportion of the habitat upstream and downstream of the dam, in addition to the efficiency of the fish ladder.

In summary, humans impact river herring populations in many ways, and, as illustrated with the example of hydroelectric generation provided above, the evaluation of fisheries, including expected yields and spawning escapements, as well as assessment and management parameters for the fisheries are dependent on these other impacts. On a positive note, population dynamics and fisheries assessment models are available that can be adapted to watershed- and population-specific situations once other impacts are quantified. In the examples above, these are upstream fish passage efficiency and downstream passage mortality for both adult and juvenile river herring.

4.0 ESTIMATING ABUNDANCES AND MORTALITY RATES

Ultimately, the goal of implementing this framework is to provide estimates of abundance, fishing mortality and total mortality that can be used to evaluate the status of the populations against the reference levels provided in the previous section. When developing this framework, two broad-scale approaches for the program were considered. The first was to evaluate the potential for and utility of monitoring an individual population and its fisheries for one or two years to evaluate its status, and then to move to another population. The major advantage of this approach is that it would provide information about the many populations within the region, albeit for a brief period of time with longer periods with no information. The second approach was to choose a set of populations and fisheries that could be monitored annually over the long term. The major advantage of this approach is that the monitoring data could be used to evaluate changes in life history parameters (e.g. stock-recruitment parameters, survival, growth) that affect the dynamics and productivity of populations to ensure status assessments are appropriate.

While developing the framework, it became evident that the first approach (short-term status assessments) would only be applicable in situations where annual abundance can be directly measured (typically only rivers with fishways where counts can be obtained). This is based in part on the need for abundance estimates to fully evaluate status using the reference point framework, and secondly, that even if status was only assessed relative to fishing mortality rates, the catch curve analyses that would be used to estimate mortality do not work well for relatively short-lived species with high recruitment variability (simulations that demonstrate this are provided in the next section). As such, implementing a one or two year assessment approach for the populations and fisheries on rivers where abundance cannot be directly estimated would be problematic, precluding assessment of some larger fisheries. For these reasons, the remainder of this document focuses on long-term data collections on individual populations in order that both abundance estimation and analyses of population dynamics are possible. With respect to the estimation of abundance and mortality, and the evaluation of status, the approaches are expected to differ depending upon whether obtaining a count is possible on the individual river. For populations for which counts can be directly obtained, fishing mortality rates can be estimated directly from the abundance estimates and landings (e.g. Gibson and Myers 2001, McIntyre et al. 2007, Bowlby and Gibson 2016). For populations for which counts cannot be directly obtained, the objective would be to develop data collections sufficient for using statistical catch-at-age models of Gibson and Myers (2003a,b) to estimate both abundance and fishing mortality rates. In the shorter term, advice on the status of fisheries for these populations would be provided via catch curve analyses subject to the uncertainties discussed below. These options are further discussed in Section 6.

4.1 CATCH CURVE ANALYSIS FOR ESTIMATING MORTALITY IN RIVER HERRING

Background

Catch curve analysis, or the use of age frequency data within a year, has a long history of use for estimating mortality in fisheries biology (Chapman and Robson 1960, Ricker 1975) and the methods for using age-frequency data continue to be studied and improved (e.g. Smith et al. 2012, Millar 2015). Assumptions underlying the analyses include (Smith et al. 2012): recruitment is constant or at least varies without trend; mortality is constant over time and age classes; above some age (which can be difficult to identify), all animals are vulnerable to the fishery and the sampling process; and there are no errors in the estimation of age composition. Despite their being several aspects of the ecology of *Alosa* (river herring and American shad) that could lead to violations of these assumptions, catch curve analysis has been widely used in assessments for these species (e.g. Melvin et al. 1983, ASMFC 2007, ASMFC 2012a,b). For example, recruitment has been shown to be highly variable (Crecco and Gibson 1990, Gibson and Myers 2001, 2003a), and, in the case of riverine alewife fisheries or monitoring, age-specific abundance is also influenced by age-at-maturity, which can also be highly variable (Figure 4.1.1). Annual exploitation rates have been shown to be variable in virtually all assessments (e.g. Crecco and Gibson 1990, Gibson and Myers 2003a). Choosing the youngest age class to use in the catch-curve analysis is problematic if the maturity schedule is not known, and even if known, can lead to the exclusion of data if the partially mature age classes are excluded. Additionally, particularly when mortality rates are high, there are relatively few age classes for fitting a model.

In the context of statistical catch-at-age models for riverine *Alosa* fisheries, Gibson and Myers (2001, 2003a,b) introduced the use of the number of previous spawnings as a metric for time (as an alternative to age). This approach has several advantages for riverine *Alosa* fisheries, including addressing the issue that total mortality rates are expected to be very different for mature and immature fish, and that the data can be modelled as a set of sub-cohorts (one for each age-at-maturity) with model parameters in common, thereby strengthening the analysis by increasing the number of observations. In the case of catch-curve analysis, adopting this approach could also strengthen the analysis by increasing the number of observations, but also has the major advantage that data from all age classes can be used (there is no need to discard the data for the youngest (partially mature) age classes).

Here, a population-simulation model was developed to evaluate the utility of catch curve analyses for estimating mortality for river herring, and to compare catch curve analyses based on age alone, or using both age and previous spawning history. The analysis is based on the simulation model developed by Gibson and Myers (2003a) for evaluating the effect of variability in age-at-maturity, based on their analysis of the dynamics of alewife in the Margaree River, NS. Random variability is introduced into the model in both recruitment and age-at-maturity.

The Simulation Model

The model equations, slightly adapted from (Gibson and Myers 2003a), are:

$$R_{t+3} = \frac{\alpha SSB_t}{1 + \frac{\alpha SSB_t}{R_{asy}}} exp\left(\varepsilon_t \sigma - \frac{\sigma^2}{2}\right) \ where\ \varepsilon_t \sim N(0,1)$$

$$N_{t,a,0} = R_{t-a+3} m_{t,a} e^{-M_{a-3}^{juv}}$$

See below for the calculation of $m_{t,a}$.

$$E_{t,a,p} = N_{t,a,p}(1 - \mu_t)$$

$$N_{t+1,a+1,p+1} = E_{t,a,p} e^{-M^{adult}}$$

$$C_{t,a,p} = N_{t,a,p}\mu_t$$

$$C_t = \sum_a \sum_p \left(N_{t,a,p}\mu_t\right)$$

$$SSB_t = \sum_a \sum_p \left(E_{t,a,p} w_a\right)$$

Here, ε_t are the annual recruitment deviates, σ is the recruitment variance, $E_{t,a,p}$ is the spawning escapement of fish in year t, of age a, that have spawned p times previously, w_a is the weight-at-age, and other parameters are as described earlier.

The parameters α, R_{asy}, and σ were obtained from an SR model (Gibson and Myers 2003a), M^{adult} and the mean maturity schedules, m_a, came from their model results for the Margaree River alewife population (Table 3 of Gibson and Myers 2003a). Following their approach for incorporating variability in the age-at-maturity, we first mapped the probability that a fish that is alive at age *a* matures at age *a* to the real line using a logistic transformation:

$$logit(m_a) = log\left(\frac{m_a}{1 - m_a}\right)$$

For each cohort, $logit(m_a)$ was calculated for age classes 2 to 5, and the mean and standard deviations of $m_a$for each age class were calculated to characterize the maturity process. A random component was introduced on the logistic scale by drawing a random number from a normal distribution with the mean and standard deviation above. This value was back-transformed to obtain the random $m_{t,a}$:

$$m_{t,a} = \frac{exp\big(logit(m_a) + \varepsilon_{t,a}\big)}{1 + exp\big(logit(m_a) + \varepsilon_{t,a}\big)}, \text{where } \varepsilon_{t,a} \sim N\left(0, var\big(logit(m_a)\big)\right)$$

The model was used to simulate data that was used to evaluate how effective catch curve analyses are for estimating mortality rates for river herring. The model was used to project numbers-at-age in the spawning run forward for 100 years. Fifty simulated population trajectories were carried out for each scenario analysed. Mortality rates were estimated for each of the final 75 years in each simulation. A total of 12 scenarios were analysed, including all combinations of four levels for the exploitation rate (0.00, 0.25, 0.50, 0.75) and three levels of recruitment and age-at-maturity variability (no variability, full variability as estimated by Gibson and Myers (2003a), and an intermediate scenario with the variances half way between the two). For each year in each simulation, datasets on which to apply the catch curve analyses where derived using all the data, and by subsampling using sample sizes of 100, 200, 500 and 1000 samples randomly drawn from the population. This approach resulted in 18,750 catch curves per scenario. The same sets of random numbers were used for each scenario to avoid the possibility of among-scenario differences occurring by chance.

For each catch curve dataset, the total instantaneous mortality rate (Z) was estimated two ways, both using a generalized linear model assuming a Poisson distribution and a log link function. The first model:

$$E(N_a) = \exp(\log(N_0) - Za),$$

is a “typical” catch curve analysis where the expected number of fish of age a (N_a), is modeled as a function of the number of fish of age-0 (N_0) and Z.

The second model is the extension to incorporate previous spawning history:

$$E\big(N_{\tau,ps}\big) = \exp\big(\log(N_{\tau,0}) - Zps\big),$$

where $N_{\tau,ps}$ is the number of fish that matured at age τ, and ps is the number of previous spawnings (zero for a first-time spawner). Here, there is a separate intercept term $N_{\tau,0}$, for each possible age-at-maturity.

Results and Discussion

Results of the simulation analyses from both analytical approaches are shown in Figure 4.1.2 (age as the independent variable) and Figure 4.1.3 (number of previous spawnings as the independent variable). Several things are evident from these figures, beginning with the observation that, at least using these methods, catch curve analyses are not really appropriate for estimating mortality rates for alewife, particularly at higher mortality rates and at the higher levels of variability characteristic of the species. Estimated total mortality rates are lower (i.e. negatively biased) than the true rates used to simulate the data. In comparing the methods, it is evident that the model incorporating the previous spawning data is both less biased and less variable, and, while these advantages are apparent in all scenarios, these are most noticeable at high levels of mortality and variability. Neither method is biased when perfect information is available about the age structure, although estimates can be quite variable even in this case. With respect to sample size, the magnitude of the bias increases with decreasing sample size. When estimating mortality when the exploitation rate is in the range of 50% in the medium variance scenario, the gain in accuracy when increasing sample size from 500 fish to 1000 fish is relatively small in comparison of increasing from 200 to 500 fish. In most cases, increasing sample size does increase precision although the precision remains low at high sample sizes when variability is high.

It may be possible to improve on these results. There are other methods available for analysing catch curves, but no one method has been demonstrated to be optimal in all situations. Millar (2015) provides an excellent comparison of these methods for fish stocks generally, using age as the independent variable. He shows that using a mixed effects model with random effects for age can help address the issues of recruitment variability (the random effect on age essentially allows a separate intercept for each age class). Initial attempts to use this approach for alewife showed some promise both with age as the independent variable and with number of previous spawnings (with random effects on number of previous spawning nested within age-at-maturity), but did not work consistently when applied in the simulations, possibility due to the small number of age classes for estimating the random effects. This approach warrants further exploration. Weighted regression approaches have also been recommended (Maceina and Bettoli (1998), as has truncating the number of age classes (as discussed in Smith et al. (2012)), both to address the issue that the proportions-at-age of older age classes (which have high influence in the regression) are the most difficult to accurately estimate. There are too few age classes in river herring data for truncation to be practical. Robust Generalized Linear Models (a weighting method that is unbiased relative to errors in the data; e.g. Cantoni and Ronchetti (2001)) using a Poisson distribution and a log link function were attempted in place of the generalized liner models (GLMs) for fitting the simulation model. The approach also showed some promise, but did not work consistently in all simulations.

With respect to this framework for assessing the status of a river herring stock, the results have several implications. First, they highlight the difficulties that would be encountered if short term (1 to 2 year) data collections were used to evaluate status in populations where only landings and age composition data are available. As an assessment approach, it is not really feasible, particularly given the other sources of variability (e.g. annual exploitation rates) not included in these simulations. In the longer term, statistical catch-at-age models (described below) have many, many advantages over catch curve analyses. While the minimum number of years of data required to fit these models has not been evaluated, depending on the types of data collected) the catch curve analyses may have a role in the interim if the results are interpreted in the context of the issues illustrated here. For example, based on these simulations, if the exploitation rates estimated from a catch curve are above the removal reference level (0.53), there is a high probability that the true exploitation rate is above that level. However, if the estimated exploitation rate is below the reference level, there is still the potential that the true rate would be above the removal reference, given the direction of the bias

4.2 STATISTICAL CATCH-AT-AGE MODELS

Long term data collections for specific populations have the advantage that they can be modelled using statistical catch-at-age models appropriate for river herring (e.g. Gibson and Myers 2003a,b). This approach has the advantages that variability in annual survival and exploitation, recruitment variability and variability in age-at-maturity are explicitly modelled and accounted for in the assessment.

Statistical catch-at-age models have a history in fisheries biology spanning more than 30 years. Fournier and Archibald (1982) and Deriso et al. (1985) developed the general theory for statistical catch-at-age models for stock assessment that allow auxiliary data to be incorporated into the model. The advances by Gibson and Myers (2003a,b) include that of extending the two dimensional matrix (age and year) typically used for marine stock assessments into a three or four dimensional space that also includes previous spawning history and, if available, sex information. This approach has several advantages, the foremost being that the life history of interest is specifically modelled and that the equations can be adapted to include data collections that are population-specific. For river herring, data such as spawning escapement counts at fish ladders, larval and juvenile abundance indices, counts of emigrating juveniles, previous spawning history, indices of the number of post-spawning fish (Olney and Hoenig 2001), and information about other sources of mortality can be incorporated into the assessment process.

One of the most useful pieces of information often collected for river herring, and that is not typically available for marine species, is the number of times that a fish has previously spawned. This is available from a fishes' scales. Riverine impacts such as fishing or turbine passage do not affect immature fish at sea. When the number of previous spawnings is known, this variable can be used to determine the number of times that the fish has been exposed to riverine impacts via the addition of an extra dimension to the catch-at-age array. Additionally, when data are partitioned by sex and age-at-maturity as well as age, the number of observations of a cohort each year increases (from one to eight for a population that matures over four years), greatly strengthening the power of the model. Assuming an adequate sample size, this increase improves the researcher's ability to estimate mortality rates or other parameters that are held constant across these categories.

Finally, a major advantage of this model is that it allows the full dynamics of a population to be analyzed, providing much more information than is often available when models are not used (see Gibson and Myers 2003b). For example, if an assessment is carried out using only an estimated *Z* (or *F*) and status is determined only relative to a reference *Z* (or *F*) as would be the case if only a spawner-biomass per recruit reference point was used, changes in productivity in a population in the pre-recruitment life stages would go undetected. In this situation, a population could crash even if the estimated values of *Z* (or *F*) were at acceptable levels.

A third method for estimating fishing mortality exists for rivers where there are both escapement estimates and estimates of the landings in the fishery. In these instances the exploitation rate in a given year (μ_t) can be calculated as:

$$u_t = \frac{C_t}{C_t + Esc_t},$$

where C_t is the harvest in year *t* and Esc_t is the escapement in year *t*. Biases may exist depending on how the data are collected. If, for example, escapement is measured at a fish ladder, then the efficiency of that ladder can introduce a bias if not all fish that escape the fishery ascend the ladder. In a case where all fishing occurs downstream of the ladder, total abundance would be $C_t + Esc_t/p$, where p is proportion of the escapement that is passed upstream by the ladder, rather than just $C_t + Esc_t$. In the context of the DFO precautionary framework, the escapement could be evaluated relative to the reference levels, because the framework is intended to incorporate all removals, not just those from the fishery (DFO 2006). However, the exploitation rate estimate from the fishery in this situation would be confounded with p and information about this parameter would be needed to fully interpret the exploitation rate (however, the ratio of the catch to the escapement would still be high, and therefore indicative of an issue).

In summary, a goal of the framework should be to work towards ongoing assessments on several individual populations in order to be able to apply better assessment methods. In the interim, catch curve analyses may have a role in providing advice, although their interpretation will be situation-specific and at times may not be clear.

4.3. MONITORING FOR EVALUATING BIOLOGICAL CHARACTERISTICS OF THE RUN

Any method of estimating mortality rates relies on a field sampling program that appropriately characterizes the run with respect to the size of the fish, its species composition, and the proportions by sex, age, and number of previous spawnings. River herring populations are known to be structured, with alewife typically running earlier in the season than blueback herring, and with older, larger repeat spawning fish also typically running earlier (see below). The number of fish migrating daily also varies markedly throughout the season. In this section, the importance of incorporating this structure into a monitoring program is evaluated. Specifically, a uniform sampling program in which a fixed number of fish are sampled per unit time (daily) throughout the run is compared with a stratified sampling program in which the number of fish sampled per unit time (daily) is proportional to the abundance in that time period. The first program is relatively easy to implement, whereas the second requires *a priori* knowledge of the number of fish moving during the time period.

Field Methods

Data used in this section were collected as part of an alewife assessment program in the Gaspereau River from 1997 to 2002. Field sampling, aging, and data processing methods were consistent throughout these years (Gibson 2000 and references therein). In brief, alewife were sampled as they ascended the fish ladder bypassing the White Rock Generating Station. Alewife were counted in 15-minute intervals from 0800h to 2000h during the majority of the run, but during the beginning and end of the run when daily abundance was low, the count interval was lengthened. For every 1000 alewife that ascended the ladder, biological data (species, fork length, weight, sex) and scale samples (age, previous spawning history) were collected for 10 randomly selected individuals. As such, daily sampling effort was stratified with many more samples collected on days when the number of fish ascending the ladder was high, with a much lower number of samples collected on days when the abundance was low. Scales for aging were randomly selected from the resulting collection. Using this approach, the assumption of independence in the sampling design was met (all fish had an equal probability of being sampled), whereas, if a uniform sampling program had been used (without information about daily abundance), fish moving on days when the abundance was high would have a lower probability of being sampled than fish moving on days when abundance is low.

Analysis

Analyses of these data proceeded first by looking for temporal patterns in the biological characteristics data, and then by weighting the data to mimic a uniform sampling protocol and comparing summary statistics for the biological characteristics and mortality rate estimates that result from the stratified and uniform sampling schemes.

Daily time series of the abundance of sub-components of the spawning run (e.g. sex, age, repeat spawners) were developed and compared with the daily time series of total abundance to identify whether the run timing of these sub-components was the same as the total or whether their run timing differed. Cumulative proportions were used to facilitate display. Samples were also grouped based on whether they occurred in the first or second half of the spawning run (based on the median date of sampling), were visualized using boxplots, and were compared using two-sided t-tests to further identify temporal patterns in the run. Other summary statistics (mean, standard deviation, and sample size) were also calculated to describe the structure of the spawning runs.

The stratified sampling data were then used to produce a dataset that approximated a uniform sampling scheme, as if the same number of Alewife had been sampled each day during each spawning run. Each observation in the stratified sampling dataset was given a weight n, equivalent to the inverse of the number of samples collected on the day of that observation.

The biological characteristics obtained using the two sampling schemes (stratified and uniform) were compared using summary statistics (mean and standard deviation) and descriptive plots. Instantaneous mortality rate estimates (Z), one of the key metrics in the reference point framework, were calculated for both sampling scheme datasets using the generalized linear model incorporating the previous spawning data, as described above.

Results

Temporal Patterns in Population Structure

Fork lengths of sampled alewife varied between 202 and 315 mm, while weights varied between 83 and 577 g (Figure 4.3.1). Alewife were significantly longer and heavier (p-values <0.005) during the first half of the spawning run in every year (Figure 4.3.10; Tables 4.3.1 and 4.3.2).

Ages of alewife sampled between 1997 and 2002 ranged from age-2 (n=2) to age-7 (n=6) with the majority of the fish being ages four or five. The cumulative proportions of alewife sampled within each age class suggested that there were temporal differences in age structure of each spawning migration. Alewife of ages three and four were generally sampled later than alewife of ages five to seven (Figures 4.3.2-4.3.6). In 1997, 1998, 2000, and 2002, mean age was greater during the first half of the spawning run than during the second half (Figure 4.3.10; Table 4.3.3).

Temporal patterns in the migration of repeat spawners were also consistent with a temporal pattern in the numbers at age. The cumulative proportions of repeat spawners were generally steeper than the cumulative proportions of all alewife sampled, suggesting that the repeat spawners migrated earlier than virgin spawners (Figure 4.3.7). The proportion of repeat spawners was higher during the first half of the run than during the second half, and this discrepancy was largest in 1997, 1998 and 2002 (Figure 4.3.10). The proportion of virgin spawners was greatest during the second half of the spawning run in every year, while the proportions of alewife that had one, two, or three previous spawnings were greater during first half of the run (Figure 4.3.8). There were relatively few alewife with three previous spawnings.

No differences in the migration timing of male and female alewife were detected using these data. The cumulative proportion of females sampled did not differ from the cumulative proportion of all alewife sampled (Figure 4.3.9) and the probability of sampling a female was similar during the first and second half of each spawning run (Figure 4.3.10).

Effects of Sampling Scheme on Population Structure

Alewife fork lengths (mm) measured under a stratified sampling scheme were larger than fork lengths measured under a uniform, same number per day, sampling scheme in all years except 2001 (Figure 4.3.11; Table 4.3.4). Similarly, weights (g) measured during stratified sampling were higher than those estimated for uniform sampling in all years except 2001 (Figure 4.3.12; Table 4.3.5).

Age structure (approximated by mean age) differed between stratified and uniform sampling schemes in all years (Table 4.3.6) but ages were not consistently older or younger using either method.

The proportion of alewife within each previous spawning history group (0, 1, 2, or 3 previous spawnings) differed between the stratified and uniform sampling schemes (Figure 4.3.13). Differences between the proportions of the population within each previous spawning group were greatest for alewife with two or three previous spawnings, particularly in 1998 and 2000 (Figure 4.3.13).

Instantaneous mortality rate estimates differed annually depending on the sampling scheme used (Figures 4.3.14-4.3.16; Table 4.3.7). Instantaneous mortality rates ranged from 1.54 to 2.40 for Alewife under the stratified sampling scheme, whereas the uniform sampling dataset produced mortality estimates of 1.53 to 2.71. In some years, estimates of instantaneous mortality were higher under a stratified sampling scheme, but in other years, they were lower. There was no consistent pattern in the differences in instantaneous mortality for each sampling scheme, but differences were observed each year.

Implications for Monitoring

These results indicate that, with respect to size, age and previous spawning history, there is considerable structure within an alewife run, but the pattern can be variable from year to year. As a result of this structure, estimates of the lengths and weights used to characterize the run are expected to be biased low if a uniform sampling methodology is used. Similarly, the age and previous spawning structure of the run each year would be expected to be biased towards younger fish. In the case of the weight estimates, this bias would be expected to produce over-estimates of the number of fish harvested in a year if the weights are used to convert landings in weight to landings in numbers. In the case of the age and previous spawning structure, it would be expected to lead to estimates of mortality that are biased high. Although not explored here, an ad-hoc sampling scheme would not be expected to produce samples that are representative of the population.

The issues described above can be avoided if sampling programs are properly designed, specifically by ensuring that sub-components of the population are not over or under-represented in the sample collection. This can be achieved either by ensuring the principle of independence is adhered to in the sampling program, or by ensuring that sampling is conducted in a way that the results can be reweighted to correct for errors or biases associated with the structure in the run. For either approach, estimates of the run size per unit time are needed in order to appropriately quantify the biological characteristics of the run. When the abundance estimates are available in "real time", the approach used on the Gaspereau River (sampling in proportion to abundance – described in Gibson (2000)) should lead to data that reasonably approximates the characteristics of the run. When abundance estimates per unit time are only available after the run is over (e.g. once video recording the run has been counted), the approach used on the Tusket River (Bowlby and Gibson 2016) of sampling a relatively large number of fish daily may be appropriate, but this does require a relatively high degree of over-sampling to ensure the results can be re-weighted (or re-sampled in the case of the ages on the Tusket) to ensure that results are representative of the characteristics of the stock.

The phrase “per unit time” has been used throughout this section, leading to the question of how frequently should sampling occur. Although not explored here, the temporal patterns in the run of older age fish indicate that the majority can move on a time scale of one or two days, as evidence by the run of age-6 fish in 1998 and 1999 (Figure 4.3.5), and age-7 fish in 1997, 1998 and 2000 (Figure 4.3.6). Abundance in the older age classes does have high influence when estimating mortality rates, and a high sampling frequency (e.g. daily) would be needed to ensure their abundance is appropriately characterized.

Finally, it should be noted that depending on the nature of the fishery, protocols for sampling the landings could be quite different than sampling at counting facilities. In cases where fish are sold frequently, frequent sampling is necessary to ensure the biological characteristics are appropriately characterized. In situations where fish are all sold at the end of the season, sampling could occur at that time, as long as the principles of sampling are met.

4.4 ESTIMATING ABUNDANCE

Options for estimating abundance for Maritimes Region river herring populations can be thought of as falling into two categories: those where abundance or spawning escapement can be directly enumerated or estimated via methods such as monitoring at fish ladders, and those where the only information (daily landings, associated biological characteristics of the run) comes from the commercial fishery. In this second instance, abundance estimates can be obtained using the statistical catch-at-age models described in Section 4.2. In this section, the focus is on sampling and analytical methods applicable in situations where the run can be directly enumerated. In nearly all cases, this occurs at fish ladders.

As discussed in Section 4.3, a goal of enumerating abundance should not be simply to obtain an estimate of the total number of fish ascending a ladder in a given year, but also to obtain estimates of the number of fish ascending the ladder in a given time period (e.g. daily) in order that the length, weight, sex, age and previous spawning composition of the run can be appropriately characterized. Visual counts of the total run do have the advantage that they do provide real time feedback about abundance that can be used to ensure that sampling for biological characteristics data is done appropriately (Jessop and Parker 1988, Gibson 2000), and also that staff are present to do the biological sampling, but have the disadvantage that they are labour intensive. Less labour intensive methods include video monitoring and electronic counters (e.g. Smith Root fish counters). Electronic counters have the advantage that real time abundance estimates are available that can be used to determine the biological sampling frequency. Video monitoring has the advantage that subsampling can greatly reduce the amount of time required to estimate run size. Nelson (2006) provides a through overview of the options for the design and analysis of partial counts. Where estimates of daily run size are required, a two-way random stratified design (using day and time periods within a day) has the advantage over a one-way random stratified design in that it greatly reduces the chance of all the counts occurring at a time of the day when abundance is high or low (Nelson 2006). The two-way design can apparently produce relatively precise estimates with relatively low effort, as shown for the Tusket River (Bowlby and Gibson 2016). Electronic methods of monitoring do require that the equipment is checked frequently to ensure that data are being recorded properly. An option for integrating video counts and biological sampling would be to do the counts daily (or on some shorter time period), and to determine the number of biological samples required daily based on counts. This approach would be expected to significantly reduce the sampling effort, particularly on days when abundance is low.

In this section, the focus has been on estimating absolute abundance. In instances where relative abundance indices could be collected, these can be used as tuning indices in the statistical catch-at-age, or other, models. For example, Chaput et al. (2001) used a larval abundance index as a tuning index in a VPA for alewife in the Margaree River, NS. Methods to develop fisheries-independent relative abundance indices include: electrofishing, seining, gillnetting, trawling, partial visual counts and use of push nets (ASMFC. In prep.: River Herring Data Standardization Meeting). Relative abundance indices would be expected to be a beneficial data input for the statistical catch-at-age models, although their development could be deferred until after fishery-dependent data collections are initiated.

5.0 WORKING WITH THE COMMERCIAL LANDINGS DATA

Using commercial landings data in this assessment framework requires:

1. that annual landings can be calculated accurately, and
2. that the landings can be assigned to the population being assessed.

This necessitates measuring the species composition in the catch and being able to estimate the river of origin for coastal or estuarine catches. Although information on commercial catches is being collected annually via a logbook program, there are several considerations that would need to be addressed to make it useable for assessment (detailed below). There is currently no assessment of the species composition or population of origin for catches in all areas, although these have the potential to be inferred from the location of the fishery in some areas. Methods to address these questions for specific rivers would need to be designed and implemented for future assessments.

5.1 LOGBOOK PROGRAM

Throughout Nova Scotia and Southwest New Brunswick, information on commercial catches of river herring comes from logbook reports submitted by individual fishermen. These are designed for individuals to record their location, gear type, gear amount, daily catch, and daily effort as well as to include identifiers like their names and (currently) their license number. The logbook program was initiated in the late 1980s, around the same time that the majority of licenses for river herring were being created. It was introduced to be able to assign catches to specific rivers rather than to larger statistical districts (typically containing multiple rivers) as had been done previously (DFO 2001). The logbook program originated as a Science initiative, where individuals involved in the river herring assessment were responsible for data input, archival and quality control. This changed in 2009 to become more similar to reporting in other fisheries, where license-holders were required to submit logbooks to dockside monitoring companies for data input, and data were subsequently archived by the Commercial Data Division at DFO. The switch in the logbook program was intended to ensure mandatory reporting of catches, where individuals would not receive license conditions for the following year without there being catch records in the database from the previous year. For the purposes of this section, we consider the old logbook program (1986 to 2008) to be separate from the new logbook program (2009 to present).

For individuals fishing in freshwater who are submitting logbook reports, it is relatively straightforward to calculate total catches and effort, barring some discrepancies between data sources as well as how information is intended to be reported vs. how it actually is reported (discussed below). However, these data are still not specific to species and they do not represent total catches in a given year. There are four issues associated with calculating total catches and effort by species for a given river related to:

1. identifying individuals who are not reporting on a specific river so as to calculate reporting rates by gear type,
2. the switch between data archiving methods from the old logbook program relative to the new one,
3. challenges related to partitioning catches by species, and
4. challenges related to partitioning estuarine or marine catches to specific populations.

For a previously unassessed river, many of the issues related to the commercial landings would need to be resolved very early in a multi-year assessment program, and would become less problematic with the length of time a particular river was assessed, because of increasing familiarity with participants and fishing practices.

Fishing Location

The idea behind the logbook program was that it would result in river-specific information on populations (DFO 2001) which would presumably be more useful than regional information for assessment. However, there was no standardized river list that an individual license-holder would use to identify their fishing location. Colloquial names, small tributary names, names of nearby features like waterfalls, and names based on fishing practices (e.g. dip stand numbers without an associated river name) were commonly reported in both old and new logbook records. There is also the opposite problem relative to extremely large river systems (where individual tributaries would be expected to contain distinct populations of river herring; McBride et al. 2014); for example, listing a location like the 'Saint John River' could be in any one of the numerous tributaries it contains. In such large systems or those with very large lakes, it is also not possible to infer locations from gear type (to differentiate in-river from estuarine catches – see below), given that gears like trap nets, and drift or set gill nets can be fished either in the estuary or in lakes. For any of the logbook data, the first step to calculating reported catch and effort is to match any variations in names to the river system(s) supporting the populations being assessed.

Reporting Rates

Estimates of reporting rates enable reported catches to be scaled to total catch estimates. For the river herring fishery, differences in average catch by gear type means that it is even more useful to have reporting rates by gear type. At a minimum, two pieces of information are required to estimate reporting rate:

1. a list of license numbers that have the potential to be fished in a particular river in a given year, and
2. a list showing the corresponding name of the license-holder in a given year.

Relative to Bullet 1, it is often not possible to unambiguously assign a license number to a particular river or gear type on the basis of license conditions. These tend to be specific to a region or county (e.g. Yarmouth County; Kings County) as well as to two types of gear (e.g. dip nets and gill nets). This means that practices like gill netting in one estuary in the morning and then dip netting in a second river in the afternoon in the same year are permitted on the basis of license conditions. In the old logbook records, the number of times that an individual moved around or switched gears varied, yet both practices were more common in the late 1990s and early 2000s. For many locations, the total number of license-holders active on a particular river in a particular year can only ever be estimated. For example, there were 115 unique license numbers that had reported catches in at least one year in the Tusket River from 1986 to 2015.

However, if the location most commonly fished was assumed for each license in years that there were no reports, this reduced 115 to a maximum of 87 licenses in a single year.

Relative to Bullet 2, the old logbook returns were not archived relative to license numbers but only to individual's names. In order to determine which license numbers the older catch records corresponded to, it became necessary to first create a list of names for a particular river and then to cross-reference these names (for specific years) with names in the licensing database (also for specific years). There were discrepancies between the names people use (logbook data) and their legal names (licensing database). Even in situations where the goal is to only use recent catch records (e.g. to report catches since 2009), reporting in the older logbook data may become the only information that is available to assign license numbers to a specific river or for calculating reporting rates. Alternatively, moving forward, this information could be developed for populations used in the assessment framework as required.

There is also the consideration that having a valid license does not necessarily mean that an individual is fishing in a particular year. Inactive licenses need to be accounted for when calculating reporting rates. Both the old and new logbook formats have check boxes where an individual can identify that they did not fish in a particular year. These data were entered each year in the old logbook program and individuals were given an 'active' or 'inactive' designation. When the older data was migrated into the new database format used by the CDD, this information was lost because it was kept separately from the catch records. For the newer logbook program, the need to enter zero catch records for individual license-holders who did not fish may not have been made explicit to the dockside monitoring companies. When completing the Tusket assessment, empty logbook reports with the 'did not fish' box checked were not entered into the electronic database (Bowlby and Gibson 2016). It is unknown if this would be true for all rivers and regions or if it is a localized issue.

Mandatory reporting is slated to be enforced for the 2016 fishing season, in that individuals will not receive their license conditions for 2016 unless the database contains catch records from 2015. This should reduce the need to estimate reporting rates into the future. However, if annual assessments incorporating the catch data are expected for river herring, all logbook reports need to be submitted, entered and quality controlled well in advance of the start of the following fishing season (e.g. by September of the same year).

Inferring Information

There are several very common issues with reporting and data entry that can be corrected provided the individual entering or working with the data is knowledgeable about local commercial freshwater fisheries and fishing practices. Many of these would have been corrected in the old logbook program where the individual responsible for the data input would have verified numbers, locations, gear types, etc. with specific license-holders on an annual basis. In the new logbook program, the dockside monitoring companies are responsible for entering data from multiple and diverse fisheries. As a result, things that would have been previously flagged as impossible or improbable are entered as they appear on the logbook record. Although there are multiple examples, the two that we will highlight relate to gear types and daily catch values, as determined by the quality control related to catches on the Tusket River.

In relation to gear types, there are freshwater fisheries for 7 species that currently use the same logbook for data entry. As such, there are 11 separate gear types listed on the log, only 5 of which are utilized the river herring fishery, and only one of which can be associated with each catch record in the database. River herring fishermen often identify both types of gears that they are licensed to fish (e.g. circling dip stand and fixed gill net) even though their catch records are specific to one of them (which can typically be inferred from the 'nets' column or comments on the form). In addition, often the wrong gear code is circled (e.g. dip net, gear code 70 vs. dip stand, gear code 08). What gets entered into the database depends on the individual at the dockside monitoring company, ranging from no gear type, gear types that are not used in the river herring fishery, or an arbitrary selection of one of the two gear types identified. Because many individuals can change gear types on the basis of license conditions, it is necessary to go back to the paper records in order to ensure that the reported gear types match local fishing practices as well as the actual reports.

In relation to daily catches, the ways in which catches are reported do not necessarily coincide with the current database structure. For example, catches may be reported as numbers of bins, numbers of truckloads, numbers of fish, or weight in kg or lbs. In the old logbook program, standard conversion factors were applied to data from all rivers to account for the different ways that catches were reported (e.g. to convert from lbs to kg, from bins to kg, from numbers to kg). In the new logbook program, there is only the option to enter catch values in lbs or kg. Therefore, catch values that are not by weight are at times entered and labeled as catches in kg (e.g. reports of 3000 fish become 3000 kg). A second example is the tendency of individual fishermen to report weekly or even monthly totals for their catches. In the current database, these tend to be entered as daily catches (e.g. 27,000 kg in one day). Both a comparison with paper records as well as knowledge of river-specific management regulations (daily closures) are needed to identify these individuals and to partition their catches appropriately.

5.2 SPECIES COMPOSITION

Daily catches from the river herring fishery would contain varying proportions of alewife and blueback herring on rivers that contain both species. For rivers that have not been previously assessed, it becomes necessary to identify which species are present and then to partition the catches by species in order to assess the status. For a previously unassessed river, it would be necessary to ascertain the presence or absence of each species in the landings. Unless it is known that only one species is present, a sampling scheme would need to be developed and implemented to sample the commercial catches for the species composition for the duration of the run. This information need could readily be addressed as part of the biological sampling program described in Section 4.

5.3 MIXED POPULATION FISHERIES

River herring exhibit substantial population structuring as a result of accurate homing to natal rivers (McBride et al. 2014, Palkovacs et al. 2013). Thus catches in fresh water can be considered to be composed exclusively of river-specific populations of alewife and blueback. Conversely, catches that occur along the coast, in estuaries (particularly if multiple rivers share an estuary), or in extremely large river systems (e.g. the Saint John River, NB), have the potential to be composed of varying proportions of returns destined for specific rivers. Previously for the Saint John River, a mark-recapture tagging study (Jessop 1994) has been used to assign catches from the lower estuary to their respective populations of origin (Jessop 2001). For populations where there is the potential for a significant portion of the landings to come from fisheries harvesting more than one population, research leading to the ability to assign catches to specific populations are needed, likely using some type of tagging study.

6.0 SYNOPSIS OF THE ASSESSMENT FRAMEWORK

6.1 STOCKS

Fishing patterns for blueback herring and alewife vary throughout the Maritimes Region. In some areas both species are harvested together in fisheries that also harvest from several stocks, while at the other extreme, some fisheries only target a single stock of a single species. From a monitoring and assessment perspective, it is advantageous to select areas where it is relatively clear that the removals can be assigned to specific populations. Results can then be interpreted in the context of the effects at the population level, within a well-established paradigm for stock assessment and fisheries advice. In rivers that support both species of river herring, the stocks of each species should be assessed separately, although monitoring for both species would likely occur concurrently.

For each of the broader geographic areas discussed in Section 1, some options for rivers that could be selected for monitoring are provided in Table 6.1. These recommendations are based to a large degree on the idea that populations or stocks selected for monitoring should function as individual demographic units. The list is not exhaustive, there are other rivers in which monitoring could occur, nor is it intended that assessment must occur on all the listed stocks.

With respect to the alewife and blueback herring populations above Mactaquac Dam, there is a considerable amount of data available for the populations, and there is the potential to re-instate monitoring for information beyond species-specific harvests and escapements at this location. However, given that spawning escapement is maintained at low levels relative to the potential that exists above the dam, it is not clear that data collected at Mactaquac would be representative of other populations (i.e. as a regional index), or that it would further benefit management of this stock given the competing management priorities for these populations. It is not clear that it should be a priority in this framework for this reason. The St. Croix River in New Brunswick is another anomaly, in the sense that abundance has been maintained at low levels due to restrictions on fish passage put in place to protect the smallmouth bass fishery in Maine. With these restrictions lifted and efforts being made to restore the alewife population in this river, it does provide an excellent opportunity to study population growth and re-colonization, of which fish passage effectiveness is a part. It could be considered a priority for monitoring for this reason, even if alewife fisheries advice is not presently a priority for this watershed.

6.2 REFERENCE POINTS

The reference points provided for evaluating status of alewife are laid out in a way that allows determination of whether overfishing is occurring, and whether a population is in an overfished state. This approach is useful because the effects of management actions such as increasing or decreasing fishing effort, can be immediately evaluated in terms of the fishing mortality rate, whereas the population-level effects in terms of abundance might not be evident for several years. The limit reference point proposed here, of 10% of the unfished equilibrium biomass is low relative to the limit reference points proposed for at least some other fisheries (DFO 2012), but reflects the idea that alewife populations can be quite productive. It is not clear that the limit reference point would be consistent with ecosystem objectives (e.g. nutrient transport; provision of prey), if such objectives were developed. Yield curves for river herring are relatively flat-topped, and the reference points are skewed to the left of this curve. This approach allows a bit of a buffer for unaccounted removals and estimation uncertainty. Similar yields can be obtained over a wide range of exploitation rates, and being slightly to the left on the curve is expected to maintain higher biomasses for the same level of yield as compared to obtaining the same yield from an exploitation rate on right of the curve. A lower exploitation rate producing the same yield while maintaining a higher biomass would likely be more consistent with the precautionary framework with respect to ecosystem objectives.

Reference points for blueback herring fisheries remain to be developed.

Monitoring

Monitoring under this framework could be considered to fall into two main categories: monitoring of stocks where it is possible to obtain abundance counts (almost always at dams and fishways), and stocks where the data collections would be fisheries dependent (Table 6.1). A key aspect of any monitoring program is that the results obtained are representative of the stocks being monitored. Given the structure that occurs in river herring spawning runs (with respect to size, age and previous spawning history), whether the sampling occurs at a fishway or from the commercial landings, estimates of abundance are needed in order to accurately summarize the biological characteristics of the run. When sampling occurs at fishways, real-time abundance estimates would be expected to markedly reduce the amount of oversampling required in the absence of the abundance data. Counting video daily during the run or the use of electronic counters are two ways that this objective could be achieved. When sampling commercial landings, appropriate protocols would depend on the nature of the fishery. If landings are stored and sold at the end of the season, a random sampling protocol could be used at the end of the run. If fish are sold regularly during the season, sampling would need to be more frequent, depending on the frequency with which fish are sold.

Commercial Landings

As discussed in Section 5, accurate accounting of total landings for each stock is necessary to implement this framework. For stocks for which biological sampling is implemented in the commercial fishery, data that can be used to estimate the quantity of the landings associated with each sample are required to be able to accurately estimate biological characteristics. An example of this type of monitoring is provided by Chaput et al. (2001) for alewife in the Maragree River, NS. When biological data are collected at counting facilities, sampling the commercial landings for species composition would avoid assumptions about lag times between the fishery and biological data collection (Bowlby and Gibson 2016) in order to quantify the species composition of the landings.

Analytical Methods

Analytical methods will vary from stock to stock depending on the data collections (Table 6.1). For stocks being monitored using fishery dependent data only, for the first few years, catch curve analyses will be used with a transition to statistical catch-at-age models once sufficient data become available. The length of time required to accumulate sufficient data to estimate abundance is not known, but likely exceeds four or five years. For stocks for which counts are available, catch curve analyses coupled with direct comparison of the counts and the landings will be used initially, with a transition to statistical catch-at-age models once sufficient data are available. In this situation the length of time is expected to be about three years.

Status Evaluation

Status evaluations would be expected to differ during the first few years of monitoring, depending on whether count data are available (Table 6.1). For stocks that would be monitored using fishery dependent data only, for the first few years, status relative to the reference point framework would only be able to be determined relative to the removal reference levels. Once sufficient data are available to estimate abundance using statistical catch-at-age models, then status determinations could be made with respect to abundance as well as removal rate reference levels. For stocks where counts are possible, status determinations with respect to removal and abundance reference levels would be available within the first year. Longer term data collections are still needed in order that: population dynamics can be evaluated to ensure than reference points remain appropriate and that changes in the dynamics can be appropriately incorporated in the advice; and to account for annual variability in the advice.

Quantifying Impacts of Activities Other than Fishing

Because river herring reproduce in fresh water, there are many activities that may affect their productivity and abundance other than fishing. DFO's precautionary framework for the management of fish stocks (DFO 2006) states that all removals and sources of mortality must be included in a framework. In this framework, activities other than fishing are primarily incorporated in the reference point system (at least those where the rates are thought to be relatively constant from year-to-year). This has the advantage that changes in the amount of habitat, the effects of barriers and of fish passage mortality can be directly incorporated. As shown in Section 3.5, methods have been developed allowing the effects of activities such as hydroelectric generation to be evaluated with respect to fisheries, both in regard to yields and also management parameters. However there is little to no information about the magnitude of these effects. Because other activities do impact on river herring populations, and because the consequences for the fishery can be significant, quantification of mortality associated with other activities is a component of this framework, consistent with DFO's framework for managing fisheries.

Reporting Under the Framework

Reporting would occur via annual updates with framework assessments occurring every five years.

7.0 ADDITIONAL RESEARCH RECOMMENDATIONS

REFERENCE POINTS

1. Alewife carrying capacity: The Gibson and Myers meta-analyses were done using the stock-recruitment data available for alewife in North America in the early 2000s. As a result of the status review in the USA by the ASMFC, more data may now be available that could be used to update this analysis, and also potentially to investigate covariates (e.g. location of habitat within a watershed, natural habitat versus reservoirs, effect of watershed size) that would lead to improved estimates of carrying capacity. This is not a small undertaking, and may require modelling the data for many watersheds in order to derive the required stock-recruitment time series.
2. Blueback herring carrying capacity: Gibson and Myers did not undertake an analysis of blueback herring carrying capacity due to the limited data availability at that time. It may be possible to undertake the analysis now, as described above. Alternately, some other method may need to be explored.
3. Derive blueback herring fishing mortality reference points.

ASSIGNING LANDINGS TO STOCKS

As outlined above, monitoring would occur in places that could be considered to support discrete stocks of one or both species.

1. Sampling of the commercial landings for species identification would improve the ability to correctly apportion landings to species-specific stocks.
2. Tagging studies: In areas where fisheries harvest from more than one stock, tagging studies could be used to correctly apportion landings to species and to river-specific stocks.

ASSESSMENT METHODS

1. The simulation results presented in Section 4 illustrate the biases that can result from routine application of some methods. Research to identify methods that lead to improved catch-curve analyses for estimating total mortality rates would improve advice provided in the short term (prior to having sufficient data for a full assessment model).
2. Statistical catch-at-age models are well-established in fisheries assessment science. Simulation testing of the models would lead to a better understanding of sampling considerations (e.g. sample size), the number of years and nature of the data required to provide robust status assessments from fisheries dependent data only, the need for tuning indices, etc.

EFFECTS OF HUMAN ACTIVITIES OTHER THAN FISHING

DFO's framework for fisheries management consistent with the precautionary approach (DFO 2006) clearly states that, in addition to fishing, other activities impacting populations need to be included in the assessment and management framework. As described in Section 3, fish passage efficiency at dams, the creation of reservoirs, and downstream passage survival at dams, all directly affect the dynamics (productivity and abundance) of river herring populations. There are other activities that would be expected to have effects as well. While the methods exist to integrate these effects into this framework, there is a paucity of information (estimates of efficiency and passage survival) to feed into the models and advice. Advice would be significantly improved with data on other activities. Additionally, this information would help in selecting appropriate management actions in situations where the status is poor.

COMMERCIAL LANDINGS

1. In the context of stock assessment, the commercial landings data are presently difficult to work with, primarily due to issues in determining where individuals are fishing. At least for stocks selected for monitoring and assessment within this framework, work that helps to better quantify the stock-specific removals is required to ensure that advice is accurate.
2. There are river herring landings in other fishery sectors (e.g. marine licenses for bait, marine by-catch, Aboriginal fisheries) that are not well quantified. Research leading to better quantification of the fishery removals for other sectors would lead to better advice under this framework.

8.0 REFERENCES

ASMFC. 2007. Stock Assessment Report No. 07-01 of the Atlantic States Marine Fisheries Commission. American Shad Stock Assessment Report for Peer Review Vol. I.

ASMFC. 2012a. Stock Assessment Report No. 12-02 of the Atlantic States Marine Fisheries Commission River Herring Benchmark Stock Assessment Vol. I.

ASMFC. 2012b. Stock Assessment Report No. 12-02 of the Atlantic States Marine Fisheries Commission River Herring Benchmark Stock Assessment Vol. II.

Bowlby, H.D., and A.J.F. Gibson. 2016. River Herring Assessment for the Tusket River, Nova Scotia. DFO Canadian Science Advisory Secretariat Research Document 2016/106.

Beverton, R.J.H., and S.J. Holt. 1957. On the Dynamics of Exploited Fish Populations. Fisheries Investigation Series II. Ministry of Agriculture. Vol. 19. 533p.

Caddy, J.F., and R. Mahon. 1995. Reference Points for Fisheries Management. FAO Fisheries Technical Paper 347. Rome, Italy.

Cantoni, E., and E. Ronchetti. 2001. Robust Inference for Generalized Linear Models. Journal of the American Statistical Association 96: 1022-1030.

Chapman, D.G., and D.S. Robson. 1960. The Analysis of a Catch Curve. Biometrics 16: 354–368.

Chaput, G., P. LeBlanc, and R. Crawford. 2001. Assessment of the Margaree River Gaspereau Fishery, 1997 to 2000. DFO Canadian Science Advisory Secretariat Research Document 2001/046.

Crecco, V.A., and M. Gibson. 1990. Stock Assessment of River Herring from Selected Atlantic Coast Rivers. Special Report No. 19 of the Atlantic States Marine Fisheries Commission. 75p.

Deriso, R.B., T.J. Quinn II, and P.R. Neal. 1985. Catch-Age Analysis with Auxiliary Information. Canadian Journal of Fisheries and Aquatic Sciences 42: 815-824.

DFO 2001. Gaspereau Maritime Provinces Overview. DFO Science Stock Status Report D3-17 (2001).

DFO. 2006. A Harvest Strategy Compliant with the Precautionary Approach. DFO Canadian Science Advisory Secretariat Advisory Report 2006/023.

DFO. 2012. Reference Points Consistent with the Precautionary Approach for a Variety of Stocks in the Maritimes Region. DFO Canadian Science Advisory Secretariat Advisory Report 2012/035.

Durbin, A.G.S., S.W. Nixon, and C.A. Oviatt. 1979. Effects of the Spawning Migration of the Alewife on Freshwater Ecosystems. Ecology 60: 8-17.

Fournier, D.A., and C.P. Archibald. 1982. A General Theory for Analyzing Catch-at-Age Data. Canadian Journal of Fisheries and Aquatic Sciences 39: 1195-1207.

Freeman, M.C., C.M. Pringle, E.A. Greathouse, and B.J. Freeman. 2003. Ecosystem-Level Consequences of Migratory Faunal Depletion Caused by Dams; pp. 255-266 In: K..E. Limburg and J. R. Waldman, editors. Biodiversity, Status and Conservation of the World's Shads. American Fisheries Society, Symposium 35, Bethesda, MD.

Garman, G.C. 1992. Fate and Potential Significance of Postspawning Anadromous Fish Carcasses in an Atlantic Coastal River. Transactions of the American Fisheries Society 121: 390-394.

Garman, G.C., and S.A. Macko. 1998. Contribution of Marine-Derived Organic Matter to an Atlantic Coast, Freshwater, Tidal Stream by Anadromous Clupeid-Fishes. Journal of the North American Benthological Society 17: 277-285.

Gibson, A.J.F. 1996. Distribution and Seaward Migration of Young-of-the-Year American Shad (*Alosa sapidissima*), Blueback Herring (*A. aestivalis*) and Alewives (*A. psuedoharengus*) in the Annapolis River Estuary. Master's Thesis. Acadia University, Wolfville, NS.

Gibson, A.J.F. 2000.The Gaspereau River Alewife Stock and Fishery: 2000. Acadia Centre for Estuarine Research Publication No 56. Wolfville, NS. 25p.

Gibson A.J.F. 2004. Dynamics and Management of Anadromous Alewife (*Alosa pseudoharengus*) Populations. P.hD. Thesis. Department of Biology, Dalhousie University, Halifax, NS. 198p.

Gibson, A.J.F., and G.R. Daborn. 1998. The Ecology of Young-of-the-Year Alewives in Gaspereau Lake with Reference to Water Management Strategies in the Black River - Gaspereau River Watershed. Acadia Centre for Estuarine Research Publication No 45. Wolfville, NS. 68p.

Gibson, A.J.F., and R.A. Myers. 2001. Gaspereau River Alewife Stock Status Report. DFO Canadian Science Advisory Secretariat Research Document 2001/061.

Gibson, A.J.F., and R.A. Myers. 2003a. Biological Reference Points for Anadromous Alewife (*Alosa pseudoharengus*) Fisheries in Atlantic Canada. Canadian Technical Report of Fisheries and Aquatic Sciences 2468. 50p.

Gibson, A.J.F., and R.A. Myers. 2003b. A Statistical, Age-Structured, Life History Based, Stock Assessment Model for Anadromous *Alosa*; pp. 275-283. In: K.E. Limburg, and J.R. Waldman, editors. Biodiversity and Conservation of Shads Worldwide. American Fisheries Society Symposium Series. American Fisheries Society, Bethesda, MD.

Gibson, A.J.F., and R.A. Myers. 2003c. A Meta-Analysis of the Habitat Carrying Capacity and the Maximum Lifetime Reproductive Rate of Anadromous Alewife in Eastern North America; pp. 211-221. In: K.E. Limburg and J.R. Waldman, editors. Biodiversity and Conservation of Shads Worldwide. American Fisheries Society Symposium Series. American Fisheries Society, Bethesda, MD.

Gibson, A.J.F., and R.A. Myers. 2004. Estimating Reference Fishing Mortality Rates from Noisy Spawner-Recruit Data. Canadian Journal of Fisheries and Aquatic Sciences 61: 1771-1783.

Ianelli, J.N., and J. Heifetz. 1995. Decision Analysis of Alternative Harvest Policies for the Gulf of Alaska Pacific Ocean Perch Fishery. Fisheries Research 24: 35-63.

Jessop, B.M. 1990. Diel Variation in Density, Length Composition, and Feeding Activity of Juvenile Alewife, *Alosa pseudoharengus* Wilson, and Blueback Herring, *A. aestivalis* Mitchill, at Near Surface Depths in a Hydroelectric Dam Impoundment. Journal of Fish Biology 37: 813-822.

Jessop, B.M. 1994. Homing of alewives (*Alosa pseudoharengus*) and blueback herring (*A. aestivalis*) to and within the Saint John River, New Brunswick, as indicated by tagging data. Canadian Technical Report of Fisheries and Aquatic Sciences 2015. 26p.

Jessop. 2001. A Brief Assessment of the Commercial Fishery on the Lower Saint John River, N.B. DFO Canadian Science Advisory Secretariat Research Document 2001/60.

Jessop, B.M., and W.E. Anderson. 1989. Effects of Heterogeneity in the Spatial and Temporal Pattern of Juvenile Alewife (*Alosa pseudoharengus*) and Blueback Herring (*Alosa aestivalis*) Density on Estimation of an Index of Abundance. Canadian Journal of Fisheries and Aquatic Sciences 46: 1564-1574.

Jessop, B.M., and H.A. Parker. 1988. The Alewife in the Gaspereau River, Kings County, Nova Scotia, 1982-1984. Canadian Manuscript Report of Fisheries and Aquatic Sciences 1992. 29p.

Loesch, J.G. 1987. Overview of the Life History Aspects of Anadromous Alewife and Blueback Herring in Freshwater Habitats; pp. 89-103. In: M.J. Dadswell, R.J. Klauda, C.M. Moffitt, R.L. Saunders, R.A. Rulifson, and J.E. Cooper, editors. Common Strategies of Anadromous and Catadromous Fishes. Proceedings of an International Symposium held in Boston, Massachusetts, USA, March 9-13, 1986. American Fisheries Society, Bethesda, MD.

Loesch, J.G., and W.A. Lund. 1977. A Contribution to the Life History of the Blueback Herring, *Alosa aestivalis*. Transactions of the American Fisheries Society 106: 583-589.

Mace, P.M., and M.P. Sissenwine. 1993. How Much Spawning per Recruit is Enough?; pp. 101-118. In: S.J. Smith, J.J. Hunt, and D. Rivard, editors. Risk Evaluation and Biological Reference Points for Fisheries Management. Canadian Special Publication of Fisheries and Aquatic Sciences 120.

Maceina, M.J., and P.W. Bettoli. 1998. Variation in Largemouth Bass Recruitment in Four Mainstream Impoundments of the Tennessee River. North American Journal of Fisheries Management 18: 998–1003.

McBride, M.C., T.V. Willis, R.G. Bradford, and P.Bentzen. 2014. Genetic Diversity and Structure of Two Hybridizing Anadromous Fishes (*Alosa pseudoharengus*, *Alosa aestivalis*) Across the Northern Portion of Their Ranges. Conservation Genetics 15: 1281-1298.

McIntyre, T.M., R.G. Bradford, T.D. Davies, and A.J.F. Gibson. 2007. Gaspereau River Alewife Stock Status Report. DFO Canadian Science Advisory Secretariat Research Document 2007/032.

Millar, R.B., 2015. A Better Estimator of Mortality Rate from Age-Frequency Data. Canadian Journal of Fisheries and Aquatic Sciences 72: 364-375.

Mills, E.L., R. O'Gorman, J. DeGisi, R.F. Heberger, and R.A. House. 1992. Food of the Alewife (*Alosa pseudoharengus*) in Lake Ontario Before and After the Establishment of *Bythotrephes cederstroemi*. Canadian Journal of Fisheries and Aquatic Sciences 49: 2009-2019.

Olney, J.E., and J.M. Hoenig. 2001. Managing a Fishery Under Moratorium: Assessment Opportunities for Virginia's Stocks on American Shad. Fisheries 26: 6-11.

O'Neill, J.T. 1980. Aspects of the Life Histories of Anadromous Alewife and Blueback Herring. M.Sc. Thesis, Department of Biology, Acadia University, Wolfville, NS. 306p.

Nelson, G.A. 2006. A Guide to Statistical Sampling for the Estimation of River Herring Run Size Using Visual Counts. Massachusetts Division of Marine Fisheries Technical Report TR-25. 33p.

Palkovacs, E.P., D.J. Hasselman, E.E. Argo, S.R. Gephard, K.E. Limburg, D.M. Post, T.F. Schultz, and T.T. Willis. 2013. Combining Genetic and Demographic Information to Prioritize Conservation Efforts for Anadromous Alewife and Blueback Herring. Evolutionary Applications 7: 212-226.

Punt, A., and R. Hilborn. 1997. Fisheries Stock Assessment and Decision Analysis: The Bayesian Approach. Reviews in Fish Biology and Fisheries 7: 35-63.

Quinn, T.J. II, and R.B. Deriso. 1999. Quantitative Fish Dynamics. Oxford University Press. New York.

Ricker, W.E. 1975 Computation and Interpretation of Biological Statistics for Fish Populations. Fisheries Research Board of Canada Bulletin 191. 382p.

Rulifson, R.A. 1994. Status of Anadromous *Alosa* Along the East Coast of North America; pp. 108-112. In: J.E. Cooper, R.T. Eades, R.J. Klauda, and J.G. Loesch, editors. Anadromous *Alosa* Symposium, Tidewater Chapter, American Fisheries Society, Bethesda, MD.

Robichaud-LeBlanc, K.A., and P.G. Amiro (editors) 2001. Assessments of Gaspereau and Striped Bass Stocks of the Maritime Provinces, 2000. DFO Canadian Science Advisory Secretariat Proceedings Series. 2001/11.

Scott, W.B., and M.G. Scott. 1988. Atlantic Fishes of Canada. Canadian Bulletin of Fisheries and Aquatic Sciences 219. 731p.

Sissenwine, M.P., and J.G. Shepherd. 1987. An Alternative Perspective on Recruitment Overfishing and Biological Reference Points. Canadian Journal of Fisheries and Aquatic Sciences 44: 913-918.

Shepherd, J.G. 1982. A Versatile New Stock-Recruitment Relationship of Fisheries and Construction of Sustainable Yield Curves. Journal du Conseil Perm. International pour l'Exploration de la Mer 40: 67-75.

Schmidt, R.E., B.M. Jessop, and J.E. Hightower. 2003. Status of River Herring Stocks in Large Rivers; pp. 171-182. In: K.E. Limburg and J.R. Waldman, editors. Biodiversity and Conservation of Shads Worldwide. American Fisheries Society Symposium Series. American Fisheries Society, Bethesda, MD.

Smith, M.W., A.Y. Then, C. Wor, G. Ralph, K.H. Pollock, and J.M. Hoenig. 2012. Recommendations for Catch-Curve Analysis. North American Journal of Fisheries Management 32: 956–967.

9.0 TABLES

Table 3.2.1. Definitions of the reference points for alewife fisheries provided by Gibson and Myers (2003a).

Theoretical basis	Reference point	Definition
Yield per Recruit	F_{max}	The fishing mortality rate that maximizes the yield per recruit.
Yield per Recruit	$F_{0.1}$	The fishing mortality rate where the marginal gain in yield is 10% that at F=0
Spawner per Recruit	$F_{35\%}$	The fishing mortality rate where the *SPR* is reduced to 35% that of $SPR_{F=0}$
Spawner per Recruit	$F_{25\%}$	The fishing mortality rate where the *SPR* is reduced to 25% that of $SPR_{F=0}$
Spawner per Recruit	F_{med}	The fishing mortality rate that produces a replacement line with a slope that equals the median survival ratio.
Spawner per Recruit	F_{high}	The fishing mortality rate that produces a replacement line with a slope that equals the 90th percentile of the survival ratio.
Spawner per Recruit	F_{low}	The fishing mortality rate that produces a replacement line with a slope that equals the 10th percentile of the survival ratio.
Life Cycle Model	F_{col}	The fishing mortality rate that would drive the population to extinction (the fishing mortality rate that produces a replacement line equal to the inverse of the maximum likelihood estimate of the slope at the origin of the stock-recruitment relationship).
Life Cycle Model	F_{msy}	The fishing mortality rate that produces the maximum sustainable yield (based on the maximum likelihood estimates of the stock recruitment parameters).
Life Cycle Model	SSB_{msy}	The spawner biomass that produces the maximum sustainable yield (based on the maximum likelihood estimates of the stock recruitment parameters).
Life Cycle Model	$SSB_{10\%}$	The spawner biomass corresponding to 10% of the unfished equilibrium spawner biomass (based on the maximum likelihood estimates of the stock recruitment parameters).

Theoretical basis	Reference point	Definition
Decision Theoretic	F_{mmy}	The fishing mortality rate that maximizes the minimum yield over a range of plausible stock recruitment parameter values.
Decision Theoretic	$F_{max.E[C]}$	The fishing mortality rate that maximized the expectation of the landings using the joint posterior probability density (for the stock-recruitment parameters) to quantify their probabilities. The joint posterior probability density is calculated using both the population-specific likelihood surface and the species-level probability densities for the stock-recruitment parameters).
Simulation Based	$F_{max.mean[]}$	The fishing mortality rate that maximized the average landings across the population simulations.
Simulation Based	$F_{max.median[C]}$	The fishing mortality rate that maximized the median landings across the population simulations.
Simulation Based	$F_{90\%.mean[]}$	The fishing mortality rate that produced 90% of the average landings across the population simulations.
Simulation Based	$F_{90\%.median[C]}$	The fishing mortality rate that produced 90% of the median landings across the population simulations.

Table 3.2.2. Biological reference points for the Margaree River, Gaspereau River, Miramichi River and Mactaquac Headpond alewife populations. Values in brackets are the corresponding exploitation rates. Definitions of the reference points are provided in Table 3.1 (from Gibson and Myers 2003a).

Theoretical Basis	Reference Point	Margaree River	Gaspereau River	Miramichi River	Mactaquac Headpond
Yield per Recruit	F_{max}	0.61 (0.50)	0.86 (0.58)	0.76 (0.53)	0.76 (0.54)
Yield per Recruit	$F_{0.1}$	>3.0 (>0.99)	>3.0 (>0.99)	>3.0 (>0.99)	>3.0 (>0.99)
Spawner per Recruit	$F_{35\%}$	0.47 (0.37)	0.54 (0.42)	0.50 (0.39)	0.51 (0.40)
Spawner per Recruit	$F_{25\%}$	0.67 (0.49)	0.77 (0.54)	0.71 (0.51)	0.72 (0.52)
Spawner per Recruit	F_{med}	0.86 (0.58)	1.12 (0.67)	0.79 (0.54)	0.42 (0.34)
Spawner per Recruit	F_{high}	1.88 (0.85)	1.98 (0.86)	1.61 (0.80)	1.03 (0.64)
Spawner per Recruit	F_{low}	0.00 (0.00)	0.93 (0.60)	0.00 (0.00)	0.00 (0.00)
Production Model	F_{col}	2.72 (0.93)	2.60 (0.92)	4.61 (>0.99)	1.82 (0.84)
Production Model	F_{msy}	0.98 (0.62)	1.01 (0.63)	4.61 (>0.99)	0.68 (0.41)
Production Model	SSB_{msy}	622.3 t	85.8 t	14.1 t	123.0 t
Production Model	$SSB_{10\%}$	883.8 t	109.3 t	865.3 t	112.5 t
Decision Theoretic	F_{mmy}	0.82 (0.56)	0.78 (0.55)	0.87 (0.58)	0.76 (0.53)
Decision Theoretic	$F_{max.E[C]}$	0.75 (0.53)	0.82 (0.56)	0.75 (0.53)	0.75 (0.53)
Simulation Based	$F_{max.mean[C]}$	0.78 (0.55)	0.94 (0.61)	0.86 (0.58)	0.53 (0.41)
Simulation Based	$F_{max.median[C]}$	0.71 (0.51)	0.91 (0.60)	0.84 (0.57)	0.53 (0.41)
Simulation Based	$F_{90\%.mean[C]}$	0.44 (0.36)	0.52 (0.41)	0.47 (0.38)	0.33 (0.28)
Simulation Based	$F_{90\%.median[C]}$	0.37 (0.31)	0.51 (0.40)	0.43 (0.35)	0.31 (0.27)

Table 3.3.1. Biomass reference levels (median plus quartiles) per square kilometer of habitat for alewife fisheries based on the meta-analysis of Gibson (2006). Spawning escapements are calculated assuming 233g per alewife (the mean value for the Gaspereau River population (McIntyre et al. 2007)). Where population-specific weights are available, they should be used.

Reference Point	Symbol	25^{th}	Median	75^{th}
Carrying capacity (mt)	R_{asy}	38.7	51.4	68.3
Spawning stock biomass in the absence of fishing (mt)	SSB_0	36.7	48.7	64.6
Spawning stock biomass at MSY (mt)	SSB_{msy}	5.45	7.23	9.60
10% of the spawning stock biomass in the absence of fishing (mt)	$SSB_{10\%}$	3.67	4.87	6.46
Spawning escapement at MSY (number of fish)	ESC_{msy}	23,378	31,034	41,197
Spawning escapement at 10% of SSB_0 (number of fish)	$ESC_{10\%}$	15,743	20,898	27,742

Table 3.4.1. Reference levels against which the assessment results for alewife can be compared for status determination. Removal reference points are exploitation rates (the proportion of the mature stock being removed).

Reference Level	Acronym	Value	Interpretation
Upper stock reference level	USR	SSB_{msy}, or an equivalent proxy, when a population specific estimate is available; 14.85% of the unfished spawner biomass otherwise	The spawning stock biomass above which the abundance which would be considered to be in the healthy zone; below which, if above the LRP, would be considered to be in the cautious zone.
Limit reference point	LRP	10% of the estimated unfished spawner biomass, based on population specific estimate if available, or the meta-analysis of carrying capacity otherwise	The spawning stock biomass below which abundance would be considered to be in the critical zone and where removal rates should be reduced to the lowest level possible.
Removal reference level	RRL	0.53	The level above which overfishing would be occurring and exploitation (removal) rates need to be reduced
Lower removal reference level	LRRL	0.35	The level above which a population would be considered fully exploited and below which it would be considered under-exploited

Table 4.3.1. Summary statistics describing fork lengths (mm) of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS. Statistics are separated by capture timing; first half or second half of the sampling period. Annual means, standard deviations (SD) and sample sizes (N) are provided for each half of the sampling period and were compared using two-sided t-tests.

	First Half			Second Half			
Year	Mean	SD	N	Mean	SD	N	p-value
1997	266.21	14.60	377	255.91	10.30	575	<0.001
1998	259.61	14.09	825	244.37	14.01	888	<0.001
1999	251.23	11.94	392	245.29	10.68	430	<0.001
2000	261.02	10.98	378	254.78	13.84	618	<0.001
2001	255.10	11.57	1017	250.72	12.24	1243	<0.001
2002	258.38	11.97	1365	250.90	11.68	1497	<0.001

Table 4.3.2. Summary statistics describing weights (g) of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS. Statistics are separated by capture timing; first half or second half of the sampling period. Annual means, standard deviations (SD) and sample sizes (N) are provided for each half of the sampling period and were compared using two-sided t-tests.

	First Half			Second Half			
Year	Mean	SD	N	Mean	SD	N	p-value
1997	246.20	40.68	393	227.06	33.93	397	<0.001
1998	252.62	45.57	799	202.32	38.62	862	<0.001
1999	217.87	33.84	409	199.22	29.34	410	<0.001
2000	250.02	35.42	365	242.64	41.30	610	<0.005
2001	225.93	35.17	963	208.65	34.22	1186	<0.001
2002	257.32	53.32	1343	221.54	37.07	1496	<0.001

Table 4.3.3. Summary statistics describing age (in years) of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS. Statistics are separated by capture timing; first half or second half of the sampling period. Annual means, standard deviations (SD) and sample sizes (N) are provided for each half of the sampling period and were compared using two-sided t-tests.

	First Half			Second Half			
Year	Mean	SD	N	Mean	SD	N	p-value
1997	4.75	0.81	179	4.17	0.46	284	<0.001
1998	4.63	0.66	204	4.18	0.43	242	<0.001
1999	4.43	0.55	242	4.36	0.48	257	0.123
2000	4.83	0.48	193	4.64	0.59	303	<0.001
2001	4.15	0.72	241	4.19	0.72	266	0.507
2002	4.26	0.55	235	3.84	0.66	285	<0.001

Table 4.3.4. Summary statistics describing fork lengths (mm) of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS, and fork lengths that would be expected from a uniform sampling scheme. Annual means, standard deviations (SD) are provided for each sampling scheme, and sample size (N) is provided for the stratified sampling scheme.

Year	N	Stratified Mean	Stratified SD	Uniform Mean	Uniform SD
1997	952	259.99	13.18	258.52	13.80
1998	1713	251.71	15.98	242.77	13.84
1999	822	248.12	11.67	247.46	11.44
2000	996	257.15	13.18	253.28	14.67
2001	2260	252.69	12.14	252.78	13.40
2002	2862	254.47	12.40	253.03	13.49

Table 4.3.5. Summary statistics describing weights (g) of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS, and weights that would be expected from a uniform sampling scheme. Annual means, standard deviations (SD) are provided for each sampling scheme, and sample size (N) is provided for the stratified sampling scheme.

Year	N	Stratified Mean	Stratified SD	Uniform Mean	Uniform SD
1997	790	236.58	38.62	225.77	37.94
1998	1661	226.51	49.03	197.97	40.51
1999	819	208.53	32.99	205.73	32.37
2000	975	245.40	39.35	236.72	41.15
2001	2149	216.40	35.69	217.27	37.29
2002	2839	238.46	48.86	234.40	58.66

Table 4.3.6. Summary statistics describing ages of Alewife sampled during spawning migrations from 1997-2002 via stratified sampling scheme in Gaspereau River, NS, and ages that would be expected from a uniform sampling scheme. Annual means, standard deviations (SD) are provided for each sampling scheme, and sample size (N) is provided for the stratified sampling scheme.

Year	N	Stratified Mean	Stratified SD	Uniform Mean	Uniform SD
1997	463	4.393	0.681	4.388	0.697
1998	446	4.383	0.591	4.180	0.439
1999	499	4.393	0.517	4.415	0.515
2000	496	4.714	0.560	4.619	0.607
2001	507	4.172	0.717	4.230	0.682
2002	520	4.029	0.643	3.963	0.698

Table 4.3.7. Instantaneous mortality rate estimates (Z) for Alewife sampled in the Gaspereau River using a stratified sampling scheme from 1997 – 2002, and the value expected had a uniform sampling scheme been used in those years.

Year	Stratified	Uniform
1997	1.54	1.53
1998	1.56	2.30
1999	1.93	1.66
2000	2.25	2.35
2001	2.40	2.71
2002	1.69	1.66

Table 6.1. Options for stocks to monitor using this framework in each of seven geographic areas within the Maritimes Region. The list is not prioritized. Participants and landings are ranked using logbook data from 2000 – 2007. Analytical methods are: 1 - catch curve, 2 – statistical catch-at-age models, and 3 – direct estimates via comparison of escapement counts and landings.

Geographic Area	River or Location	Number of Participants Relative to Other Rivers the Region (low/medium/ high)	Commercial Landings Relative to Other Rivers in the Region (low/medium/ high)	Escapement Counts	Quantification of Commercial Landings	Sampling for Biological Characteristics	Potential for Partnering (past or present program, or active association)	Analytical Methods (short term)	Analytical Methods (long term)	Comparison Against Removal Reference Points	Comparison Against Biomass Reference Points
Eastern Cape Breton, NS	South Aspy	low	high	no	effect of local intercept fisheries uncertain	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Eastern Cape Breton, NS	Mira	high	medium	no	effect of local intercept fisheries uncertain	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Eastern Shore, NS	Milford Haven	low	high	no	effect of local intercept fisheries uncertain	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Eastern Shore, NS	Ship Harbour / Lake Charlotte	high	medium	no	effect of local intercept fisheries uncertain	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Lunenburg and Queens County, NS	Medway	high	high	no	effect of local intercept fisheries uncertain	Fishery dependent only	yes	1	2	yes (better in the long term)	long term only
Lunenburg and Queens County, NS	LaHave	high	high	no (potential to develop index at Morgans Falls??)	effect of local intercept fisheries uncertain	Fishery dependent only	yes	1	2	yes (better in the long term)	long term only
Yarmouth and Shelburne Counties, NS	Tusket River	high	high	yes	can be accounted for	both	yes	1,3	2,3	yes (better in the long term)	yes (better in the long term)

Geographic Area	River or Location	Number of Participants Relative to Other Rivers the Region (low/medium/ high)	Commercial Landings Relative to Other Rivers in the Region (low/medium/ high)	Escapement Counts	Quantification of Commercial Landings	Sampling for Biological Chracteristics	Potential for Partnering (past or present program, or active association)	Analytical Methods (short term)	Analytical Methods (long term)	Comparison Against Removal Reference Points	Comparison Against Biomass Reference Points
Yarmouth and Shelburne Counties, NS	Eel Lake/Kiack Brook	high	high	no	effect of local intercept fisheries uncertain	Fishery dependent only	yes	1	2	yes (better in the long term)	long term only
Bay of Fundy rivers, NS	Gaspereau	high	high	yes	yes	both	yes	1,3	2,3	yes (better in the long term)	yes (better in the long term)
Bay of Fundy rivers, NS	Shuben-acadie	high	high	no	yes	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Other Rivers, SW NS	Salmon River (Digby)	high	high	no	yes	Fishery dependent only	not known	1	2	yes (better in the long term)	long term only
Saint John River, NB	Maqctaquac Dam	low	high	yes	difficult	both	yes	1,3	2,3	yes (better in the long term)	yes (better in the long term)
Saint John River, NB	Oromocto River	medium	medium	no	difficult	Fishery dependent only	yes	1	2	yes (better in the long term)	yes (better in the long term)
Saint John River, NB	Kennebec-asis	medium	medium	no	difficult	Fishery dependent only	yes	1	2	yes (better in the long term)	yes (better in the long term)
Other rivers, SW NB	St. Croix	low (no?)	low (no?)	yes	effect of local intercept fisheries uncertain	Fishery Independent	yes	1,3	2.3	yes (better in the long term)	yes (better in the long term)

10.0 FIGURES

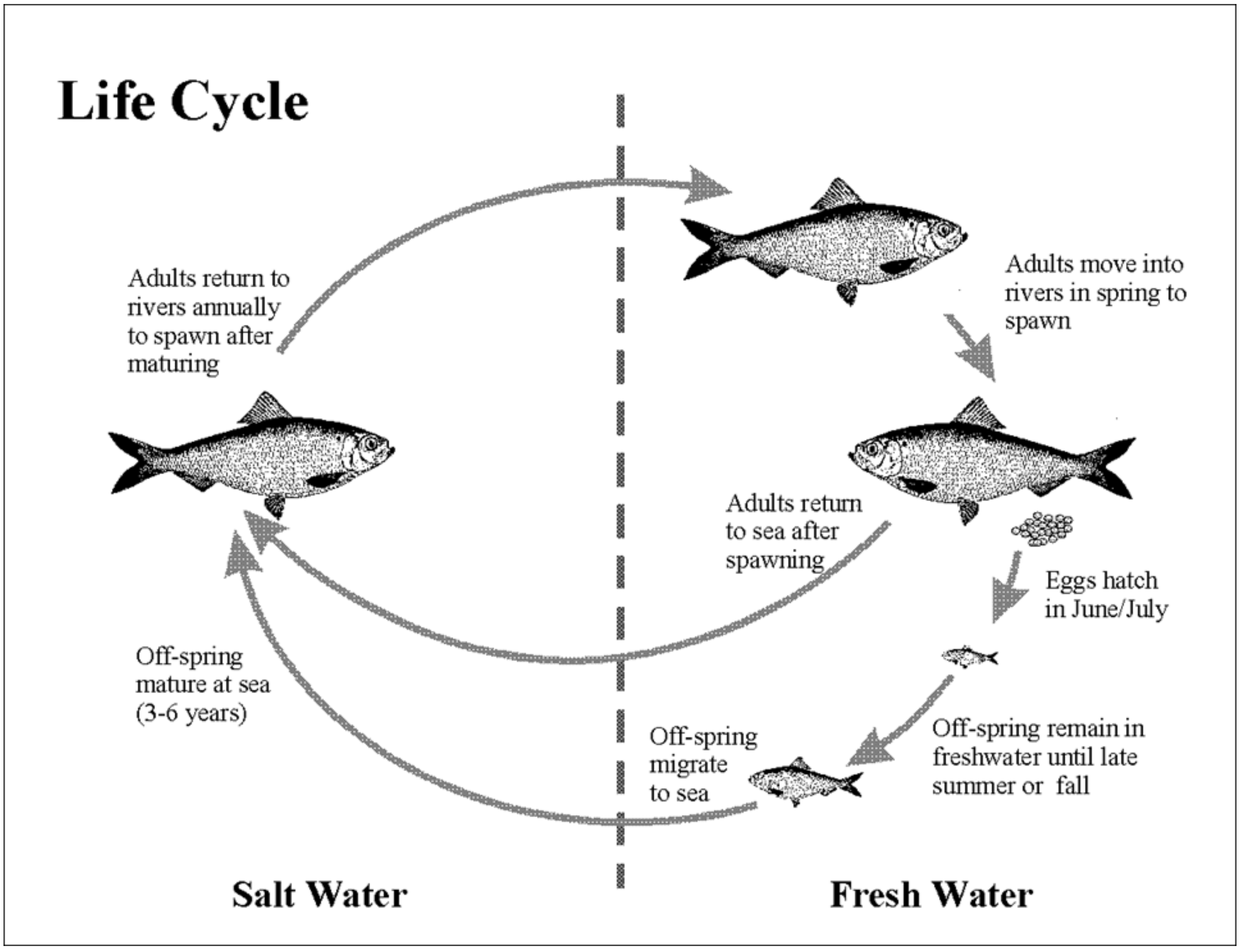

Figure 1.1. Life cycle of anadromous alewife.

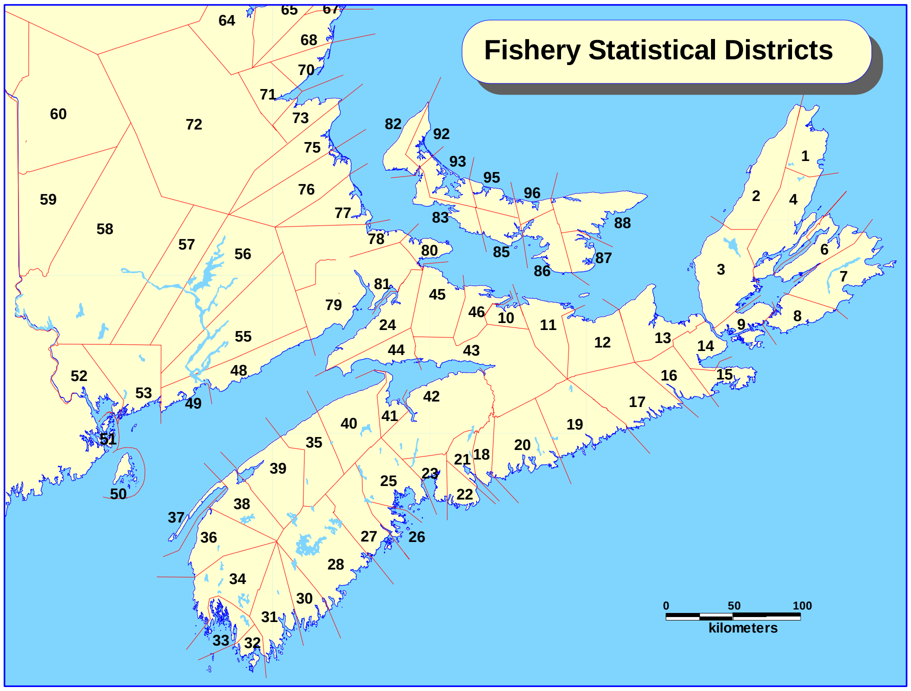

Figure 1.2.1. Map showing the boundaries of the Fishery Statistical Districts for Nova Scotia, Prince Edward Island and Southwest New Brunswick.

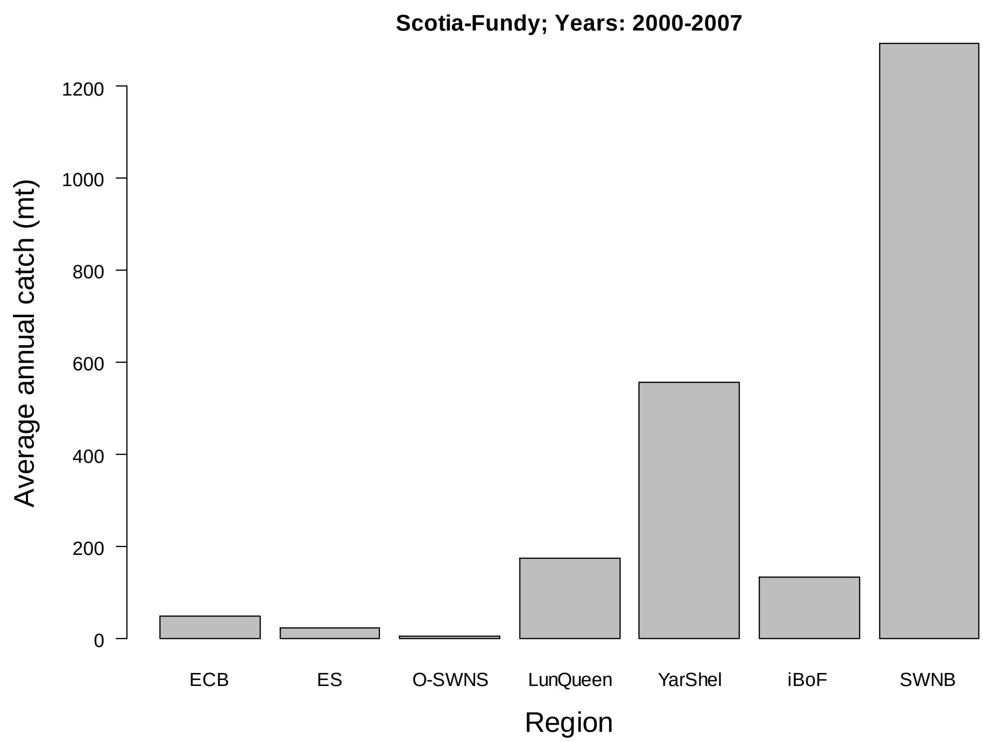

Figure 1.2.2. Average reported landings of river herring in the Maritimes Region for the years 2000 to 2007. ECB = Eastern Cape Breton (FSDs: 1, 3-9); ES = Eastern Shore, NS (FSDs: 14-20); O-SWNS = other rivers in southwest NS (FSDs 21-23, 36-39); LunQueen = Lunenburg and Queens Co., NS (FSDs 25-28); YarShel = Yarmouth and Shelburne Co., NS (FSDs 30-34); iBoF = Inner Bay of Fundy, NS (FSDs 24, 41-44); SWNB = Southwest New Brunswick (FSDs 48-53, 55-60, 79, 81).

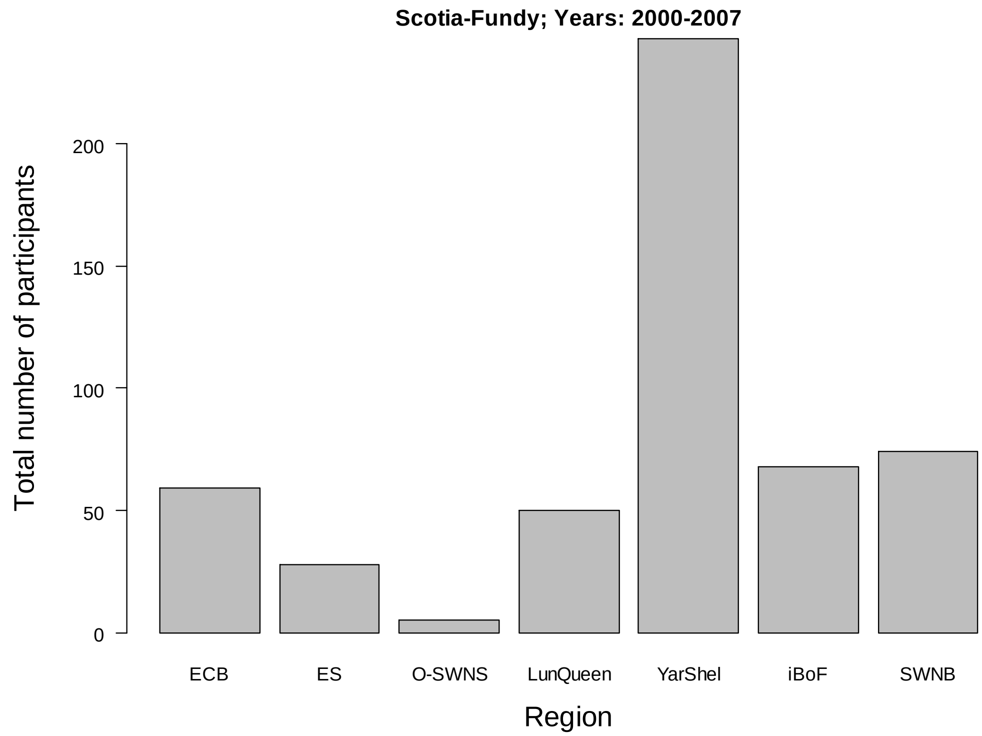

Figure 1.2.3. Total number of individuals reporting landings via the logbook data for the years 2000-2007. The values give an indication of the number of people who have participated in the commercial fishery during this time period, but do not represent the number of licenses that exist or that have been fished annually in specific rivers because the numbers are not corrected for non-reporting and because licenses may have changed hands during this time period. Statistical districts for each bar correspond to those in Figure 1.2.2.

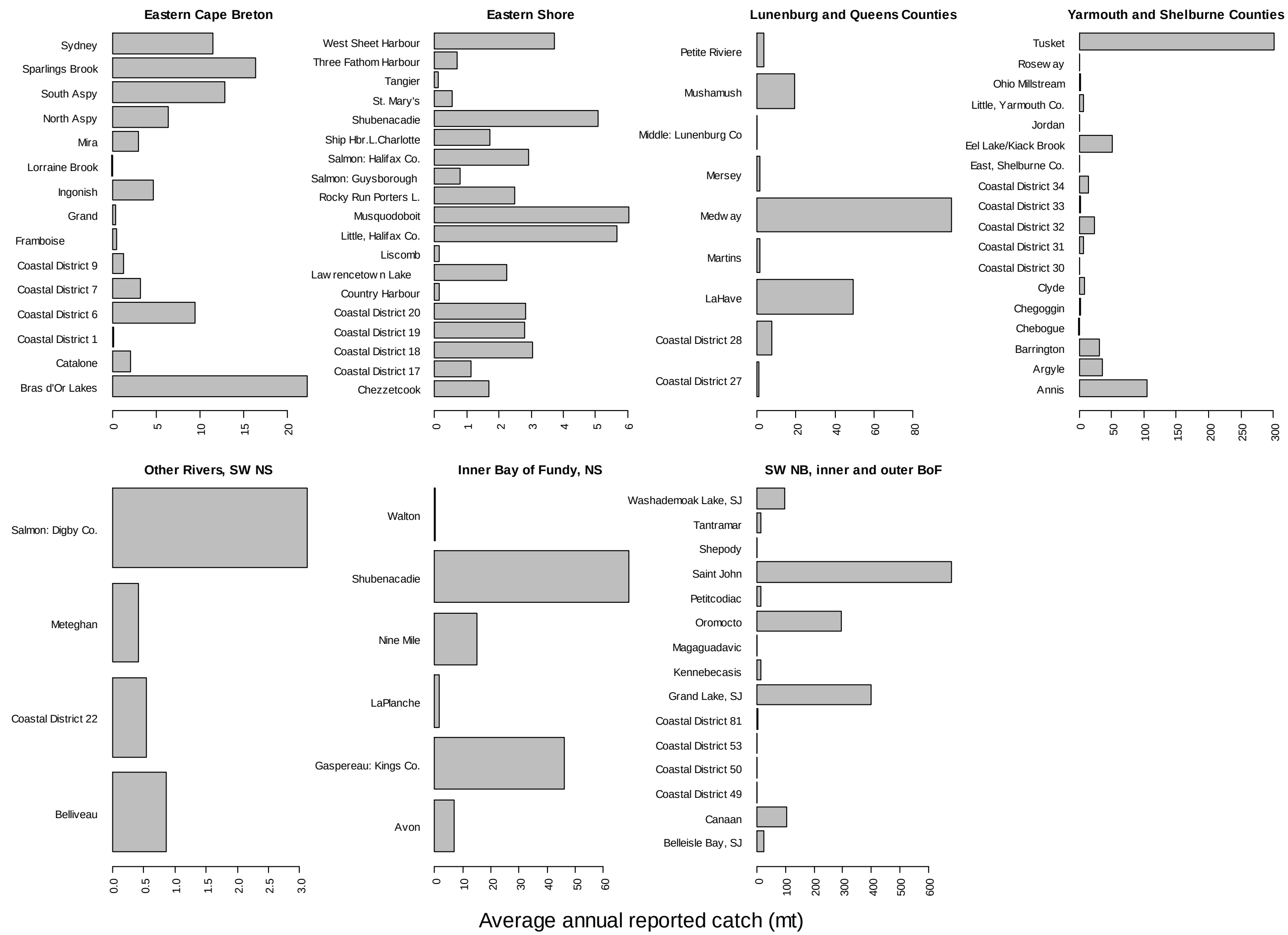

Figure 1.2.4. Average annual reported landings by geographic area, from the logbook data for the years 2000-2007. Values are not corrected for non-reporting. Statistical districts in each plot correspond to those in Figure 1.2.2. Note that the landings from some rivers are reported in different statistical districts and are proportioned accordingly.

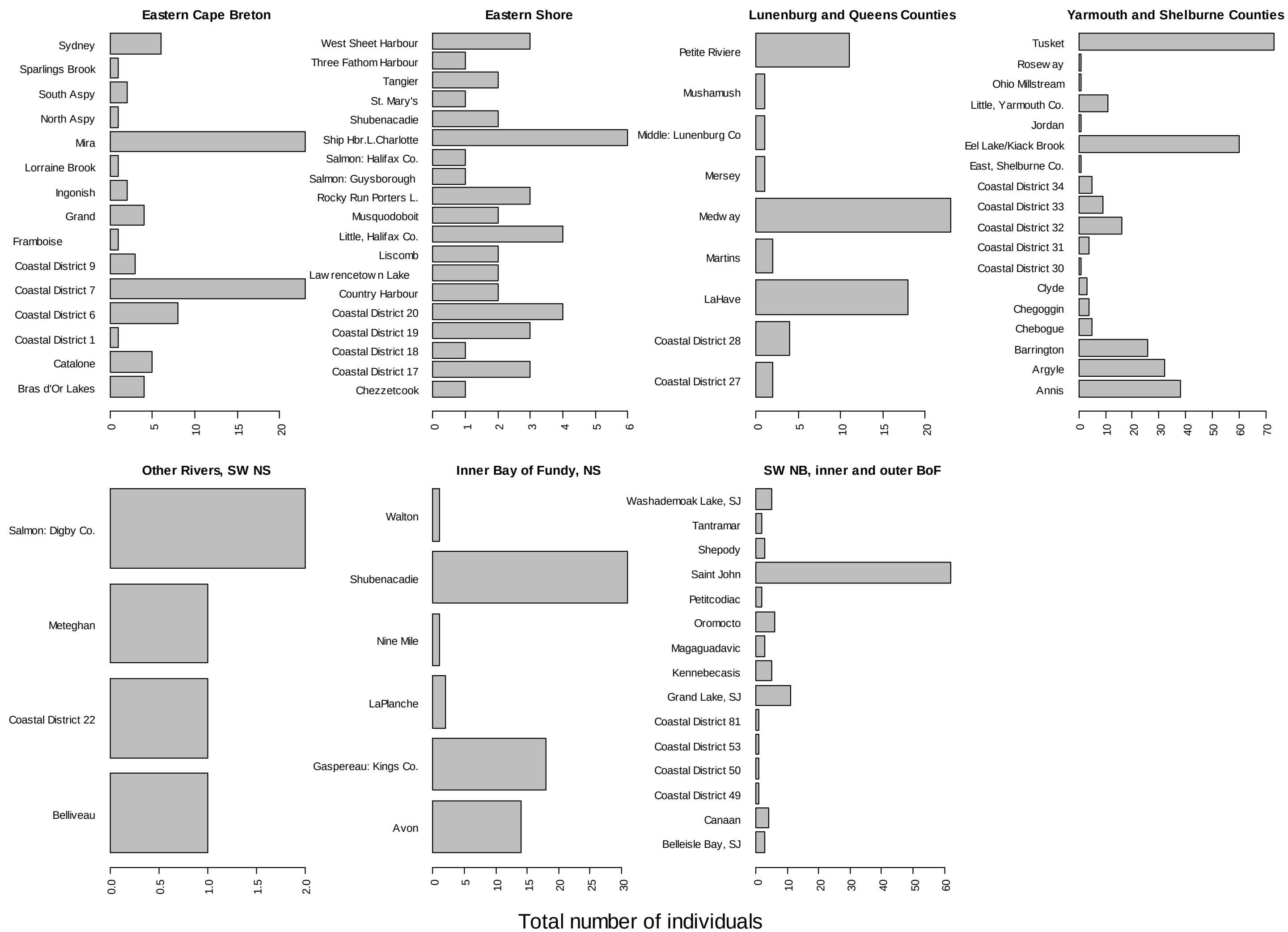

Figure 1.2.5. Total number of individuals reporting landings via the logbook data for the years 2000-2007 by the area where they reported they fished. The values give an indication of the number of people who have participated in the commercial fishery during this time period, but do not represent the number of licenses that exist or that have been fished annually in specific rivers because the numbers are not corrected for non-reporting and because licenses may have changed hands during this time period. Statistical districts in each plot correspond to those in Figure 1.2.2.

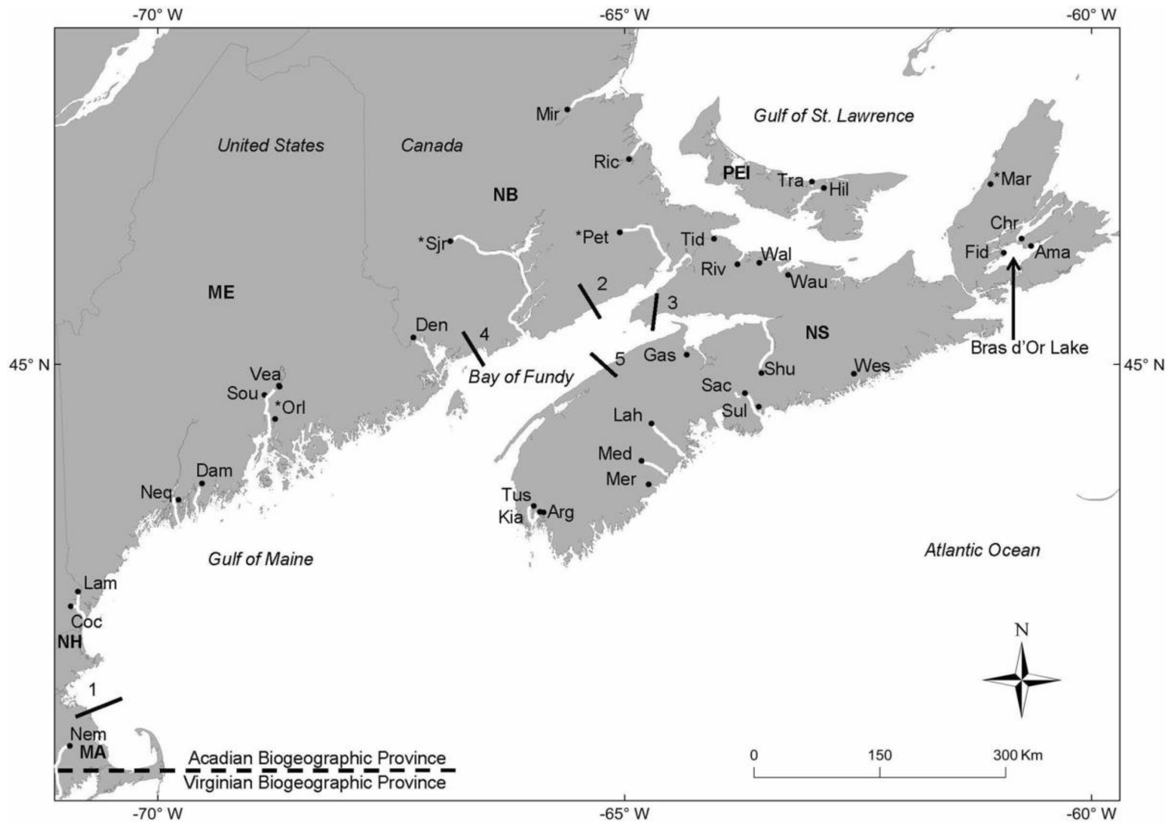

Figure 2.1.1. Map showing collection sites for alewife and blueback (*), as well as discontinuities in gene flow (1 most important; 5 least important) revealed using BARRIER. (Permission granted to reproduce from McBride et al. (2014)).

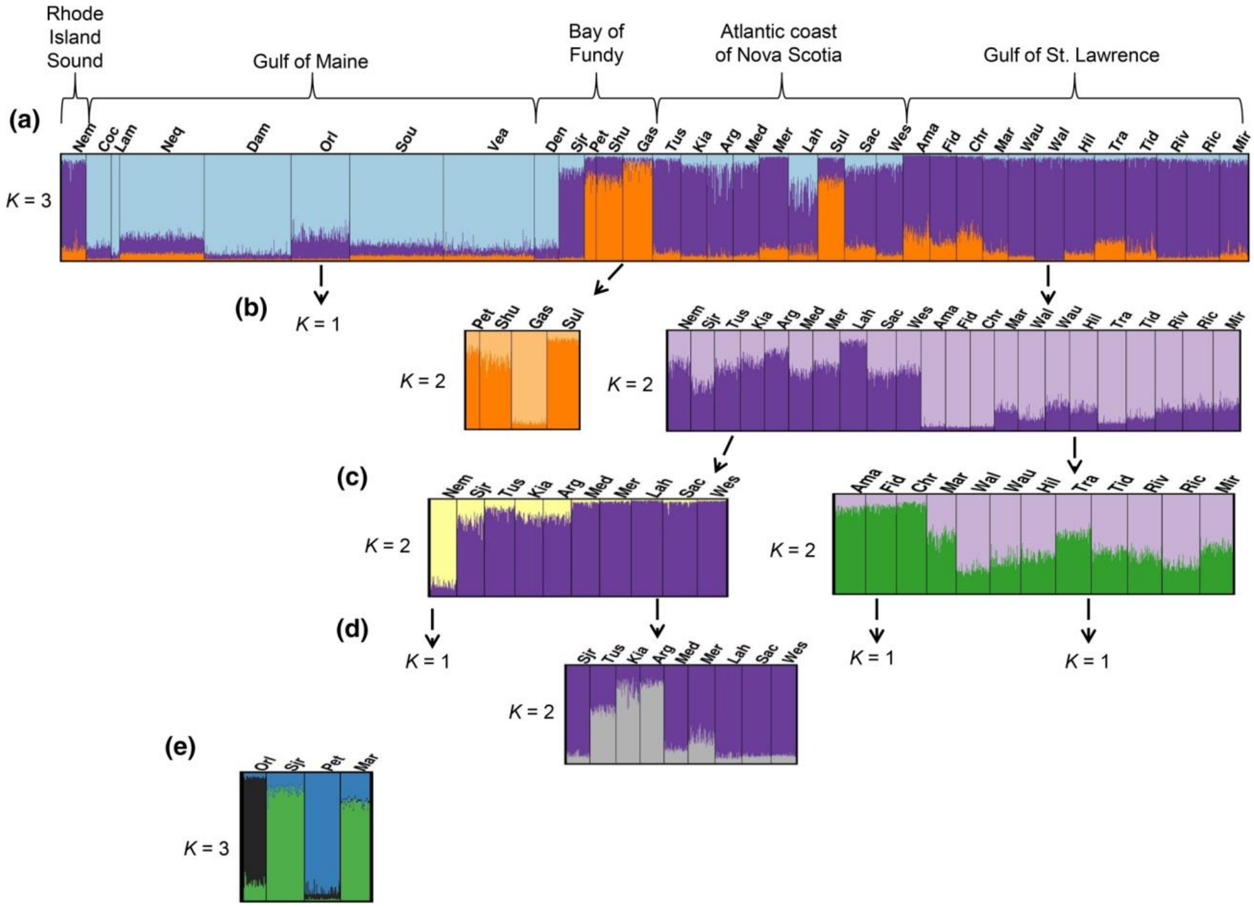

Figure 2.1.2. Bar plots showing results of hierachical STRUCTURE analysis of alewife. Each vertical line represents one individual; coloured segments indicate the estimated membership of that individual in each inferred genetic cluster. a K = 3 clusters were initially inferred, associated with GoM (blue), BoF (orange), and Atlantic coast and GoSL populations (purple). b Hierarchical analysis revealed further structure for BoF and Atlantic coast/GoSL clusters (K = 2). c Further analysis recovered K = 2 within the Atlantic coast and GoSL populations. d Further structure (K = 2) was identified in the Atlantic coast populations. STRUCTURE analysis of blueback identified K = 3 genetic clusters. (Color figure online). (Permission granted to reproduce from McBride et al. (2014)).

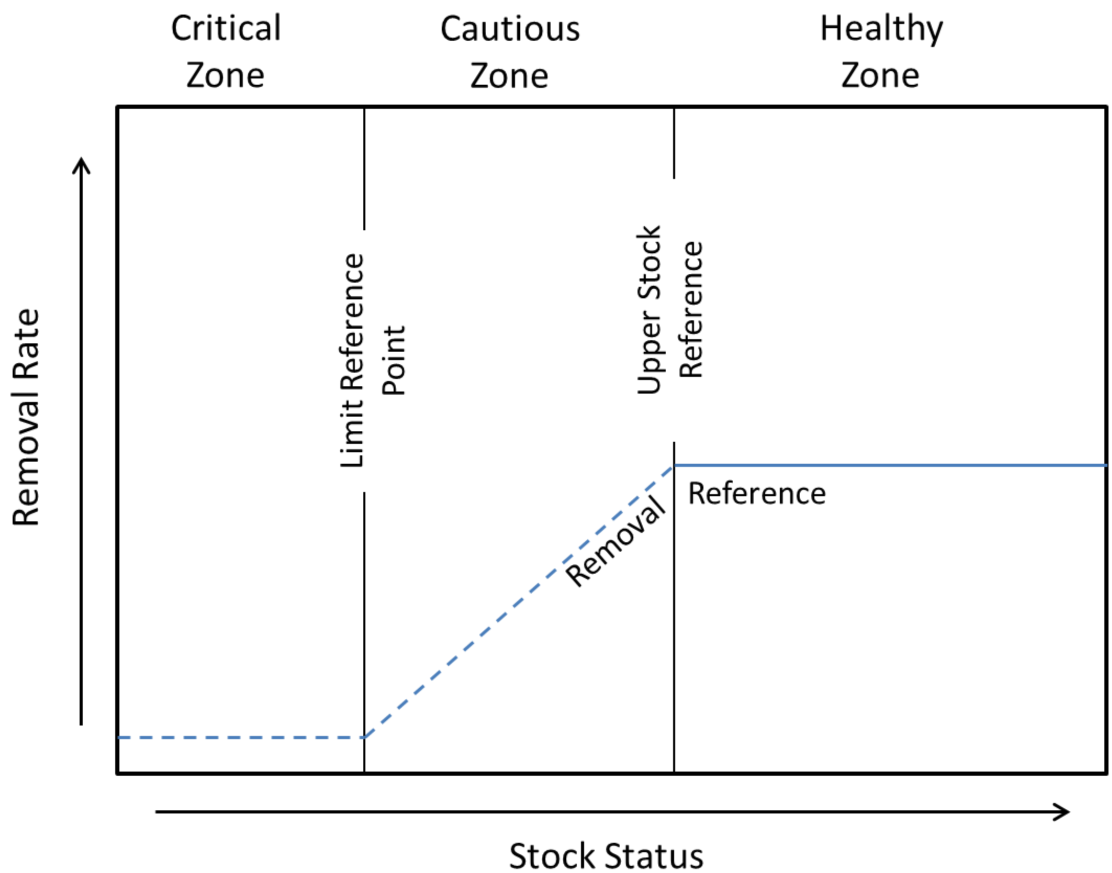

Figure 3.1.1. Fisheries management framework consistent with the precautionary approach (redrawn from DFO 2006a).

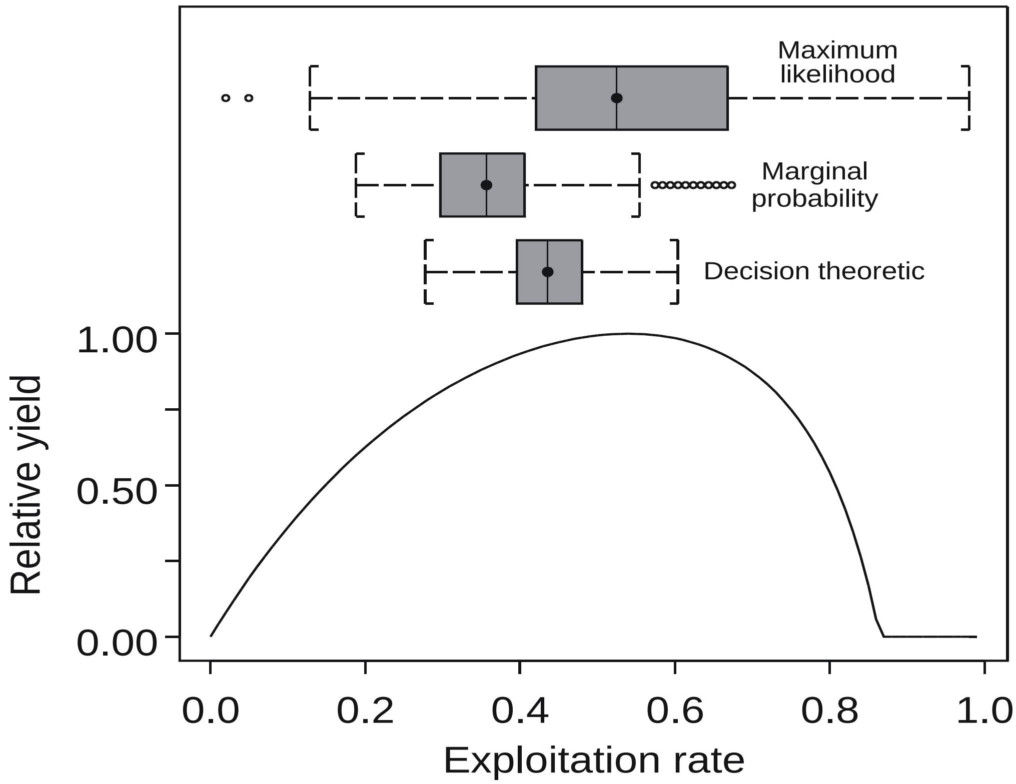

Figure 3.2.1. A summary of reference point estimation simulation results, based on the life history and fisheries for alewife, using stock recruitment values of alpha = 50 and sigma = 0.9. The solid line gives the known equilibrium yield, scaled to a maximum of one, as a function of the exploitation rate based on the assumed population dynamics. The box plots show the distribution of the reference fishing mortality rates estimated from 500 simulated spawner-recruit datasets (based on the same dynamics) using the three methods. The maximum likelihood method produces the maximum likelihood estimate of F_{msy}, the marginal probability method uses the mode of the marginal probability density for the maximum reproductive rate to obtain a reference F (F_{marg}) and a reference F is found using the decision theoretic method by maximizing the expected yield ($F_{max.E[C]}$). For each method, the solid line is the median value and the grey shaded region shows the inter-quartile range. The whiskers are drawn to the nearest value within 1.5 times the inter-quartile range. Points beyond these limits are plotted as points (from Gibson and Myers 2004). Note: Their marginal probability fishing reference mortality rate is not discussed in this document.

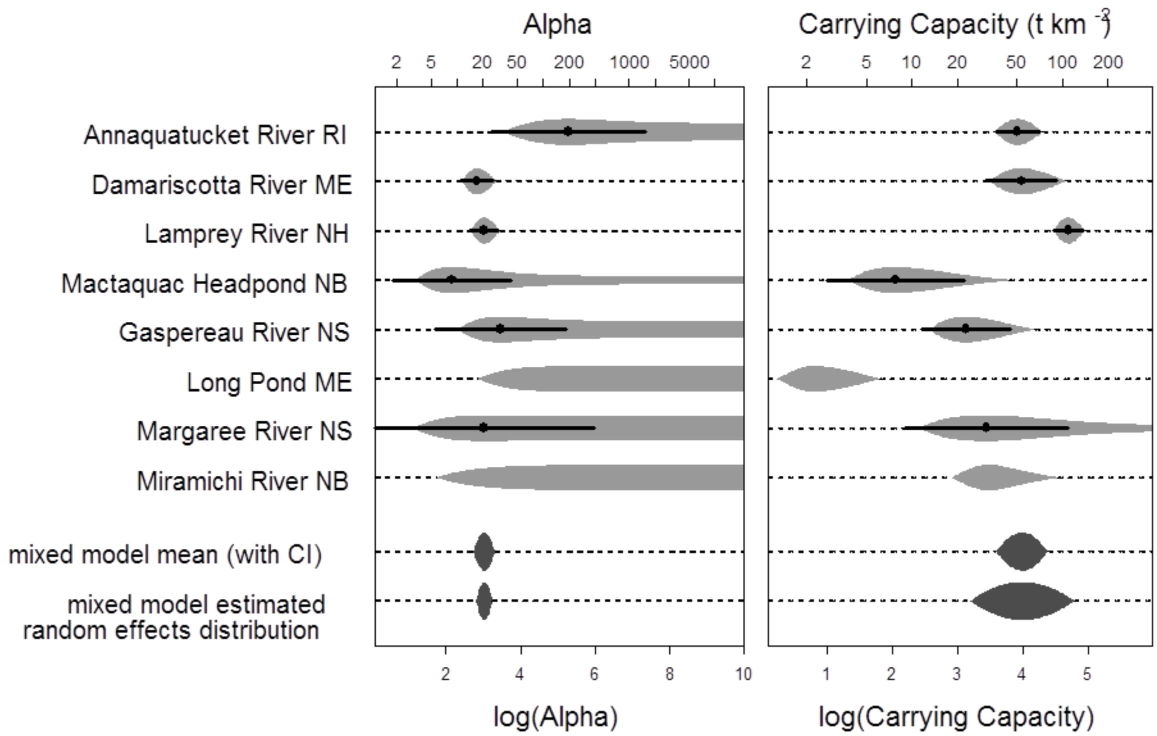

Figure 3.3.1. A meta-analytic summary of the maximum lifetime reproductive rate (alpha) and the habitat carrying capacity for eight alewife populations. The light grey shaded regions are individual fits that depict the profile likelihood for each parameter, truncated to show the 95% confidence interval. The profile is used to gauge the relative plausibility of different values (wider is more plausible). The black dot is the maximum likelihood estimate for each parameter. Convergence of the nonlinear least squares algorithm was not obtained for the Miramichi and Long Pond stocks. Where convergence was obtained, approximate asymptotic 95% confidence intervals are shown (black line). The dark grey shaded regions show summaries of the mixed model results. The "mixed model mean" represents the estimated mean of the logarithm of each parameter with a 95% confidence interval. The "mixed model estimated random effects distribution" is the normal distribution for the logarithm of each parameter based on its mean and variance estimated with the mixed effects model (from Gibson 2004).

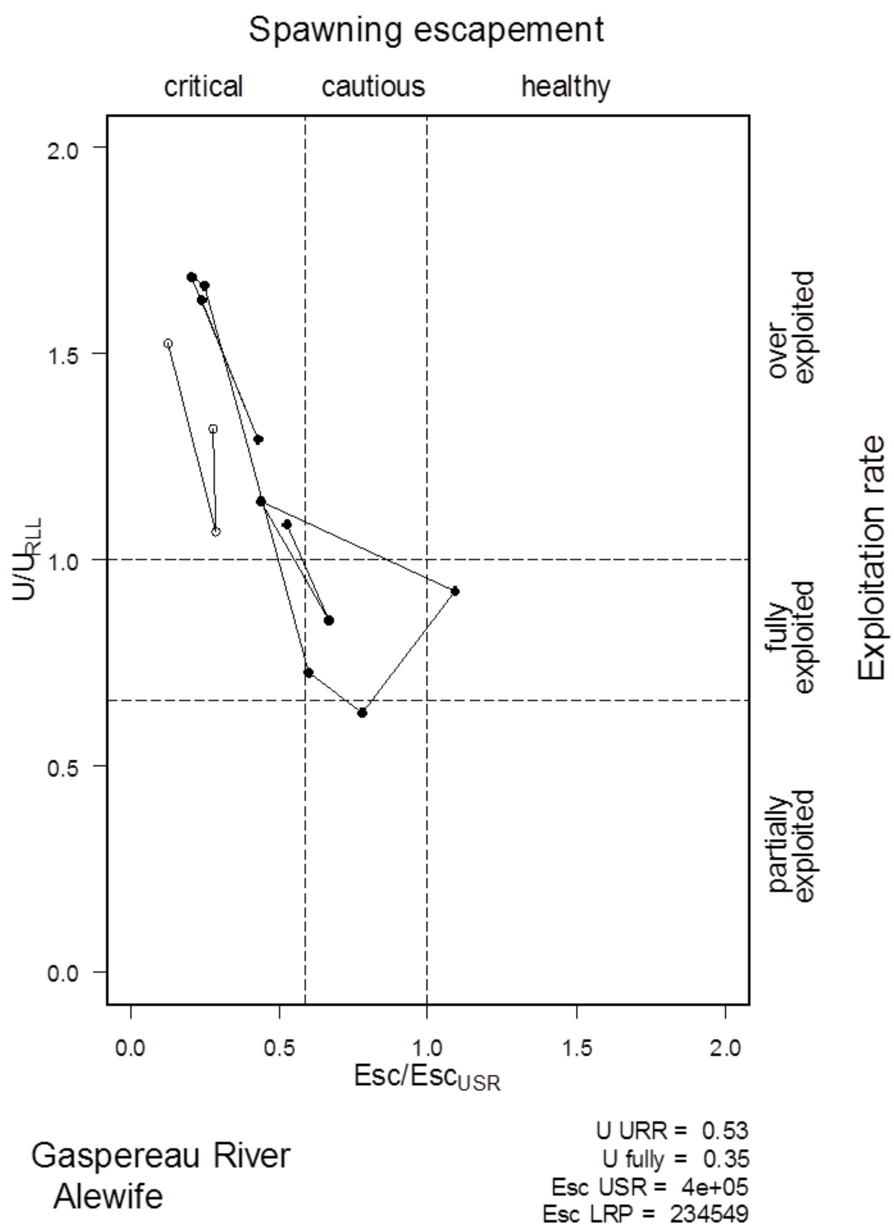

Figure 3.4.1. Example of an assessment summary figure showing the status of the stock and fishery relative to the spawning escapement and fishing mortality reference levels. Abbreviations are: U_{RLL} – the exploitation rate at the removal reference level, above which the overfishing would be considered to be occurring; U_{fully} – a lower removal reference level, below which a population would be considered underexploited; Esc_{USR} – the spawning escapement at the upper stock reference level, above which escape is considered good; and Esc_{LRP} – the spawning escapement at the limit reference point, below which mortality should be reduced to as low a level as possible. For illustrative purposes, results from the most recent Gaspereau River assessment (McIntyre et al. 2007) are shown for two time periods: 1982-1984 (open circles) and 1997 to 2006 (closed circles). Fishing mortality reference points are those proposed in this document. The upper stock reference level is the value currently being used for this stock, and the limit reference point corresponds 10% of the unfished spawner biomass as estimated by Gibson and Myers (2003a).

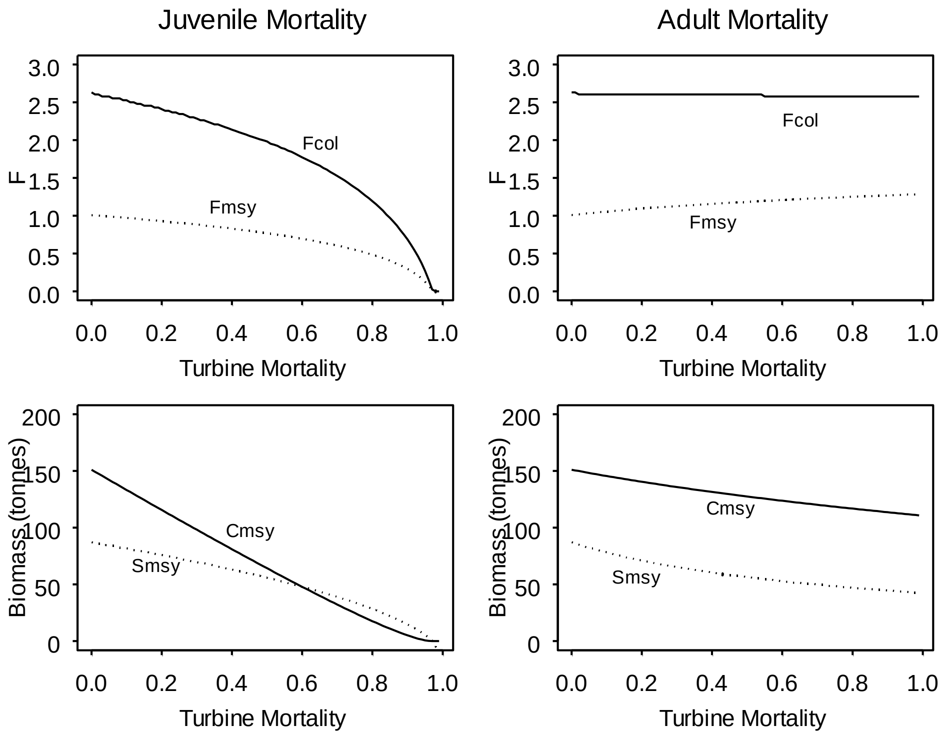

Figure 3.5.1. The relationship between juvenile (left column) and adult (right column) and fishery reference points for Gaspereau River alewife. F_{msy}, S_{msy} and C_{msy} are the equilibrium fishing mortality rate, spawner biomass and catch at MSY. F_{col} is the fishing mortality rate that will drive the population to extinction. Juvenile turbine mortality was assumed to occur after compensatory mortality. Adult turbine mortality occurs after spawning (from Gibson 2004).

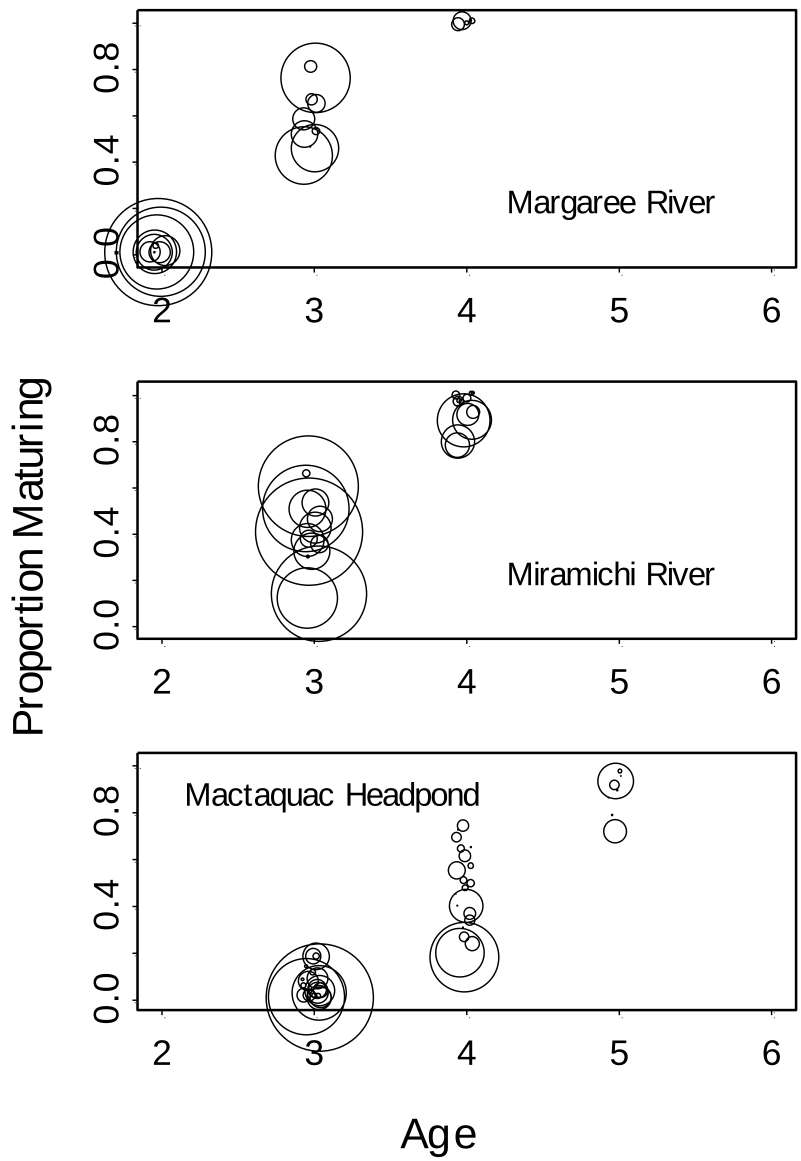

Figure 4.1.1. Variability in the age at maturity for the Margaree River, Miramichi River and Mactaquac Headpond alewife populations. Each point (or circle) represents the proportion of fish within a cohort that were alive at a given age and matured at that age. The size of the circle is proportional to the number of immature fish in the cohort at that age. Points are jittered slightly to facilitate display (from Gibson and Myers 2003a).

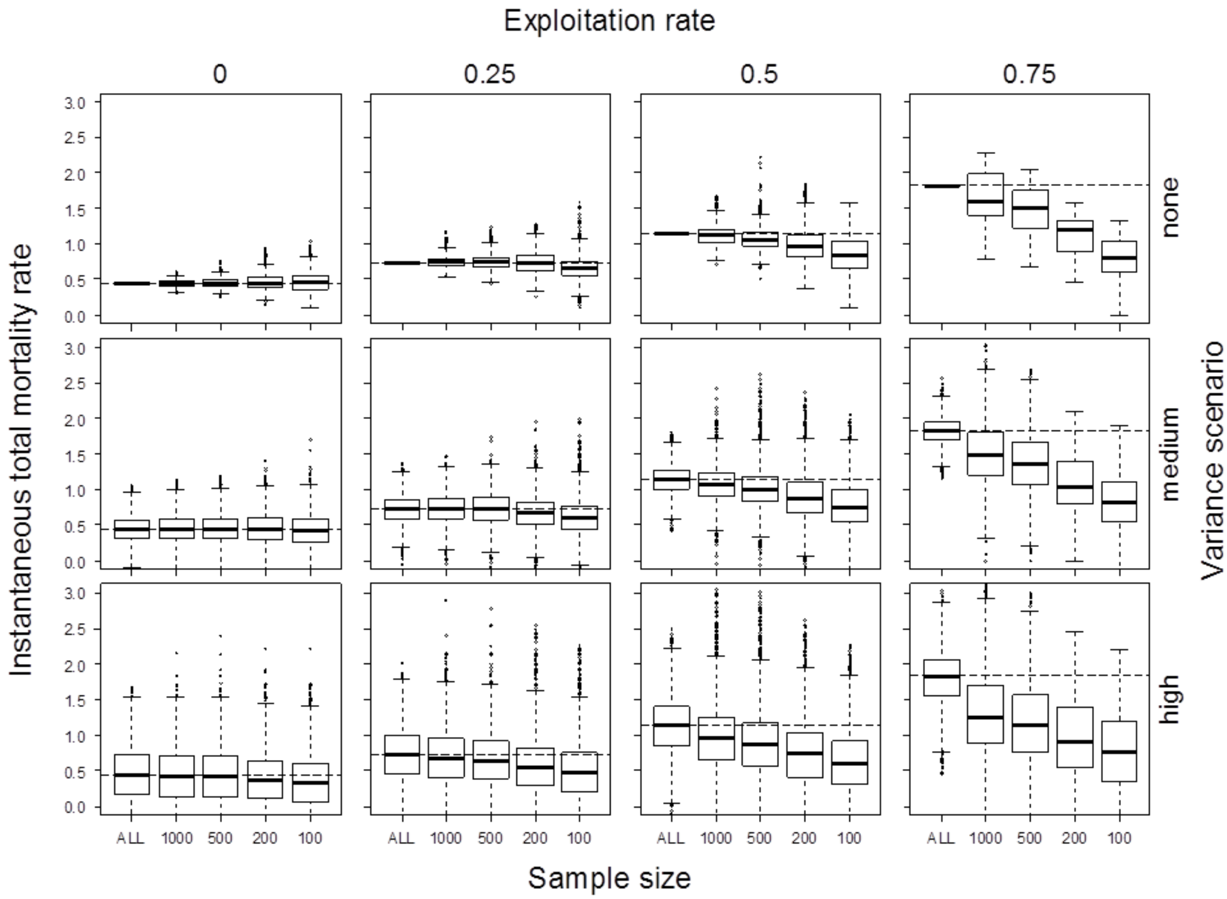

Figure 4.1.2. Boxplots summarizing the results of the simulations of catch curve analyses to estimate the instantaneous total mortality rate (Z) for alewife, using a generalized linear model using age with independent variable. Three variance scenarios were analysed: no variability (top row), medium variability (middle row) and high variability (bottom row). Different levels of mortality were simulated using exploitation rates of 0.00, 0.25, 0.50, and 0.75 (left to right columns, respectively). In each panel, the horizontal dashed line is the true value of Z, and each boxplot summarizes 3750 catch curve analyses with randomly drawn sample sizes of all fish, 1000, 500, 200 and 100 fish respectively.

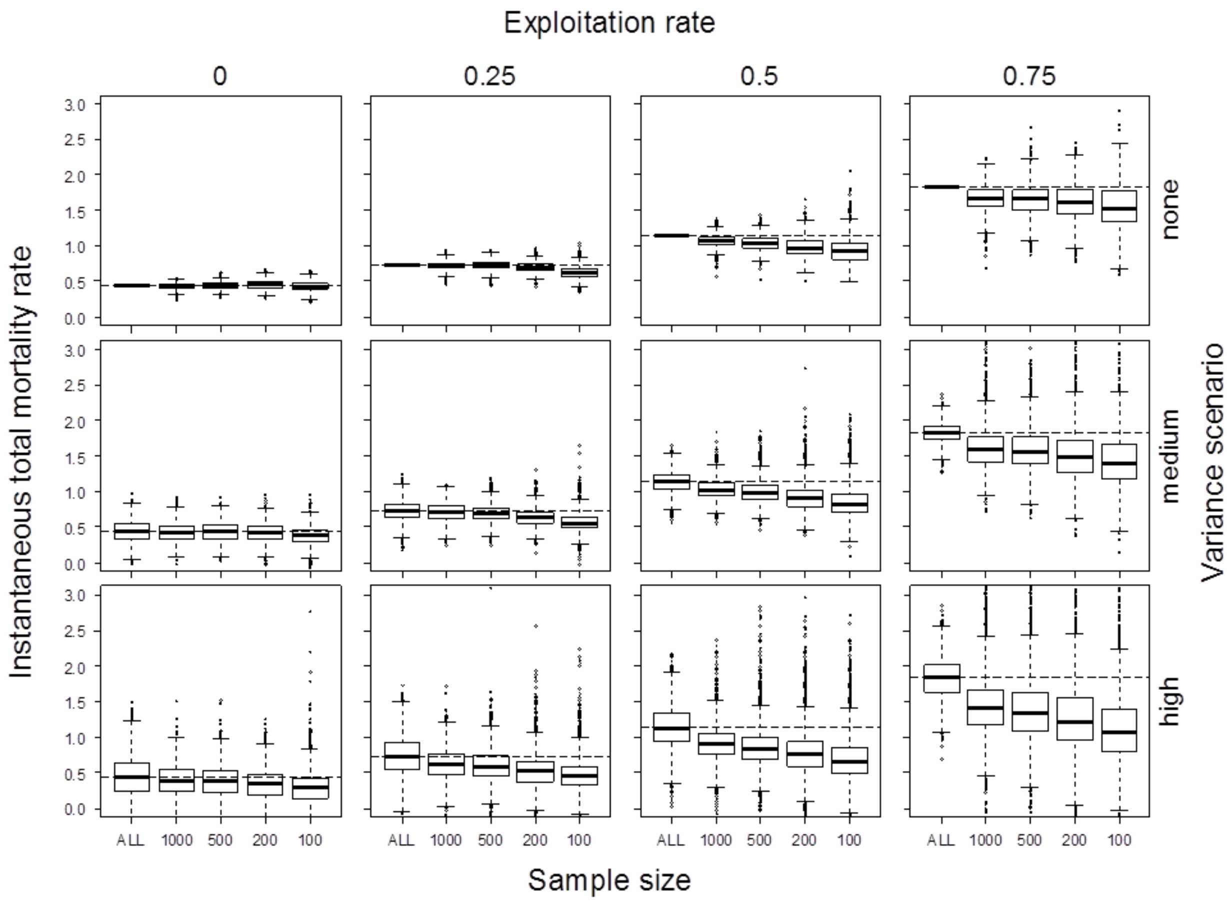

Figure 4.1.3. Boxplots summarizing the results of the simulations of catch curve analyses to estimate the instantaneous total mortality rate (Z) for alewife, using a generalized linear model using previous spawning as the independent variable with separate intercepts for each age-at-maturity category. Three variance scenarios were analysed: no variability (top row), medium variability (middle row) and high variability (bottom row). Different levels of mortality were simulated using exploitation rates of 0.00, 0.25, 0.50, and 0.75 (left to right columns, respectively). In each panel, the horizontal dashed line is the true value of Z, and each boxplot summarizes 3750 catch curve analyses with randomly drawn sample sizes of all fish, 1000, 500, 200 and 100 fish respectively.

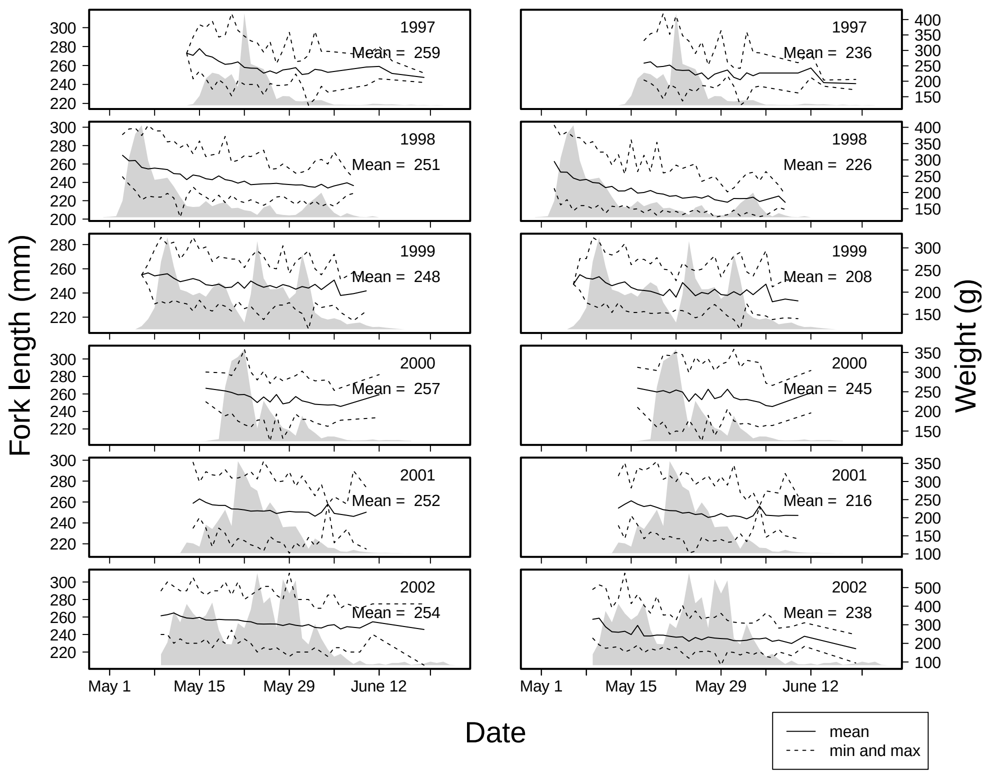

Figure 4.3.1. Fork lengths (left panels) and weights (right panels) of fish collected each day during annual sampling of spawning runs from 1997 to 2002. Solid line shows mean daily fork length. Dashed lines show daily minimum and maximum fork lengths. Grey shading represents the relative daily abundance of fish sampled.

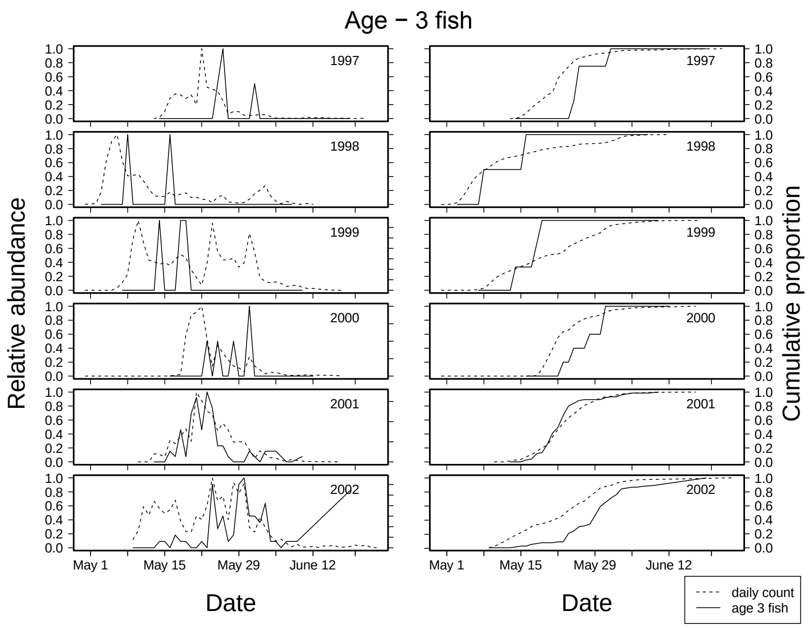

Figure 4.3.2. Relative abundance (left panels) and cumulative proportions (right panels) of age-3 fish collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of age-3 fish. Dashed lines represent the daily count of fish of all ages.

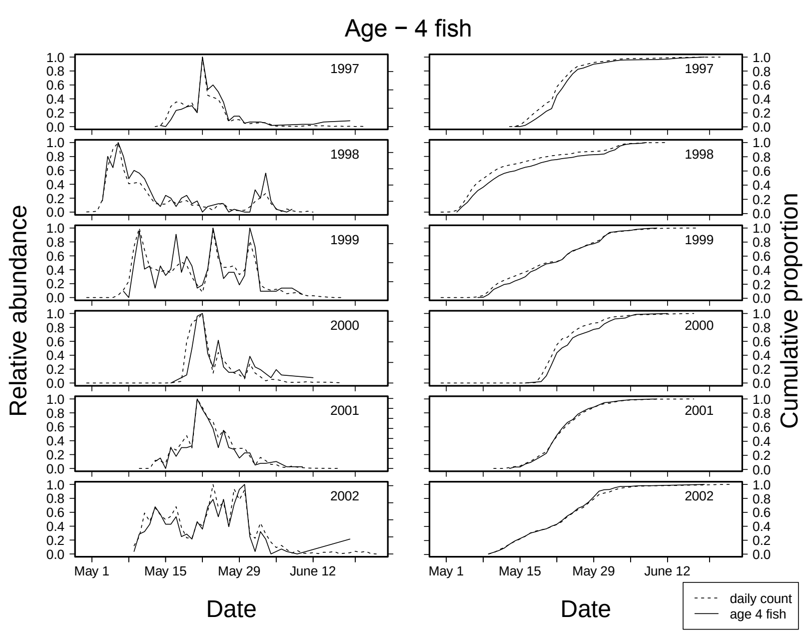

Figure 4.3.3. Relative abundance (left panels) and cumulative proportions (right panels) of age-4 fish collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of age-4 fish. Dashed lines represent the daily count of fish of all ages.

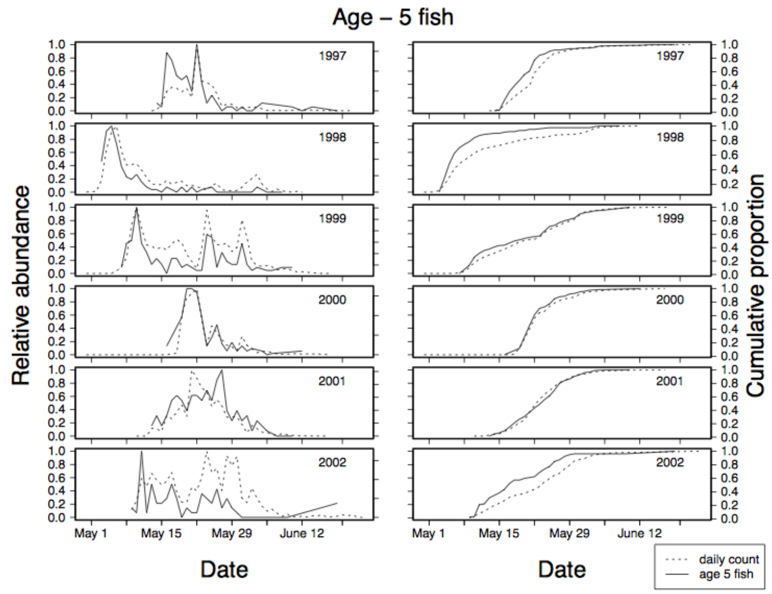

Figure 4.3.4. Relative abundance (left panels) and cumulative proportions (right panels) of age-5 fish collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of age-5 fish. Dashed lines represent the daily count of fish of all ages.

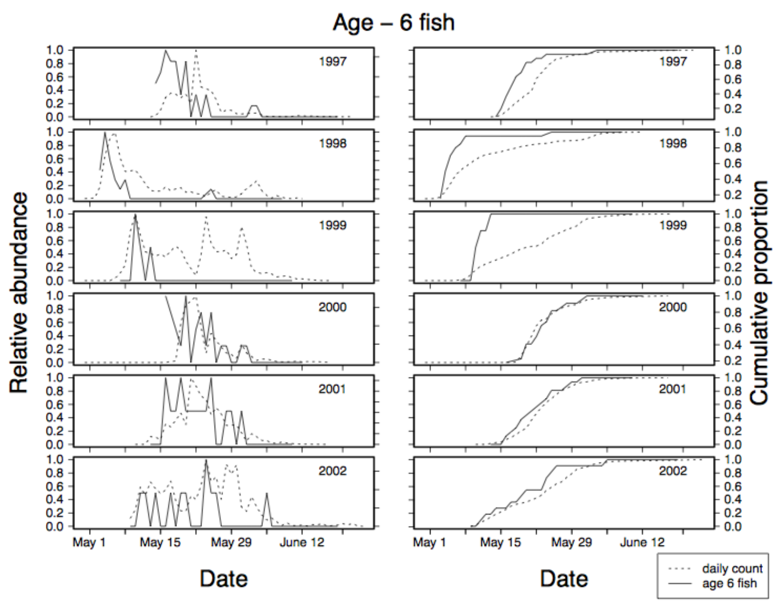

Figure 4.3.5. Relative abundance (left panels) and cumulative proportions (right panels) of age-6 fish collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of age-6 fish. Dashed lines represent the daily count of fish of all ages.

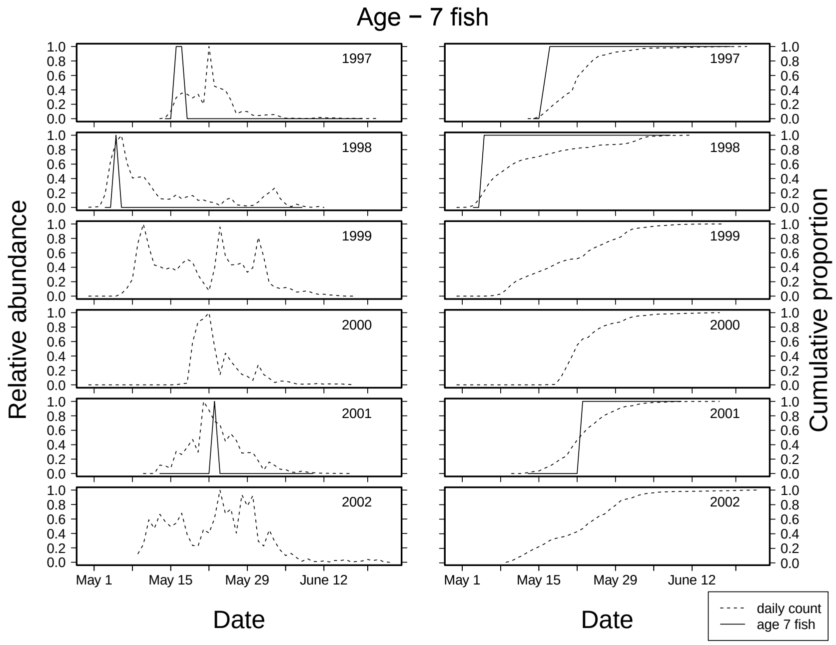

Figure 4.3.6. Relative abundance (left panels) and cumulative proportions (right panels) of age-7 fish collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of age-7 fish. Dashed lines represent the daily count of fish of all ages.

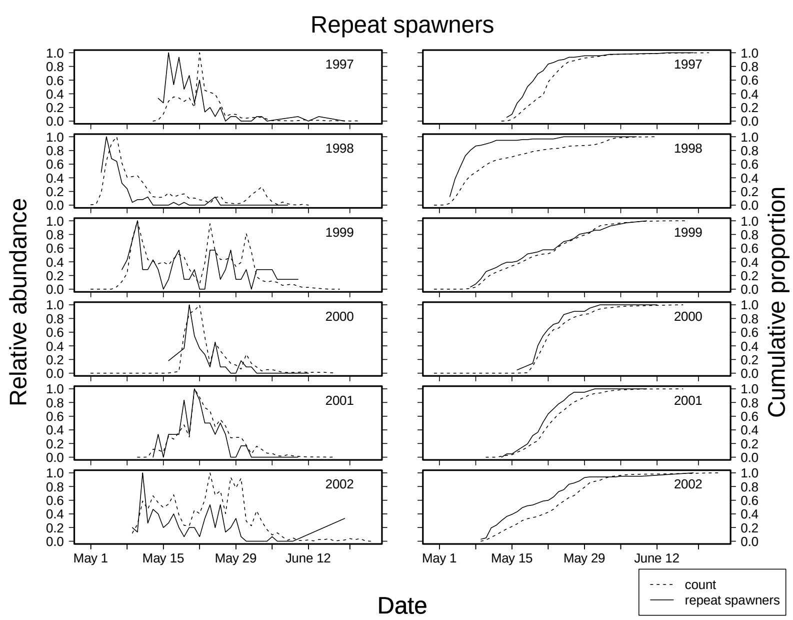

Figure 4.3.7. Relative abundance (left panels) and cumulative proportions (right panels) of repeat spawners collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of repeat spawners. Dashed lines represent the daily count of all fish.

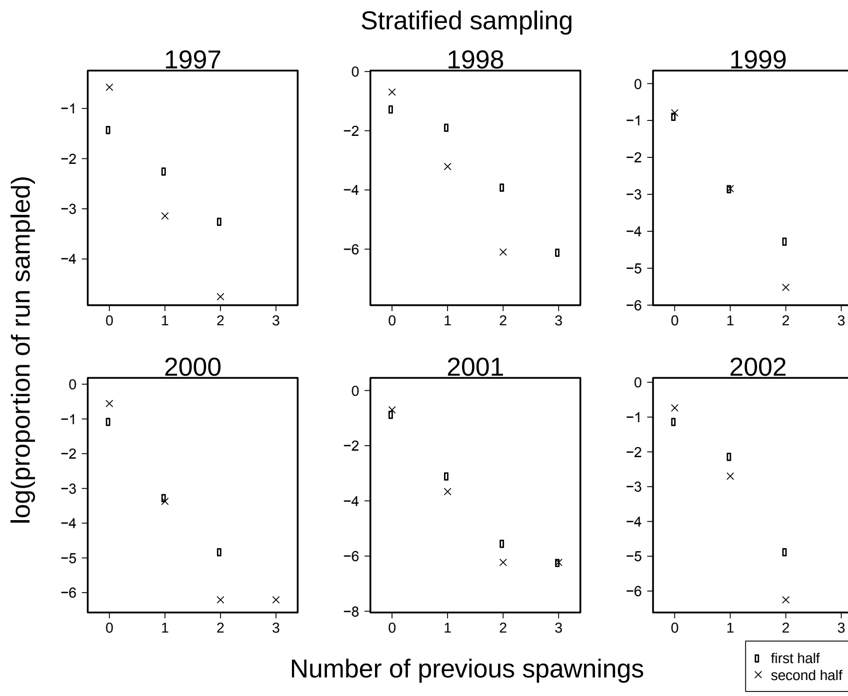

Figure 4.3.8. The proportion of alewife sampled (log-transformed) during each year's spawning migration based on number of previous spawnings. The proportions of Alewife sampled during the first half of each spawning migration are represented by circles, while proportions sampled during the second half are represented by Xs.

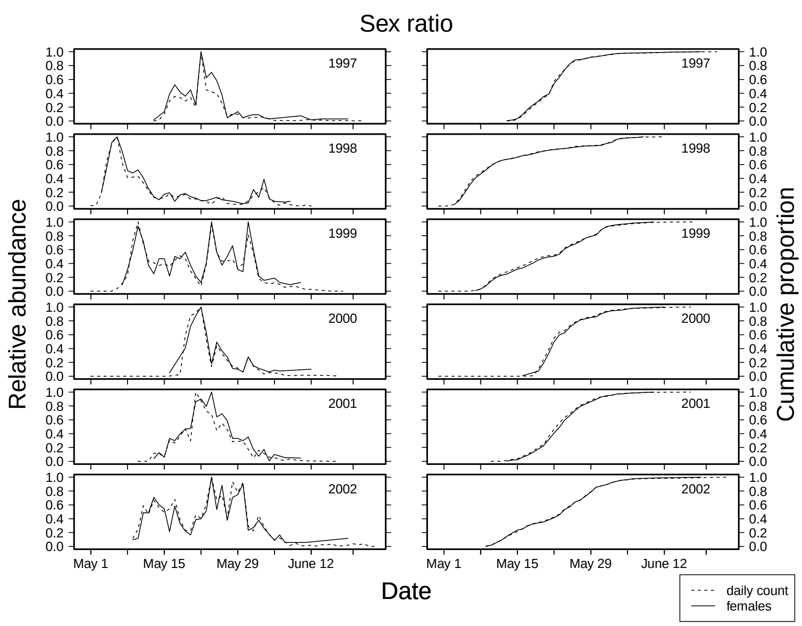

Figure 4.3.9. Relative abundance (left panels) and cumulative proportions (right panels) of females collected each day during annual spawning run sampling. Relative abundance is the number of fish collected each day divided by the maximum number of fish collected on a single day during sampling. Solid lines represent daily counts of females. Dashed lines represent the daily count of all fish.

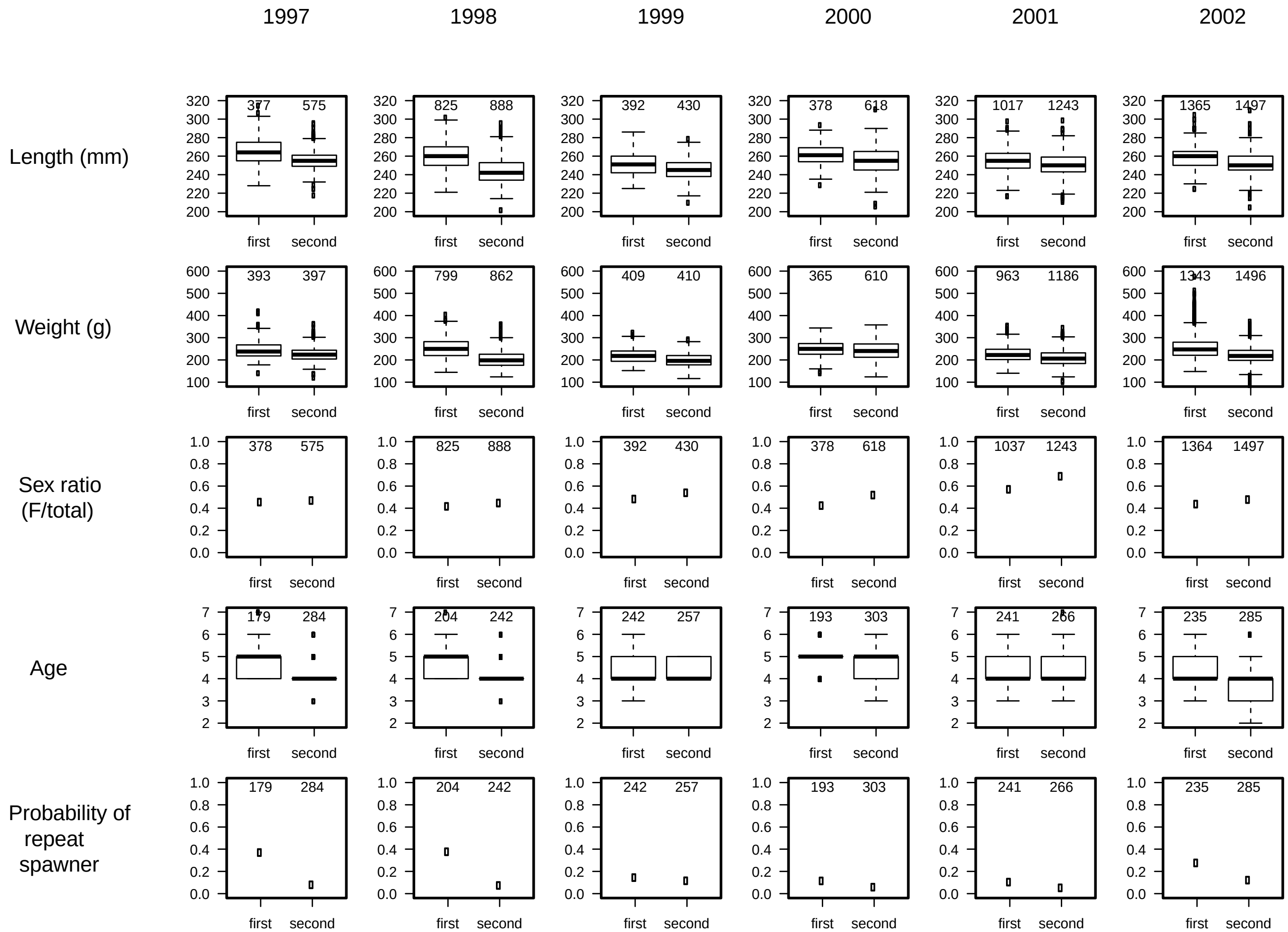

Figure 4.3.10. Summary plots of Alewife biological characteristics during the first and second half of spawning migrations in 1997-2002. Sex ratio represents the proportion of females out of total population sampled in each half of each year, and the probability of repeat spawner is the proportional likelihood of sampling a repeat spawner in each half of the spawning migration each year.

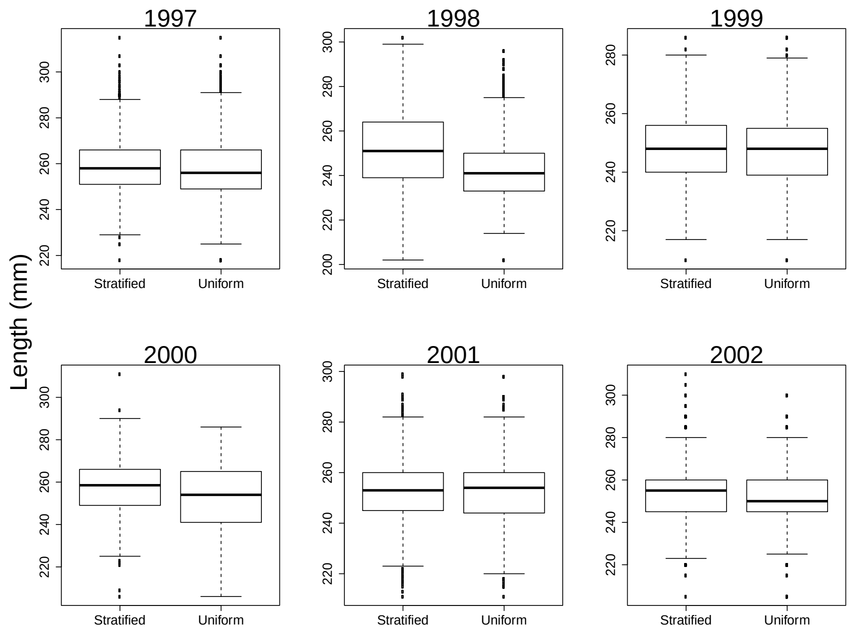

Figure 4.3.11. Fork lengths (mm) of Alewife sampled during the Gaspereau River spawning migration each year (stratified, left boxplot), and fork lengths that would be expected if a uniform sampling scheme had been used (uniform, right boxplot).

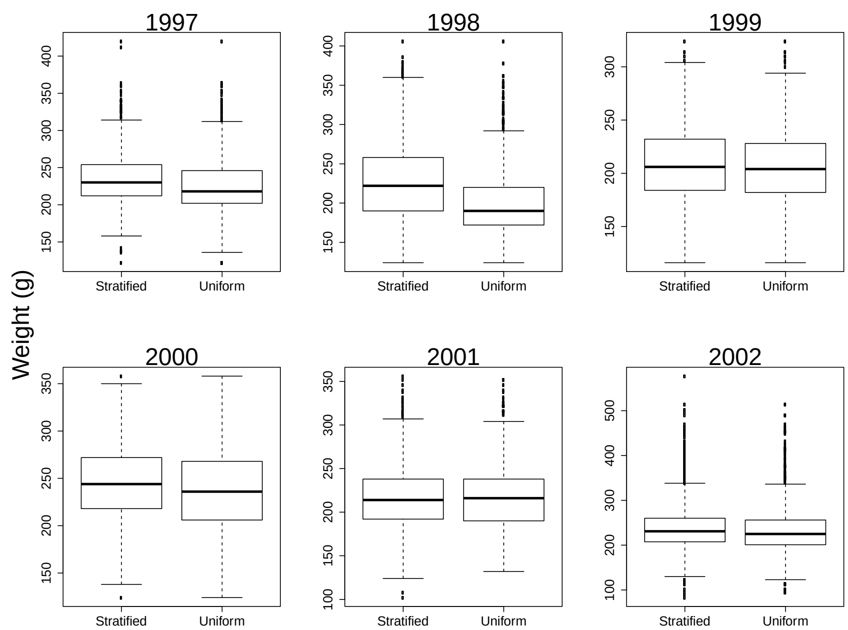

Figure 4.3.12. Weights (g) of Alewife sampled during the Gaspereau River spawning migration each year (stratified, left boxplot), and weights that would be expected if a uniform sampling scheme had been used (uniform, right boxplot).

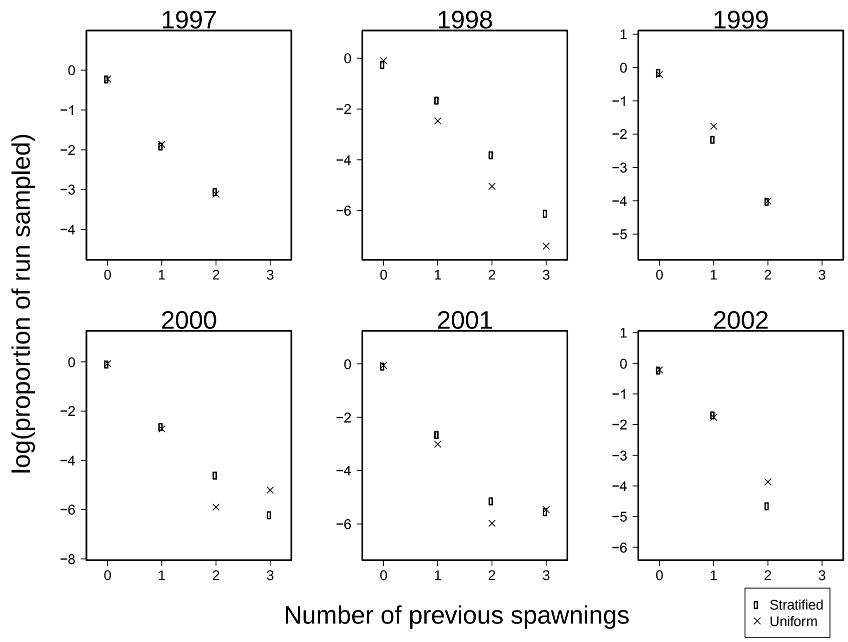

Figure 4.3.13. Relative number of alewife (log-transformed proportions) that had previously spawned 0, 1, 2, and 3 times observed using a stratified (circles) and uniform (X) sampling scheme.

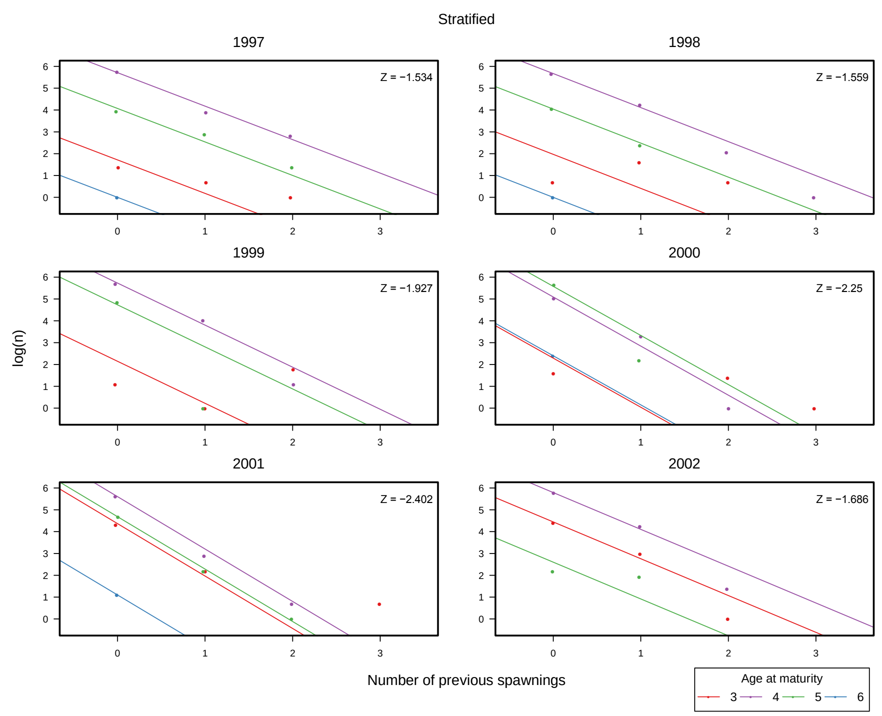

Figure 4.3.14. Log-linear relationship in Alewife age structure for the stratified sampling dataset from the Gaspereau River. Regression lines are shown for each maturation age, based on the number of previous spawnings.

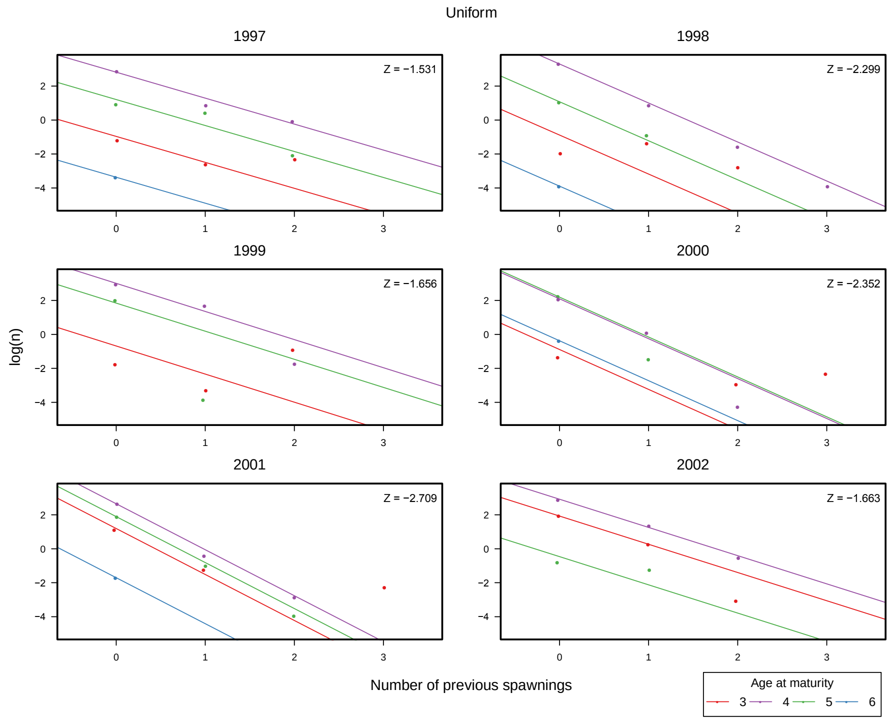

Figure 4.3.15. Log-linear relationship in Alewife age structure for the expected uniform sampling dataset based on the Gaspereau River stratified dataset. Regression lines are shown for each maturation age, based on the number of previous spawnings.

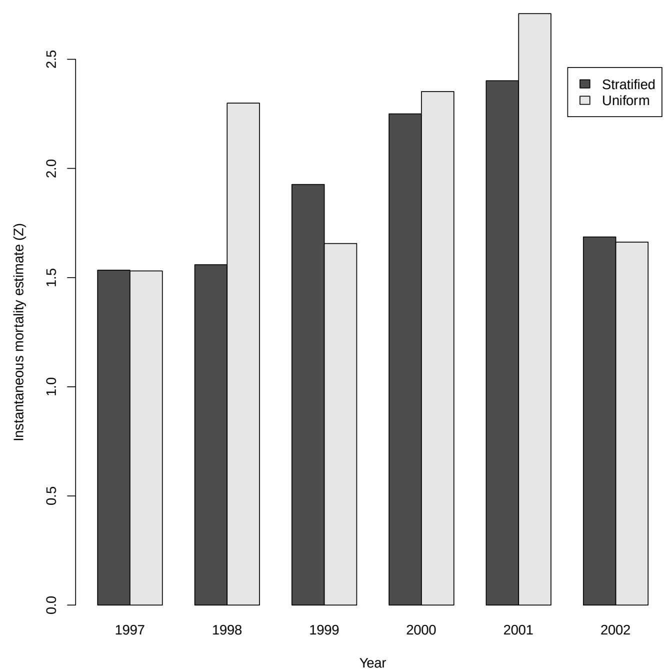

Figure 4.3.16. Instantaneous mortality estimates for the stratified (dark grey) and uniform (light grey) datasets.