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Review of approaches and methods to assess Environmental Flows across Canada and internationally

Examen des approches et des méthodes d'évaluation des débits environnementaux au Canada et à l'échelle internationale

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TABLE OF CONTENTS

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ABSTRACT

Freshwater resources are under increasing threat from anthropogenic activities, both in terms of consumptive and non-consumptive use. The increasing societal demands for water have led to substantial flow alterations in rivers in Canada. Flow alteration can be directly linked to impacts on the physical and chemical attributes and processes of rivers and subsequent ecological changes. In addition to increasing demand of water, the ecological, social and cultural values of rivers are increasingly being recognized. With so many competing needs for water, there is an urgent need to develop sustainable environmental flow management guidelines to manage the risk associated with alterations to the flow regime in Canadian rivers. A national environmental flow framework would also support the habitat protection provisions of the *Fisheries Act* (S.35).

This document is intended to serve as an input for a scientific review process at a national level and to provide background information on *i*) conflicting terminology used in the environmental flows literature, *ii*) environmental flow assessment methodological approaches and *iii*) current status of environmental flow guidelines used in different jurisdictions in Canada. The information contained in this document was critically discussed by an assembled group of experts who will provide further scientific advice on environmental flow regimes for the Canadian context in a separate document.

The terminology used in environmental flow assessment literature is variable; "instream flow needs" and "environmental flows" seem to be the most widely used and inclusive terminology. The endorsement of these terms for wider use in Canadian context needs to be further verified by an assembled group of experts.

The techniques used in environmental flow regime assessment were classified into four general categories (hydrological, hydraulic rating, habitat simulation and holistic methodologies and frameworks) and the benefits and weaknesses are reviewed. The four methodological categories differ drastically in the scope and implementation costs and therefore, are suited for different level of assessment of environmental flow regimes. Most assessment methods are not based on tested relationships between the extent of flow regime alteration and ecological response. A recent trend seems to suggest that there is an increasing recognition that any environmental flow method used alone will not be sufficient for determining environmental flows in all situations; the holistic methods and frameworks (wherein a combination of other methods are used) are increasingly common especially in large scale projects.

The examination of the current methodologies used in different jurisdictions in Canada revealed that many provinces do not have an established guideline to be used for determining environmental flow regimes. Some provinces have established guidelines to be used in uncontroversial situations (i.e. cases that are believed to lead to no harmful alteration, disruption or disturbance to fish habitat, "HADD"). None of the jurisdictions appeared to have an established environmental flow framework that is used in larger-scale projects (i.e. "potential or incurring HADD") but evaluation is carried out on a case-by-case basis.

Various options for establishing a national environmental flow regime framework are available but at least two suggestions are to be examined more closely by an assembled group of experts. The first option is related to incorporating a framework similar to the process characterized by the Ecological Limits of Hydrologic Alteration (ELOHA; Poff et al. 2010) to at least some degree in Canada. The second option is to establish a two-tiered framework wherein the more general Level 1 ("no HADD") assessment would be based on some combination of hydrologically-based guidelines to protect the natural variability in river flow regimes with some cut-off values to terminate water withdrawal during the lower flows. The more specific Level 2 ("potential HADD") assessment would be based on a holistic approach for which a detailed protocol would need to be drafted to ensure the ecological integrity, comparability and transparency across Canada. Regardless of the type of framework to be established, it is fundamental that the established environmental flow standards are preceded and followed by a controlled monitoring program and the possibility to refine the environmental flow regime standards by adaptive management in an iterative process.

The need to establish a system to categorize Canadian rivers (or their segments) into ecological management classes needs to be further discussed by an assembled group of experts. The potential benefit of such classification would be the possibility to design different environmental flow standards based on the ecological or societal "value" of various river sections. Finally, a time-frame must be determined to ensure that a national environmental flow framework will be established in an expedited manner.

RÉSUMÉ

Les ressources d’eau douce sont de plus en plus menacées par les activités anthropiques, soit pour des fins de consommation ou non. L’accroissement de la demande en eau des sociétés a entraîné la modification importante du débit des rivières au Canada. Cette modification peut être directement liée aux incidences sur les processus et les caractéristiques physiques et chimiques des rivières ainsi qu’aux changements écologiques subséquents. En plus de la demande accrue en eau, les valeurs écologiques, sociales et culturelles des rivières sont de plus en plus reconnues. Comme il y a tant de besoins concurrentiels en eau, il est urgent d’élaborer des lignes directrices sur la gestion des débits environnementaux durables afin de gérer le risque lié à la modification du débit des rivières canadiennes. Un cadre national sur les débits environnementaux appuierait également les dispositions relatives à la protection de l’habitat de la *Loi sur les pêches* (article 35).

Le présent document a pour but de documenter un processus d’examen scientifique à l’échelle nationale et de fournir des renseignements généraux sur *i*) la terminologie contradictoire utilisée dans les textes sur les débits environnementaux, *ii*) les approches méthodologiques utilisées pour évaluer les débits environnementaux et *iii*) l’état actuel des lignes directrices sur les débits environnementaux utilisées dans les différentes provinces du Canada. Les renseignements contenus dans ce document ont été discutés par un groupe d’experts rassemblés. Ces experts vont ensuite fournir des avis scientifiques sur les débits environnementaux qui s’appliquent au contexte canadien dans un document distinct.

La terminologie utilisée dans les textes sur l’évaluation des débits environnementaux varie. Les termes « norme de débit minimal » et « débits environnementaux » semblent les plus utilisés et généraux. Avant qu’une utilisation plus répandue de ces termes dans le contexte canadien soit recommandée, un groupe d’experts devra en réaliser un examen approfondi.

Les techniques d’évaluation des débits environnementaux utilisées ont été divisées en quatre grandes catégories (hydrologique, l’évaluation hydraulique, la simulation de l’habitat et les méthodes et les cadres holistiques); on examine actuellement les avantages et les inconvénients de ces techniques. La portée et les coûts de mise en œuvre des techniques diffèrent nettement d’une catégorie à l’autre. Par conséquent, ces techniques permettent d’évaluer les débits environnementaux à différents niveaux. La majorité des méthodes d’évaluation ne sont pas fondées sur les relations éprouvées entre la portée de la modification des débits et la réaction écologique. Selon une tendance récente, il semblerait que l’on reconnaisse de plus en plus le fait que toute méthode utilisée seule ne permettra pas de déterminer les débits environnementaux dans tous les cas; les méthodes et les cadres holistiques (en combinaison avec d’autres méthodes) sont de plus en plus utilisés, plus particulièrement dans le contexte de projets de grande envergure.

L’examen des méthodes actuellement utilisées dans les différentes provinces du Canada révèle que plusieurs provinces ne possèdent aucune ligne directrice relative à la détermination des débits environnementaux. Certaines provinces ont établi des lignes directrices pour les situations qui ne prêtent pas à controverse (c.-à-d. les cas qui ne devraient pas entraîner la détérioration, la destruction ou la perturbation de l’habitat du poisson). Il semble qu’aucun cadre sur les débits environnementaux n’ait été établi dans les provinces pour les projets de grande envergure (c.-à-d. les cas pouvant entraîner la détérioration, la destruction ou la perturbation de l’habitat), mais une évaluation est effectuée au cas par cas.

Il existe plusieurs options pour établir un cadre national sur les débits environnementaux, mais au moins deux propositions doivent faire l'objet d'un examen approfondi par un groupe d'experts. La première option vise l'intégration, au Canada, d'un cadre semblable au processus décrit dans le livre *Ecological Limits of Hydrologic Alteration* (Poff et coll. 2010), dans une certaine mesure. La deuxième option consiste à établir un cadre à deux volets. L'évaluation générale de niveau 1 (aucune détérioration, destruction ou perturbation de l'habitat) serait fondée sur une combinaison de lignes directrices axées sur l'hydrologie afin de protéger la variabilité naturelle du débit des rivières et comporterait des valeurs limites imposant l'arrêt de l'extraction d'eau lorsque le débit est faible. L'évaluation précise de niveau 2 (détérioration, destruction ou perturbation potentielle de l'habitat) serait fondée sur une approche holistique pour laquelle un protocole détaillé devrait être rédigé afin de garantir l'intégrité écologique, la comparabilité et la transparence dans l'ensemble du Canada. Peu importe le type de cadre qui doit être établi, il est essentiel que l'établissement des normes sur les débits environnementaux soit précédé et suivi par un programme de surveillance contrôlée et la possibilité de perfectionner ces normes en fonction d'une gestion adaptative dans le cadre d'un processus itératif.

Un groupe d'experts doit discuter en profondeur de la nécessité de mettre en place un système permettant de classer les rivières canadiennes (ou leurs parties) dans différentes catégories de gestion écologique. Cette classification permettrait d'élaborer différentes normes sur les débits environnementaux en fonction de la « valeur » écologique ou sociale de diverses parties d'une rivière. Enfin, un échéancier doit être établi afin de garantir que le cadre national sur les débits environnementaux sera élaboré rapidement.

1. INTRODUCTION

1.1 NEEDS FOR ENVIRONMENTAL FLOW FRAMEWORK IN CANADA

Freshwater resources are under increasing threat from anthropogenic activities, both in terms of consumptive and non-consumptive use (Poff et al. 2010; Richter et al. 2011). The increasing societal demands for water have led to substantial flow (i.e. discharge) alterations in rivers both in Canada and internationally. Flow alteration can be directly linked to impacts on the physical attributes of rivers and the resulting ecological changes (Figure 1, Clarke et al. 2008; Poff and Zimmerman 2010). Furthermore, the risk of adverse changes will increase with increasing magnitude of flow alteration (Poff and Zimmerman 2010). Moreover, the effects of extreme daily fluctuations such as hydropeaking operations have also been shown to lead to potentially deleterious effects on numerous biotic and physical components of riverine ecosystems (Bain 2007).

Parallel to increasing demands to consume water, the ecological, social and cultural values of rivers are increasingly being recognized, together with the ecosystem services provided by "environmental flows" (Richter 2010; see terminology in Section 2 for detailed definitions). According to the Brisbane Declaration (2007, p. 1), "*Freshwater ecosystems are the foundation of our social, cultural, and economic well-being*".

With so many competing needs for water, the question is "*How much water does a river need?*" (Richter et al.1997). Moreover, in order to ensure the conservation and protection of fish and other valued ecosystem components, it has been recognized that the riverine ecosystem is not only dependent on available water quantity but also on the natural dynamic character of the flow regime (Poff et al. 1997). The Instream Flow Council (Annear et al. 2004) also stress the need to maintain the integrity of inter- and intra-annual flow patterns that mimic the natural hydrograph. Thus, there is an urgent need to develop sustainable, environmental flow management guidelines to regulate the alterations to the flow regime.

In Canada, the consideration of environmental flow requirements is indirectly mandated by federal legislation. The federal government, acting through Fisheries and Oceans Canada (DFO), administers the *Fisheries Act*. Section 32 of the Fisheries Act prohibits "*destruction of fish by means other than fishing*", and Section 35 prohibits "*harmful alteration, disruption or destruction of fish habitat*" (i.e. HADD). Despite this legislation, there are currently no federal guidelines on how to determine environmental flows in Canada, and current recommendations have been decided *ad hoc* at regional and provincial levels (Peters et al. 2012). The fact that there is no existing national framework to set environmental flow standards has led to a situation where fisheries resources, fish habitat and the supporting freshwater ecosystems may not be consistently protected across Canada.

With increasing water demand, and potentially changing background levels in water availability (as predicted from current consensus on the long-term effects of global climate change; IPCC 2007), there is an urgent need to establish such an environmental flows framework in Canada. This is a challenge, given the large geographic area to be covered which encompasses a substantial range of hydrological, geomorphological and ecological variety across the country. Moreover, although many different environmental flow methods exist (over 200; Tharme 2003), determining an effective approach for the application of environmental flow thresholds across Canada has proved elusive.

1.2 PURPOSE AND SCOPE OF REVIEW

The purpose of this report is to:

1) Distinguish and comment on the conflicting terminology used in the environmental flows literature;
2) Provide a summary and comparative review of current environmental flow assessment methodological approaches; and
3) Collate information on the current status of environmental flow guidelines used in different jurisdictions in Canada and internationally.

This report served as a draft discussion document for an assembly of national and international experts in the area of environmental flows science in order to *provide formal scientific advice* for the Canadian context (see accompanying Science Advisory Report published by DFO's Canadian Science Advisory Secretariat).

It must be noted that the intent of this report is not to review questions related to the implementation of any potential environmental flow framework or to consider any aspect related to federal or provincial legislation, or legal responsibilities in various jurisdictions as may be related to environmental flows. These are critically important elements of effective environmental flow regime management that will be addressed in other venues.

Due to the wealth of literature concerning environmental flow assessment methods, the information collated in this report has largely relied on numerous existing review articles. Greater emphasis was put on the more recent articles describing different approaches. While not discounting the value of older literature, it was believed that *i*) the information described in the older literature would be contained within the numerous review articles on the subject and *ii*) the more recent articles are building upon the former knowledge and have potentially addressed any deficiencies that the older methods may have included. Here we aim to provide a concise description of general groupings of environmental flow methods. In doing so, we identify and briefly describe the more commonly-used or emerging methods and frameworks.

As per the status quo of environmental flow guidelines in Canada and the trends internationally, we limited the scope of this report to considerations that could be collated with "reasonable effort". This means that the report does not include a listing of every method or case study used in different jurisdictions, rather, the intent is to provide a general overview of the similarities, differences and trends related to environmental flow considerations in various jurisdictions.

It is also recognized that defining environmental flow requirements could be very different in perennial (a stream which flows continuously all year) and intermittent (a stream that flows only for a part of the year; also seasonal or ephemeral stream) streams and rivers. Environmental flow frameworks are typically designed with a consideration that some flow remains in the river at any given time. This general approach to include perennial streams has been applied in this report and no special consideration is given to intermittent streams.

Finally, the requirements for environmental flow standards can be quite variable depending on the planned (or current) anthropogenic developments in various rivers. For example, different environmental flow "rules" would need to apply in cases where planned water use is related to: *i*) consumptive use requiring only *abstraction* of water; and *ii*) releases of water from impoundments leading to *augmentation* of water. Hydropeaking (defined as an artificial rapid and frequent alteration in water discharge) can be considered to be another special case of flow augmentation that requires special consideration for environmental flow standards. Therefore, the requirements and design of an environmental flow framework in the specific case with frequent hydropeaking was considered to be beyond the scope of this report.

For a more thorough review of environmental flow literature, the reader is advised to exhaust the following data sources;

1) The Instream Flow Council (IFC) website (www.instreamflowcouncil.org). The IFC is an organization that represents the interests of state and provincial fish and wildlife management agencies in the United States and Canada dedicated to improving the effectiveness of their instream flow programs.
2) The eFlowNet website (www.eflownet.org). The eFlowNet is an independent network whose overall goal is to integrate environmental flows into standard practices for the management and use of river basins.

2. CONFLICTING TERMINOLOGY OF "ENVIRONMENTAL FLOWS"

The science of flow management in the context of the protection of natural ecosystems is necessarily complex, and this is reflected in the plethora of terminology surrounding even its most basic concepts. It could be argued that this has been a significant obstacle to the development of an overarching framework for flow management: while practitioners struggle with definitions of basic concepts, then progress towards unifying framework will be impeded. The cross-cutting nature of flow management which brings together scientists from different disciplines is one cause of the current "etymological dissonance". Here we sidestep this problem by not reviewing all variants of terminology, but rather provide a definition and comparison of the commonly used terms that are currently used.

Instream Flow Need / Requirement

This term is defined by the Instream Flow Council (Annear et al. 2004) as: *"The amount of water flowing through a natural stream course that is needed to sustain, rehabilitate, or restore the ecological functions of a stream in terms of hydrology, geomorphology, biology, water quality, and connectivity at a particular level."*. The term was popular early in the fisheries literature (e.g. Annear et al. 2004). One key element when instream flow is invoked is the requirement to integrate the water use needs among all stakeholders. This includes ecosystem services provided to people living within the watershed. Some criticism of the term is related to the use of wording "instream": it is widely recognized that the river ecosystem form and function is also dependent on the lateral connectivity onto the floodplain, and "instream" *could* be interpreted as if the floodplain is excluded from the consideration.

Environmental Flow

The term is defined by the Brisbane Declaration (2007) as: "*Environmental flow describes the quantity, quality and timing of water flows required to sustain freshwater ecosystems and the human livelihoods and well-being that depend on these ecosystems*". The term was endorsed at International River Symposium (held in Brisbane, Australia, in 2007) by more than 750 delegates from 50 nations. It appears the term provides the most inclusive definition for the science of flow management in the context of the protection of natural ecosystems and water use needs among all stakeholders.

This term is used within the context of the present document.

Ecological Flow:

Ecological flows are intended to indicate flows required to maintain the character of an ecosystem. Implicit in the definition is the knowledge of what constitutes a freshwater ecosystem in its natural state and therefore, the ecological focus of the term varies in practice. Overall, the term is conceptually more limited than the terms above because stakeholder requirements are excluded from this definition. The term is well defined by the New Zealand Ministry for the Environment (2008): *"The flows and water levels required in a water body to provide for the ecological function of the flora and fauna present within that water body and its margins".* By this definition, ecological flows are considered a component of overall environmental flow (New Zealand Ministry for the Environment 2008).

Base Flow:

Base flow which is often confused among the terms 'minimum flow', 'low flow', 'low-water discharge', 'base runoff', and 'fair-weather runoff') has been the focused target of early "instream flow" science. The current science opinion is much more complicated in relation to hydrological regimes as we describe herein. This term is correctly defined by the United States Geological Survey (2005): *"That part of the stream discharge that is sustained primarily from groundwater discharge. It is not attributable to direct runoff from precipitation or melting snow."*

Due to the reviewing nature of the current report, we further note that some terminology related to environmental flows is maintained as it was originally used (i.e. in the source literature) when describing certain assessment methods or concepts (i.e. to not confuse a method to another by changing the terminology). It seems likely, therefore, that some historical baggage is carried forward until the terminology becomes established as the environmental flow science matures as a discipline.

3. REVIEW OF ENVIRONMENTAL FLOW ASSESSMENT METHODOLOGY

Flow management is an ecological imperative, and this is reflected in a vast array of assessment methodologies internationally. Tharme (2003) described over 200 individual environmental flow assessment methodologies and classified these techniques into four general categories; 1) Hydrological, 2) Hydraulic rating, 3) Habitat simulation and 4) Holistic methodologies. In order to avoid further complicating the terminology within environmental flow assessment methodology literature (see above), Tharme's (2003) four-category classification is used throughout this review, and the necessary use of divergent terminology within these general categories for the purposes of review is highlighted where appropriate.

Commonly-used methods within each category are described in detail below, where they are associated with their primary reference (Table 1). To contrast the pros and cons of each of the four methodology categories, their specific attributes (i.e. purpose, scale, scope, duration of assessment, relative cost and use) are listed in Table 2.

Tharme's (2003) four categories differ considerably, based on differing viewpoints regarding how to sustain the biotic integrity of rivers. Specifically, hydrological and hydraulic rating categories assume that a reduction in water availability will also reduce available habitat and/or impair ecosystem function, while the habitat simulation techniques suggest that there is an "optimum" flow where the ecosystem function is sustained (Jowett 1997). The conceptual difference in biotic response to a flow change between these three methodological categories is visualized in Figure 2. Holistic methods suggest that the environmental flow regimes are best designed so that the altered flow regime follows the variability in the natural hydrograph (see details in Section 3.4)

3.1 HYDROLOGICAL METHODS

Hydrological methods are based on analysis of historic (existing or simulated) streamflow data, do not operate at a species-specific level, and provide an overall flow level that aims to conserve the biotic integrity of a stream. This is based on the general assumption that more water provides the best insurance for river biota (to a point), and sustaining some low threshold reduces risk to the biota.

Hydrologically-based methods are still the most widely used approaches internationally (Tharme 2003) most probably because of their ease of use and low cost, i.e., the methods use stream flow data series, real or simulated, and don't require field visits. Hydrological methods are also referred to as "historic flow" or "discharge" methods (Jowett 1997, Caissie and El-Jabi 2003), "look-up table" and "desk-top" analysis (Acreman and Dunbar 2004), "fixed-percentage" methodologies (Tharme 2003) or "office" methods (Wesche and Rechard 1980). Quantitative comparisons of common methods were made by Caissie and El-Jabi (1995) and Caissie et al. (2007). In the next section, we describe some commonly used hydrological methods in more detail.

<u>3.1.1 The Tennant method and its derivatives</u>

The Tennant method (also called the "Montana" method) assumes that some proportion of the average annual flow (AAF; synonymous with mean annual flow - MAF - which is used hereafter) is required to sustain the biological integrity of a river ecosystem. Based on original field data collected from 11 rivers (58 cross sections, 38 different flows) in Montana, Nebraska and Wyoming and further supplemented with additional data from hundreds of gauged flow regimens in 21 states, Tennant (1976) recommended percentage values of MAF predicted to sustain predefined ecosystem attributes (Table 3). Specifically, 10% of the MAF was considered to be the lowest instantaneous flow to sustain short-term survival of aquatic life while > 30% MAF was considered to provide flows where the biological integrity of the river ecosystem as a whole was sustained.

Tennant (1976) recommended the proportionate flow of MAF for "low" and "high" flow periods, or October-March and April-September, respectively, for the region where the method was developed (i.e., North-Central USA). In other regions, temporal matching or seasonal adjustments of low and high flow periods have been assessed, e.g., Orth and Maughan (1981) adjusted the low flow period (10 % of the MAF) to be July-December in Oklahoma, USA.

Variations of the original Tennant thresholds are also used in other jurisdictions, e.g., 25% MAF is regularly used as the minimum flow level required to maintain aquatic life across the Atlantic provinces of Canada (Caissie and El-Jabi 1995).

Another modification of the Tennant method is to use a more frequent time step. Tessman (1980) recommended a monthly time step to determine the flow thresholds, i.e., Mean Monthly Flow (MMF). The Tessman rule recommends minimum flow guidelines as follows:

1) MMF, if MMF < 40 % MAF;

2) 40 % of MAF, if 40 % MAF < MMF < 100% MAF; and,

3) 40 % of MMF, if MMF > MAF.

The Tessman rule has been applied in Manitoba, Canada for use in perennial streams (Anon. 2007). Additional derived methods use a median monthly (or annual) discharge instead of the MMF or MAF, e.g., the Texas Method and Lyons Method (Anon. 2005). Many other locally used modifications have been described, and mostly to better accommodate the variations in hydrologic regime in various geographic areas (Tharme 2003).

Mann (2006) tested the validity of the original Tennant predictions in seven western states of the USA, including the region where the original data was collected. Mann (2006) concluded that the Tennant's original dataset was most applicable in low gradient streams <1%, but not representative of high gradient streams in the west (>1 %).

3.1.2 Flow Duration (Exceedence) curves, and statistical low-flow frequency methods

Flow duration curve methods define the proportion of time a certain flow threshold level is equaled or exceeded in the particular river or region. The duration curve is calculated based on multiple years of data, preferably using records >20 years (Caissie et al. 2007). Flow thresholds can then be extracted from field data (filtered by expert opinion) describing the flow levels that are required to support biotic integrity. Typically, the indices based on flow duration curves are referred to using a Q_x notation, where the subscript *x* indicates the exceedence percentile (or in some cases amount of time in days, see below). For example, Q_{95} refers to a relatively low flow level (i.e. flow that is exceeded 95% of the time) and Q_{50} to a much higher flow level (i.e. flow exceeded 50% of the time).

The Q_{50} method, or median monthly flow method, was developed by the New England U.S. Fish and Wildlife Service for catchments with good hydrological records (USFWS 1981). For smaller ungauged catchments, an Aquatic Base Flow (ABF) was suggested and was described as the median flow in August. The scientific support for selecting the August flow was based on the hydrological regime in the New England region because it typically is the month with lowest flow of the year with high water temperatures. Currently, the New England ABF method is used on a more seasonal basis than the previous August Q_{50}. For example in the state of Maine, the "Seasonal ABF" is determined as the median flow for six different time periods or "seasons";

(1) Winter (January 1 to March 15): a flow equal to the February Q_{50}

(2) Spring (March 16 to May 15): a flow equal to the April Q_{50}

(3) Early summer (May 16 to June 30): a flow equal to the June Q_{50}

(4) Summer (July 1 to September 15): a flow equal to the August Q_{50}

(5) Fall (September 16 to November 15): a flow equal to the October Q_{50}.

(6) Early winter (November 16 to December 31): a flow equal to the December Q_{50}

It has to be noted that the above ABF flows do not describe the environmental flow that needs to be maintained in different rivers in Maine but indicate the level of flow when no further abstraction of water is allowed (so called "hands-off flow"; Maine Dep. 2006). The actual environmental flow is determined using a percentage of natural flow and the allowed withdrawal percentage varies between different stream condition classes (ranging from best condition of grade AA to the most altered grade C class). The Q_{50} method has been applied also in some Atlantic Canadian provinces, with the peculiarity that certain proportion (e.g. 70%) of the Q_{50}

has been proposed as the minimum environmental flow (see Section 4). The scientific basis for the use of some proportion of Q_{50} indices remains unproven (D. Caissie, Fisheries and Oceans Canada, pers. comm.).

Other commonly used exceedence indices include much lower percentiles of the long-term exceedence curve, most commonly the Q_{95} and Q_{90}, which are typically used on a monthly time-step (Caissie et al. 2007). These metrics, like the ABF, are often used to indicate a flow level when all the abstraction should be ceased or to mark a benchmark where different level of abstraction is allowed. In the UK, for example, the Q_{95} has been proposed as a threshold when water withdrawal should be ceased or much reduced, depending on river type (Acreman and Ferguson 2010). The Q_{95} and Q_{90} have been criticised as not providing an adequate level of ecosystem protection for rivers (Caissie and El-Jabi 1995, Annear et al. 2004). For example, salmonid growth rates have been shown to be much reduced at flows exceeding the Q_{95} (Armstrong and Nislow 2012).

It is also worth noting that in some jurisdictions, the Q_x notation is applied to a flow that is exceeded by a certain number of days of the year, for example, the Q_{330}, Q_{355} and Q_{364} indices used in the Czech Republic and Slovakia (see Appendix 2).

Another commonly used hydrological index that is used to determine minimum flow thresholds is the 7Q10 (and the 7Q2 variant), based on a statistical flow-frequency analysis. The 7Q10 is calculated as the lowest flow for seven consecutive days within a 10-year return period (and with a 2-year return period for 7Q2; e.g. Caissie et al. 2007). Although the 7Q10 was originally designed to protect water *quality* under the USA Federal Clean Water Act (Richter et al. 2011) it is errantly used in some places to make inference on environmental flow assessment, *i.e.water quantity,* (Bradford 2008). The 7Q10 is commonly used in Brazil (Tharme 2003) and at one time was widely used in the eastern USA (Richter et al. 2011) while the 7Q2 has been mostly applied in Quebec (Belzile et al. 1997; Caissie et al. 2007). The 7Q2 methods results in somewhat higher flow thresholds than the 7Q10 because of the 2-year rather than 10-year recurrence interval (Caissie et al. 2007). In Quebec's rivers, 7Q2 represents approximately 33% of the MAF (Caissie and El-Jabi 2003).

Although frequently used, the 7Q10 and 7Q2 approaches have been strongly criticized as lacking any scientific support for their use in setting environmental flow standards for fisheries, and could lead to severe degradation of fishery resources (Annear et al. 2004; Caissie et al. 2007). When rigorous statistical testing was carried out comparing a variety of hydrological methods, the 7Q10 and 7Q2 consistently produced the lowest instream flows (Caissie et al. 2007). Therefore, their use for fisheries protection is not appropriate.

3.1.3 Indicators of Hydrologic Alteration (IHA) and the Range of Variability Approach (RVA)

The use of quantitative hydrological variables to support the development of ecologically-sound environmental flow strategies is well accepted in the scientific literature (see review by Poff and Zimmerman 2010). Developed by Richter et al. (1996), the Indicators of Hydrologic Alteration (IHA) represent a subset of 33 ecologically-important hydrological parameters based on variability of the annual flow regime e.g., magnitude and frequency (Olden and Poff 2003).

Calculated from daily flow data using the Nature Conservancy's IHA software (http://conserveonline.org/workspaces/iha), the parameters quantify the magnitude (size), frequency, timing, duration, and rate of change/flashiness of the annual flow regime (Table 4).

The IHA software also calculates an additional 34 parameters for five different types of Environmental Flow Components (EFCs), namely low flows, extreme low flows, high flow pulses, small floods, and large floods. The IHA variables can be used for long-term trend analysis or for comparative statistical analysis to quantify change pre- versus post-activity, e.g. dam construction or water abstraction.

The Range of Variability Approach (RVA) identifies flow targets as ranges for each of the IHA variables (Richter et al. 1997, 1998). The RVA analysis divides each IHA variable under natural flow (or before a change in water use) into three categories (low, middle, and high). Boundaries can be percentile values (non-parametric approach) or the 1-2 standard deviations from the mean (parametric approach). The Nature Conservancy recommends using the non-parametric statistics because an equal number of pre-impact values will fall into each category in most analyses allowing for easier understanding and interpretation. Ideally, RVA is based on 20+ years of daily hydrological data because this amount of data is required to capture the natural variability of a system (e.g. Kennard et al. 2010). In addition to change in IHA variables, a Hydrological Alteration (HA) factor can also be calculated for the three categories for each of the IHA variables (Equation 1). A positive HA value indicates that the frequency of values in the category has increased in the test period and vice versa.

Equation 1: Hydrological Alteration factor as calculated in the IHA software

$$HA = \frac{(Frequency_{Observed} - Frequency_{Expected})}{Frequency_{Expected}}$$

Another diagnostic tool to support environmental flow assessments has been developed in Ontario (Streamflow Analysis and Assessment Software (SAAS); http://people.trentu.ca/rmetcalfe/SAAS.html). The distinct difference with IHA/RVA is the ability to analyse hourly flow records, making SAAS suited for assessing large changes in flow regimes over very short temporal scales (i.e. hourly) similar to those observed downstream of hydropeaking facilities. (R. Metcalfe, Ontario Ministry of Natural Resources, pers. comm.).

3.1.4. Percentage of Flow (POF) methods, Sustainability Boundary Approach (SBA) and Presumptive Standards

Percentage of flow (POF) methods define environmental flows in terms of the proportion of natural flow which can be abstracted instantaneously without compromising ecosystem integrity. POF methods have been increasingly used to define regional environmental flow regimes in lieu of more detailed methods and various proportions of natural flow have been suggested depending on different river classification criteria (see section 4 for examples internationally and in Canada). The POF approach recognizes the importance of natural intra- and inter-annual flow variability and the concept is easy to comprehend and implement. Environmental flow levels that are determined using % MAF methods are tied to historic flow levels and become fixed for the incremental time period (week, month, season). Thus, interannual variability is effectively removed. POF methods, however, use the percentage of the flow that is currently experienced and therefore, the nuances between dry and wet years are incorporated into the flow regime. Such capacity is important if flow regimes change (e.g. due to the global climate change).

Building on the POF concept, Sustainability Boundary Approach (SBA) defines the extent to which changes in the natural hydrograph can occur without impairing the flow-dependent ecosystem benefits (Richter 2010). Thus, the SBA is a framework and not a method to determine environmental flows; the original intent was that the boundaries within which the flow should remain would have to be determined using other environmental flow assessment techniques (but see below for Richter et al. 2011).

The basic approach in SBA is to use the natural hydrograph as a basis and then modify the flow by a certain percentage of allowable augmentation or depletion of the flow (Figure 4A). The percentage of allowable alteration can be variable throughout the year, as determined by applying some combination of environmental flow assessment methods and further dialogue between stakeholders and water managers. In essence, the framework does not prescribe a certain volume of flow that is to be maintained, but an allowable deviation (expressed as a %) from the natural condition is used instead (Richter 2010).

Some of the key benefits of this framework are (Richter 2010):

1) The natural hydrograph is maintained
2) Guidelines are set for both low and high flows
3) Scientific uncertainty is reduced as only the magnitude of flow is altered leaving many other flow characteristics unchanged (e.g. timing, duration, frequency of flows)
4) Easy to implement and comprehend

Soon after the SBA was published (Richter 2010), a new environmental flow framework, the "Ecological Limits of Hydrologic Alteration" was suggested (ELOHA; Poff et al. 2010; see description of the ELOHA framework in section 3.4). While a promising approach, some difficulties have already emerged in application of the ELOHA in some jurisdictions, mostly related to the implementation cost of the ELOHA framework (ranging from $100k to $2M to develop relationships between hydrologic patterns and ecological response; Richter et al. 2011) or time (i.e. need for an instant remedy). Thus, building upon the SBA, Richter et al. (2011) suggested a coarse guideline to provide an interim protection of river flows, termed a Presumptive Standard, that should be used "*until ELOHA or some variation can be applied*".

After a review of case studies in the USA and UK (see Richter et al. 2011), the presumptive standard (Figure 4B) suggests that;

1) a high level of ecological protection is provided when flow alterations are within 10% of the natural flow
2) a moderate level of protection is provided when daily flow alterations are with in 10-20%
3) moderate to major *changes* in riverine ecosystem are to be expected if alterations are > 20% of the natural flow, with an increasing risk for alterations with a higher deviation from the daily natural flows.

The guideline is considered to be conservative and precautionary (Richter et al. 2011). However, authors remind that the standard may be insufficient to protect the riverine ecosystem in hydropeaking facilities where more specific guidelines should be applied. In addition, minimum flow levels when all water abstraction should stop may be required to the above standards during the low-flow periods as indicated by some case studies reviewed in Richter et al. (2011).

As the presumptive standard is a relatively new suggestion, it is not currently known if the standard has been adopted in some jurisdictions.

3.1.5 Strengths, weaknesses and data requirements of hydrological methods

Provided that a record of hydrological record can be obtained for a number of years, the hydrological methods are the simplest, quickest and most inexpensive way to provide information on threshold flow levels, but by themselves they do not produce credible flow regimes that mimic the natural hydrograph. They may be used with other methods, however, as part of a methodological approach to generate reasonably natural hydrographs. Hydrological methods do not necessarily require as much fieldwork as other methods.

Despite the widespread use of various hydrological methods, quantitative studies comparing the different hydrological methods are surprisingly few. However, Caissie and El-Jabi (1995) and Caissie et al. (2007) provide useful comparisons of commonly used methods in a Canadian context. Specifically, Caissie and El-Jabi (1995) compared two exceedence methods (Q_{50} in the form of ABF and Q_{90}), two Tennant derivatives (25% MAF and Tennant's "excellent" scenario; Table 3) and 7Q10 in 70 rivers in Atlantic Canada. They concluded that Q_{90} and 7Q10 methods lead to low flow predictions that could have serious adverse consequences (Caissie and El-Jabi 1995) and arrived at the same conclusion for 7Q2 in a later study (Caissie et al. 2007). The Q_{50} method applied at a monthly time step was recommended to be used in watersheds equipped with a stream gauge and 25% MAF was to be used in ungauged watersheds (Caissie and El-Jabi 1995). In a later study using a jackknife resampling technique, Caissie et al. (2007) re-iterated some of the above conclusions and noted that of the simple hydrological methods, 25% MAF is best suited for regional instream flow recommendations. This was stemming from the observation that the length of the hydrological dataset had a great effect on variability of instream flow estimates, and generally, 25% MAF showed lower variability than the other methods even when the length of the data record was only 10-20 years (Caissie et al. 2007).

In some settings, hydrological methods have been suggested to be used at the planning level or to set up preliminary flow targets in low risk, low controversy situations but are not recommended for studies requiring a high level of detail (Tharme 2003; Acreman and Dunbar 2004). Lately some instream flow specialists recommend against offering preliminary flow recommendations as they often become institutionalized by the regulators and are difficult to recant or replace (Tom Annear, Wyoming Game and Fish Department, pers. comm.; Allan Locke, Alberta Fish and Wildlife Division, pers. comm). Recently, hydrological methods have been used to set the most general level of flow protection in hierarchical or tiered environmental flow frameworks. Current scientific understanding in setting general environmental flow standards emerges to be to use a combination of 1) percent MAF or exceedence curve methods (or other static methods mentioned below) to determine a flow threshold level when ALL abstraction has to stop (i.e. the so-called "hands-off flows") and 2) POF method which is used to determine the actual percent of the current flow that can be abstracted each day (see Section 4 for examples).

Because hydrological methods are easy to use, these methods should always be used to check the suggested environmental flow regimes derived using other assessment (e.g. habitat suitability) methods as an increased safety measure or a benchmark (Caissie and El-Jabi 2003). For example, if habitat suitability methods suggest very low flow levels that are, for example, much lower than 10% MAF or Q_{95} for the river being considered, it would be advisable to re-analyse the habitat simulation data in a context of overall riverine health instead of the theoretical well-being of some selected target species. Hydrological methods may also be used as an input within holistic framework assessments (see section 3.4).

Hydrological methods have been criticized for their lack of ecological validity and high uncertainty with regard to hydrology-ecology relationship (Acreman and Dunbar 2004). If flow-ecology relationships are not known for the type of river under consideration for flow modifications, rendering a rule based on hydrological methods will be "a shot in the dark". Many hydrological methods lead to stable (i.e. flat-lined) environmental flow regime, which is known to lead into degradation over time (Poff et al. 1997; see also the criticism of habitat simulation techniques in Section 3.3.1). If hydrological methods are used for flow recommendations, appropriate validation in the target region must be carried out, and different flows should be assigned at different times of the year (and even between years; see e.g. Alfredsen et al. 2011) to mimic the natural hydrograph, and to better accommodate seasonal biological needs (Bradford 2008).

3.2 HYDRAULIC RATING METHODS

3.2.1 General description and common methods

The hydraulic rating methods (also known as habitat retention or hydraulic geometry methods; Tharme 2003, Moyle et al. 2011) are based on a relationship between some hydraulic measure of a river (usually wetted perimeter or depth) and discharge (e.g. Jowett 1997). Leopold and Maddock (1953) described simple power functions (e.g. wetted width = aQ^b, where Q is the discharge, and a and b are constants) that can be used in describing changes in hydraulic variables as a function of discharge. The constants and the exponents in these equations should be empirically developed for each river or region, as the general form of river channels is variable.

The methods next assume that the hydraulic measure is directly or indirectly related to habitat quantity for a target species, almost exclusively fish (e.g. Bovee 1982, Reiser et al. 1989) or in some instances the ecological function of the river (e.g. Gippel and Stewardson 1998). For example, depth will determine fish presence because of body size and wetted area will affect primary and secondary production. The most commonly used method is the “**wetted perimeter method”** that predicts wetted area of a cross-section as a function of discharge at a location (one point) in the river (Tharme 2003). It has been frequently used in the USA (Reiser et al. 1989) and Canada (Kilgour et al. 2005).

The task, therefore, becomes to establish a relationship between the river discharge and typically, the amount of wetted perimeter and then use this relationship to identify a "break-point"; this is, finding a discharge below which a drastically increasing amount of river bed becomes exposed (Figure 5). In a typical case, the response in hydraulic variable is measured across a single or a number of "representative" cross-sections of the river channel across a range of different discharges (or is simulated using a 1D hydrodynamic model).

One problem in defining the break point from a wetted perimeter - discharge graph is the fact that the break-point in the curve is strongly dependent on the scale of the axis used to graph the data (Figure 5; Gippel and Stewardson 1998; Annear and Conder 1984). For a remedy, they showed that mathematical methods can be used to determine the critical discharge or the breakpoint in the relationship, and suggested two methods (a Slope method and a Curvature method) to define the break-point in the shape of the curve (Gippel and Stewardson 1998).

Shang (2008) further evaluated these two mathematical methods and concluded that the two methods lead to inconsistent values for minimum environmental flows. The recommendation was to not use the Curvature method but instead the Slope method with unity slope or an Ideal Point Method was recommended (Shang 2008). Using these two methods, the minimum environmental flow recommendations resulted in 21% of MAF in a case study river in China (Shang 2008). In Minnesota, O'Shea (1995) found that the flow standards recommended based on the wetted perimeter method corresponded to 39% to 122% of MAF, and in general, the point of inflection decreased (expressed as % MAF) with increasing river size.

As hydraulic rating methods are highly dependent on the channel form, it is sometimes difficult or impossible to find a break point that can be used to establish flow level standards (Jowett 1997). In particular, in uniform channels (consider e.g. a box shaped river bed) very small and shallow flows can result in a very high wetted perimeter, while in reality, most of river may be unsuitable for majority of biota (Jowett 1997) and is prone for other cascading effects like increase in water temperature or harsh ice conditions greatly affecting the suitability of the habitat. In rivers with triangular channel geometry, it is often impossible to find an inflection point altogether (Jowett 1997). For example, Gippel and Stewardson (1998) failed to find an optimum environmental flow using the wetted perimeter method in a case study in Australia. However, they found a relationship between discharge and *flowing* water (i.e. in contrast to just wetted) perimeter which they further recommend to be used instead of the wetted perimeter method (Gippel and Stewardson 1998).

3.2.2. Strengths, weaknesses and data requirements of hydraulic methods

Hydraulic rating methods require some limited field data from the target river in order to establish the relationship between the desired hydraulic feature (e.g. wetted perimeter) and the discharge. The data can be either collected by multiple visits to sample the hydraulic feature across a range of discharges or it is possible to utilize a 1-D hydraulic model (e.g. R-2 cross, Parker et al. 2004, see also Habitat simulation methods). At a minimum, the data need to be collected at one cross-section of the river, but is often commonly measured in a number of transects and reaches. Because of the moderate amount of associated field work, the costs of application of hydraulic methods to set environmental flow standards are intermediate (i.e. higher than hydrological methods, less than habitat simulation or holistic methods).

Generally, hydraulic methods are designed to be used in rivers with well defined single channels. Method is not well suited for braided rivers as inflections point cannot typically be found in such channels (Jowett 1997). In channels where an inflection point can be observed, the hydraulic rating methods result in a single flow recommendation based on the inflection point; seasonal (or monthly/daily) or species-specific adjustments cannot be accommodated using this methodology. On the contrary, these methods never result in a zero flow recommendation, which is theoretically possible for habitat simulation methods. Hydraulic methods can also be easily applied in areas where historic flow records do not exist (but see below).

Similarly to the hydrological methods, the hydraulic methods are recommended in situations with insufficient information of the river systems. The premise is that these methods are coarse and should be used with caution to set a conservative protection limit (Shang 2008). While not implicit, it is advisable to check the environmental flow recommendations derived using hydraulic methods against recommendations using hydrological methods to ensure the consistency and robustness of flow standards. Otherwise, it is possible to get unsustainable recommendations that lead to flows below historical base flows (Gippel and Stewardson 1998).

The main critique of the technique is similar to the habitat simulation methodology (see below). The assumption that the wetted perimeter *per se* corresponds to biological requirements of a species is overly simplistic and perhaps flawed. Also, these methods are estimating a proxy for the amount of physical habitat for riverine biota, and therefore, links to population abundance cannot necessarily be drawn (habitat quantity and population abundance are two fundamentally separate issues, as will be discussed below; see also Conder and Annear 1987). Another criticism is related to the selection of the transect(s) where the hydraulic variable (e.g. wetted width) is measured. Often the selected transects are subjectively determined and there is no guarantee that the measured transects indeed are representative for the whole river or reach (see e.g. Fonstad and Marcus 2010).

Hydraulic rating methods are generally considered as a precursor to the more detailed habitat simulation methods (Section 3.3), and the popularity of hydraulic methods has decreased as a result. However, Booker and Acreman (2007) suggest that the generalized measurements of channel form and river hydraulics are a viable trade-off for the habitat simulation methods when a basis for further flow evaluation is needed. The information derived from hydraulic methods can also easily be used as input tools for holistic methods (see below; Tharme 2003) as is the case for example in Alfredsen et al. (2011) in a Norwegian Building Block Methodology case study.

3.3 HABITAT SIMULATION METHODS

3.3.1 Habitat simulation methods at the microhabitat scale

Different from the categories mentioned above, the habitat simulation methods aim to conserve specific and pre-selected target species for which the habitat requirements can be reasonably estimated in the case study area or are believed to be known from previous studies elsewhere. As mentioned above, the theory is based on the belief that there is an underlying relationship between the level of flow and "optimum" physical habitat conditions for the target species (Figure 2). By using simulations of the discharge conditions, the method, in its typical and simplest form, aims to find this optimum and set a target flow (a typical recommendation includes a static minimum flow level) such that the amount of physical habitat for the selected group of target species does not decline beyond a subjectively determined conservation level (see e.g. Figure 6, but also see other approaches for determining flow thresholds based on habitat time series below).

The habitat simulation methods have become extremely popular and even a legal requirement in many jurisdictions in North America and globally (Tharme 2003). The popularity stems from the establishment of the Instream Flow Incremental Methodology (IFIM) framework that was developed for assessing the effects of flow manipulation on river habitats (Bovee 1982; Bovee et al. 1998; Stalnaker 1995). IFIM is a holistic decision-making tool that includes, among other steps, quantifying of the incremental differences in physical habitat that result from alternative flow regimes (Table 5; Stalnaker 1995; Hudson et al 2003). Indeed, it is the physical habitat tool (Physical Habitat Simulation, PHABSIM; Milhous et al. 1989) that has been most widely used when reference to incremental methodology is made and often without the rest of the IFIM framework tools (Hudson et al. 2003; Acreman and Dunbar 2004). This has lead to the fact that the IFIM process is often confused with PHABSIM (Hudson et al. 2003). Technically, an IFIM study does not need to employ the PHABSIM component if it is recognized that the habitat conditions are not the limiting factor but, for example, the issues in a particular river are related to water quality issues.

However, as it is commonly the PHABSIM-type component (and not full IFIM) that is meant when habitat simulation tools are discussed, the description provided below refers only to the physical habitat simulation component of the IFIM framework.

3.3.1.1 Generic approach

Habitat simulation methods consist of two integral parts that are linked together: 1) physical, or hydraulic modeling providing information of changes in the physical habitat as function of discharge and 2) modeling of the biological associations with their physical environment (assumed to be fixed across a range of discharges).

While a suite of different nuances in habitat simulation methods can be identified (details below), the general approach to evaluate effects of flow on habitat quantity between different habitat methods is the same.

The habitat simulation methods have a few general assumptions (e.g. Armstrong 2010):

1. Local density of individuals (with adjustments for habitat availability) reflects local habitat quality (i.e. "preference");
2. Preference of physical habitat attributes (for each species) is constant across discharges;
3. Individuals are free (and/or willing) to move in response to change in physical habitat with discharge.

Basic steps (Figure 6) of the habitat simulation methods are described as (Hudson et al. 2003):

1. "Representative" or “critical” study sites are selected
2. Hydro-geomorphology of the study site(s) are surveyed and hydraulic models (see below) are calibrated so that changes in depth and velocity can be simulated at different streamflows (substrate size distribution is considered to remain the same across all simulated discharges)
3. Species-habitat association models (a.k.a. abundance-environment relationships, habitat preference or habitat suitability index) are selected from existing literature or are developed within the study system to represent how ‘suitability’ of a particular stream location for a species and/or life stage is related to physical habitat variables (most often depth, velocity, and substrate).
4. The hydraulic model is combined with either suitability or preference curve information to simulate how "Weighted Usable Area", WUA (an index of habitat quality-quantity), varies with streamflow.
5. WUA-streamflow relationships for individual species and life stages are calculated. These relations can then be used to develop a flow recommendation.

A large number of habitat models and techniques have been described, but all operate within the ideological template as described above, and they largely share the same assumptions, benefits and pitfalls. While the conceptual basis of the different hydraulic-habitat models is the same, there are differences between the habitat simulation models (both in the hydraulic and biological models) in the detailed calculations or derivation methods of various indices used in the process (e.g. Ahmadi-Nedushan et al. 2006; Dunbar et al. 2011). Some commonly used hydraulic-habitat models are listed in Table 3. A recent in-depth review of habitat-simulation methods used for setting environmental river flow is provided in Dunbar et al. (2011).

3.3.1.2 Biological component

The purpose of the biological component of habitat simulation models is to describe the physical habitat conditions where individuals are found during different life stages and/or seasons. The traditional habitat simulation methods assume that the spatial distribution of the individual in the river is determined by the measured physical variables independently of other factors (it is assumed that the "other" factors are at play at the time of collecting the biological field data and thus are taken into account). A large variety of statistical methods has been used to analyse the species-environment relationships and include both univariate and multivariate functions (e.g. Ahmadi-Nedushan et al. 2006; Dunbar et al. 2011).

Traditionally, the species-environment relationships are represented in a form of a Habitat Suitability Index (HSI; e.g. Heggenes 1990). HSI is composed as a proportionate ratio between the *habitat use* of the individuals (i.e. data on where the individuals are found) for the target species and the *availability* (general composition of physical variables) of the habitat within the studied site(s) and are typically scaled from 0 (unsuitable habitat) to1 (preferred habitat). For most applications, HSI are composed separately for velocity, depth and substrate, but many other additional variables have also been used (Dunbar et al. 2011).

The data collection for habitat use data is often a time consuming process and the methodology to observe the habitat use of individuals varies between species and type of habitat under study. Common methods for fish include electrofishing (Mäki-Petäys et al. 2002), direct observations from the river banks (e.g. Heggenes et al. 1991) or using snorkeling (Keenleyside 1962) or SCUBA diving (Linnansaari et al. 2010), and various fish telemetry methods, including use of acoustic or radio transmitters (Scruton et al. 2002) and Passive Integrated Transponder methods (Linnansaari et al. 2009).

Habitat availability data needed to calculate the habitat suitability indices within the studied sites are typically collected simply using cross sectional transects, and point measurements of velocity, depth and substrate (or some other additional variables) are taken. These point data are sometimes used as such to calculate proportions of available habitat in the study reach (e.g. Mäki-Petäys et al. 2002) but ArcGIS and other spatial interpolation programs (e.g. Surfer) can be used to transfer spatially explicit point data into interpolated surface data for the study reaches (e.g. Muotka et al. 1999; Linnansaari et al. 2010). It is inevitable that the estimation of habitat availability in a study reach introduces an error as both using proportionate point data or upscaling from point data to surface will be associated with some interpolation error. Sheehan and Welsh (2009) report that by sampling of 5% of the total area of the study site, the achieved accuracies of the availability are in the range of 57-95% depending on the interpolation method (corresponding numbers for 2.5% sampling were 49-92% with the best results observed using natural neighbor interpolation method for both 2.5% and 5% datasets). It is also possible to obtain the habitat availability data using a hydraulic model (see below), but because the availability data (for calculating a HSI) is only required for the discharge conditions when the habitat use data is collected, the static methods described above are more commonly used. Other methods also exist, and may include collection of habitat use data and then convene an expert panel where professional judgment is added to develop the suitability indices, like is a common practice in Alberta (Allan Locke, Alberta Fish and Wildlife Division, pers. comm.).

In addition to the traditional univariate HSIs, various multivariate methods have been used that are based on different generalized linear models including logistic regression methods (Guay et al. 2000) or generalised additive models (Ahmadi-Nedushan et al. 2006 and references therein). Another recent development in biological modelling within habitat simulation method family is the utilization of fuzzy logic (e.g. Ahmadi-Nedushan et al. 2008) and Demonstration Flow Assessment techniques (Railsback and Kadvany 2008). Instead of collecting *in-situ* based observational data on species habitat use, the species-environment relationships are determined based on expert opinion (Figure 6B).

Ahmadi-Nedushan et al. (2006) provides a comparison of the advantages and disadvantages of the different statistical methods used for estimating the species-environment relationships. In general, the species-environment relationships have often been considered to be a weak link in the habitat simulation methods (Moyle et al. 2011; see criticism of habitat simulation methods below).

3.3.1.3 Hydraulic component

The purpose of the hydraulic component in the habitat simulation methods is to provide the information how the physical environment is changing as a function of discharge. Hydraulic modeling can range from relatively simple solutions (1D models) to very complex multi-dimensional models (2D and 3D solutions; Dunbar et al. 2011). Common to all models is that the wetted perimeter of the river is divided into compartments, or cells, within which each of the physical variables is given a simulated value for each discharge (Figure 6A). The process involves a geo-referenced survey of the bed topography including bed roughness, detailed measurements of water velocities and depths on cross-sectional transects (or anywhere within the study site for 2D and 3D models) and measurement of wetted widths for at least two discharge levels for model calibration purposes. Model validation datasets are (or should be) also collected during the field survey (velocities, depths, wetted widths; e.g. Dunbar et al. 2011)

Traditionally, the 1D models were most commonly used due to lack of computational capacity needed for higher dimension models, however, this is no longer a limitation. In general, multidimensional models are used to model more complex river reaches (e.g. high-gradient or braided river reaches) because of their denser and more dynamic cell-structure. Some 2D models have the capacity to also consider ice covered channels (e.g. River2D; Blackburn and Steffler 2002). However, the multidimensional models require more extensive field data collection and thus, can be more expensive to apply (see below).

Recently, Jowett and Duncan (2011) compared the practicality and accuracy of 1D and 2D modeling approaches in New Zealand. They concluded that the difficulty in acquiring sufficient and accurate bed topography and the skill required in calibrating 2D models is a practical limitation to their utility, and it cannot be assumed that they are better simply because they require more data; the time and effort required to develop a good 2D model is not warranted in many situations (Jowett and Duncan 2011). Furthermore, the main advantage of 2D models over 1D models is that they should provide more accurate predictions outside the calibration range especially at high flows in braided rivers, but improved calibration and validation techniques are required (Jowett and Duncan, 2011).

An important step when using any hydraulic model is that after the model is calibrated to simulate the physical conditions in a study reach, the model predictions must be validated using a separate (independent) dataset. Such validation provides the means to understand the accuracy and associated error of the modeling output. Surprisingly, the validation is not always carried out even in large scale applications and furthermore, the associated error is almost never incorporated into final habitat-discharge relationships.

3.3.1.4 Combining hydraulic and biological components

Once the hydraulic model has been calibrated and the species-environment relationships have been established, the two separate components need to be combined into composite flow-habitat relationship (sometimes referred to as a habitat-discharge rating curve). In a case that univariate HSIs are used, the information regarding all the suitability values of each parameter need to be combined into one composite number in each modeled cell of the hydraulic model (e.g. Vadas and Orth 2001). Various techniques for establishing the composite value have been used, and include direct multiplication, arithmetic and geometric mean and lowest SI value (reviewed in Ahmadi-Nedushan et al. 2006). In addition, individual SI values may be weighted if some variables are deemed more important than others (e.g. Guay et al. 2000). When each cell is assigned a composite suitability value, a Weighted Usable Area (WUA) can be calculated by multiplying the composite suitability value by the area of each cell of the hydraulic model and by summing the values together for the given discharge. Alternatively, a composite suitability value within a certain range is assigned a category, typically referred to as preferred, indifferent or neutral, and avoided (or unsuitable) condition (e.g. Linnansaari et al. 2010) and the area of cells belonging to each category is summed separately (e.g. Forseth et al. 2009). Finally, the summation is carried over repeatedly across the range of discharge conditions to produce the habitat-discharge rating curves (Figure 6).

Variability in the combined modelling output stems from both the error to accurately reflect the species-environment relationship, and from the error associated with accurately predicting the true value of each physical variable in each cell in the hydrological models. Typically, however, the error is not reflected in the habitat-discharge curves and the modelling output is envisioned as "absolute".

The habitat-discharge rating curves can be used directly to identify environmental flow thresholds, but they can also be used to translate a flow series of a river (water per unit time) into a habitat time series (suitable area per unit time; Bovee et al. 1982; 1998). Habitat time series for different flow alternatives can then be compared to natural ("no flow change") habitat time series and therefore, the consequences of each alternative flow scenario can be analysed as proportional habitat change compared to the natural flow. Such approach is fundamentally different from setting target flows based on habitat-rating curves; in many aspects, the approach is similar to hydrological methods (described in section 3.1) but units have changed from flow to habitat. The habitat time series analysis has been used in large environmental flow projects in Canada (e.g. in the South Saskatchewan River Basin and Athabasca River in Alberta; Clipperton et al. 2003; Paul and Locke 2009, respectively; see also below).

3.3.1.5 A Canadian Example

A recent Canadian example of an application of habitat simulation method in a high-profile case study is the Lower Athabasca River (LAR) Phase 2 Water Management Framework which was developed to prescribe when, and how much, water can be withdrawn from the LAR for cumulative oil sands mining water use (Ohlson et al. 2010). The determination of the environmental flow regime (a combination of both fixed-flow and percent-of-flow reduction factors and an Ecological Base Flow as referred to in the project descriptions) was based on seven evaluation criteria (Ohlson et al. 2010), one of which was the consideration fish habitat (Paul and Locke 2009).

The LAR case study goes to show the complexity and extensive effort that is required when habitat simulation methods are used as a tool to provide information to be used as a part of environmental flow recommendation. In the LAR project, habitat simulations were carried out for six fish species (considered to be characteristic, of traditional importance or abundant in the river), each including four life stages during two different seasons resulting in a theoretical 48 different considerations for habitat requirements, of which 12 combinations were deemed not to be applicable (e.g. "burbot / spawning / summer" is not applicable as burbot spawn in winter). Nevertheless, habitat information (as are required for habitat simulation) was needed for 36 considerations, but could be determined for 24 combinations (Paul and Locke 2009). A prominent problem in the LAR project was the general lack of information regarding species-environment relationships for the juvenile life stages during winter (Paul and Locke 2009), which also is a period when the relatively low flow conditions naturally occur, and it is reasonable to assume that the impacts of water withdrawal on fisheries during this period may be the largest. Studies by the Wyoming Game and Fish Department that showed losses of juvenile trout up to 90% during the winter in tailwater habitats of dams support this tendency (Zafft et al. 1995).

River2D hydraulic simulations were carried out in four river reaches, and as per the preamble of the habitat simulation approach, the information based on these reaches were considered to be representative for the whole LAR. The WUA information was calculated for the different species-life stage-season combinations and used to develop habitat time series for natural flows and any proposed water management alternative. Metrics for acute and chronic habitat loss (from natural) were calculated for each combination. Because of the large number of outputs, the calculated habitat loss for the most flow sensitive life-stages in the matrix were considered to represent biologically-relevant habitat changes within each river segment for each season (Franzin 2009). These habitat-loss metrics were further used, alongside of the other six evaluation criteria, to evaluate the effects of various water management alternatives (Ohlson et al. 2010).

3.3.2 Strengths, weaknesses and data requirements of habitat simulation methods

In many jurisdictions, habitat simulations are considered more accurate than hydrological and hydraulic methods to determine flow thresholds levels, and habitat simulation is recommended in high-risk projects (Hatfield et al. 2003). Habitat simulation methods and today’s highly developed computer platforms easily produce WUA-discharge curves and make the interpretation of the results straight-forward. It is also possible, at least in theory, that habitat simulations can be used to design modified flows with the goal to improve habitat conditions for a target species because the flows are "tailored" to accommodate a particular objective (e.g., Hvidsten 1993; Jowett et al. 1995).

Generally, traditional habitat simulation methods require a considerable amount of field work and expertise to collect the hydraulic and biological data. They can be time consuming and expensive projects. There is no shortage of debate surrounding criticisms of the method, e.g., Lancaster and Downes (2010a, 2010b), Lamouroux et al. (2010), Hudson et al. (2003), Hatfield et al. (2003), Moyle et al. (2011), Dunbar et al. (2011) and Armstrong (2010). The criticisms were synthesized by Moyle et al (2011):

"Habitat association models such as PHABSIM that infer habitat quality from AERs [Abundance-Environment Relationships] *are based on outdated concepts and unsupported assumptions, do not deal with the processes that actually control populations, and have dubious utility for estimating the future abundance of biomass of the target organisms*

(Anderson et al. 2006, Armstrong 2010, Lancaster and Downes 2010a, 2010b), especially in response to changes in flow. Published tests claiming to show strong relationships between populations or biomass and the habitat index estimated by PHABSIM are mostly flawed, and better tests show weak or no relationships" (Moyle et al. 2011, p. 79).

It is very important to recognize that the habitat simulation methods estimate only the amount of physical habitat as a function of discharge. This "optimal" or "suitable" physical habitat is assumed to be linked to relative abundance or biomass of a species, but such an assumption is unproven (e.g. Bradford et al. 2011). Regardless, the knowing of the physical habitat still is a necessary step in a process to understand the potential for perpetuation of a population of aquatic animals (Milhous 1999).

There is considerable risk associated in implementing decisions based solely or primarily on studies of microhabitat (Hatfield et al. 2003) and thus, habitat simulations can't be used alone to recommend environmental flow standards in rivers (e.g. Ohlson et al. 2010).

Finally, for habitat simulation method to be useful, it is important that confidence intervals are reported in the flow-habitat rating curves to address the inherent uncertainty that accumulates from the biological and the hydraulic models (Castleberry et al. 1996).

3.3.3 Generalized (statistical) habitat models

Because applications of conventional hydraulic-habitat simulation models require considerable field effort and experience, generalized (a.k.a. statistical) hydraulic-habitat models have been proposed as an alternative (Lamouroux and Jowett 2005). The notable difference between the traditional habitat models and generalized habitat models is that the latter are not based on hydraulic model, but the change in hydraulic variables with varying discharge are computed statistically and as an average distribution at a study reach scale (Lamouroux et al. 1995, Lamouroux 1998). The biological data input is similar to traditional habitat models (i.e. in the form of a Habitat Suitability Index) and the data can be either collected within the studied river or can be obtained from literature. The use of statistical habitat models require only little experience and field effort (some measured points of velocity, depth and substrate and wetted width are required at minimum on two different flows to establish the statistical relationships and a "mixing parameter"; Lamouroux et al. 1995) and the model output is in a format of WUA curves that can be interpreted similarly as the WUA curves derived from traditional habitat simulations. However, as the distributions of habitat variables are calculated as average reach conditions, the models are spatially non-explicit. The generalized models have been shown to perform well outside their calibration range (Lamouroux and Jowett 2005).

Statistical habitat models (e.g. STATHAB; http://www.irstea.fr/stathab; ESTIMHAB http://www.irstea.fr/estimhab) have been widely used in France where they were developed (e.g. Lamouroux and Capra 2002), but also in New Zealand (Lamouroux and Jowett 2005), Ecuador (Girard 2009), Norway (Forseth et al. 2009) and Sweden (Harby and Sundt 2007). Recently, generalized habitat models have been used in New Zealand to develop regionalized environmental flow regimes (Snelder et al. 2011). The method links regionalized flow duration curves, at-station hydraulic geometry, and generalized habitat models to make assessments at regional scale in New Zealand. The method was also used to assess the hydrological methods that are used in New Zealand to determine minimum environmental flows and allocation limits. Snelder et al. (2011) concluded, not surprisingly, that in a spatially variable environment, uniform application of a hydrological rule to define minimum flows will have spatially varying consequences for environmental protection and reliability of supply for abstractors and recommend the use of the method based on generalized habitat models to define regionally varying rules for minimum flows.

Generalized habitat models share essentially all the fundamental and theoretical uncertainties and deficiencies for which other habitat modeling approaches have been criticized (except, of course, the uncertainty related to hydraulic modeling which is replaced by uncertainty in statistical procedures to estimate the same variables but at the reach scale). Additional questions rise from the non-spatially explicit nature of these models. As the statistical habitat models are very easy to use, it is relatively easy to extrapolate the calculations into range where no field data were collected for either biological preferences or physical variables. Naturally, the quality of such calculations needs to be validated.

It is possible that statistical habitat models yield the same level of confidence in estimating the amount of physical habitat as a function of changing discharge for a target species than the more traditional habitat simulation methods, but with much less effort. However, more rigorous testing in wider geographic area is recommended before these methods are substituted for more traditional spatially explicit habitat models.

3.3.4. Mesohabitat models

3.3.4.1 General description

Similarly to the generalized habitat models, mesohabitat models have been developed in the past 15 years in response to some of the criticism towards the habitat simulation methodology at the microhabitat level.

The basic idea is that instead of associating an individual of a species into a small microhabitat unit wherein physical parameters are described in detail, the mesohabitat models try to reveal patterns of species association at a larger, mesohabitat, or hydromorphologic unit level. In other words, the models make a general case that mesohabitat types with some associated cover structures (for example, "riffle", or "pool") are related to species presence/absence or abundance. As the quantity and the spatial distribution of the mesohabitat classes change as a function of discharge experienced, the changes will cascade (given a discharge change that is permanent in comparison to the natural condition) into fluctuations in target species populations and this information can then be used to make inference on habitat availability and further, environmental flow standards.

A few modeling platforms at the mesohabitat scale have been described in primary literature including MesoHABSIM (Parasiewicz 2001), MesoCASiMiR (Eisner et al. 2005), the Norwegian mesohabitat classification approach (Borsanyi et al. 2004) and Rapid Habitat Mapping (Maddock et al. 2001). The different methods vary in their field effort-intensity (Eisner et al. 2005). The MesoHABSIM platform seems to be the most commonly applied mesohabitat approach, and it has been applied in a number of rivers, mostly in the New England region of the US, and a few case studies are described in http://www.mesohabsim.org/articles/articles.html. Conceptually, the different mesohabitat models are similar in their basic approach, and the basic steps can be summarized:

1) Mesohabitats (quantity and spatial distribution) are mapped across a range of discharge conditions (minimum of 3, typically 4 or 5; e.g. Parasiewicz 2001) along the length of the stream or the area of management interest (Figure 7). Depending on the modelling platform, some field measurements are also collected to ensure the classification of mesohobitat type.

2) Mesohabitat-level biological associations are quantified. These data are collected using standard methods (e.g. electrofishing, snorkeling etc.) depending on the size of the river. In some cases, existing datasets can be used. Binary models (presence-absence and presence–abundance) are then used to distinguish between unsuitable vs suitable mesohabitats and suitable vs optimal mesohabitats, respectively, for different aquatic species or assemblages (e.g. Vezza et al. 2011).

3) Habitat–flow rating curve is constructed and reference minimum discharge can be determined (Figure 8). Unlike the microhabitat models that typically are associated with a hydrodynamic model making calculations of availability of physical habitat at any discharge, the mesohabitat models resort to fitted curves (e.g. non-linear regression) to extrapolate between discharges.

The schemes how to classify the different hydromorphologic units are variable (see Table 6 for just two examples), but can easily be standardized for use in a regional assessment. Eisner et al. (2005) showed how mesoscale habitat models can be prone to subjectivity in identifying mesohabitats (i.e. two observers could map the same mesohabitat and produce a different description), however, the mesohabitat classification can and should be based on established criteria that can be easily measured if necessary during the mesohabitat assessment or survey, to reduce any subjectivity that may be otherwise introduced when determining the habitat composition (see e.g. Borsanyi et al. 2004; Table 6).

3.3.4.2 Comparison with habitat simulation methods at the microhabitat scale

Modelling at the mesohabitat scale has some apparent benefits in comparison to microhabitat modeling (Parasiewicz 2001). Microhabitat models are often criticized because of the fact that the individual associations to their physical habitat are measured at a such detailed scale that the models become sampling time dependent and thus, prone for error and often biologically unrealistic (e.g. " for a species x, areas with depth of 10-20 cm are preferable over areas with depth 20-40 cm"). In mesohabitat approach, the associations are considered at a larger scale that is more robust across discharge variations (e.g. "for species x, riffle type habitat is preferable over pool type habitat"). Generally, it is believed that an individual would be found at the same mesohabitat unit for example throughout the diurnal cycle, whereas the microhabitat model could be sensitive to even such small-scale temporal sampling biases (Paraziewicz 2001).

Mesohabitat models are also much less resource-intensive in comparison to microhabitat models, at least in terms of physical habitat measurements. No cross-sectional velocity-depth-substrate (v-d-s) data are collected reducing the field effort significantly. Some mesohabitat modeling platforms collect a limited amount of detailed data, often in a binary format or using actual random measurements (e.g. MesoHABSIM; 7 - 30 measurements of v-d-s in each HMU; scale dependent). The most time consuming aspect in mesohabitat modeling is the collection of biological data in each of the mesohabitat types (e.g. electrofishing for fish, snorkeling for mussels/fish; or standard benthic sampling methods for macroinvertebrates). According to Parasiewicz (2007), sampling of 500 mesohabitat units equals to 25 days of fieldwork using a three people crew.

Stemming from the reduction in the detail of the physical habitat data collection within a "study site", the greatest advantage of the mesohabitat approaches is their ability to quickly collect information about physical conditions from long river sections (Parasiewicz 2007). Because the effort can be invested to examine the river at a larger spatial scale, typical mesohabitat applications range up to tens of kilometers within a studied river (Dunbar et al. 2011). Even in projects where environmental flows are planned at a regional scale, mesohabitat methods have been used to survey 5% to 10% of the stream length in one day (Vezza et al. 2011). Sampling at a whole river length removes the problem of selecting a "representative" study reach that is an inherent problem in microhabitat scale. Even in cases where a section of a river is sampled, the sampled sections may be used to represent a large proportion of each individual river (and at any rate, larger section than if a microhabitat modeling approach was used) making it more likely that the sampled section is representative of the non-sampled parts of the river. Thus, upscaling of the physical habitat is better justified and more likely to address issues relevant at management levels.

Mesohabitat models have been criticized on the grounds that they do not rely on hydrological modeling to make predictions of habitat availability at the discharges where direct measurements are not carried out, but instead, the interpolation is accomplished by curve fitting (Dunbar et al. 2011). While this makes the data gathering more laborious (i.e. data cannot be simulated but it has to be directly collected), it also means that the extrapolation is only carried out for discharges where the habitat association data is valid (Vezza et al. 2011). Furthermore, mesohabitat models may perform better in environments where hydrological modeling is difficult (e.g. small streams with high gradient, boulder rich reaches characterized by a high degree of flow complexity; Vezza et al. 2011).

In general, however, mesohabitat models share the same conceptual deficiencies than microhabitat models. Similarly to the microhabitat models, the mesohabitat models are intended as a method to estimate the amount of physical habitat available as a function of flow. Mesohabitat models, thus, share the same incapability to take biological interactions into account or predict changes other than the amount of physical habitat as a function of discharge. These models can be used, for example, to evaluate if flow thresholds exist where dramatic changes in the amount of physical habitat for a target species would likely occur. However, this does NOT mean that such identified thresholds could be used to establish a static flow regime for a river and assume this will maintain existing or desired biotic integrity in the river; this is not the case. Therefore, mesohabitat models can be used to determine threshold flows under or above which no water abstraction or augmentation should be allowed; however, the method is not designed to recommend environmental flow regimes to preserve biological integrity. Like micro-habitat modelling approach, environmental flow regimes based on the meso-habitat approach don't address the inter-annual flow variability needs.

For such purpose, some comparisons between mesohabitat methods and other methodology is available. One case study compared the output from two microhabitat models to a mesohabitat model (Parasiewicz and Walker 2007). While only the mesohabitat model correlated with fish abundance, and the three modelling methods suggested different amounts of habitat available, the break point in the habitat-discharge curve at the segment scale would have provided similar recommendation to the management (note that this is not what Parasiewicz and Walker 2007 conclude from this dataset). Recently, the MesoHABSIM approach has been used to set environmental flows at a regional scale in Italy (Vezza et al. 2011). When compared to Q_{95} hydrological method, the MesoHABSIM approach resulted in average environmental flows of 1.42 times Q_{95} (Vezza et al. 2011). Output from this study provides also a possibility of subjective comparison of the mesohabitat approach and hydraulic method (wetted perimeter) approach; while calculations are not provided, the general conclusion for minimum flows would have been, in this sample case, somewhat similar (Figure 8). However, the author informs that the result is largely co-incidental as in general the wetted perimeter in the high-gradient reaches where the study was carried out (P. Vezza, Dept. of Land, Environment and Geo-engineering, Politecnico di Torino, Italy, pers. comm.)

3.3.5 Bioenergetic models

Since many deficiencies have been identified in the use of habitat simulation methods, alternatives to habitat-simulation models have been developed. These models have been recently reviewed in the Canadian context by de Kerckhove et al. (2008).

Models based on bioenergetics have been used increasingly to predict spatial distribution of fish in streams. The bioenergetic models are based on the estimation of the Net Energy Intake as cost-benefit ratio of different stream locations (i.e focal points) and require the estimation of gross energy intake (from nutrition) and the energy spent maintaining the foraging position (e.g. Urabe et al. 2010). Conceptually, the models based on bioenergetics are more realistic (i.e. take into account individual fitness) than the models based on physical habitat simulations. Bioenergetic models have been shown to be able to predict profitable stream locations (Fausch 1984; Hughes and Dill 1990; Guensch et al. 2001; Hayes et al. 2007), and relative fish abundance (Hayes et al. 2007; Jenkins and Keeley 2010). In comparison to habitat simulation methods, the bioenergetic models have been shown to better predict fish abundance in streams (Urabe et al. 2010).

However, bioenergetic models also make a number of dubious or even falsifiable assumptions, (see Urabe et al. 2010 for a discussion). The fact that bioenergetic models require estimates of drifting prey adds another level of complexity when comprising the model output and also makes the models expensive and time consuming to use. In addition, bioenergetic models have only been shown to be plausible for drift-feeding fish species. Application of bioenergetic models to make predictions of profitable stream locations may be especially difficult in winter as the foraging behaviour of many stream fish changes seasonally (e.g. Cunjak 1996; Annear et al. 2002; Linnansaari 2009).

3.4 HOLISTIC METHODS AND OTHER ANALYTICAL FRAMEWORKS FOR DEVELOPING ENVIRONMENTAL FLOW STANDARDS

Holistic environmental flow frameworks have been vigorously developed during the past two decades (Tharme 2003). Holistic methods are a group of methods, or rather, environmental flow frameworks which are based on the need to maintain some resemblance to the natural hydrological regime in order to sustain healthy river and riparian ecosystems. Holistic methods aim to merge human and ecosystem flow requirements into a seamless assessment framework (Arthington 1998). Holistic frameworks integrate social, cultural and economic values within ecosystem protection goals. Holistic methods are sometimes referred to as expert panel approaches, where environmental flow standards are developed in a workshop setting where river-specific data is considered by a multi-disciplinary team of experts (typical areas including hydrology, geomorphology, water quality and various disciplines of ecology) and importantly, other stakeholders as the basis for consensus recommendations (Arthington 1998).

Holistic methods can be categorized into two main approaches based on either a bottom-up or top-down strategy to describe environmental flow regime (Tharme 2003). The bottom-up procedures are based on the supposition that it is possible to prescribe the critical components of flow regime that needs to remain in the river (Arthington 1998). In comparison, top-down methods assume that the entire natural flow regime is ecologically important but some flow components can be modified or removed without ecological risk (Arthington 1998).

Whether bottom-up or top-down, all holistic approaches share some common properties regarding achievement, or maintenance, of ecological sustainability (Gippel 2005):

1) some components of the natural flow regime cannot be scaled down, and must be retained in their entirety
2) other components of the natural flow regime can be scaled down
3) other components of the natural flow regime can be omitted altogether
4) the variability of the regulated flow regime should mimic that of the natural flow regime

Many holistic frameworks have been described; four commonly used or emerging frameworks are reviewed herein in some detail. Arthington (1998) and Tharme (2003) provide thorough reviews of various holistic methods.

3.4.1 Building Block Methodology (BBM)

BBM was developed in South-Africa in the early 1990s to produce rapid advice on the environmental flow standards using limited amounts of data (Arthington 1998). BBM is based on a prescriptive bottom-up approach, designed to construct a flow regime for maintaining a river in a predetermined condition (Tharme and King 1998). To obtain the predetermined condition, the following assumptions are made (Tharme and King 1998):

1. The river biota can cope with frequent, naturally-occurring low flow conditions, and may be reliant on higher flow conditions that naturally occur at certain times (i.e. specific floods).
2. Identification of the most important components, or "building blocks", of the natural low flows and floods, and combining them as the modified flow regime, will facilitate maintenance of the river’s natural biota and processes.
3. Certain flows influence channel geomorphology more than others, and incorporating such flows into the modified flow regime will aid maintenance of natural channel structure, and diversity of physical biotopes.

There are three main parts to BBM that take place in a sequence, each of which are described in great detail in a comprehensive BBM manual (Tharme and King 1998; King et al. 2008):

1) *A comprehensive information gathering / preparatory phase*

   A structured set of activities is followed to collect and display the best available information on the river for consideration by the workshop participants. The collected information includes social use of riverine resources, flow regime evaluations (historic and present), hydraulic analysis, geomorphology, water chemistry, groundwater and biological surveys for vegetation, aquatic invertebrates and fish. The BBM manual includes detailed instructions on how the data is collected for each criteria. The information is collected in a "Starter Document" that is provided to the participants of BBM workshop (see below).

2) *BBM Workshop*

   The BBM workshop typically involves ~20 people comprising of water managers, engineers and river scientists. The workshop consists of four main sessions and typically takes 2 - 4 days to complete. The first session is a visit to the field sites that are being considered followed by another session where all the gathered information is presented. In the third session, the actual modified environmental flow regime is designed based on monthly flows and special purpose flows and reported as %MAF (Figure 9). Finally, further research needs are identified to address major uncertainty and to improve the environmental flow regime. A technical report is produced after the workshop that outlines the environmental flow regime and describes the reasoning for the different flow components.

3) *Follow-up activities linking the workshop with the engineering and planning concerns*

Following the workshop, the flow regime described in the workshop is incorporated in a hydrological yield analysis. This reveals whether or not the EFR can be met without conflict with potential consumptive users. If conflicts are identified, adjustments are made until a compromise is achieved.

The BBM method does not examine alternative flow scenarios as it is designed to build one consensus-based flow regime that supposedly results in a predefined river condition based on best available scientific data (Tharme 2003). Of course, the weakness of this approach is the assumption that the experts have a comprehensive knowledge of what constitutes a critical flow event within the river in question.

While the applications of comprehensive BBM framework can be resource intensive and time consuming (1-2 years; Tharme 2003), the conceptual BBM model can be used in a simplified setting in situations where considerable data on the river system already exists. For example, Alfredsen et al. (2011) used a simplified BBM approach to identify important flow components needed for establishing environmental flow suggestion in a Norwegian salmon river in a situation where limited resources were available. While not self-identifying as a BBM application, many other projects have recognized the need to identify important "building blocks" that are needed in order to establish a functioning environmental flow regime in modified rivers. For example, Enders et al. (2009) used a BBM-type approach to establish guidelines for flow regulation in an eastern Canadian Atlantic salmon river. Also, some similarities to BBM methodology can be identified in a project in South Saskatchewan River, Alberta, where the flow regime was compiled by a technical team by considering four ecosystem components (water quality, fish habitat, riparian vegetation and channel maintenance). A flow regime was determined as a consensus of the technical team and the final recommendation included adaptive management to validate the predictions of the models similarly to the BBM approach (Clipperton et al. 2003).

3.4.2 Downstream Response to Imposed Flow Transformation (DRIFT)

The DRIFT methodology was developed from the foundations of the BBM method in South Africa (King et al. 2003). Unlike BBM, the DRIFT methodology is a top-down, interactive, scenario-based approach, designed for use in environmental flow negotiations (Tharme and King 1998). The DRIFT framework is comprehensive and includes all major abiotic and biotic components that constitute the ecosystem to be managed (King et al. 2003). The methodology employs experienced scientists from different biophysical disciplines which include hydrology, hydraulics, fluvial geomorphology, sedimentology, chemistry, botany and zoology (King et al. 2003). While DRIFT methodology makes extensive use of expert knowledge, the guidelines for selecting scientific panel members for DRIFT projects are based upon the well-established protocols of the BBM (King et al. 2008). Moreover, the roles, responsibilities and interactions of DRIFT panel members are governed by the step-by-step procedures built into the DRIFT methodology and the possibility for any one member to dominate the workshops or bias the outcomes of the scenario evaluations are removed (King et al. 2003; Arthington et al. 2003).

The DRIFT framework consists of four modules (King et al. 2003; Figure 10):

1) Biophysical module:

The component is used to describe the present ecosystem condition in the river and to collect data on all aspects of the biota so that predictions can be made how it would change with flow changes. Data collection is multidisciplinary and includes similar components as described for the BBM, including informed study site selection. Analysis includes the use of 10 hydrological statistics that are used to summarize daily flow records (*sensu* 32 IHA statistics), and hydraulic modelling (either 1D or 2D) that translates flow changes into variables that are needed to evaluate the flow-related impacts on biota (wetted width, velocity, depth etc).

2) Sociological module:

The component identifies the groups of people directly affected by flow alteration (i.e. "Population At Risk“, defined as those people who live along the river and use its resources for subsistence) and describes the potential social impacts.

3) Scenario development:

The different environmental flow regime scenarios are drafted (typically less than five). Each discipline represented in the biophysical module is then assessed using direction and severity ratings that are decided in each project and the effects of each scenario on subsistence users are also described. The flow regime is then negotiated using the different severity rating trade-offs in an expert workshop environment.

4) Economics:

The component is used to calculate the costs of mitigation and compensation for people who are directly depended on the riverine ecosystem to be affected by the proposed alterations.

Modules 2 and 4 are omitted if the case study does not involve subsistence users (King et al. 2003) but in addition to the basic modules DRIFT framework should be run in parallel with two other exercises which are external to it: 1) an economic assessment of the wider regional implications of each scenario, and 2) a Public Participation Process whereby people other than subsistence users can indicate the level of acceptability of each scenario (King et al. 2003).

Arthington et al. (2003) provide a clear description of how specific biophysical disciplines are considered within the Biophysical module; the example is given by using a fish component for DRIFT framework. To illustrate the extent each discipline is (or should be) represented within the Biophysical module, the basic steps of the DRIFT-fish, as reported in Arthington et al. (2003), are shown:

"The basic steps in the fish component of DRIFT are the following:

Step 1. Review of literature to produce a compilation of published flow-related information on each fish species in the study rivers.

Step 2. Selection of study sites to characterize river reaches likely to be affected by existing and future water resource developments.

Step 3. Seasonal field surveys at each site to determine fish species composition, abundance and habitat use in relation to flow conditions.

Step 4. Analysis of field data to generate habitat preference curves for each fish species.

Step 5. Tabulation of field data and information from literature review to produce summary of flow-related data on each fish species.

Step 6. Development of scenarios of flow regime change for evaluation using DRIFT.

Step 7. Development of protocols to document the consequences of flow regime change for each fish species at each study site.

Step 8. Prediction of the ecological and social consequences of flow regime change for each fish species at each study site.

Step 9. Preparation of a monitoring strategy to assess the outcomes of environmental flow provisions.

Step 10. Implementation of monitoring programme, evaluation of ecological outcomes of any environmental flow provisions, and adjustment of those provisions in the light of new knowledge generated by monitoring (and research)." Arthington et al. (2003), p. 643-644).

Considerable uncertainty in the decision-making is inevitable when the ecological consequences are predicted for the different species within each biotic component (e.g. step 8 for fish above). The DRIFT framework accounts for this uncertainty by using "severity ratings" and the predicted direction in change while the confidence level in all these decisions is also reported. When the different flow scenarios are contrasted, patterns in the direction of change and severity emerge, and can be used for making a decision between the scenarios despite the uncertainty. Like any other holistic framework, DRIFT must also be followed up with adaptive management that is based on quality monitoring data (Arthington et al. 2003).

The implementation costs of the DRIFT framework can be significant depending on the scope of the research carried out in the biophysical module (Acreman and Dunbar 2004). However, DRIFT is more suitable for trade-off negotiations than the BBM method as implications of not meeting the environmental flow targets can be assessed (Tharme 2003).

3.4.3 Benchmarking and the derived frameworks

The idea of "benchmarking" was originally developed in Fitzroy Basin, Australia (Arthington 1998). The benchmarking methodology was designed to link information on alterations of natural flow regimes to ecological consequences of flow regime change (Arthington and Pusey 2003).

The main idea is to evaluate the condition of a range of rivers (or river reaches) that have been subjected to various degrees of flow regulation and water resource development (Arthington and Pusey 2003). The flow regime of the study river is described using a set of key flow statistics that are thought to have important ecological relevance (12 variables in the original method; Arthington 1998). The next step is to describe the percentage of change in each flow statistic from its natural (pre-regulation) value and link that to the observed ecological (or geomorphologic) impact. These relationships (% change in flow statistic versus ecological impact) can be used for making probability statements about the ecological implications of altering a river's flow regime by specified amounts compared to the natural regime (Arthington and Pusey 2003). This can be further used to establish limits, or "benchmarks", for the maximum "allowable" change in each flow statistic in comparison to the natural condition.

The concept of benchmarking was further adopted within a new framework that was designed to fight a " *growing temptation to ignore natural system complexity in favor of simplistic, static,* *environmental flow "rules" to resolve pressing river management issues*" (Arthington et al. 2006, p. 1311). They suggested a new approach that incorporates essential aspects of natural flow variability common across particular classes of rivers that can be validated with empirical biological data and other information in a calibration process (Arthington et al. 2006). Therefore, the framework was designed to provide sustainable flow management criteria for larger groups of "similar" rivers.

In this framework, four steps were suggested (Figure 11; Arthington et al. 2006):

Step 1: *Develop classification for reference streams*;

> Groups of similar streams are defined using ecologically-relevant flow statistics that are identified in an analysis of the respective natural hydrographs (Figure 11a)

Step 2: *Develop frequency distributions selected flow variable in each class*;

> Rivers within a similar reference group have natural variability with respect to the selected ecologically-relevant flow statistics. By combining the information from multiple rivers within the same reference class, a composite hydrographs is developed and can be used as a "norm" within each class of rivers (Figure 11b)

Step 3: *Frequency distributions are compared between flow modified and natural streams within the same class;*

> A measure of deviation from the norm is obtained for each ecologically-relevant flow statistic (Figure 11c)

Step 4: *Develop flow-response relationships using selected ecological health indicators from reference and flow modified steams for each flow variable.*

> Threshold conditions for the selected flow variables are identified based on a number of river health indicators (e.g. aquatic macroinvertebrate diversity, density of a select fish species etc.). These thresholds become the "benchmarks" that are used to design an environmental flow regime within each river class.

Arthington et al. (2006) is a clear precursor for the ELOHA framework (see below).

3.4.4 The Ecological Limits of Hydrologic Alteration (ELOHA) Approach

The ELOHA framework represents a recent consensus view from a group of internationally recognized environmental flow scientists (Poff et al. 2010), building on a previous framework described in Arthington et al. (2006). ELOHA can be used to determine ecological limits of flow alteration at a regional scale and, therefore, simultaneously for a large number of "similar" rivers (Poff et al. 2010). The ELOHA framework does not reveal any new environmental flow assessment techniques *per se* but rather provides a consistent approach for analysis and synthesis of available information (i.e. using existing hydrologic techniques and environmental flow methods) to achieve environmental flows (Poff et al. 2010).

The goal of the process is to develop regional environmental flow standards that are rooted in a science-based, quantified evaluation of the effects between different categories of flow alteration and the consequent effects on riverine biota.

The ELOHA framework consists of five basic steps divided into scientific (steps 1-4) and social processes (step 5) (Figure 12):

Step 1: **Hydrologic modeling** for baseline and status quo hydrographs in the region;

Step 2: **Classification of rivers** (or river segments) based on flow regime and geomorphic types;

Step 3: Determination of the **extent of alteration**;

Step 4: Development of **flow-ecology relationships** for the different river types; and

Step 5: Establishing **environmental flow standards** with subsequent **monitoring** and **adaptive management**.

Because the ELOHA framework provides a possibly promising approach to be used in the Canadian context, each of the steps are described in more detail with reference to their current status in Canada.

Step 1: Hydrologic modelling

The first step of the framework relies on the use of a hydrologic database which is explored to extract daily flow hydrographs for simulated baseline and developed conditions. Baseline conditions consist of rivers that have been minimally altered (i.e. "reference sites") whereas the developed conditions refer to rivers where the flow regime has been altered. The classification of the hydrographs into two categories should be carried out using a single time period (to be able to separate human and climatic influences) of at least 10-20 years with flow record (and the process may have to be iteratively renewed depending on the potential effects of climate change; see below). Statistical techniques and hydrological rainfall-runoff models can be used to generate data for ungauged locations (Poff et al. 2010, and references therein).

In Canada, the Water Survey of Canada (WSC) maintains a national database (HYDAT) for gauged daily hydrological data (rivers, lakes and reservoirs) collected by provincial and federal agencies. Currently, WSC releases an updated Access database (free to download) approximately every three months containing the most recently available hydrometric data in addition to station information. An examination of the available daily hydrometric data, the stations and the length of available data records emphasises the variability in both spatial and temporal coverage across Canada. Figure 13 shows that the majority of sites have 20 years or fewer of available data.

Examination of Figure 14 demonstrates the spatial disparity of data collection with a strong southern bias to station location. This is not unsurprising because the majority of the population lives in southern Canada and it is easier to maintain these gauges. However, recently additional data is being collected in northern Canada (Figure 14a).

Established by the WSC (Environment Canada), a subset of 255 hydrometric gauging stations has been allocated to the Reference Hydrometric Basin Network (RHBN) (Figure 14b). The RHBN stations provide a network for monitoring longer-term changes and impacts of climate. Brimley et al. (1999) and Harvey et al. (1999) defined the original station selection criteria: (i) sites should represent near-natural conditions with less than 10% modification from natural flow conditions; (ii) absence of significant regulations or diversions upstream of the gauging station (less than 5% of the area regulated); (iii) minimum of 20 years of hydrological data; (iv) longevity of the station in its current pristine or stable state in the future; and (v) highly accurate data records. The final criterion refers to the breadth of coverage of the different types of available hydrometric stations (seasonal, continuous, streamflow, and lake level). Of the 255 RHBN stations, 223 record river discharge.

Coupled reference and developed conditions hydrologic time series should be developed for all locations in the region where environmental flow protection is anticipated (Poff et al. 2010). The hydrological data for both reference and developed are also needed to establish the flow alteration-ecological response relationships (i.e. step 4 of the ELOHA framework). The existing HYDAT database with the prescribed reference stations provides a good starting point for implementing the step 1 of the ELOHA framework in Canada. However, it must also be stated that unresolved issues remain concerning the minimum hydrometric series length required to capture climate variability in temperature and precipitation, which drive runoff processes. This issue is further exacerbated by the influence of anthropogenic climate change as a further forcing factor.

Step 2: Classification of hydrologic regimes

The rivers that have been identified to represent the reference conditions (step 1) are classified into similar groups or "river types" (similar to the Arthington et al. (2006) framework). The region within which the classification is spatially and temporally variable from areas defined by jurisdictional boundaries (e.g. provinces) to more natural biophysical domains within a larger geographic area (see below). Classifying is based on ecologically-relevant characteristics of the flow regime and geomorphology. Identifying river types is based on flow-geomorphology relationships and corresponds to different ecological characteristics. Therefore, the form and the direction of an ecological response to flow alteration is hypothesised to be similar within, but vary between, river types (Poff et al. 2010). This allows the generalization of environmental flow thresholds (step 5) for all the rivers within the same type (Arthington et al. 2006). It is notable that the river types not necessarily are geographically contiguous, meaning that different segments of a river (known as "analysis nodes" in ELOHA) may be classified as different river types; headwater tributaries and a mainstem river in the same basin would likely be classified as different river types (Kendy 2009). This will also depend on the geomorphology component because for example, a given flow level may have different ecological consequences in a bedrock vs an alluvial bed river. Various methods and tools exist for both hydrological (reviewed in Olden et al. 2011) and geomorphological classifications (e.g. Elliot and Jacobson 2006; Thompson et al. 2001).

In the Canadian context, Monk et al. (2011) provide an initial hydrological regime classification for rivers across the country. Using the RHBN sites with daily data available for hydrological years 1970-2005, the authors quantified the timing (shape) of the annual hydrological regime, which allows spatial (between-station) patterns to be examined. Based on standardised weekly runoff averages, an agglomerative hierarchical classification method (using Ward's approach) modified from Hannah et al. (2000) and Harris et al. (2000) was applied to identify homogenous hydrological regions with similar timing of the annual flow regime, regardless of absolute magnitude. Six hydrological regime shape classes were identified reflecting known geographical and climatic variability across Canada (Figure 15). Following part of the standardised approach to classification proposed by Olden et al. (2011), the classification approach adopted by Monk et al. (2011) provides a starting point for the development of an ELOHA framework in Canada for future environmental flows assessment. The method could be expanded to include non-RHBN stations in addition to developing a method to identify the river type for hydrologically-impacted hydrometric stations (to be used in step 3) or non-gauged basins.

Step 3: Determination of the extent of alteration

In the third step of the ELOHA framework the extent of flow alteration is determined as the deviation of the flows between the historic (baseline) and the current day condition in each individual river (or analysis node) for each hydrological metric and is expressed as % of deviation from the baseline (Poff et al. 2010). These data are used as an input when developing flow alteration-ecological response relationships (step 4). A variety of tools exists for calculating the extent of flow alteration; IHA and SAAS are described above (Section 3.1.3) in addition to another commonly used software the Hydrologic Alteration Tool (HAT) of the U.S. Geological Survey's Hydroecological Integrity Process (HIP) package is described in Henriksen et al. (2006).

The key part of the ELOHA framework is carried out in step four where biological and hydrological data are combined into flow alteration-ecological response relationships (flow-ecology relationship for short), similarly to the framework described in Arthington et al. (2006, Figure 12). The process of developing flow-ecology relationships begins by formulating testable hypotheses how flow is expected to alter the ecological response variables that are chosen for evaluation. The hypothesis formulation is carried out as a collaborative process by scientists familiar with the ecology and hydrology of the region that is being considered (e.g. Haney et al. 2008).

Step 4: Development of flow-ecology relationships

The next stage is to collate existing ecological data to quantify the relationships. The data needs to be collected within each river type identified at step 2 of the framework because globally transferable flow-ecology relationships have not yet been developed (Poff and Zimmerman 2010). The ecological response variables ("health indicators" in Arthington et al. 2006) that are used to establish flow-ecology relationships should be: 1) sensitive to existing or proposed flow alterations; 2) can be validated with monitoring data; and 3) are valued by society (Poff et al. 2010). The response variables can be simple metrics (e.g. presence-absence of a species, changes in relative abundance of a species) or based on composite indices. A number of different indices have been developed for fish (e.g. Fausch et al. 1984; Pirhalla 2004; Bain and Meixler 2008; examples from the USA) and lotic macroinvertebrates (e.g. Extence et al. 1999; Monk et al. 2007; Armanini et al. 2011a; examples from the UK and Canada).

While no comparable fish metric for the development of flow-ecology relationships currently exists in Canada, a composite index for macroinvertebrates has been developed. The Canadian Ecological Flow Index (CEFI) was developed using paired benthic macroinvertebrate community samples and flow velocity data extracted from the Canadian Aquatic Biomonitoring Network (CABIN) database (Armanini et al. 2011a). The index approach summarises flow velocity preferences for common benthic macroinvertebrate taxa within a sample. Therefore, CEFI offers a quantitative method for linking hydrological regime variability (both natural and anthropogenic) to compositional variation in the ecological community. To test CEFI and its response to hydrological variability, Armanini et al. (2011b) demonstrated how runoff regime type (using the classification of Monk et al. (2011) influenced CEFI community response. In addition, Peters et al. (2012) highlighted a practical approach to utilise CEFI in an environmental flows assessment. Using a reference condition approach, observed CEFI values for potentially impacted sites can be compared with an expected value for regional hydrologically-similar rivers. The index values have not yet been collected across a gradient of flow-altered sites in order to establish a flow-ecology relationship. Nevertheless, CEFI seems as a good candidate for initiating such undertaking in a nationally standardized context.

In ideal situation, the flow-ecology relationships are quantified as % change in health indicator vs % change in flow metric (Arthington et al. 2006) but may have to be expressed in a more simple format such as categorical or binomial relationships (Poff et al. 2010). In some cases, data are abundant to develop flow-ecology relationships across a gradient of flow alterations (Apse et al. 2008). More often, this may not be the case. In cases of limited data, different approaches have been used to advance in the ELOHA framework. Different studies have resorted to expert opinion (Haney et al. 2008) and statistical analysis (Konrad et al. 2008; Webb et al. 2010) to establish the flow-ecology relationships. Compilation of data will at any case identify conditions where ecological data are missing; such information can be used to direct future field studies into strategic locations (Arthington et al. 2006). Finally, whether data are abundant or scarce, scientists must account for confounding factors such as changes in water quality and temperature, ice regime, physical habitat degradation, and invasive species, which may cause substantial impact even with minimal flow alteration (Kendy et al. 2009); however, it is possible to limit the influence of confounding variables using appropriate statistical approaches (e.g. Armanini et al. 2011a).

Step 5: Establishing environmental flow standards

Once flow-ecology relationships have been established for key hydrological variables, environmental flow standards can be established through a stakeholder-driven process. The benchmarking approach (Arthington et al. 2006) has been adopted as a tool to help establish ecologically and societally acceptable flow thresholds (Poff et al. 2010). Societally acceptable flow thresholds may be related to public concern around the "value" of each individual river (or segment) within the river type and/or analysis node, and the framework could benefit if analysis nodes were assigned to different "management classes" (e.g. pristine, modified, highly modified etc.), as has been done in other approaches (see Section 4). The ecological goals and therefore, e.g. maximum limits for water abstraction, could then be differently assigned within the management classes (intuitively this would be stringent "rules" in pristine waters, and more relaxed thresholds in modified catchments). The ELOHA framework does not pre-determine the societal process by which environmental flow standards are finally derived, but case studies show that this can take place through stakeholder/expert consultation and committee meetings (Poff et al. 2010). Thus the ELOHA framework, like other holistic frameworks, is highly suitable to establish an initial set for regional environmental flow standards that can be refined in an iterative loop of adaptive management based on monitoring data (Poff et al. 2010).

Examples of ELOHA projects

A number of projects using at least some parts of the ELOHA template have recently been initiated (described in the ELOHA website: http://conserveonline.org/workspaces/eloha). Several projects within the United States are currently applying elements of ELOHA to accelerate the integration of environmental flows into regional water resource planning and management (Kendy et al. 2009). In the US, ELOHA has been applied in Pennsylvania, Tennessee, Michigan, Arizona, Colorado and Washington at least in some parts of the state (Apse et al. 2008; Kendy et al. 2009; Sanderson et al. 2011). ELOHA has also been used in Australia (Arthington 2009). Many projects are at a stage where only some steps of the ELOHA framework have been accomplished or adopted. For example, a project in Huai River basin in China used ELOHA steps 1 to 3 to provide a foundation for development of hydro-ecological relationships in the region (Zhang et al. 2011). In Canada, Peters et al. (2012) recommended a framework to be used to establish environmental flows in agricultural regions of Canada with many similarities to ELOHA.

While implementing full ELOHA framework across Canada would probably take substantial amount of time to complete, it provides a number of benefits over more traditional methods used for establishing environmental flow regimes. The main benefits are that environmental flow standards are developed for large, "similar" regions simultaneously, and the decisions are based on scientifically driven, testable, and quantified relationships that link hydrological change to meaningful ecological variables.

3.4.5 Strengths, weaknesses and data requirements of holistic methods

Holistic methods have several advantages in comparison to other methodologies. These methods consider all aspects of the flow regime, retain the natural-like hydrological regime and address all relevant components of the river ecosystem and take the associated societal needs into account (Tharme 2003). The methods, in general, operate under the premise that ecosystem must be assessed and satisfied before humans can take water (Arthington 1998).

Holistic methods rely considerably on professional judgment and expert opinion. It can be debated who qualifies as "an expert" and there is always a risk that expert panels are dictated by a few dominant personalities or that decision process becomes flawed due to interpersonal dynamics (Gippel 2005). Depending on the depth of evaluation, data collection, and the extent of expert consultation, applications of holistic framework can be time consuming and very expensive. Also, if single flow regime is prescribed by an expert panel, the natural inter-annual variation in flow regime will be lost; however, recent holistic applications have recognized the importance of inter-annual variability and describe different flow regime depending on the "water availability" in each year (i.e. dry, normal, wet year; Alfredsen et al. 2011).

Data requirements for holistic methodologies are not easily specified. Many holistic methods rely on expert opinion and therefore, well informed specialists within each ecosystem component to be addressed are needed (King et al. 1999). The holistic methods can utilize data collected using any other environmental flow assessment methodology as an input and generally, the expert panel benefits from receiving as much supporting data as possible, including the data need concerning recreational use and subsistence requirements of the local people (King et al. 1999).

Data requirements for ELOHA-type framework are considered above. The intention in the ELOHA framework is to initially invest in simple tools and rely on existing data and continue to more complex and expensive approaches at later stages (Poff et al. 2010). This will include additional data collection depending on where the largest data caps are identified after the initial models have been formulated. It is possible that flow alteration - ecological relationships are difficult to establish for some river types if the gradient in terms of flow alterations is limited (e.g. Arctic Canada). In such cases, alternative approaches have to be considered (e.g. DRIFT framework).

4. SUMMARY OF ENVIRONMENTAL FLOW GUIDELINES CURRENTLY USED IN CANADA AND INTERNATIONALLY

4.1 CURRENT ENVIRONMENTAL FLOW GUIDELINES IN CANADA

The information represented herein is collected from three general sources; 1) current (i.e. Oct 2011 - Jan 2012) www-sites of the Agency/Ministry responsible for water allocation in each province, 2) personal communications with provincial experts (Table 7) and 3) a review by Katopodis (2009). The list of provincial contacts is intended as a source for an *initial* point-of-contact in the respective provinces with regard to environmental flows and does not imply an all-inclusive list of provincial experts. In addition, an Agency / Ministry responsible for water allocation is reported below for each jurisdiction in order to facilitate further enquiries regarding environmental flows.

Some general similarities were identified between the different jurisdictions with regard to environmental flow guidelines. The shared features can be summarized as:

- All jurisdictions share a similar intent to protect water resources from anthropogenic impacts in rivers; this is typically contained in provincial act related to water resources ("Water Rights", "Water Protection Act", "Water Act" etc.). The provincial legislation protects water resources using relatively general description (i.e. reference to protection of aquatic biota/ habitat), without specifying the means how to e.g. determine environmental flow thresholds. In general, overarching rules describe that alterations must be sustainable, should not cause any significant adverse effects to the watercourse and that "some" amount of water is required in river to maintain a healthy aquatic habitat. Environmental flow guidelines, if these exist, tend to be recommendations or "best advice" to the regulator and not legally binding.
- A provincial permit (e.g. "water withdrawal approval", "a watercourse and wetland alteration permit", "permit to take water" etc.) from an agency responsible for water resources (i.e. typically provincial government) is generally required if flow alterations are considered. Some small amounts of water for consumptive use can be withdrawn without a permit (typically in the range of 20 to 50 m^3 / day). Many different branches of local government are typically consulted in the permitting process. Federal agencies (i.e. DFO) are involved in cases where a potential for "Harmful Alteration, Disruption or Destruction of fish habitat" (i.e. HADD) needs to be considered.
- In terms of HADD considerations, many provinces aim to a set of rules that can be used to differentiate between "no HADD" and "potential HADD". This translates into a 2-tiered structure (Figure 16) with some general guiding rule for distinguishing the "no HADD" cases (referred to as Level 1 assessment below). In projects where the general guiding rules cannot be met, site-specific studies are to be carried out to determine more specific environmental rules (referred to as Level 2 assessment below)
- Definitive frameworks or protocols to set environmental flow standards do not seem to exist for cases where potential HADD is invoked (i.e. site specific regarding Level 2 assessments; but see Lewis et al. 2004 for British Columbia). The protocol is determined by case-by-case basis, and environmental flow standards are typically developed in a public participation decision making process.

- Cumulative effects of water abstraction are very poorly managed and no central databases seem to exist for small scale (i.e. Level 1 or "no HADD") consumptive projects. This may possess a potential problem because a large number of small consumptive projects may constitute a large cumulative effect.

4.1.1 British Columbia

Responsible agency: British Columbia Ministry of Environment

Level 1 assessment: The current environmental flow guideline(s) for British Columbia are comprehensively described in Hatfield et al. (2003), and the related habitat assessment methods in Lewis et al. (2004). Environmental flow guidelines are determined separately for fishless and fish-bearing streams, although the maximum diversion rate is determined as the Q_{80} for both stream types. For fishless streams, the environmental "cut-off" flow is determined as the Q_{50} during the low flow month (the low flow month is defined as the calendar month with the lowest median flow, based on natural mean daily flows).

The recommended environmental flow thresholds for fish-bearing streams are adjusted on a monthly basis. The environmental flow for the lowest flow month is set as Q_{90}, and for the highest flow month as $Q_{20.}$ The environmental flow thresholds for all other months are calculated as a percentile between Q_{90} and Q_{20} using a weighted function (Hatfield et al. 2003). As a result more water is available for diversion during high flow months than during low flow months.

Some practitioners have reported that the Level 1 rules are conservative to such an extent that they are rarely used - because many projects requiring water would be deemed not economically viable if based on guidance of these Level 1 environmental flow rules (R. Ptolemy, British Columbia Ministry of Environment, pers. comm.).

Level 2 assessment: Lewis et al. (2004) provides a guideline for carrying out detailed assessments for various environmental criteria that should be used as part of the process to determine environmental flows at a more detailed level. Habitat simulation methods are frequently used as part of the assessment, but are not used to determine static environmental flow levels, but the analysis is based on "building blocks" that are identified based on biological understanding life cycle of some species that are considered to be valued for special conservation.

4.1.2 Alberta

Responsible agency: Alberta Environment and Water

Level 1 assessment: After a comprehensive review, the province of Alberta recently established a guideline for environmental flow needs that is to be used in the absence of having site-specific information that could otherwise be used to establish an environmental flow (Locke and Paul 2011). The guideline uses a combination of POF and exceedence methods. The guideline prescribes the greater of either:

1) A 15% instantaneous reduction from natural flow or
2) The lesser of either the natural flow or the Q_{80} natural flow based on a weekly or monthly (depending on the availability of hydrology data) time step.

In other words, no water abstractions are allowed for the lowest flows that occur up to 20% of the time, and for the remaining 80% of the time, up to 15% of the natural flow can be withdrawn. The guideline is based on the objective to fully protect the aquatic environment relative to natural conditions. Approximately a dozen licences have been issued based on the desktop guideline (as of Dec. 2011; A. Locke, Alberta Fish and Wildlife Division, pers. comm.).

Level 2 assessment: No guideline currently exists. Environmental flows have been recently determined in a number of large-scale projects (e.g. Clipperton et al. 2003, Goater et al. 2007, Ohlson et al. 2010). The approach in these projects has been holistic, using multiple environmental criteria and a range of different methodologies for assessment. The assessments have not, however, followed any fixed framework, but have been carried out on a case-by-case basis.

4.1.3 Saskatchewan and Manitoba

Responsible agency: Saskatchewan Watershed Authority; Manitoba Conservation and Water Stewardship

Level 1 assessment: Neither province currently has public guidelines for establishing environmental flow standards, although development for provincial guidance in underway in both provinces. In Manitoba, the Tessman rule (see Section 3.1) has been used.

Level 2 assessment: No guideline exists, and environmental flows are determined case-by-case.

4.1.4 Ontario

Responsible agency: Ministry of Natural Resources environmental flows related to hydropower developments), Ministry of Environment (consumptive use).

Level 1 assessment: There is currently no guideline and various rules have been used. However, both the Ministry of Natural Resources and Ministry of Environment are drafting science advice related to environmental flows to support respective approvals processes. Recent suggestions have considered Q_{70} as a general guideline in rivers where species at risk are present, and Q_{75} in other cases.

Level 2 assessment: No guideline exists, and the assessment methods are chosen by the proponents requiring flow alteration.

4.1.5 Quebec

Responsible agency: The Le ministre du Développement durable, de l'Environnement et des Parcs

Level 1 assessment: An "ecohydrological method" (Belzile et al. 1997; Berube et al. 2002) has been described to determine conservation flows in the rivers of southern Québec (south of 52^{o} N). The method has some conceptual similarities to the ELOHA framework. Different environmental flows are set for 10 different ecohydrological regions, wherein valued target (fish) species were evaluated. Critical life phases of the selected target species were determined and different seasonal flow exceedence rule was applied (not based on flow-ecology relationships) in each ecohydrological/species composition area. It is not known if the method is supported by the responsible agency, and if any guidelines apply in northern parts of Quebec.

Level 2 assessment: There is no current guideline and the proponent of a project must justify the chosen method that is used to establish environmental flows. Habitat simulation methods have commonly been used to establish environmental flows on a case-by-case manner.

4.1.6 Atlantic Provinces (New Brunswick, Nova Scotia, Prince Edward Island and Newfoundland and Labrador)

Responsible agencies: New Brunswick Department of Environment; Nova Scotia Environment and Labour; Prince Edward Island Department of Environment, Labour and Justice; Newfoundland and Labrador Department of Environment and Conservation

Level 1 assessment: All the Atlantic provinces have some general guideline regarding environmental flows, although the derivation of the is not well documented. The guideline in all Atlantic provinces consist of a rule when all water abstraction must stop. In New Brunswick and Prince Edward Island, 70 % of the monthly Q_{50} is used (in PEI, another cut-off threshold is added such that the above rule cannot lead to withdrawal below monthly Q_{95}). In Nova Scotia, 25% MAF rule is used and in Newfoundland and Labrador, the use of "low quartile of mean monthly lows method" has been suggested (i.e. Q_{25} of MMF; Rollings 2011).

Level 2 assessment: There is currently no guideline; habitat simulation methods have been used.

4.1.7. Eastern and Western Arctic

Responsible agencies: In Northwest Territories various land and water boards depending on the region; Nunavut Water Board. No information was obtained for Yukon.

Level 1 assessment: In Northwest Territories, DFO drafted a guideline for winter water withdrawal in 2005, which allowed a 5% instantaneous reduction from natural flow (Cott et al. 2005). The guideline was revised, however, in 2010 and no fixed allowable reduction was described (DFO 2010). The current assessment is carried out on a case-by-case basis and if withdrawal is allowed, the recommendation is typically 5 - 10 % of the instantaneous flow by the time of withdrawal. No specific guideline exists in Nunavut.

Level 2 assessment: N/A.

4.2 ENVIRONMENTAL FLOW GUIDELINES IN OTHER SELECT COUNTRIES

4.2.1 USA

Similarly to Canada, there is no nationwide framework for establishing environmental flows in the USA and the different states describe limits to flow alteration independently. Traditionally, habitat simulation methods have been extensively used to determine suitable environmental flows, targeting some valued species (Tharme 2003), and this still is the preferred method in many states. However, an increasing number of states have adopted various ways to classify rivers based on their ecological or societal values, and establish environmental flow standards based on some combination of hydrological methods, within the river classes or types. In many states, these steps have some resemblance to the ELOHA framework, and a few states are endorsing it fully (e.g. Minnesota, Pennsylvania).

Details for current environmental flow assessment methodologies for altogether 18 different US states can be obtained by reviewing Locke and Paul (2011, p. 59-72) and the case studies in ELOHA toolbox website [http://conserveonline.org/workspaces/eloha/documents/template-kyle].

4.2.2 European Union

The European Union member states are mandated by the EU Water Framework Directive (WFD) to achieve good ecological status in all waterbodies by 2015 (e.g. Acreman and Ferguson 2010). While the WFD does not implicitly make reference to environmental flows, it is generally accepted that ecologically appropriate hydrological regimes are necessary to meet the WFD requirements (Acreman and Ferguson 2010). The various methods to describe environmental flows in different EU member states were reviewed prior to the 2nd Workshop on Water Management, Water Framework Directive and Hydropower, held in Bryssels in September 2011; Kampa et al. 2011). The review showed a very wide range of guidelines concerning environmental flows in the EU (Appendix 2). Most countries have a recommendation for setting environmental flow regime but some countries determine environmental flow guidelines a case by case basis. To capture the range of different level of environmental flow protection in the EU, three cases are highlighted.

In Norway, no nationwide environmental flow guideline has been established, and the evaluation of flow standards is carried out on a case-by-case basis. As a general rule, the Q_{95} is used and is calculated separately for summer and winter seasons. More water is required for rivers that have a special "National salmon river" status (52 rivers) or where species of special concern exist. Habitat simulation methods are commonly used for the case by case evaluations. Recently, a holistic BBM method was applied in Norway (Alfredsen et al. 2011). [http://www.ecologic-events.de/hydropower2/documents/Norway.pdf]

In France, the requirements to provide environmental flows in rivers are required by law using the Tennant (1976) 10% of MAF rule. The rule is relaxed to 5% of MAF for rivers with hydropeaking operations. In practice, project proponents are often mandated to carry out a case study based on habitat simulation methods, using the EVHA model (Table1). [http://www.ecologic-events.de/hydropower2/documents/France.pdf]

In the United Kingdom, two major projects have been carried out (i) to define water abstraction limits that maintain a healthy river ecosystem and (ii) to define ecologically appropriate flow releases from reservoirs (Acreman and Ferguson 2010). The limits for water abstraction were developed for a number of river types (classification based on the requirements of fish, macroinvertebrates and aquatic macrophytes), and different percentages of natural flow can be abstracted in each river type. Moreover, the allowed abstraction limits vary between seasons, and more water can be taken during higher flows than low flows (e.g. for river type "A1" in winter, 35 % can be abstracted when flows $>Q_{60}$, down to 20 % when flows $< Q_{95}$; Acreman and Ferguson 2010). Overall, allowable withdrawal ranges between 7.5% and 35 % of the natural flow. For water releases from impoundments, a Building Block Methodology was adopted (see Section 3.4.1).

4.2.3 Australia, South-Africa and New-Zealand

The environmental flows in Australia have strongly centered on holistic methodologies. Similarly to Canada, Australia is divided into many jurisdictions who each describe environmental flows based on separate criteria. All the jurisdictions have to provide environmental flows, and although they have different legislation, every jurisdiction uses holistic methods and subscribes to monitoring and adaptive management (A. Arthington, Griffith University, Australia, pers. comm.). The common holistic methods used in Australia are reviewed in detail in Arthington (1998). Environmental flows in South-Africa are also prescribed using holistic methods (e.g. King and Brown 2006). In New Zealand, the environmental flow management is largely a responsibility of regional councils (Snelder et al. 2011). The national environmental flow standards are based on hydrological methods and the current set of guidelines was proposed in 2008. The New Zealand national guideline defines minimum flows and total allocation based on proportions of the mean annual seven-day low flow (MALF). Environmental guidelines are described separately for small (MAF < 5 m^3/s) and large rivers (MAF > 5 m^3/s), and are 90 % and 30% of MALF (minimum flow / total allocation), and 80% and 50% of MALF for small and large rivers, respectively (Snelder et al. 2011). The functionality of the general national guideline has been criticized (Snelder et al. 2011).

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TABLES

*Table 1. Commonly used environmental flow assessment methods, listed by a general category. A source for each method is provided; * indicates that the reference is not the original source of the method, but provides a comprehensive description to understand the basics of the method.*

Category	Method	Source
Hydrological	Tennant (Montana)	Tennant (1976)
	Aquatic Base Flow (ABF)	*Caissie and El-Jabi (1995)
	7Q10	*Caissie and El-Jabi (1995)
	7Q2	Belzile et al. (1997)
	Median Monthly Flow (Q50)	*Caissie and El-Jabi (1995)
	Range of Variability Approach (RVA)	Richter et al. (1997)
	Sustainability Boundary Approach and Presumptive Standard	Richter (2010); Richter et al. (2011)
Hydraulic Rating	Wetted Perimeter Inflection Point Method	*Gippel and Stewardson (1998)
	Flowing Perimeter Method	Gippel and Stewardson (1998)
	R-2 Cross	Espegren (1996)
Habitat Simulation	PHABSIM (Physical HABitat SIMulation system)	Bovee (1982)
	RYHABSIM (River hYdraulic and HABitat SIMulation)	Jowett (1989)
	EVHA (EValuation de HAbitat)	Ginot (1995)
	RSS (River System Simulator)	Alfredsen et al. (1995)
	CASIMIR (Computer Aided SIMulation of habitat In Regulated streams)	Jorde (1996)
	River2D	Blackburn and Steffler (2003)
	MesoHABSIM	Parasiewicz (2001)
	MesoCASIMIR	Eisner et al. (2005)
	Generalized Habitat models (e.g. STATHAB)	Lamouroux and Jowett (2005)
Holistic frameworks	Building Block Method (BBM)	Tharme and King (1998)
	DRIFT (Downstream Response to Imposed Flow Transformation)	King et al. (2003)
	Benchmarking	Arthington (1998; et al. 2006)
	ELOHA	Poff et al. (2010)

Table 2. Comparison of four general categories of environmental flow assessment methodologies.

Method Category	General purpose	Scale	Scope	Duration of assessment (months)	Relative Cost	Relative frequency of use*
Hydrological	Examination of historic flow data to find flow levels that naturally occur in a river and can be considered "safe" thresholds for flow abstraction	Whole rivers, applicable for regional assessments	Reconnaisence, "low risk situations"	0.5	$	+++
Hydraulic rating	Examination of change in a hydraulic variable, often wetted width, as a function of discharge; the change in variable is taken as a proxy for general quantity of habitat in a river	Applied at a study site / river segment scale, upscaling to whole river level based on the assumption of "representative" sites. River specific.	Medium	2-4	$$	+ (decreasing)
Habitat simulations	Examination of change in the amount of physical habitat for a selected set of target species as a function of discharge	Applied at a study site / river segment scale, upscaling to whole river level based on the assumption of "representative" sites. River specific.	Detailed	6-18	$$$	+++
Holistic frameworks	Examination of flows in an expert opinion workshop leading to recommendation of flows for all components of the riverine ecosystem, including societal and recreational uses	Whole rivers, applicable for regional or river specific scales	Flexible	12-36	$ - $$$$	+ (increasing)

Table 3. Flow recommendations as per the Tennant method. Based on Tennant (1976.)

Description of flows	Recommended flow regimen (% of Mean Annual Flow)	
	Oct - Mar	Apr - Sept
Flushing or maximum	200%	
Optimum range	60%-100%	
Outstanding	40%	60%
Excellent	30%	50%
Good	20%	40%
Fair or degrading	10%	30%
Poor or minimum	10%	10%
Severe degradation	< 10%	

Table 4: Summary of Indicators of Hydrologic Alteration (IHA) variables and ecological influences (adapted from IHA help file)

IHA Parameter Group	Hydrologic Parameters	n	Ecosystem Influences
1. Magnitude of monthly water conditions	Mean or median value for each calendar month	12 parameters	Habitat availability for aquatic organisms Soil moisture availability for plants Availability of water for terrestrial animals Availability of food/cover for fur-bearing mammals Reliability of water supplies for terrestrial animals Access by predators to nesting sites Influences water temperature, oxygen levels, photosynthesis in water column
2. Magnitude and duration of annual extreme water conditions	1-day, 3-day, 7-day, 30-day and 90-day mean annual minimum 1-day, 3-day, 7-day, 30-day and 90-day mean annual maximum Number of zero-flow days Base flow index: 7-day minimum flow/mean flow for year	12 parameters	Balance of competitive, ruderal, and stress- tolerant organisms Creation of sites for plant colonization Structuring of aquatic ecosystems by abiotic vs. biotic factors Structuring of river channel morphology and physical habitat conditions Soil moisture stress in plants Dehydration in animals Anaerobic stress in plants Volume of nutrient exchanges between rivers and floodplains Duration of stressful conditions such as low oxygen and concentrated chemicals in aquatic environments Distribution of plant communities in lakes, ponds, floodplains Duration of high flows for waste disposal, aeration of spawning beds in channel sediments

IHA Parameter Group	Hydrologic Parameters	n	Ecosystem Influences
3. Timing of annual extreme water conditions	Julian date of each annual 1-day maximum and 1-day minimum	2 parameters	Compatibility with life cycles of organisms Predictability/avoidability of stress for organisms Access to special habitats during reproduction or to avoid predation Spawning cues for migratory fish Evolution of life history strategies, behavioral mechanisms
4. Frequency and duration of high and low pulses	Number of low pulses within each water year Mean or median duration of low pulses (days) Number of high pulses within each water year Mean or median duration of high pulses (days)	4 parameters	Frequency and magnitude of soil moisture stress for plants Frequency and duration of anaerobic stress for plants Availability of floodplain habitats for aquatic organisms Nutrient and organic matter exchanges between river and floodplain Soil mineral availability Access for waterbirds to feeding, resting, reproduction sites Influences bedload transport, channel sediment textures, and duration of substrate disturbance (high pulses)
5. Rate and frequency of water condition changes	Rise rates: Mean or median of all positive differences between consecutive daily values Fall rates: Mean or median of all negative differences between consecutive daily values Number of hydrologic reversals	3 parameters	Drought stress on plants (falling levels) Entrapment of organisms on islands, floodplains (rising levels) Desiccation stress on low-mobility streamedge (varial zone) organisms

Table 5. The main steps identified in the Instream Flow Incremental Methodology (IFIM) framework (Source: Hudson et al. 2003, based on Stalnaker et al. 1995).

Step	Activity
1 Problem Identification	
Institutional Analysis	Identify/contact legitimately interested parties.
Physical Analysis	Determine: • Geographic extent • Possible factors limiting success.
2 Study Planning	
Strategy Design	Define: • Temporal and spatial scale • Important variables (e.g. water quality, habitat availability) • Plan data collection and analysis.
Technical Scoping	Describe: • Hydrologic time series and biological reference conditions (historic records or simulated) • Historic critical events (e.g. droughts, spills). Identify: • Critical reaches (e.g. impediments to migration) • River segment types (e.g. braided, meandering) • Representative reaches. Select: • Target species or guild • Select or develop habitat suitability criteria
3 Study Implementation	
Data collection	Measure within representative and critical reaches: • Velocity, depth, substrate and cover • Temperature, pH, dissolved oxygen, biological parameters …
Model calibration	• Verify model assumptions with site data • Calibrate models with site data
Simulation	• Describe relation between stream flow and stream habitat utility • Determine baseline habitat time • Evaluate water quality and temperature effects
Habitat Models	• Develop weighted usable area (WUA) vs. discharge function (PHABSIM) • Generate time series of daily or monthly WUA.
Interpretation	Determine habitat bottlenecks
4 Analysis of Alternatives	Test alternatives • Factor in physical and institutional constraints on water management • Compare scenarios with baseline habitat time series.
5 Problem Resolution	Negotiate multiple-use, water budget approach • Trade-off habitat cost/benefits, feasibility, risk and economics • Agreement → implement flow regime • No agreement → RE-START
6 Verification and Validation	Use adaptive management approach • Post-project monitoring and evaluation • Reassess flow objectives/flow regime

Table 6. Examples of classification schemes used in mesohabitat modeling. A) Classification used in the MesoHABSIM model (Parasiewicz 2007) and B) in the Norwegian mesohabitat model (Borsanyi et al. 2004). The Norwegian classification system uses thresholds to differentiate the choices between different categories (surface pattern: wave height above or below 5 cm; gradient: above or below 4 %; velocity: above or below 0.5 m/s; depth: above or below 0.7 m).

A)

Mesohabitat	**Description**
Riffle	Shallow stream reaches with moderate current velocity, some surface turbulence and higher gradient. Convex streambed shape.
Rapid	Higher gradient reaches with faster current velocity, coarser substrate, and more surface turbulence. Convex streambed shape.
Cascade	Stepped rapids with very small pools behind boulders and small waterfalls.
Glide	Moderately shallow stream channels with laminar flow, lacking pronounced turbulence. Flat streambed shape.
Run	Monotone stream channels with well determined thalweg. Streambed is longitudinally flat and laterally concave shaped.
Fast run	Uniform fast flowing stream channels.
Pool	Deep water impounded by a channel blockage or partial channel obstruction. Slow flow. Concave streambed shape.
Plunge pool	Where main flow passes over a complete channel obstruction and drops vertically to scour the streambed.
Backwater	Slack areas along channel margins, caused by eddies behind obstructions.
Side arm	Channels around the islands, smaller than half river width, frequently at different elevation than main channel.

B)

surface pattern (SP)	surface gradient (SG)	surface velocity (SV)	water depth (WD)	Code	Name
smooth/little waves	steep	fast	deep	A	Run
			shallow		
		slow	deep		
			shallow		
	moderate	fast	deep	B1	Deep Glide
			shallow	B2	Shallow Glide
		slow	deep	C	Pool
			shallow	D	Walk
broken/riffling	steep	fast	deep	E	Rapid
			shallow	F	Cascade
		slow	deep		
			shallow		
	moderate	fast	deep	G1	Deep Splash
			shallow	G2	Shallow Splash
		slow	deep		
			shallow	H	Rill

*Table 7. A list of initial points-of-contact in the different provinces of Canada with regard to environmental flows. * indicates that the contact does not represent a province-specific association, but general environmental flow expertise.*

Province	**Name**	**Affiliation**	**Email**
British Columbia	M. Bradford	Fisheries and Oceans Canada, Pacific Region	Mike.Bradford@dfo-mpo.gc.ca
British Columbia	R. Ptolemy	British Columbia Ministry of Environment, Aquatic Conservation Science Section	Ron.Ptolemy@gov.bc.ca
Alberta	A. Locke	Government of Alberta, Sustainable Resource Development, Fish and Wildlife Division	Allan.Locke@gov.ab.ca
Alberta	A. Paul	Government of Alberta, Sustainable Resource Development, Fish and Wildlife Division	andrew.paul@gov.ab.ca
Saskatchewan	M. Pollock	Saskatchewan Watershed Authority	Michael.Pollock@swa.ca
Manitoba	J. Long	Manitoba Water Stewardship, Ecological Services Division, Fisheries Branch	Jeff.Long@gov.mb.ca
Ontario	R. Metcalfe	Ontario Ministry of Natural Resources, Aquatic Research and Development Section	robert.metcalfe@ontario.ca
Ontario*	A. Bradford	University of Guelph	abradfor@uoguelph.ca
Ontario	K. Smokorowski	Fisheries and Oceans Canada, Central and Arctic Region	Karen.Smokorowski@dfo-mpo.gc.ca
Ontario	T. Kondrat	Ontario Ministry of Environment	Todd.Kondrat@ontario.ca
Ontario	M. S. Khan	Ontario Ministry of Environment	mohammad.khan@ontario.ca
Ontario*	M. Lebel	World Wildlife Fund	MLebel@WWFCanada.org
Ontario*	T. Maas	World Wildlife Fund	tmaas@wwfcanada.org
Quebec	A. St-Hilaire	INRS	Andre.St-Hilaire@ete.inrs.ca
New Brunswick*	A. Curry	Canadian Rivers Institute / University of New Brunswick	racurry@unb.ca
New Brunswick*	D. Baird	Environment Canada / University of New Brunswick	djbaird@unb.ca
New Brunswick	K. Collet	Government of NB, Department of Natural Resources	kathryn.collet@gnb.ca
New Brunswick	H. Pelkey	Government of NB, Department of Environment	Howard.Pelkey@gnb.ca
New Brunswick	D. Caissie	Fisheries and Oceans Canada, Gulf Region	Daniel.Caissie@dfo-mpo.gc.ca
Prince Edward Island	B. Raymond	Government of PEI, Department of Environment, Labour and Justice	bgraymond@gov.pe.ca
Nova Scotia	K. Garroway	Nova Scotia Environment, Water & Wastewater Branch	GARROWKG@gov.ns.ca
Newfoundland and Labrador	H. Khan	Government of NFL, Department of Environment and Conservation, Water Resources Management Division	hkhan@gov.nl.ca
Newfoundland and Labrador	K. Rollings	Fisheries and Oceans Canada, NL Region	Ken.Rollings@dfo-mpo.gc.ca
Western Arctic Area	P. Cott	Fisheries and Oceans Canada, Western Arctic Area	Pete.Cott@dfo-mpo.gc.ca
Western Arctic Area	B. Hanna	Fisheries and Oceans Canada, Western Arctic Area	Bruce.Hanna@dfo-mpo.gc.ca

FIGURES

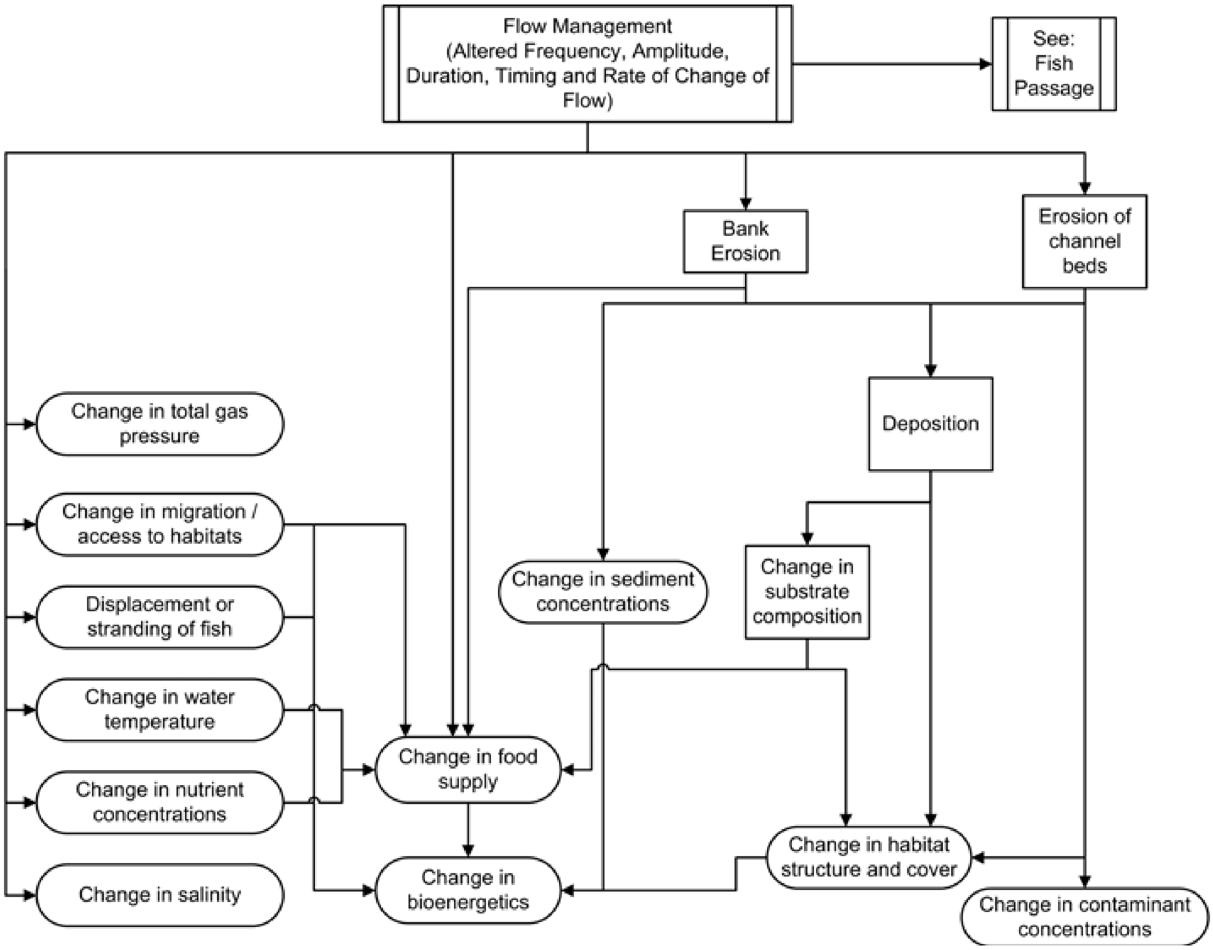


Figure. 1. Cascading effects of flow alteration on physical and biological aspects of the river ecosystem. In-water linkages affecting fish habitat are shown in boxes and endpoints of concern to fish are in bubbles. Source: Clarke et al. (2008). [The reproduction is a copy of an official work that is published by the Government of Canada and has not been produced in affiliation with, or with the endorsement of the Government of Canada]

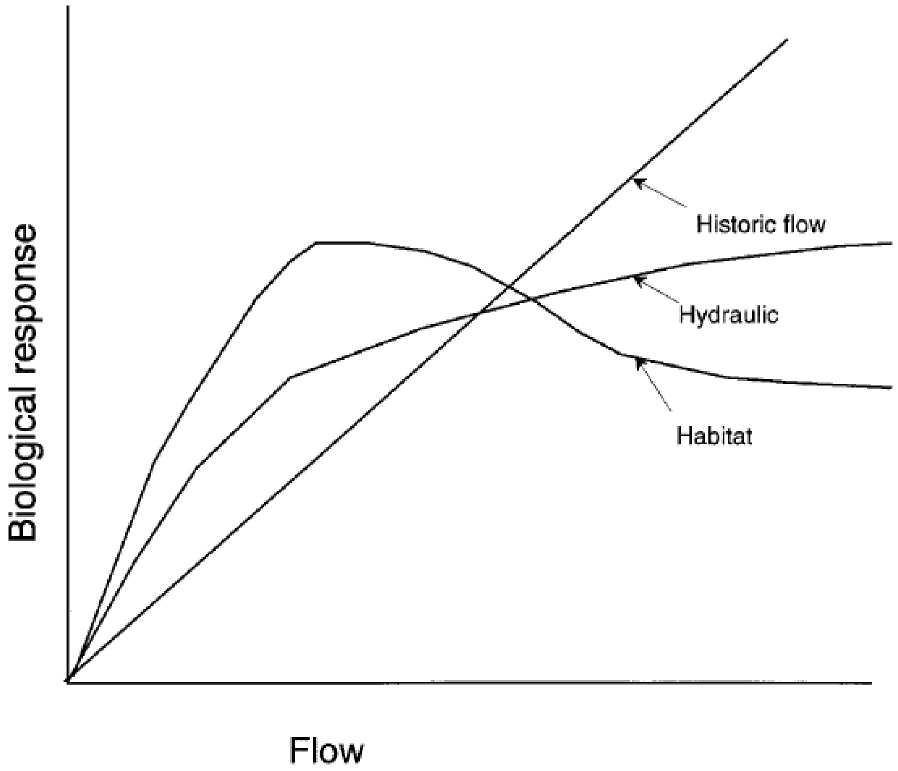


Figure 2. Conceptual relationships between flow and biological response as is assumed by different environmental flow assessment methodologies. Historic flow refers to the duration curve exceedence methods within the hydrological methodology group, and not to % MAF methods for which the conceptual response between flow and biological response is assumed to be similar to the Hydraulic methods (see Figure 3). Source: Jowett (1997). Copyright (2012): Wiley, reproduced with permission.

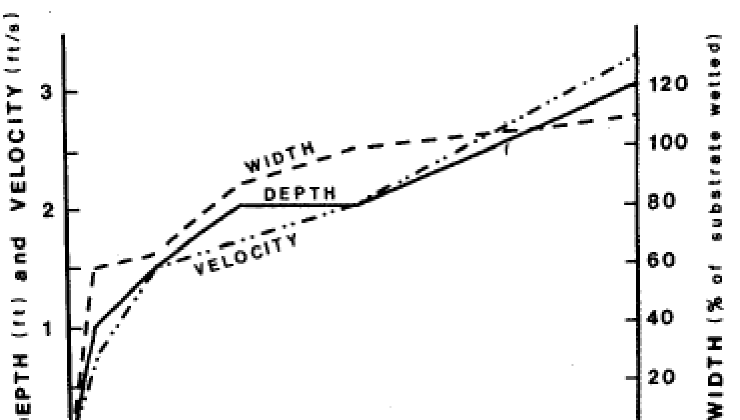
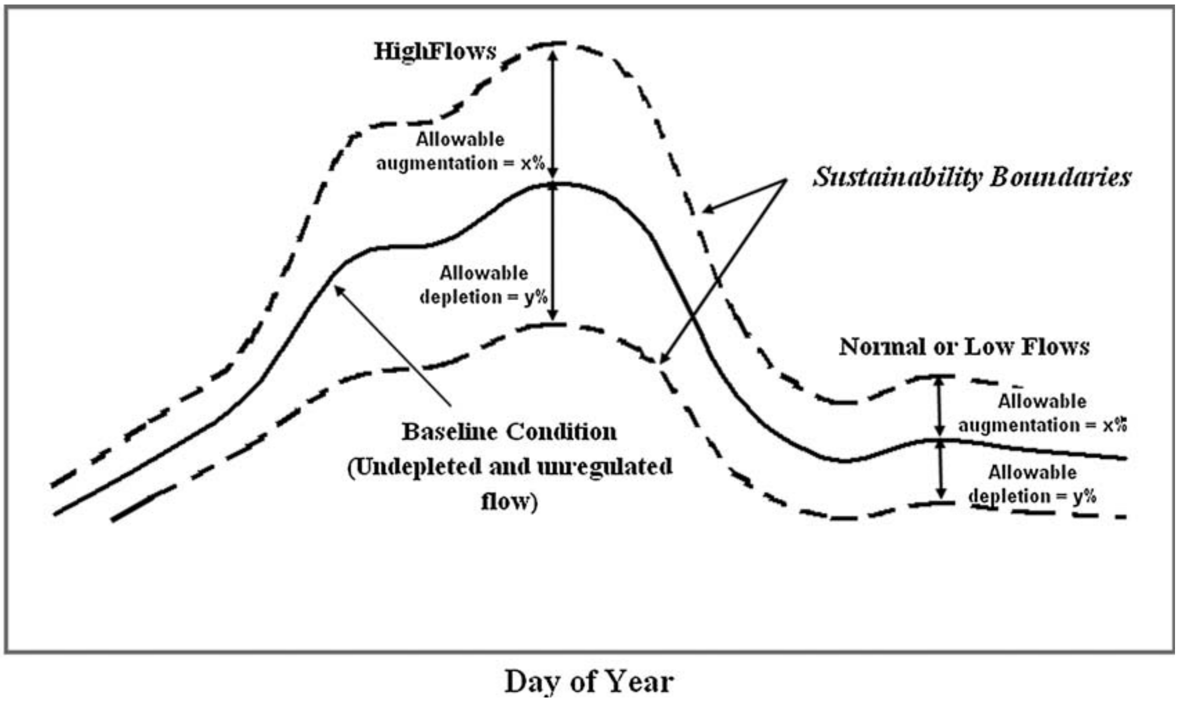


Figure 3. Tennant's (1976) relationship between proportion of mean annual flow and depth, velocity and wetted width, based on 11 streams (38 different flows, 58 cross-sections) located in Montana, Wyoming and Nebraska, USA. Copyright (2012): Taylor and Francis, reproduced with permission.

A)



B)

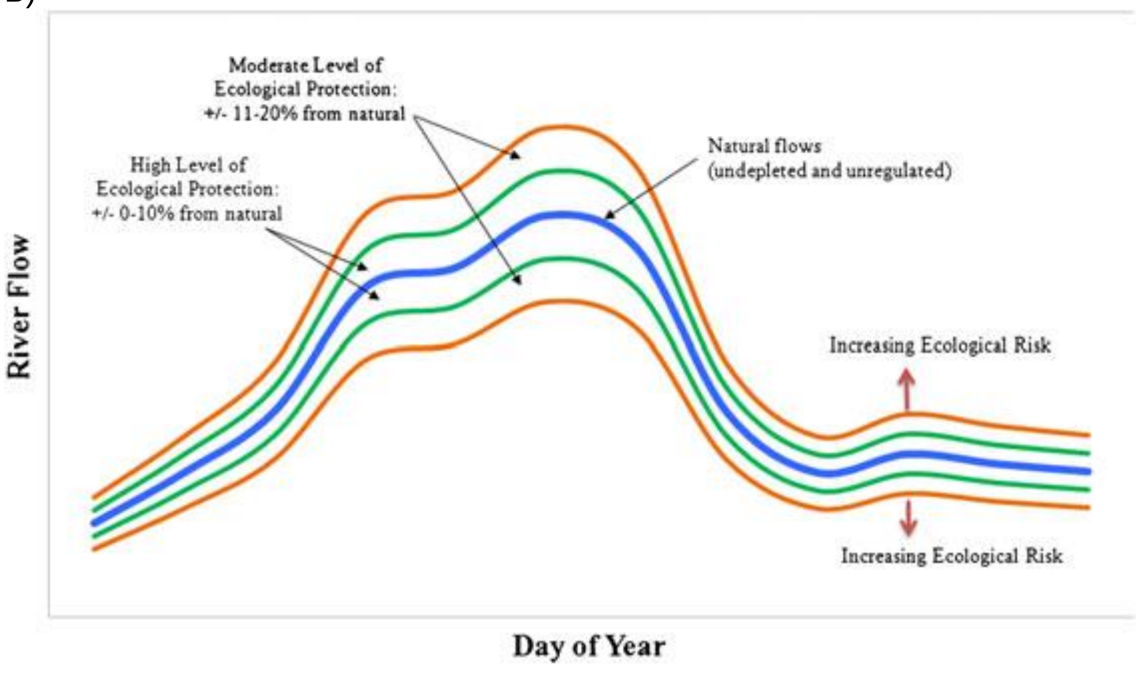


Figure 4. A) An illustration of the basic idea of Sustainability Boundary Approach (Source: Richter 2010, Copyright (2012): Wiley, reproduced with permission) and B) SBA applied as the Presumptive Standard for environmental flow protection. Source: Richter et al. 2011, Copyright (2012): Wiley, reproduced with permission.

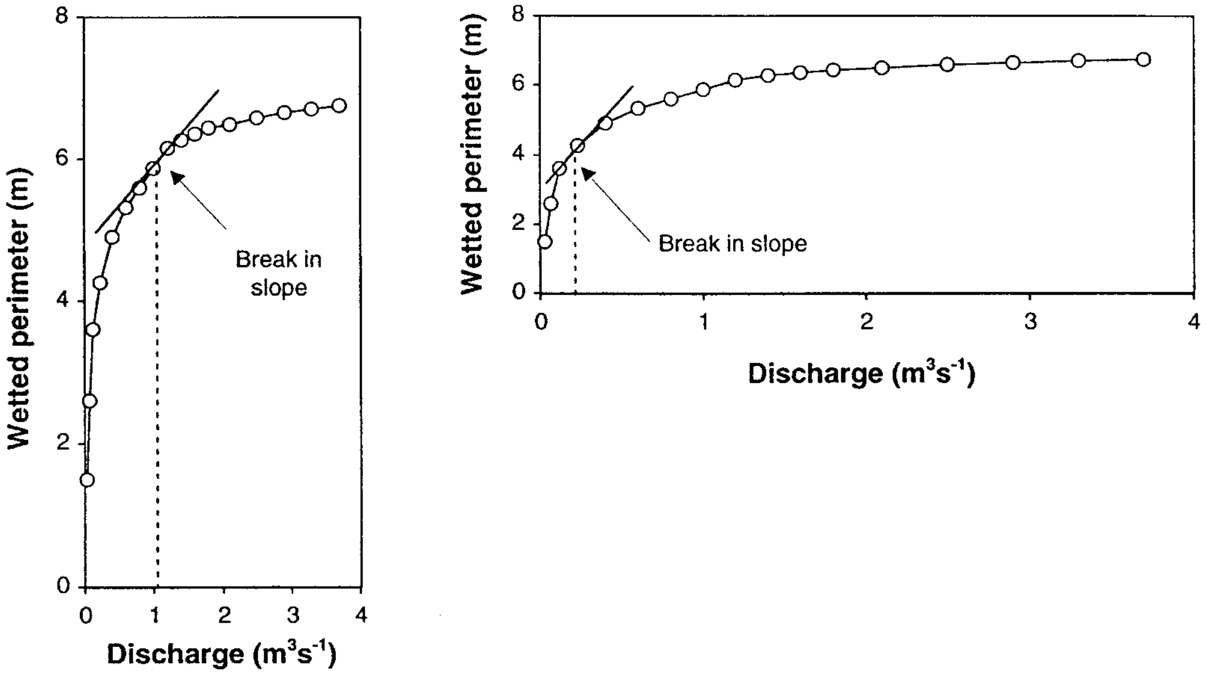
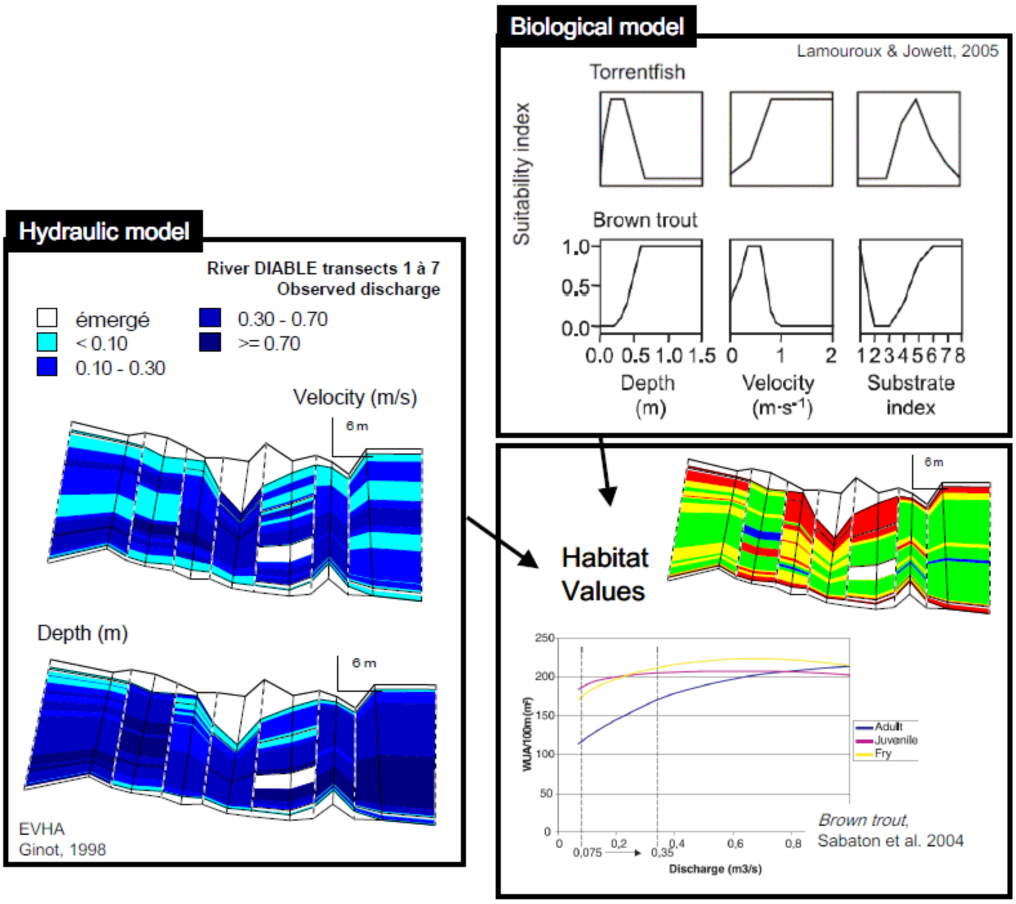


Figure 5. Hypothetical wetted perimeter–discharge relationship plotted on two differently scaled sets of axes. Different breakpoints are apparent, even though the same data are plotted on both graphs. Source: Gippel and Stewardson (1998), Copyright (2012): Wiley, reproduced with permission.

A)



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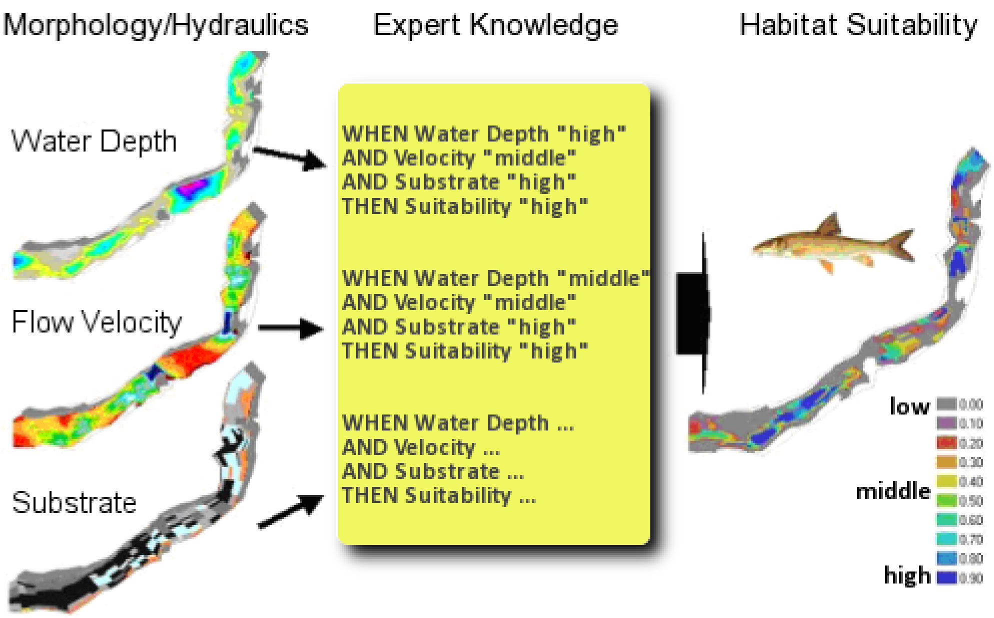

Figure 6. A) Generalized approach used in habitat simulation methodology. (a) Hydro-geomorphology of the study site(s) are surveyed and hydraulic models are calibrated, (b) Species-habitat association models are developed for chosen variables, (c) weighted usable area is estimated as a function of discharge for the select species and/or lifestages. Source: Girard (2009), reproduced with permission.

B) Habitat modelling approach using a fuzzy-logic based approach. The Habitat Suitability Indices are replaced by expert derived fuzzy rules. Source: CASiMiR website http://www.casimir-software.de/aufbau_eng.html, Copyright (2012): Schneider & Jorde Ecological Engineering, reproduced with permission.

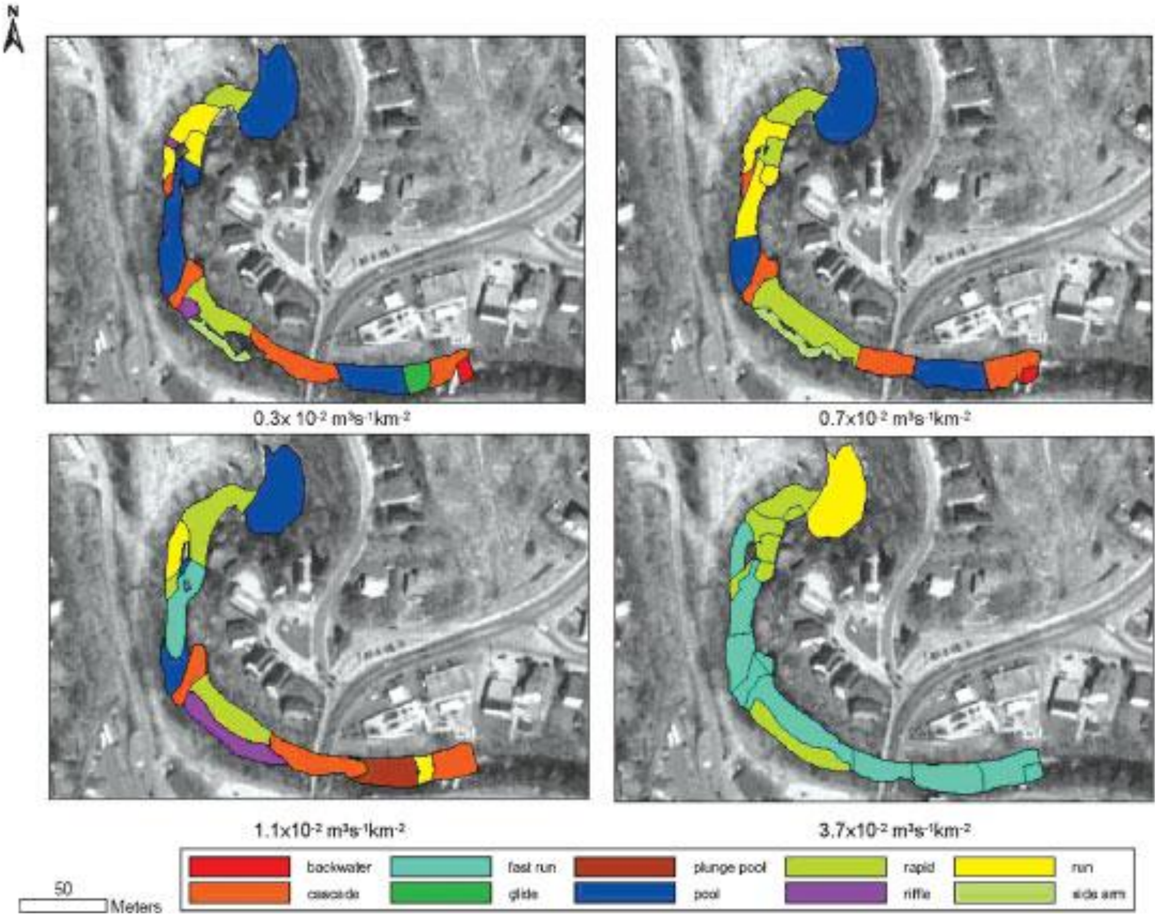

Figure 7. An example of the mesohabitat approach showing mapping results of different hydromorphological units at four different discharges in a study site in Quinebaug River, Connecticut, USA. Source: Parasiewicz (2008), Copyright (2012): Wiley, reproduced with permission.

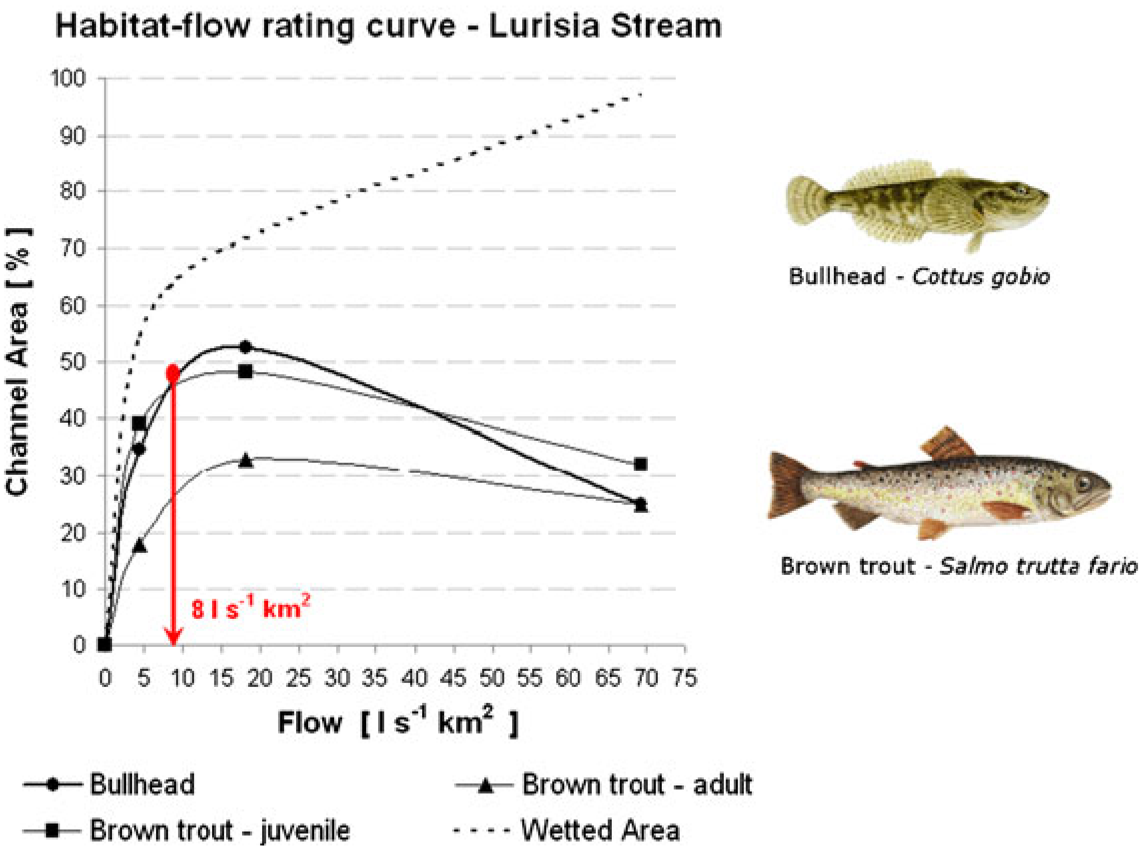

Figure 8. A chart showing flow-habitat relationship derived using a mesohabitat modelling (MesoHABSIM) approach. Source: Vezza et al. (2011), Copyright (2012): Wiley, reproduced with permission.

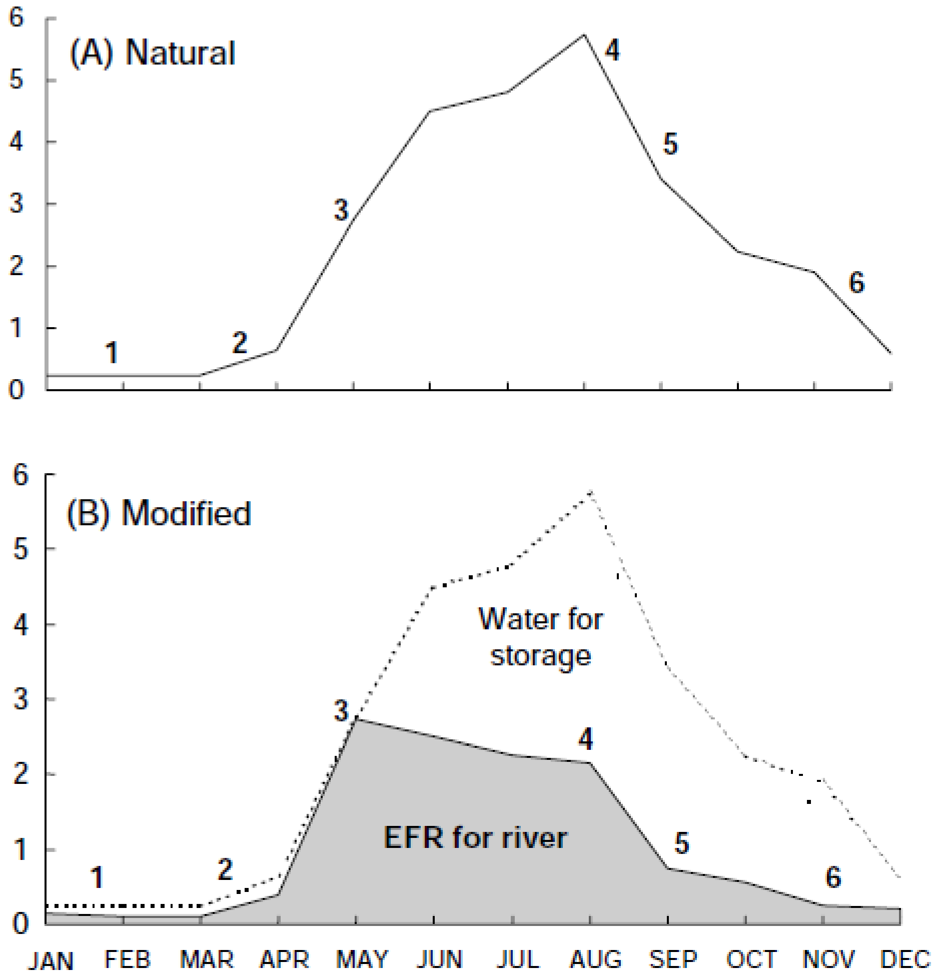

Figure 9. Conceptual model of the Building Block Methodology approach. Using a natural hydrograph, key components of the river flow are identified (A), and some proportion of these building blocks are retained to provide an environmental flow regime. For example, blocks 1 and 6 are used to retain perennial low flows of the hydrograph, blocks 2, 4 and 5 are used to separate wet and dry seasons, and block 3 is used to introduce a first larger flood of the wet season. Source: King et al. (2008), Copyright (2012): Water Research Commission, South-Africa, reproduced with permission.

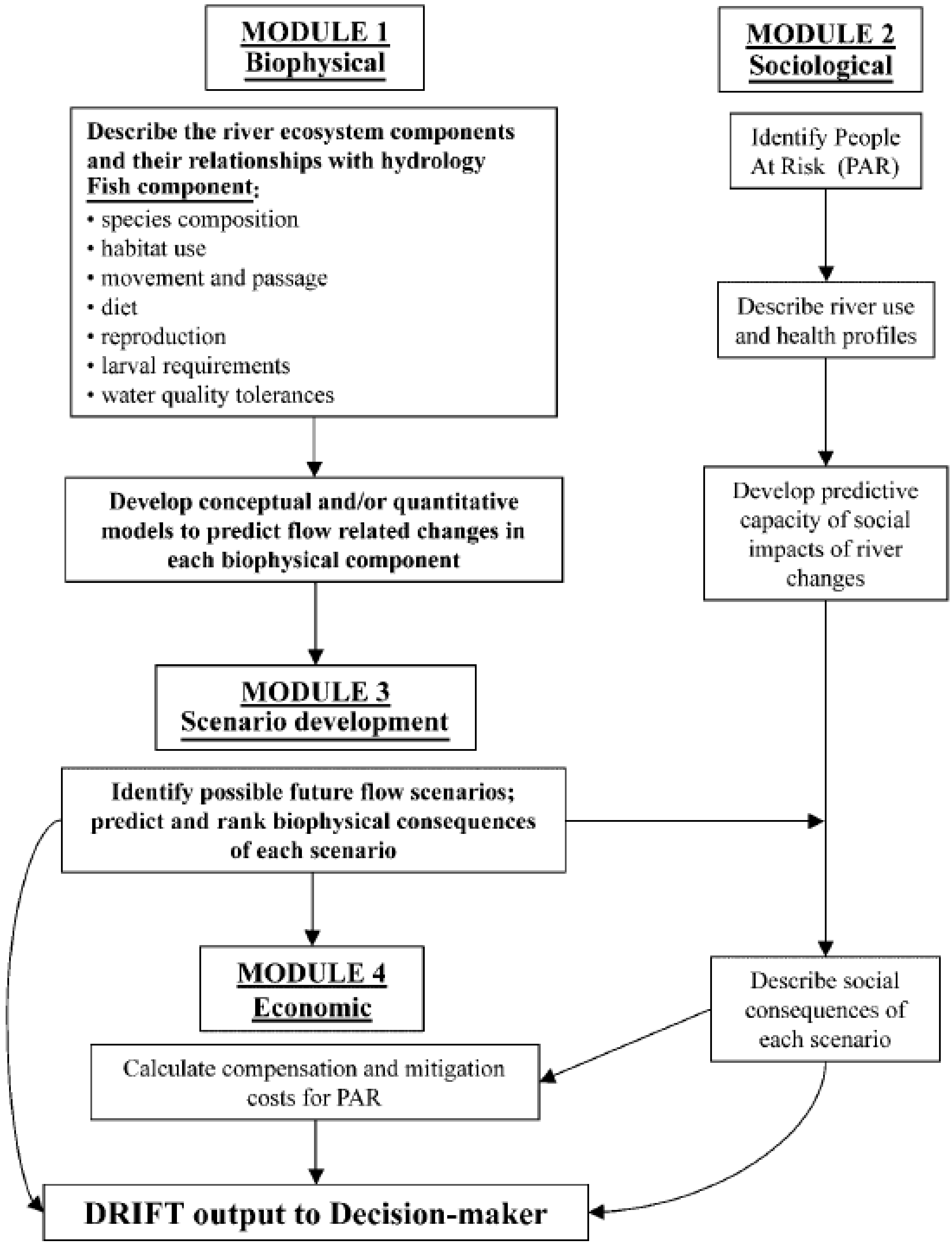

Figure 10. A schematic of the four modules of the DRIFT framework. The example shows the fish component of the Biophysical module; the specific sub-tasks within this module are different for each biophysical discipline which all are contained within the module. Source: Arthington et al. (2003), Copyright (2012): Wiley, reproduced with permission.

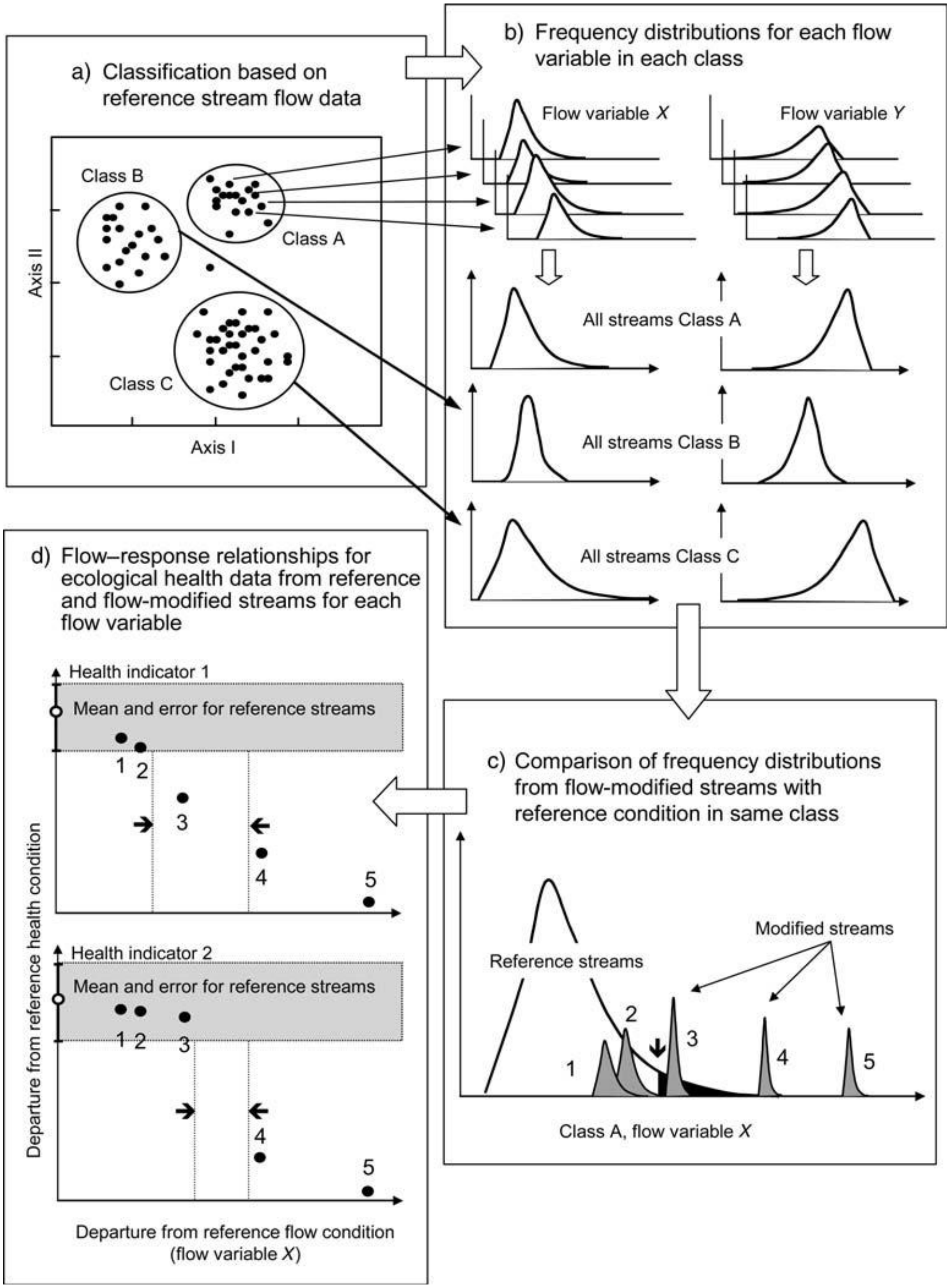

Figure 11. Conceptual model of the framework suggested by Arthington et al. (2006).Benchmark conditions (vertical lines in graph (d)) can be identified for each flow variable using a number of different river health indicators. See text for description of the different steps in the framework. Source: Arthington et al. (2006), Copyright (2012): Ecological Society of America, reproduced with permission.

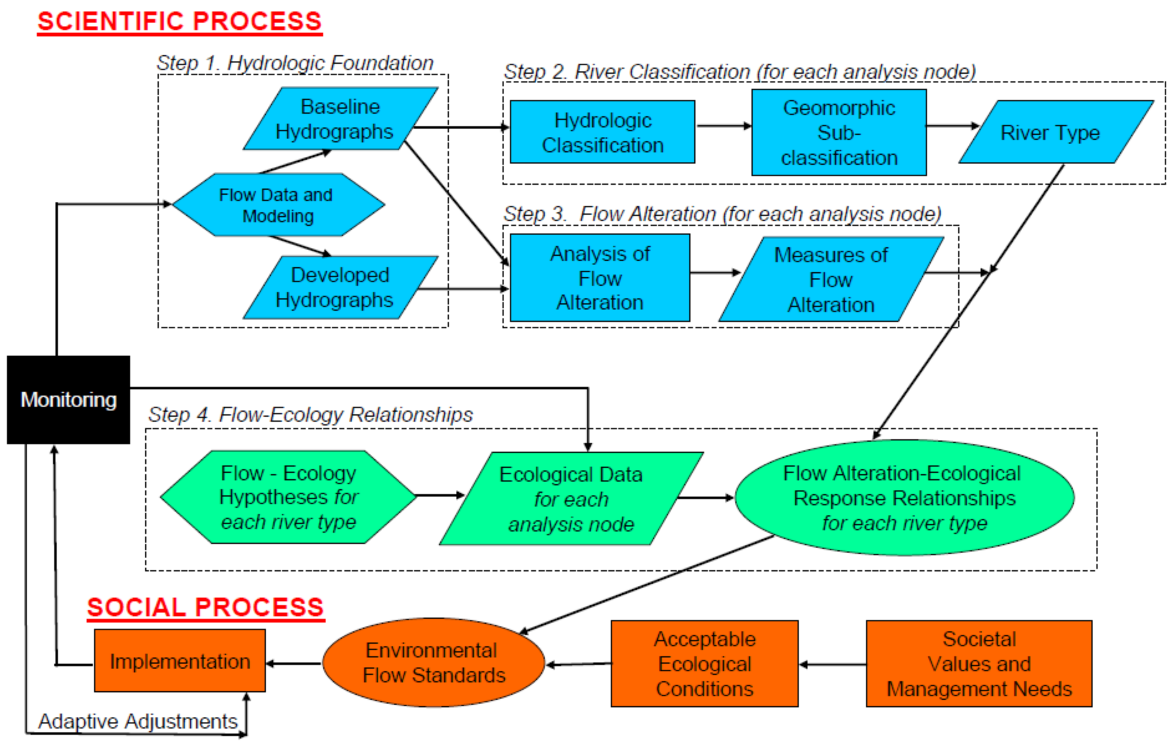

Figure 12. Conceptual figure of the ELOHA framework. See text for description of the different steps. Source: Poff et al. (2010), Copyright (2012): Wiley, reproduced with permission.

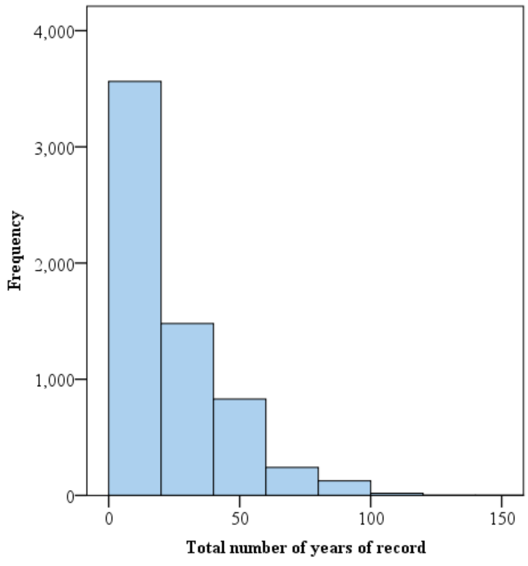

Figure 13: Histogram displaying the number of stations against total number of years of record for river gauging stations available through HYDAT.

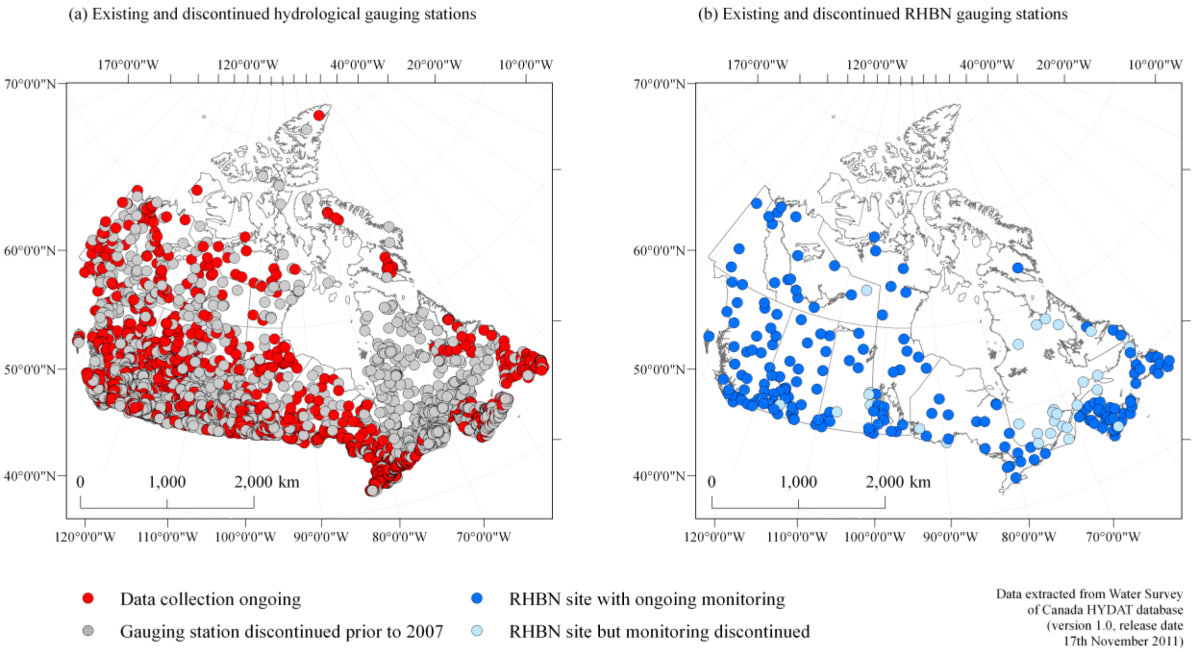

Figure 14: (a) Map of existing and discontinued hydrometric gauging stations (rivers only); (b) map of existing and discontinued RHBN stations.

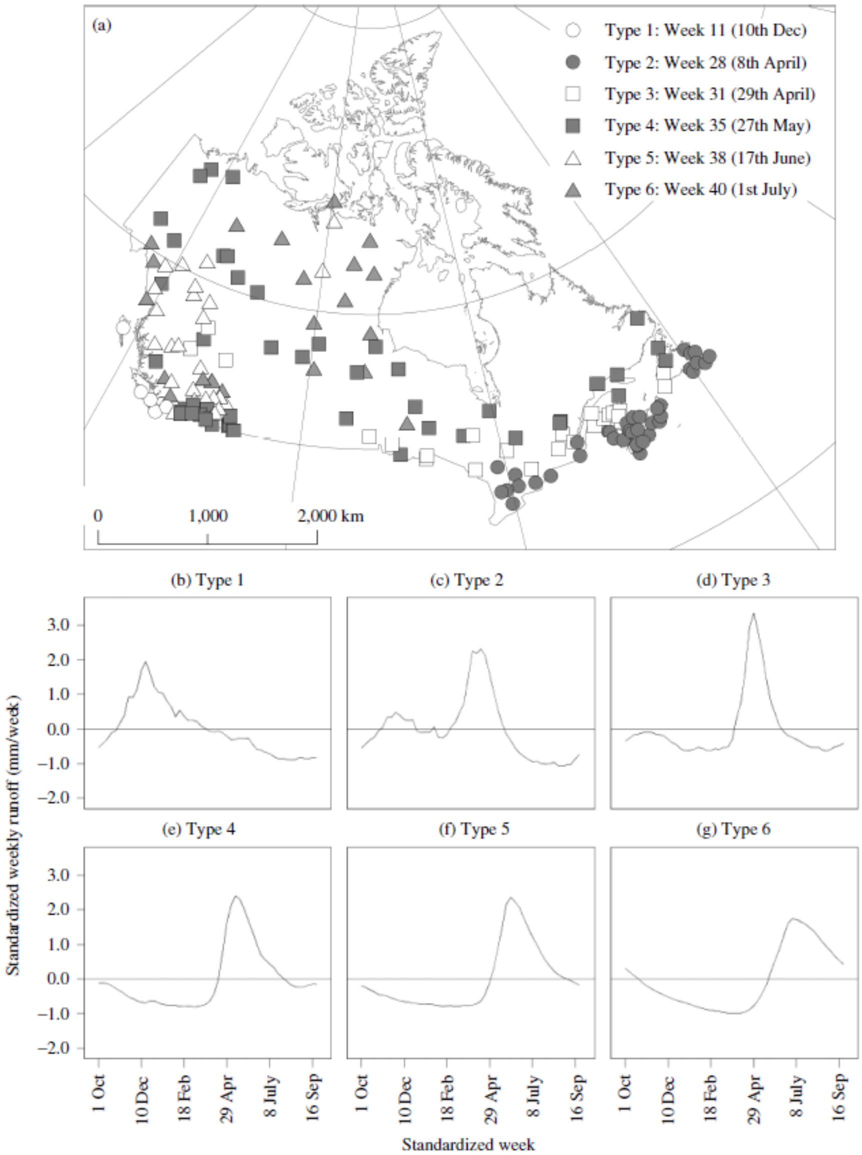

Figure 15: (a) Map of six identified hydrological regime classes in Canada; (b) standardised weekly runoff hydrographs for each of the six regime classes. Source: Monk et al. (2011), Copyright (2012): Wiley, reproduced with permission.

A)

B)

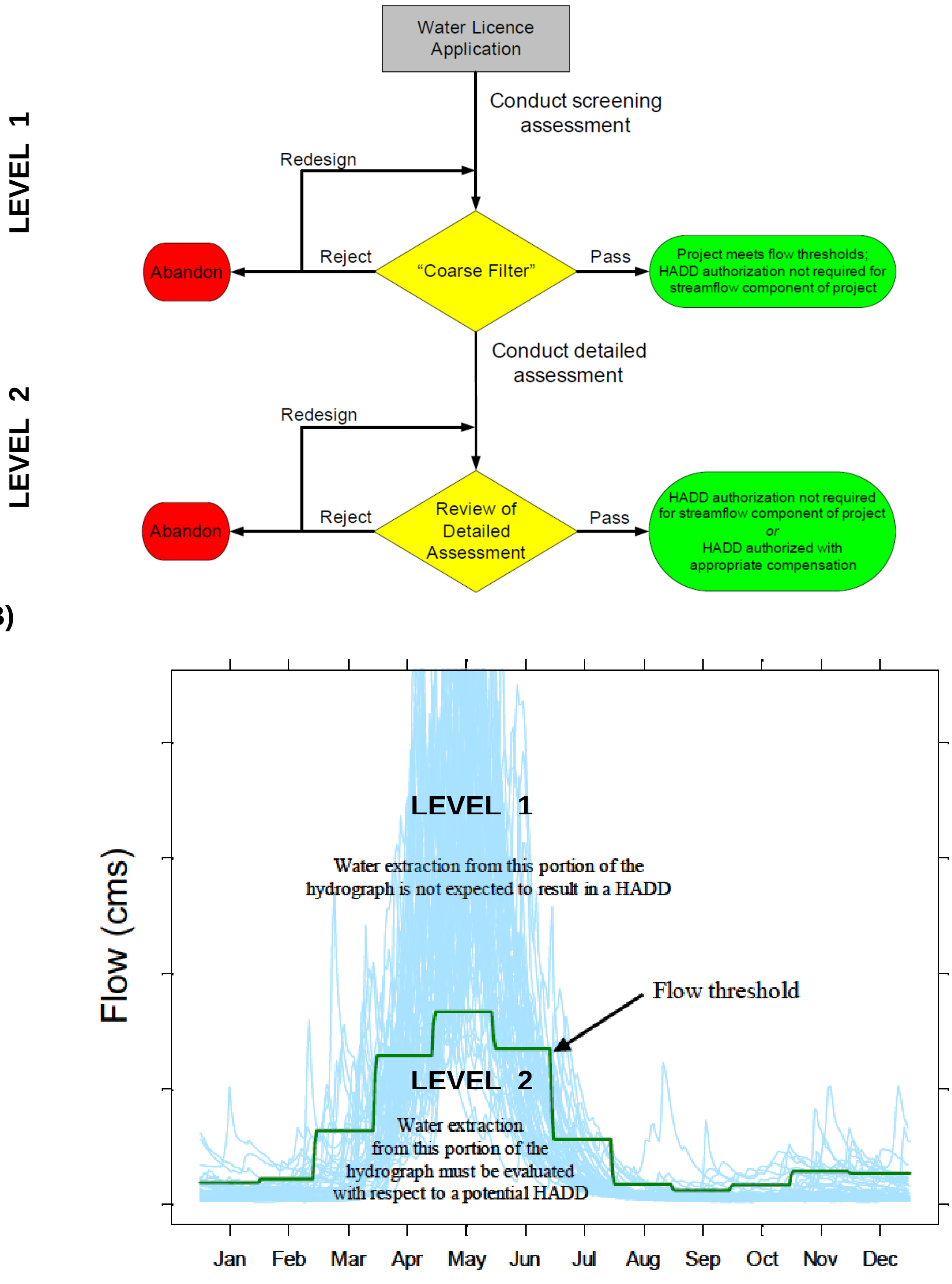

Figure 16. Conceptual model of a 2-tiered environmental flow framework. The Level 1 assessment serves as a coarse filter and identifies projects with "no HADD". Level 2 assessment requires case specific analysis due to potential HADD. Source: Hatfield et al. (2003), reproduced with permission.

Appendix A: List of Abbreviations

ABF:	Aquatic Base Flow
BBM:	Building Block Methodology
DFO:	(Department of) Fisheries and Oceans Canada
DRIFT:	Downstream Response to Imposed Flow Transformation
EFC:	Environmental Flow Components
ELOHA:	Ecological Limits of Hydrologic Alteration
HADD:	Harmful Alteration, Disruption or Destruction of fish habitat
HSI:	Habitat Suitability Index
HYDAT:	National hydrological database maintained by Water Survey of Canada
IFIM:	Instream Flow Incremental Methodology
IHA:	Indicators of Hydrologic Alteration
LAR:	Lower Athabasca River
MAF:	Mean Annual Flow
MMF:	Mean Monthly Flow
PHABSIM:	Physical HABitat SIMulation
POF:	Percentage Of (natural) Flow
Q_X:	Flow that is exceeded X% of the time
RHBN:	Reference Hydrometric Basin Network
RVA:	Range of Variability Approach
SBA:	Sustainability Boundary Approach
WSC:	Water Survey of Canada
WUA:	Wetted Usable Area

Appendix B. The current method of establishing minimum or environmental flows in some European countries.

Table is originally published as the Annex II, p. 68-70 in Kampa et al. (2011). The attached table is a summary of a longer questionnaire directed to the government officials in each country; the full response to questionnaire of each country is available at http://www.ecologic-events.de/hydropower2/background.htm. Abbreviations of the various countries are used in the Table are: AT, Austria; BE, Belgium; BG, Bulgaria; CH, Switzerland; CZ, Czech Republic; DE, Germany; ES, Spain; FI, Finland; FR, France; HU, Hungary; IS, Iceland; IT, Italy; LT, Lithuania; LU, Luxembourg; LV, Latvia; NL, The Netherlands; NO, Norway; PL, Poland; PT, Portugal; RO, Romania; SE, Sweden; SI, Slovenia; SK, Slovakia; UK, United Kingdom.

8.11 What method/approach is (are) applied to defined minimum ecological flow in your country?

	Static defin.	Dynamic def.	Modeling	Other methods	Explain methods	Comments
AT	x	x	x			Guide values for a "basic" minimum flow and additional "dynamic flow" (Ordinance on Ecological Status Assessment) or determination by modelling which proofs that good status for all biological elements is achieved - see Annex 2 of AT questionnaire
BE	x		x			Depends of the type of the watercourses (navigable or unnavigable)
BG	x					10% of annual mean flow
CH	x	x	x			within a catchment area the minimum's can be optimised e.g., one rive no water - the other more water.
CZ	x				According to value of Q355 category is chosen and after minumum residual flow (MRF) is calculated which is based on values Q330, Q355 and Q364. If flow Q355 is < 0,05 m3.s-1; MRF=Q330 0,05 - 0,5 m3.s-1; MRF=(Q330 + Q355)*0,5 0,51 - 5,0 m3.s-1; MRF=Q355 > 5,0 m3.s-1; MRF= (Q355+ Q364)*0,5	
DE	x	x	x			

ES					For relevant locations double studies are carried out, using hydrological and ecological (IFIM) data. According to the results obtained, the most adequate results are used, on a case by case basis	
FI		X	X		In some cases: fish habitat and other habitat modelling based on the relationship between flows, water depth, substrate and the quality and quantity of available habitats.	
FR	x	x	x		The lower limit is 10% or 5% (law) The adapted ecological minimum flow is generally based on "micro-habitats méthod, EVHA" (Souchon & al.), which can be completed or remplaced whith modeling or extrapolations. But there are other possible methods adapted when this one doesn't fit whith the type of river	It depends on the type of the river. This ecological minimum flow can be differently distributed over the year, in a compatible way between use and species interests
HU						
IS		x				
IT	x	x	x	Baseflow method and similar		Details on minimum ecological flow can be found in the regional water protection plans, included in the RBMPs.
LT	x					
LU	x	x			10% AMF or 30% MMF	
LV	x	x				Guaranteed rate of flow; summer 30 day period mean minimal flowrate with 95% coverage.
NL		x				
NO		x				Different methods are being assessed.Ongoing R & D. A specific technical requirement to set minimum flow is not recommended in Norway due to large variations in river basins and the purpose of setting minimum flow. However, hydrological indexes are commonly used (e.g Q95 summer/winter in small scale HP)
PL						

PT		x	x			INAG has develop a simple method (based on historical flows) than can be used when there is not many knowledge about an area. The IFIM method is is recommended
RO	x					
SE	x					Commonly static byt in some cases defined from fish migration
SI		x			Ecological acceptable flow considers hydrological baseline, type of water abstraction, hydrological, hydromorphological and biological characteristcs and information on protection regimes. Hydrological baseline considers value of mean minimum flow and mean flow at the location of water abstraction. Qes=f*sQnp (Qes - ecological acceptable flow, f-factor depend on ecological type of watercourse, sQnp - mean minimum flow) It is also possible to choose interdisciplinary holistic approach.	
SK	x				Q355 - Average daily water discharge during the reference period, achieved or exceeded during 355 days in the year	
UK	x	x	x		The national guidance is available at: http://www.wfduk.org/UKCLASSPUB/LibraryPublicDocs/gep_hmwb_final. This includes a list of good practice mitigation measures. In relation to flow, the list includes maintenance of a proportion of the flow that would have naturally been exceeded 95 % of the time. The proportion depends on the river type but is typically about 85 %. It also includes provision of variable higher flows, depending on the needs of the site-specific ecological characteristics. These flows are defined on a case-by-case basis.	The minimum flow requirements may differ from scheme to scheme depending on ecological needs (eg whether or not fish migration needs to be supported) and on what flow can be provided without a significant impact on electricity generation