



Fisheries and Oceans
Canada

Pêches et Océans
Canada

Ecosystems and
Oceans Science

Sciences des écosystèmes
et des océans

Canadian Science Advisory Secretariat (CSAS)

Research Document 2022/080

National Capital Region

Estimating Impacts and Offsets for the Death of Fish

Marten A. Koops¹, Cody J. Dey², Simon Fung¹, Sebastian Theis³, Tyler D. Tunney⁴ and Adam S. van der Lee¹

¹Great Lakes Laboratory for Fisheries and Aquatic Sciences, Fisheries and Oceans Canada, Burlington, ON

²Fisheries and Oceans Canada, Ottawa, ON

³Department of Renewable Resources, University of Alberta, Edmonton, AB

⁴Gulf Fisheries Centre, Fisheries and Oceans Canada, Moncton, NB

Foreword

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

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

[http://www.dfo-mpo.gc.ca/csas-sccs/
csas-sccs@dfo-mpo.gc.ca](http://www.dfo-mpo.gc.ca/csas-sccs/csas-sccs@dfo-mpo.gc.ca)



© His Majesty the King in Right of Canada, as represented by the Minister of the
Department of Fisheries and Oceans, 2022

ISSN 1919-5044

ISBN 978-0-660-46285-1 Cat. No. Fs70-5/2022-080E-PDF

Correct citation for this publication:

Koops, M.A., Dey, C.J., Fung, S., Theis, S., Tunney, T.D., and van der Lee, A.S. 2022.
Estimating impacts and offsets for the death of fish. DFO Can. Sci. Advis. Sec. Res. Doc.
2022/080. v + 94 p.

Aussi disponible en français :

Koops, M.A., Dey, C.J., Fung, S., Theis, S., Tunney, T.D., et van der Lee, A.S. 2022. Estimation des effets néfastes sur le poisson et des mesures de compensation pour la mort du poisson. Secr. can. des avis sci. du MPO. Doc. de rech. 2022/080. v + 106 p.

TABLE OF CONTENTS

ABSTRACT	v
1. INTRODUCTION	1
2. QUANTIFYING MORTALITY LOSSES AND OFFSET GAINS.....	6
2.1. COUNT.....	6
2.2. BIOMASS.....	6
2.3. EQUIVALENT AGES (EA)	6
2.4. REPRODUCTIVE POTENTIAL.....	7
2.5. PRODUCTION FORGONE	8
2.6. HABITAT PRODUCTIVITY INDEX (HPI).....	8
2.7. FORGONE FISHERIES YIELD	9
2.8. TOTAL BIOMASS LOST	9
2.9. POPULATION MODEL	9
2.10. EXAMPLE	11
2.11. RISK AND UNCERTAINTY IN EQUIVALENCY CALCULATIONS	15
3. ASSESSING COMMUNITY IMPACTS	20
3.1. QUALITATIVE NETWORK MODELS (QNM).....	20
3.2. BAYESIAN BELIEF NETWORKS (BBN)	22
3.3. INDIVIDUAL-BASED MODELS (IBM).....	22
3.4. SIZE-SPECTRA MODELS	23
3.5. DYNAMIC MULTISPECIES SIZE SPECTRUM MODELS	23
3.6. MINIMUM REALISTIC MODELS	24
3.7. WHOLE ECOSYSTEM MODELS	24
3.8. STRUCTURAL EQUATION MODELS	25
3.9. MULTI-SPECIES BIOENERGETICS MODELS	26
3.10. COMMUNITY MODEL EXAMPLE USING QUALITATIVE NETWORK MODEL	26
4. CONSEQUENCES OF FISH MORTALITY AND CONSIDERATIONS FOR DECISIONS RELATED TO THE DEATH OF FISH	40
4.1. AMOUNT OF FISH MORTALITY	40
4.2. POPULATION SIZE AND TRAJECTORY.....	40
4.3. SPATIAL STRUCTURE OF POPULATIONS.....	40
4.4. LIFE HISTORY, LIFE STAGE IMPACTED, AND DENSITY DEPENDENCE	41
4.5. INTERACTIONS WITH OTHER SOURCES OF MORTALITY	42
4.6. DURATION, TIMING, AND MECHANISM OF MORTALITY	42
4.7. BIOTIC INTERACTIONS AND ECOSYSTEM IMPACTS	43
4.8. MANAGEMENT OBJECTIVES	44
4.9. DOES THE EFFECT ON LOCAL FISH POPULATION OR COMMUNITIES CHANGE WITH RESPECT TO WHEN AND HOW FREQUENTLY FISH ARE KILLED?.....	45
4.10. WHAT IS THE IMPACT OF BEING WRONG ABOUT EITHER THE AMOUNT OF MORTALITY OR THE SENSITIVITY OF THE POPULATION OR ECOSYSTEM?.....	46

4.11. CAN A PRECAUTIONARY FRAMEWORK BE USED TO SUPPORT MANAGEMENT OF WUA RELATED RESIDUAL MORTALITY?	46
4.11.1. Summary	51
5. OFFSETTING MORTALITY	52
5.1. WHAT ARE CURRENT OFFSETTING PRACTICES FOR RESIDUAL MORTALITY? ..	52
5.1.1. Habitat creation	53
5.1.2. Habitat restoration	55
5.1.3. Biological and Chemical Manipulation	58
5.2. GENERAL MONITORING INDICATORS FOR OFFSETTING SUCCESS	61
5.3. OFFSETTING DISCUSSION	62
6. KEY UNCERTAINTIES	64
7. CONCLUSIONS	65
8. REFERENCES CITED	69
APPENDIX A. EQUIVALENCY CALCULATIONS	79
REFERENCES	80
TABLES	81
APPENDIX B. OFFSETTING SYSTEMATIC REVIEW	90

ABSTRACT

The Fish and Fish Habitat Protection Program (FFHPP) has requested science advice on the potential consequences and how to quantify impacts from works, undertakings or activities (WUAs), other than fishing, that cause the death of fish, and ways that the death of fish can be offset. To support this request, this document is organized into four main sections. First, the available metrics for quantifying impacts and offsets for fish mortality are reviewed. There are differences in the data requirements and equivalency objectives among metrics; total biomass lost is the recommended metric for most cases. Second, modelling approaches that could be used to evaluate the community or food web consequences of fish mortality are briefly reviewed. An example of simple community models is provided, demonstrating that the outcomes for fish communities from fish mortality events are likely to be more than the sum of single species responses. Third, an overview is provided of the considerations that should be included in management decisions about fish mortality, providing information on the components that could be included in a risk management framework for WUA-related residual mortality. An adaptation of a precautionary fisheries management framework is presented as an example of a risk management framework for decisions about fish mortality impacts and offsets. Finally, the literature on current practices for offsetting fish mortality events is reviewed. There is evidence that habitat creation, habitat restoration and enhancement, and biological and chemical manipulation can all provide approaches to offsetting fish mortality, but consideration of the specifics of the offset application and sufficient monitoring, including pre-impact assessment, are needed to effectively achieve an offset for fish mortality.

1. INTRODUCTION

Mortality is one of the most important parameters determining the dynamics and productivity of fish populations and fisheries. Fisheries science has a long history of studying the impacts of fishing mortality on productivity. Fundamentally, increasing mortality will reduce abundance. This can be simply expressed as:

$$N_t = N_0 e^{-Zt} \quad (1.1)$$

where N_t is abundance at a point in time (t), N_0 is the initial abundance, and Z is the total instantaneous mortality rate. The survival from one year to the next is expressed as:

$$S = e^{-Z} \quad (1.2)$$

From these equations, it is easily seen that an increase in mortality (or a decrease in survival) results in lower abundance. Reductions in population abundance have been shown to increase the vulnerability of populations to local extinction with perturbations. This increased vulnerability applies to both selective perturbations such as fishing for larger individuals (e.g., Anderson et al. 2008) and non-selective perturbation such as catastrophes (e.g., Vélez-Espino and Koops 2012). However, this does not mean that all species are equally sensitive to mortality. Instead, population sensitivity to mortality is dependent on a number of traits such as life history (e.g., Vélez-Espino et al. 2006) or body size (e.g., Pope et al. 2006, van der Lee and Koops 2016).

Working with instantaneous mortality rates, it is common to simply parse total mortality (Z) into components, such that:

$$Z = M + F + A \quad (1.3)$$

where M is natural mortality, F is fishing mortality, and A is other anthropogenic mortality. Ultimately, increasing mortality from anthropogenic sources will reduce the ability of a population to sustain pressures from natural and (or) fishing sources of mortality.

Natural factors that result in fish mortality include interactions with other organisms including disease, pathogens, parasites, and predators, or lack of prey. Environmental conditions, or changes in conditions, that exceed physiological tolerances can also result in mortality. Even when environmental conditions do not directly lead to mortality, they can have sub-lethal effects that reduce the capacity for fish to withstand other stressors (e.g., reduced swimming performance). Potential environmental conditions that stress or kill fish include temperature (low, high, or large changes), turbidity, hypoxia, and salinity. Changes in these environmental conditions can be driven by weather events (e.g., cold fronts, heat waves, storms, etc.), winter ice cover, and algal blooms (e.g., harmful algal blooms or HABs). Mortality can also occur from stress associated with completing life processes (e.g., post-migration or post-spawning stress) and from old age. While these causes of mortality occur naturally, they can also be driven by anthropogenic activities.

While there are both natural and anthropogenic causes of mortality, many works, undertakings, or activities (WUAs) that take place in or near water have the potential to directly or indirectly increase fish mortality. A survey of recent cases involving fish mortality managed by the Fish and Fish Habitat Protection Program (FFHPP) across multiple regions revealed a few characteristics that are relevant to considerations for the management of such WUAs. First, most WUAs could be organized into a set of seven categories (Table 1.1) which cover most but not all the pathways of effects that can result in fish mortality (Brownscombe and Smokorowski 2021). The causes of mortality often include impingement and entrainment, physical trauma up to and including severe lacerations, accelerative and shear forces, barotrauma, rupture of swim bladder and (or) internal organs, stranding and (or) exposure of fish (including eggs), or

increased predation. Second, depending on the WUA, the frequency of mortality events can range from discrete or isolated instances to occasional (either regularly or unpredictably), annual, seasonal, monthly, or continuous (daily). Third, in all cases identified, mortality affected multiple species in the fish community; there were no cases identified where only a single species was affected. Fourth, some cases had the potential to affect species at risk (SAR). Potentially affected SAR were both marine and freshwater, small and large bodied species. Examples include American Eel (*Anguilla rostrata*), Atlantic Salmon (*Salmo salar*), Lake Chubsucker (*Erimyzon sucetta*), Lake Sturgeon (*Acipenser fulvescens*), Redside Dace (*Clinostomus elongatus*), and whales. Finally, very few fish mortality cases have been offset to date, so there are limited examples of offsets accepted for fish mortality. Those that have been accepted include additional monitoring to address assessment gaps, fish stocking, habitat creation (specifically coastal wetlands or spawning shoals), and improved downstream fish passage.

The *Fisheries Act* prohibits the killing of fish other than by fishing:

s.34.4 (1) No person shall carry on any work, undertaking or activity, other than fishing, that results in the death of fish.

The *Species at Risk Act* further prohibits the killing of individuals of listed species:

s.32 (1) No person shall kill, harm, harass, capture or take an individual of a wildlife species that is listed as an extirpated species, an endangered species or a threatened species.

Further, for listed species, the *Species at Risk Act* requires that issuing a permit or authorizing an activity that affects a listed species is only possible if the activity will not jeopardize the survival or recovery of the species (s.73(3c)).

While the *Fisheries Act* prohibits the death of fish by means other than fishing, a distinction will be made in this document between the death of fish and mortality. Specifically, most of this document will refer to mortality, which is a probabilistic population process that can occur by natural or anthropogenic causes. Death of fish will be considered as the outcome of mortality events or processes, and this terminology will be used specifically when referring to the *Fisheries Act* and management decisions. Furthermore, unless species are identified, general references to fish should be interpreted *as per* the *Fisheries Act* (s. 2(1)) which includes all life stages of fish, shellfish, crustaceans, and marine animals.

The successful protection and conservation of fish and fish habitat requires managing WUAs that affect fish mortality. To achieve this, FFHPP needs science advice on the available approaches to quantifying impacts from and offsets for fish mortality, the factors that determine the consequences of fish mortality, and the options for offsetting fish mortality. This document aims to provide information to support answers for the following questions:

1. What approaches can be used to quantify the impacts of WUA-related residual mortality, and associated offsetting requirements?
 - a. What are the advantages and limitations of the different approaches?
2. What determines local fish population or community responses to WUA-related residual mortality?
 - a. Does the effect on local fish populations or communities change with respect to when and how frequently fish are killed?
 - b. What criteria should be considered when quantifying or describing impacts from WUA-related residual mortality?

-
3. What are the current domestic and international practices for offsetting the effects of WUA-related residual mortality?
 - a. What are the options for offsetting WUA-related residual mortality?
 - b. What is the effectiveness of the available offsetting options?
 - c. What are the rationales for selecting certain offsetting options?

The information provided in this document assumes that the avoidance and mitigation steps of the mitigation hierarchy (DFO 2019a, 2019b) have been applied and that it is the residual mortality that must be managed. While the expectation is that the information provided will most often be used during the authorization process, some of the material covered will be relevant to managing accidents and monitoring and evaluating the effectiveness of offsets for fish mortality impacts. This document will consider direct (e.g., caused by the WUA and can happen immediately or following a delay) and indirect (i.e., the WUA pre-disposes the fish to another mortality source such as increased predation) mortality as part of the residual mortality to be managed. For the rest of the document, the term fish mortality will be used to refer to WUA-related residual mortality. Sub-lethal effects (reduced capacity or performance in traits related to the fitness of the individual) can result from many activities that also result in fish mortality. However, the issue of sub-lethal effects requires more attention than can be provided here, and while identified as an important issue, will need to be dealt with separately.

Table 1.1. A summary of the types of activities from a survey of recent cases managed by FFHPP that can result in fish mortality, as well as examples of specific activities, causes of mortality, and the typical frequencies of mortality events considered.

Type of activity	Examples of activities	Possible mortality causes	Frequencies
Water intakes	<ul style="list-style-type: none"> • hydroelectric stations • thermal generating stations • nuclear generating stations • marine water intakes 	<ul style="list-style-type: none"> • impingement • entrainment • supersaturation • physical trauma (e.g., turbine strikes) • barotrauma • accelerative and shear forces 	<ul style="list-style-type: none"> • continuous
Changing water levels or water diversions	<ul style="list-style-type: none"> • reservoir operations • reservoir maintenance (or construction) • water diversions • canal operations and drawdowns • invasive plant control 	<ul style="list-style-type: none"> • stranding • entrainment • inability to complete life cycle • reduced habitat capacity 	<ul style="list-style-type: none"> • continuous • seasonal • annual • occasional
Removal, addition, and (or) relocation of benthic materials	<ul style="list-style-type: none"> • dredging • infilling • sedimentation 	<ul style="list-style-type: none"> • physical trauma • stranding • exposure • burying and suffocation 	<ul style="list-style-type: none"> • occasional
Barriers to passage	<ul style="list-style-type: none"> • dams (with or without fishways) • tidal barriers • reduced flows 	<ul style="list-style-type: none"> • inability to complete life cycle • supersaturation • increased predation 	<ul style="list-style-type: none"> • continuous • occasional

Type of activity	Examples of activities	Possible mortality causes	Frequencies
Explosives	<ul style="list-style-type: none"> • construction barriers (e.g., coffer dams) • underwater explosives 	<ul style="list-style-type: none"> • physical trauma • rupture of swim bladders or internal organs 	<ul style="list-style-type: none"> • occasional • discrete
Shipping	<ul style="list-style-type: none"> • ship operation 	<ul style="list-style-type: none"> • physical trauma (e.g., ship strikes) 	<ul style="list-style-type: none"> • seasonal • occasional • discrete
Deleterious substances	<ul style="list-style-type: none"> • mining effluents • aquaculture operations • toxic or caustic substances dumped down storm sewers 	<ul style="list-style-type: none"> • toxicity • deformities • organ damage • tumours or cancers 	<ul style="list-style-type: none"> • continuous • seasonal • occasional • discrete

2. QUANTIFYING MORTALITY LOSSES AND OFFSET GAINS

Fish mortality that results from WUA typically impact multiple species and life-stages simultaneously. As a result, the loss of one individual fish (e.g., a larva) is not equivalent to the loss of another (e.g., a reproductive adult). An additional complication is that the species and life-stages produced through an offset for the fish mortality may not be the same as those impacted (i.e., an “out-of-kind” offset). It is therefore necessary to quantify the losses from the mortality event in units that value the affected life-stages appropriately and allow for a direct comparison to the implemented offset. A variety of metrics have been used to quantify fish mortality events. These metrics attempt to provide a “common currency” that equates losses across species and life-stages and allows for direct comparison between the mortality and offset. The metrics differ in which population characteristic is used to equate the value of the losses to the gains from the proposed offset. The population characteristics, and therefore equivalency metric, selected should reflect specific management goals. Generally, when multiple species are affected, the metrics are applied to individual species and summed. Various metrics are summarized with the benefits and limitations of each highlighted (Table 2.1).

2.1. COUNT

The simplest method to quantify losses is to count the number of dead fish. Count alone, however, has limited applicability when multiple age-classes and species are impacted by a project since it would consider all individuals killed to be equivalent. As well, it is difficult to link the benefits of an offset directly to counts of mortality. In most scenarios, count should not be used as a quantification method for fish mortality.

2.2. BIOMASS

Quantifying the loss from a mortality event as the biomass lost, B , is an alternative to simple counts. With biomass, heavier, older fish will have more of an impact on the total value of the loss than young fish. Biomass has also been shown to correlate well to productivity (Randall et al. 1995) suggesting that it may be a good metric to measure impacts. Biomass alone, however, quantifies the immediate, direct losses but does not account for the future production of the fish that were killed. It may also be difficult to directly evaluate an offset with use of total biomass. For example, is 200 kg of stocked age-1 fish equivalent to 200 kg of larvae entrained at a power generating station water intake? These are likely not equivalent and other aspects, such as future production, should be taken into consideration.

2.3. EQUIVALENT AGES (EA)

Equivalent ages (EA, Horst 1975) translates losses across multiple age-classes into a single “age-of-equivalence” (AOE). If the AOE is younger than the affected fish then the EA value is the number of individuals at the AOE required to produce the equivalent number of harmed fish. If the AOE is older than the affected fish then the EA value represents the number of individuals at the AOE that would have resulted from the number of harmed fish. The final equivalency estimate will depend on the selected AOE and therefore it is important that the selected AOE be relevant to the impact assessment goals. An AOE of age-1 has often been invoked in EA analysis (i.e., age-1 equivalents; EA-1). Age-1 is convenient as it allows for a more rapid assessment of the effectiveness of an offset relative to choosing an older age-class.

In order to calculate EA there must be an accounting of the number of fish deaths at the age-class level. In addition, the survival rate between the AOE and affected age-classes must be known. EA at age, t , is calculated from:

$$EA_t = \begin{cases} N_t s_{t \rightarrow AOE} & \text{if } t < AOE \\ N_t \frac{1}{s_{AOE \rightarrow t}} & \text{if } t \geq AOE \end{cases} \quad (2.1)$$

where N_t is the age-specific number of fish deaths, $s_{t \rightarrow AOE}$ and $s_{AOE \rightarrow t}$ is the cumulative survival from age- t to AOE or from AOE to age- t depending on if age- t is $<$ or $>$ the AOE. For example,

$$s_{AOE \rightarrow t} = \prod_{i=AOE}^t s_i, \quad (2.2)$$

where s_i is the survival rate between successive age classes. The total EA is calculated by summing age-specific estimates:

$$EA_T = \sum_{i=t_{min}}^n EA_i. \quad (2.3)$$

The survival rates incorporated in the estimate should reflect the time of year when mortality occurred in order to correctly back-calculate to the number of age-AOE fish.

EA is a convenient metric for quantifying fish deaths in a “common currency” to equate fish number of different age classes to a single value. It is also straightforward to apply an EA valuation to some offset types, particularly stocking. It is less straightforward to assign an EA value when applying a habitat creation/alteration offset. Any offset based on EA will conserve the age structure of the fish mortality, after enough time has passed.

EA, however, has been criticized for over-valuing older fish and not properly accounting for the value of an offset (Barnhouse et al. 2019). When a fish older than the AOE is killed the EA value will be > 1 and this will increase with greater time between AOE and t . Any production generated by the EA fish between AOE and t is not accounted for by the metric. For example, if a 15 year old fish is killed, and average annual survival is 70%, then ~ 210 age-1 fish will be required to replace that single age-15 fish. The 210 age-1 fish required in the offset will then live out their natural life cycle contributing to the ecosystem with 1 fish surviving into their 15th year. The production generate by these fish is not credited to the proponent in the offset. Instead, only the single fish that lives into its 15th year counts towards offsetting the harm. This undervalues the offset.

2.4. REPRODUCTIVE POTENTIAL

The impact of losses from fish mortality can be evaluated by quantifying the forgone reproductive potential of the killed fish. One measure of forgone reproductive potential is eggs-per-recruit (EPR; Goodyear 1988, Boreman 1997) where:

$$EPR_t = \sum_{i=t}^n f_i MO_i R_i \prod_{j=0}^{n-1} s_j. \quad (2.4)$$

f_i is the mean age-specific fecundity, MO_i is the proportion mature at age- i , R_i is the proportion that are female, s_j is the survival rate between ages and n is the number of age classes in the population. Total reproductive potential (RP) loss from fish mortality is then:

$$RP = N_t EPR_t, \quad (2.5)$$

where N_t is the number of mortalities of each age-class for a species. RP quantifies fish mortality as the number of eggs lost and an offset would need to be quantified in the same units. Barnhouse et al. (2019) suggested using EPR as a method for quantifying EA-1 by using the ratio of EPR at age- t to EPR at age 1:

$$EA1_{RP} = \sum_{i=t}^n N_i \frac{EPR_i}{EPR_1}. \quad (2.6)$$

When $t < AOE$ Equation 2.6 reduces to the standard EA equation (Equation 2.1). This formulation of EA ensures an equivalency in egg production between the offset and the fish mortality rather than an equivalency in age structure. $EA-1_{RP}$ will always be $<$ standard EA-1 as the ratio $\frac{EPR_i}{EPR_1} < \frac{1}{s_{1 \rightarrow t}}$. With $EA-1_{RP}$, however, the same drawbacks as standard EA-1 remain. In addition, calculating RP requires age-specific fecundity values which may not be available for some species.

2.5. PRODUCTION FORGONE

Production forgone is a measure of the biomass that would have resulted from the survival and growth of the fish lost from the fish mortality (Rago 1984). Production forgone is estimated as the product of the mean biomass of an age-class or stage, b_t , and the instantaneous growth rate, G_t , of that age or stage:

$$PF_t = \sum_{i=t}^n G_i \bar{b}_i = \sum_{i=t}^n \frac{G_i b_i (e^{G_i - M_i} - 1)}{G_i - M_i}. \quad (2.7)$$

Where G_t is estimated from mean weights at the beginning and end of a stage or age:

$$G_t = \ln \left(\frac{W_{t+1}}{W_t} \right), \quad (2.8)$$

M_t is the instantaneous mortality rate, and b_i is the expected mean biomass at an age or stage based on the initial number harmed (N_t) and the survival schedule (s_j):

$$b_i = N_t W_t \prod_{j=t}^i s_j. \quad (2.9)$$

Equation (2.7) is the estimate of production forgone for a single stage or age-class. Total production forgone that results from fish mortality is the summation of PF_t across stages/age-classes:

$$PF_T = \sum_{i=t_{min}}^n PF_i. \quad (2.10)$$

Production forgone provides an estimate of all the future production lost to an ecosystem. The same unit of measure can be applied to potential offsets. The expected lifetime production from stocking can be estimated in the same manner as production forgone making the harm and offset directly comparable. The value of a habitat offset can be measured in units of production per year which can be expected to match a lifetime or annual estimate of production forgone depending on how potential surplus production of new habitat is expected to affect the population. Production forgone, however, does not account for the biomass of the killed fish directly, only the future production of the fish, nor does it account for the lost reproductive potential.

2.6. HABITAT PRODUCTIVITY INDEX (HPI)

The habitat productivity index is a surrogate of fish production used to estimate the expected productivity of a unit of habitat per year (Randall and Minns 2002). HPI is calculated as the product of species-specific biomass, B_s , and an estimate of species-specific production, P/B_s :

$$HPI = B_s \frac{P}{B_s}. \quad (2.11)$$

P/B_s is an estimate of species-specific production and relates to the potential rate of change in biomass expected annually. P/B_s estimates can be measured directly or estimated from an allometry, such as for Canadian freshwater fishes (Randall and Minns 2000):

$$\frac{P}{B} = 2.64W^{-0.35}, \quad (2.12)$$

where W is the mean weight-at-maturity of the species. It has been suggested that HPI could be applied to fish mortality as a measure of forgone production when growth and survival rates are unknown by calculating HPI and summing across species. This estimate would equate to a one-year estimate of production forgone, not lifetime production forgone which is more typical. It is not clear how well HPI approximates annual production forgone and it is therefore a less preferred method. Furthermore, when considering a habitat offset, the function of the offset needs to be demonstrated; whereas the HPI is typically applied using an estimate of P/B from an allometric relationship, which would assume rather than demonstrate that the habitat offset functions as natural habitat.

2.7. FORGONE FISHERIES YIELD

Forgone fisheries yield (FY; Dey 2002) provides an estimate of the harm to the ecosystem solely based on losses to commercial and recreational fisheries. This analysis has been employed to assess the losses of fish mortality on an economic basis rather than the ecological significance of the event. Forgone fisheries yield for species that are the target of a commercial or recreational fishery is estimated as:

$$FY_t = \sum_{i=t}^{t_{max}} \frac{N_i W_i V_i S_i F_i}{Z_i}, \quad (2.13)$$

N_i is losses at each age; V_i is the vulnerability to fishing gear; F_i is instantaneous fishing mortality; Z_i is the instantaneous total mortality rate for age- i ; S_i is the total mortality rate for age- i , and W_i is the average weight at age- i . Harm to non-target species are accounted for by multiplying their production forgone PF_T (Equation 2.10), and biomass, B , by an estimate of trophic transfer efficiency, TTC , and fishing mortality of the target species:

$$FY = (PF_T + B) \times TTC \times F. \quad (2.14)$$

As this metric only accounts for losses to commercial or recreation fisheries it is not relevant to the *Fisheries Act* where the death of fish prohibition applies to all species. As such, forgone fisheries yield will not be considered further.

2.8. TOTAL BIOMASS LOST

Total biomass lost is the summation of biomass of the fish mortality, B , and production forgone, PF_T (Equation 2.10), and provides a total estimate of the biomass lost to the ecosystem from fish mortality of all species impacted:

$$TB = PF_T + B. \quad (2.15)$$

Total biomass lost provides the most complete estimate of loss to an ecosystem from a mortality event accounting for both the current and future losses.

2.9. POPULATION MODEL

With more detailed data on the life-history and population structure of the affected species populations models such as matrix population models (Caswell 2001) or individual-based models (IBM; Rose et al. 1996) can be constructed to quantify fish mortality. Matrix models

model age or stage groups on annual time steps. The impact of mortality events can be assessed with a sensitivity analysis where expected changes in population growth rate can be determined based on a change in mortality rate. Gibeau et al. (2020) used a matrix population for Coho Salmon (*Oncorhynchus kisutch*) to quantify the size of an offset necessary to compensate for mortality rates of 2-20% per year. When the impact of fish mortality is assessed as a rate, and population size is unknown, population models may represent useful tools for quantifying harm. IBMs follow individuals over short time steps and may be able to provide insight into potential sub-lethal effects of various forms of anthropogenic harm.

The major drawbacks of population modelling are the greater data requirements, greater time requirement to put together the models, and greater technical skill requirement to implement them; however, where populations models already exist, such as species at risk or fisheries species, use of population models may be more straightforward.

Table 2.1. Pros and cons of various common currency metric for quantifying fish mortality.

Metric	Pros	Cons
Counts	<ul style="list-style-type: none"> • Simple • Equivalency: number of fish 	<ul style="list-style-type: none"> • Does not account of 'value' of different life stages • Does not account for future loss of production
Biomass	<ul style="list-style-type: none"> • Simple • Related to production • Equivalency: standing stock 	<ul style="list-style-type: none"> • Does not account for future loss of production • Difficult to relate to offset
Equivalent Ages	<ul style="list-style-type: none"> • Equate losses of different age-class • Can be measured as counts or biomass • Easily compared to stocking offset • Equivalency: age structure 	<ul style="list-style-type: none"> • Does not credit proponent with future production when fish age > age-of-equivalence • Difficult to compare to a habitat creation offset
Reproductive Potential	<ul style="list-style-type: none"> • Can be converted to equivalent age-1 (EA-1) • Equivalency: egg production 	<ul style="list-style-type: none"> • Can be difficult to measure offset as egg production • Requires fecundity values • Does not credit proponent with future production when fish age > age-of-equivalence • Difficult to compare to a habitat creation offset
Production Forgone	<ul style="list-style-type: none"> • Can be compared to habitat or stocking offset • Credits proponent with future production of offset • Equivalency: lifetime biomass production 	<ul style="list-style-type: none"> • Does not account for direct loss of biomass from mortality • Does not account for lost reproductive production
Habitat Productivity Index	<ul style="list-style-type: none"> • Requires little species-specific life-history data • Equivalency: annual biomass productions 	<ul style="list-style-type: none"> • P/B often not known and may require use of an allometric estimator • May not provide accurate species-specific estimates

Metric	Pros	Cons
Total biomass lost	<ul style="list-style-type: none"> Accounts for direct loss in biomass and future production forgone Equivalency: Standing stock and lifetime biomass productions (current and future production) 	<ul style="list-style-type: none"> Does not account for lost reproductive production
Population models	<ul style="list-style-type: none"> Can be used to estimate long-term impacts of harm Model exist for many species 	<ul style="list-style-type: none"> Requires detailed species-specific life-history data Requires estimates of harm as rates More difficult and time-consuming to develop

2.10. EXAMPLE

An example of an annual impingement and entrainment mortality event through water intake at a power plant is presented to demonstrate the equivalency calculations and compare and contrast results of the available metrics (Table 2.2). The example is hypothetical and does not reflect real data from any specific power plant nor do the parameters used in the calculations reflect recommended values for the species considered.

Table 2.2. Fish mortality example. Represents hypothetical impingement and entrainment mortality at a power plant water intake. Three species are impacted by the mortality with individuals counted, weighed, and assigned to life-stage.

	Count				Biomass (kg)			
	YOY	Juvenile	Adult	Total	YOY	Juvenile	Adult	Total
Alewife	187,500	50,000	12,500	250,000	30	635	535	1,200
White Sucker	1,250	875	375	2,500	0.3	50	180	230
Gizzard Shad	22,500	6,000	1,500	30,000	65	1,000	635	1,700

The example represents total annual mortalities of three species: Alewife (*Alosa pseudoharengus*), White Sucker (*Catostomus commersonii*), and Gizzard Shad (*Dorosoma cepedianum*). The fish impact by the fish mortality are counted, weighed, and assigned to life-stage: young-of-the-year (YOY), juvenile, or adult.

The various equivalency metrics have different data requirements (Table 2.3). Most equivalency metrics require that losses are assigned to individual age-classes within species. Ideally, this would be done by direct aging of the fish, however, this would be the most labour intensive and costly. Alternatively, length-age-keys could be used to assign ages to fish if lengths are taken. Lacking length-age-keys, growth curves, such as the von Bertalanffy growth curve (VBGF), could be used to find the expected age from fish lengths. If no information on age is available, the stable-age-distribution from a life table can be used to assign mortalities to ages. This method assumes that the mortalities occur in proportion to the relative number of fish of each age-class expected in the environment. This assumption can be modified if certain life-stages are expected to be disproportionately susceptible to the mortality event.

Table 2.3. Data requirements for the various common currency metrics.

Metric	Mortality data	Life history data
Counts	<ul style="list-style-type: none"> • Total counts 	<ul style="list-style-type: none"> • NA
Biomass	<ul style="list-style-type: none"> • Total biomass 	<ul style="list-style-type: none"> • NA
Equivalent Ages	<ul style="list-style-type: none"> • Species and age-specific counts 	<ul style="list-style-type: none"> • Species and age-specific survival rate
Reproductive Potential	<ul style="list-style-type: none"> • Species and age-specific counts 	<ul style="list-style-type: none"> • Species and age-specific survival rate • Species and age-specific fecundity
Production Forgone	<ul style="list-style-type: none"> • Species and age-specific counts 	<ul style="list-style-type: none"> • Species and age-specific survival rate • Species and age-specific growth rate
Habitat Productivity Index	<ul style="list-style-type: none"> • Species-specific biomass 	<ul style="list-style-type: none"> • Species-specific P/B
Total biomass lost	<ul style="list-style-type: none"> • Species and age-specific counts • Species-specific biomass 	<ul style="list-style-type: none"> • Species and age-specific survival rate • Species and age-specific growth rate

Additional life-history data may be required for estimation of the equivalency metrics. The majority of metrics will require knowledge of age-specific mortality rates to estimate cumulative survival rates among ages or stages. Depending on data-availability, parameters may be estimated from different data sources, but there is a hierarchy for sourcing data or estimates based on the availability of species- and ecosystem-specific the data sources. Ideally, survival rates would be estimated directly from the affected populations from field surveys prior to impact. In many circumstances this may not be feasible, and data will need to be sourced from other areas. If data from neighbouring populations are available these may provide reasonable estimates for the impacted populations. Literature reports of species-specific values represent the next alternative. If there are no relevant published data for the species, predictive relationship may be the next best alternative for providing estimates for the species. Many estimators for natural mortality exist (Kenchington 2013) using a variety of predictors such as VBGF coefficients or species' longevity (Then et al. 2015). Finally, surrogate species may be used to fill in missing data if all other avenues have failed. When considering mortality rates for inclusion in equivalency metrics the rates should be scaled to account for timing of mortality event(s) within the year. This is important when mortality rates change among life-stages or ages or vary at different times of the year. Uncertainty in life-history parameters can be incorporated into equivalency metrics by using a range of potential values.

Other data needed to estimate some metrics include age-specific growth rate and (or) age-specific egg production and stage duration data such as longevity and age-at-maturity. Further life-history data that could be used to fill in data gaps include growth curves (i.e., VBGF) and length-weight relationships. When parameterizing the equivalency metrics, it is prudent to follow the hierarchy suggested above when sourcing data.

Table 2.4. Comparison of equivalency metric estimates for a hypothetical impingement and entrainment fish mortality (Table 2.2). Units of each metric are in brackets. EA-1 represent age-1 equivalents; RP represents reproductive potential, EA-1 RP represents age-1 equivalents based on reproductive potential; and HPI represents habitat productivity index.

	Count (no.)	Biomass (kg)	EA-1 (no.)	EA-1 Biomass (kg)	RP (eggs x1000)	EA-1 RP (no.)	EA-1 RP Biomass (kg)	PF - annual (kg)	PF - Lifetime (kg)	HPI (kg)	Total Biomass Lost (kg)
Alewife	250,000	1,200.00	485,655.7	1,666.59	944,949.5	256,634.64	880.67	637.30	1,060.02	968.51	2,260.02
White Sucker	2,500	230.25	6,422.39	44.63	21,057.9	3,600.06	25.02	80.89	376.05	99.71	606.30
Gizzard Shad	30,000	1,700.00	27,734.53	1,851.13	1,303,811.6	14,739.10	983.75	1,652.46	3,085.33	617.92	4,785.33
Total	282,500	3,130.25	519,812.7	3,562.34	2,269,819.0	274,973.81	1,889.44	2,370.65	4,521.40	1,686.1	7,651.65

Description of the methods used to estimate the equivalency metrics for the example data are presented in Appendix A.

Estimates of losses from the equivalency metrics (Table 2.4) vary substantially; however, as the units and population characteristic of equivalency differ most are largely not directly comparable. Some metrics, however, are directly comparable. For example, age-1 equivalents (EA-1) and age-1 equivalent from reproductive potential (EA-1_{RP}). EA-1 values were consistently greater than EA-1_{RP} values with total EA-1 almost double (1.9 times) the EA-1_{RP} values. Annual estimates of production forgone (PF_{annual}) and HPI are also directly comparable, representing estimates of losses of 1 year of biomass production to the environment. HPI produced greater values than PF_{annual} for both Alewife and White Sucker (1.5 and 1.2 times respectively); however, HPI was significantly less than PF_{annual} for Gizzard Shad (0.4 times). The use of HPI does not appear to be a consistently good approximation of annual production forgone. The largest equivalency values were produced from total biomass lost; however, these values may best represent losses to the environment as they capture both the loss in standing stock and future biomass production.

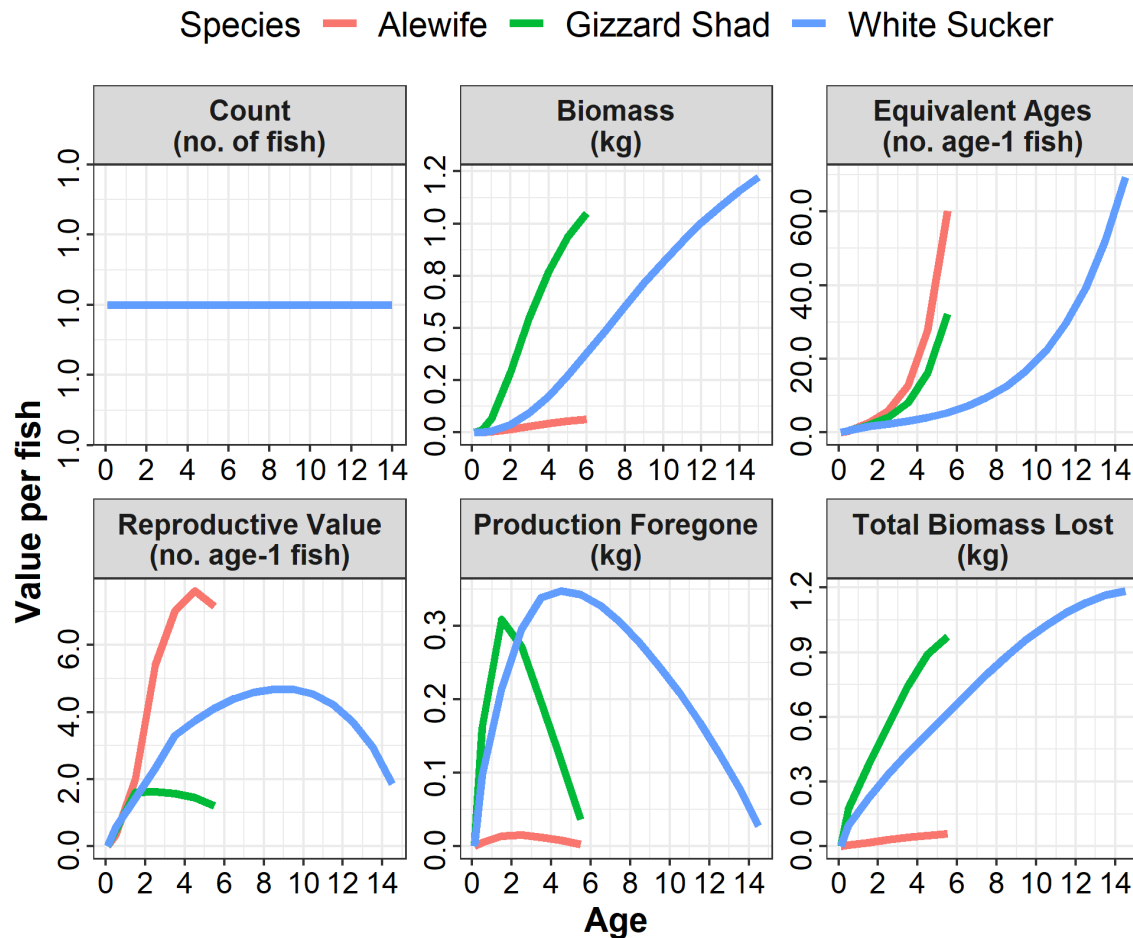


Figure 2.1. The amount an individual fish of each age-class contributes to the total estimate from the equivalency metric for all species in the hypothetical impingement/entrainment example. Representative of how the metrics value an individual of each age-class. Units differ among figure panels and are displayed in brackets in the strip text.

Direct comparison of the metrics can be made by examining how each values fish of different age-classes (Figure 2.1). Figure 2.1 shows the value of an individual fish of each age-class of Alewife, Gizzard Shad, and White Sucker in the units of the metric. In general, count-based metrics (count, equivalent ages, and reproductive value) value different species more similarly than biomass-based metrics (biomass, production forgone, and total biomass lost), which assign greater value to larger species. Consequently, Alewife mortality accounts for much less of the total estimate, relative to White Sucker and Gizzard Shad, when using a biomass-based metric than a count-based metric. Count values all ages of each species equally demonstrating why it is not a preferred method. Biomass values older fish much more than young fish with the value of a maximum aged fish relative to an age-1 fish depending on growth rate. For example, an age-6 Alewife's and age-6 Gizzard Shad's value were ~ 18 and 16 times that of an age-1 fish while an age-15 White Sucker's value was 176 times that of an age-1 White Sucker. Equivalent ages values age-classes based on how many age-1 fish are required for one fish to reach a certain age, leading to older fish having significantly greater value. Maximum aged Alewife, White Sucker, and Gizzard Shad were 60, 69, and 32 times more valuable than an age-1 fish. Reproductive value equivalent ages values age-classes on their future reproductive potential relative to an age-1 fish. Maximum value occurred at ages-4.5, 8.5, and 2.5 for Alewife, White Sucker, and Gizzard Shad, decreasing thereafter. Peak values, relative to an age-1 fish, were 7.6, 4.7, and 1.6 times, respectively. Production forgone values age-classes based on the expected future lifetime biomass production. As a result, older fish are less valuable than younger fish and are likely undervalued. Value peaked at age-2.5, 4.5, and 1.5 for Alewife, White Sucker, and Gizzard Shad, respectively. The peak value relative to an age-1 fish were 1.6, 2.2, and 1.3 times greater for the three species, respectively. Total biomass lost values older individuals the most (peak value occurred at maximum age) but younger age-class were considerably more valued than based on biomass alone. The relative value of a maximum age-fish to an age-1 fish was 5.0, 7.4, and 3.6 times. Total biomass lost values age-classes based on their mean biomass and future production (sum of biomass and production forgone). This balance of current and future biomass may provide the best age-specific valuation.

2.11. RISK AND UNCERTAINTY IN EQUIVALENCY CALCULATIONS

Inherent in the calculation and application of an offsetting plan is uncertainty. Uncertainty exists in the initial measurement of the extent of harm to the environment, as well as in the calculation of equivalency metrics. Additionally, uncertainty exists in the efficacy of the proposed offset. Time delays in the delivery and functionality of the offset can also result in an inequality between harm and offset. These uncertainties must be accounted for in equivalency calculations. Dealing with risks and uncertainties has been previously reviewed by Clarke and Bradford (2014), however, the main concepts will be reiterated here.

Time delays represent delays in the implementation of offset measures and (or) when it takes time for an offset to become fully functional and effective. These time delays can be incorporated with proper accounting of the impact and offset schedules for a particular project. First, a time horizon, representing the length of time the effects of the impact and offset will be measured, must be established. There is no clear objective way to select the time horizon. Selecting a time horizon of infinity or such that compensation measures must be applied immediately are impractical and therefore a balanced approach must be taken. Minns (2006) recommended as options: twice the duration of the project or twice the time taken to get the last of the compensation measures in place. Ultimately, choice of the time horizon will be project specific and can be negotiated while developing offset plans, but must be explicitly selected.

It is also recommended that discounting be applied to accounting of the impact and offset through time. Discounting is widely used in economics to weigh past and future benefits such

that they become comparable. Discounting in relation to the environment reflects society valuing access to natural resources in the present more than in the future. Discounting is applied by multiplying the magnitude of the impact or offset at a future date by the discounting weight, d_t , which represents the relative value of the resource at the future date to its present value, where:

$$d_t = (1 + r)^{t_0 - t}. \quad (2.15)$$

t_0 represents the initial impact year, t the year being valued, and r is the discount rate. The typically accepted discounting rate applied to ecological assets is 3% (NOAA 1999; Clarke and Bradford 2014). This value was initially based on an economic assessment of the real rate of interest (NOAA 1999) and is subject to ongoing debate (Minns 2006). The impact of discounting is muted when offsets are implemented and quickly become functional.

Time-lags are accounted for by calculating the time-lag compensation ratio, CR_{tl} , the multiplier (increase in size of the offset) needed to account for the time-lag:

$$CR_{tl} = \frac{\sum_{t=1}^H p_{i,t} d_t}{\sum_{t=1}^H p_{o,t} d_t}. \quad (2.16)$$

H is the time horizon, t is year, p_i is the proportional impact of harm, p_o the functionality of the offset relative to the maximum (i.e., the impact or offset schedule), and d is the discounting weight.

As an example of accounting for time-lags, a hypothetical example is presented using the equivalency example data from above. The total biomass lost metric is used with a total annual impact estimate of 7,651.65 kg. A time horizon of 30 years is selected, representing the length of the fish mortality authorization. The impact is expected to be constant over the 30-year time period. A habitat creation offset is proposed but construction won't begin for five years, and the habitat won't become completely functional for an additional 5 years after that, with a linear increase in functionality assumed. Discounting is applied with a discount rate of 3%. The ratio of impact to compensation based on these impact and offset schedules (CR_{tl}) is 1.46. Therefore, the offset will need to be 1.46 times greater than the impact ($7,651.65 \times 1.46 = 11,197.36$ kg) to account for the time-lag at the start of the project (Table A.15; Figure 2.2). Had construction begun immediately, a CR_{tl} of only 1.11 would have been necessary.

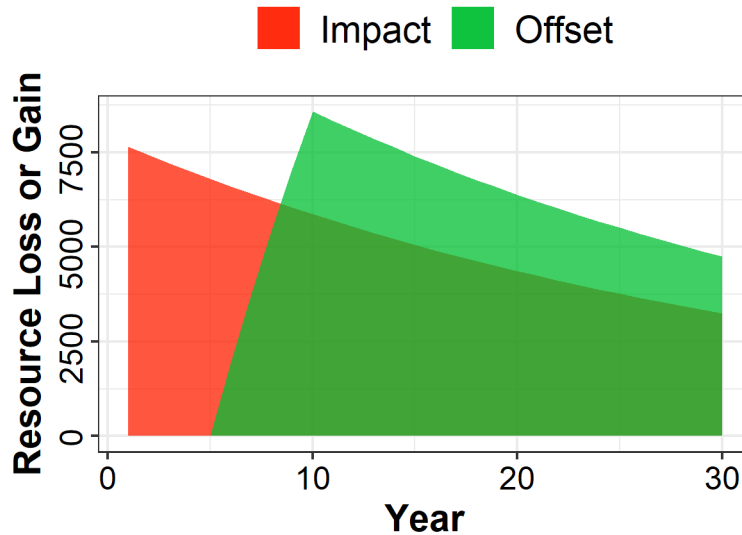


Figure 2.2. Hypothetical example of an impact and offset accounting for a time-lag. This representation incorporated 3% discounting. The summed area of the impact polygon (red) is equal to the summed area of the offset polygon (green).

Uncertainties are also often accounted for with compensation ratios, CR_u ; however, they are less straightforward to estimate. Favaro and Olszynski (2017) found that for the majority of projects in DFO's Pacific and Central and Arctic regions in 2012, compensation (in area) was less than the size of the impact resulting in a net loss of fish habitat and a likely reduction in productivity. The magnitude of the CR_u required to prevent a loss in productivity is not clear. Moilanen et al. (2009) found that very high ratios (>100) could be necessary to guarantee equivalency in value between an impact and its offset. Bull et al. (2017) reviewed implementation of offset multipliers around the world and found they typically ranged from 1 to 10. Quigley and Harper (2006) found a net loss in productivity after compensation in the majority of projects examined with a mean CR_u of 0.7:1. They determined that a CR greater than 2:1 would be required to adequately offset harm and that only when the CR exceeded 4.5:1 was there a net gain in productivity.

Bradford (2017) performed Monte Carlo simulations to determine CR_u s required to account for uncertainty in predictions of resource losses and gains from a project's impact and offset. Required ratios were between 1.5:1 and 2.5:1 to account for the risk. These values were based on an 80% equivalency threshold where there is a 1 in 5 chance the offset does not adequately account for the impact. A greater threshold would require larger CR_u s. Similar Monte Carlo analyses should be performed for WUAs when determining appropriate CR_u s. This would force the explicit quantification of the amount of uncertainty in both the estimation of the extent of impact and the efficacy of the proposed offset.

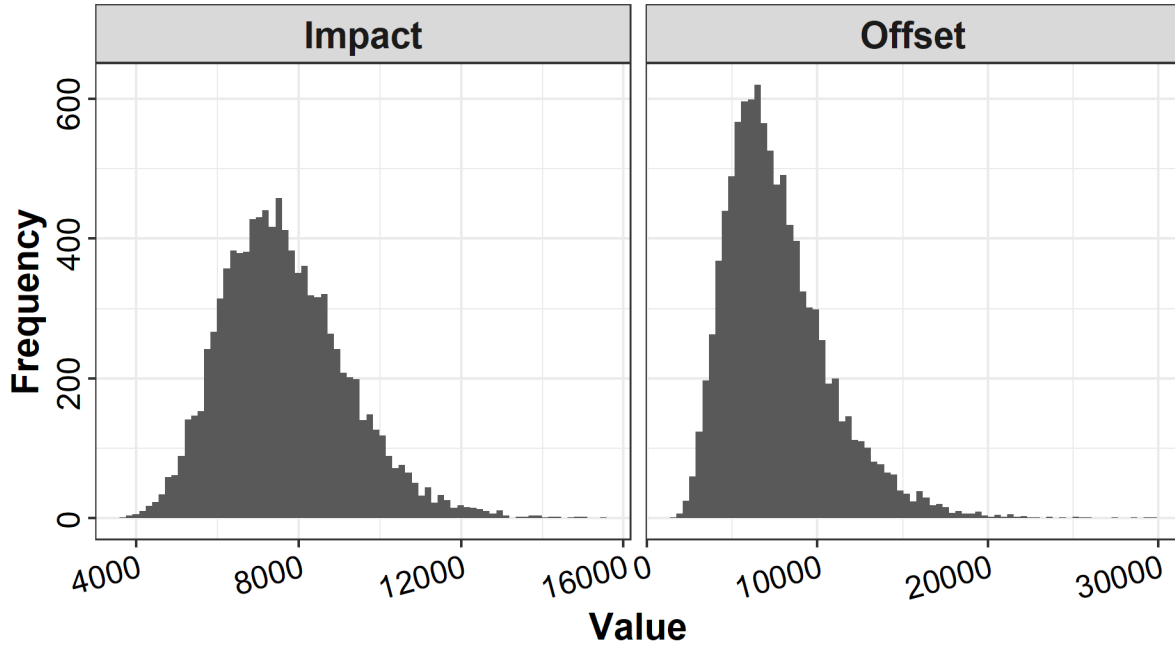


Figure 2.3. Monte Carlo simulations of the impact and offset distributions generated to estimate uncertainty compensation ratio.

Monte Carlo simulations to estimate the uncertainty compensation ratio can be conducted if the mean, standard deviations, and probability distribution of the impact and offset can be estimated. These distributions represent uncertainty in the extent of the impact of the mortality and effectiveness of the offset. The ratio of these two distributions generates a frequency distribution of compensation ratios M (Bradford 2017):

$$M = m_i / m_o, \quad (2.17)$$

where m_i is the distribution of impacts and m_o is the distribution of offsets. The risk associated with allowing an offset that may not adequately compensate for the extent of the impact is balanced with selection of the equivalency threshold, q . The equivalency threshold, represents the percentile of the distribution of M taken as the CR_u :

$$CR_u = M_{(q)}. \quad (2.18)$$

An example of Monte Carlo simulation to estimate CR_u is applied to the hypothetical fish mortality and offsetting plan presented above. The means of the impact and offset are assumed to be equal and set to the total biomass lost equivalency metric: 7,651.65 kg. The error estimates are assumed to be 20% for the impact and 40% for the offset (S.D. of 1,530 and 3,060, respectively). The error applied to the impact can account for both error in measuring the amount of mortality and error in quantifying the equivalency metric (e.g., error in life history estimated use in calculations). A log-normal distribution is assumed for both. The log-normal distribution was selected to ensure negative values were not generated; however, other distributions, such as the gamma, truncated-normal, Poisson (for discrete counts), etc., could be used. The variation in the level of impact and effectiveness of the offset are assumed to be uncorrelated. If the impact and offset are located in the same area, they may be subject to similar stochastic events which would result in some level of correlation between them. Positive correlation between the impact and offset uncertainty results in reduced CR_u estimates. 10,000 draws from the distributions are made giving uncertainty distributions for the impact and offset

(Figure 2.3). The distribution of potential CR_u s is generated following equation 2.17 (Figure 2.4). The choice of equivalency threshold is a management decision to balance the risk that the offset does not adequately compensate for the level of impact. The amount of risk is represented by $1 - q$. In this example, a q value of 0.8 is used, following Bradford (2017), leaving a 20% chance the fish mortality is not completely offset. Taking the 80th percentile of M gives a $CR_u \sim 1.53$.

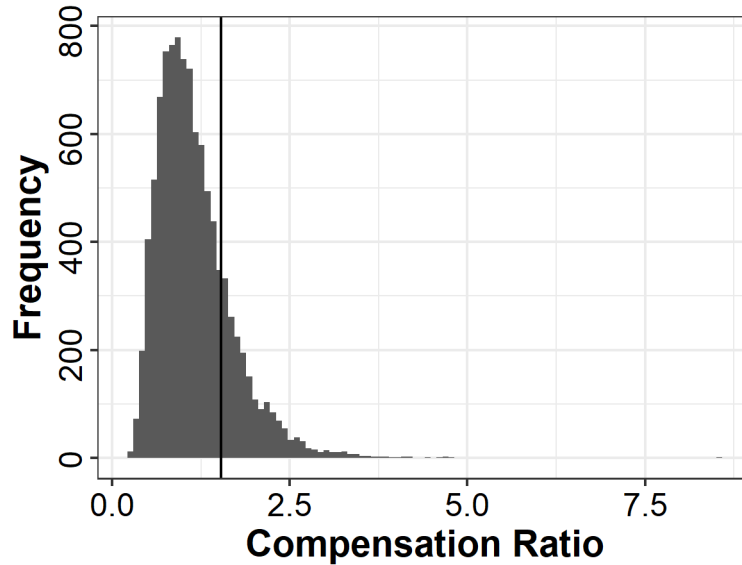


Figure 2.4. Distribution of uncertainty compensation ratios generated from Monte Carlo simulations. The vertical line indicates the 80th percentile which is used as the equivalency threshold.

The majority of studies have examined CR_u s with respect to habitat creation or alteration offsets; however, CR_u will likely be necessary for stocking offsets as well. Fitness of stocked fish is often less than the native population (Loughlin and Clarke 2014), therefore, it cannot be assumed that stocking biomass equal to the impact will be sufficient and CRs should be employed.

Full accounting of the offsetting for fish mortality requires the choice and calculation of equivalency metric (to get the equivalency value), determination of the impact and offset schedule over the appropriate time horizon, quantification of the time-lag multiplier, CR_{tl} , and finally selection of the uncertainty multiplier, CR_u , to account for uncertainties potentially estimated through Monte Carlo simulation. The size of the required offset is then calculated as:

$$\text{Offset size} = \text{Equivalency Value} \times CR_{tl} \times CR_u. \quad (2.19).$$

From the above example, this would give a required offset size of $7,651.65 \times 1.46 \times 1.53 = 17,092.26$ kg.

3. ASSESSING COMMUNITY IMPACTS

There is an increased interest for managers to consider the community-level impacts of fish mortality rather than only on the species that are most valued by humans. Projections of impacts of human activities on populations can often be incomplete, as harm applied to a single species can impact whole communities and may cause counter-intuitive results due to indirect effects and feedback loops. This can include species that are not directly affected by mortality events. However, predicting how fish mortality impacts the whole fish community has many challenges. Not least of these challenges is that information on the network of species interactions that governs community dynamics (i.e., the food web) is typically scarce for all but the most well studied ecosystems. The state of knowledge for ecosystems can be represented as a progression from no site-specific information with lower certainty to site-specific data on species abundances and the strength of linkages (Fig. 3.1). Mechanistic multi-species models and whole system experiments can be effective for identifying community change, and yet these approaches can require large investment in resources and time, both of which may be limited when a management decision must be made.

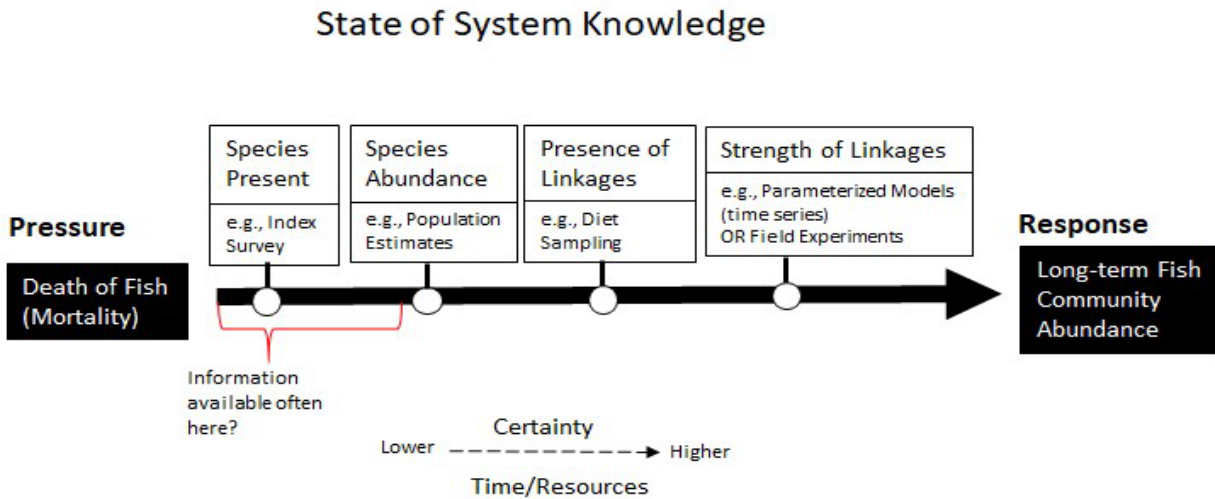


Figure 3.1. An illustration of the state of system knowledge associated with predicting the long-term fish community response to a sustained pressure of fish mortality (death of fish). The large black arrow begins in a state of no site-specific information for predicting and sources of information are added along the line (a simplified view as the way information is added is likely not linear). For the majority of cases where rapid decisions are required for fish and fish habitat management, knowledge may fall somewhere between no information and some information on the species present and the abundance of a few fish species.

A number of community models are available to evaluate community impacts from fish mortality, depending on data availability, modelling difficulty, and the type of advice needed (summarized in Table 3.1). More in-depth examinations of the various models can be found in Plaganyi (2007), DFO (2008) and model-specific references in Table 3.1. Depending on the questions to be answered and considerations such as time and data availability, one or more appropriate models described below may be applied to inform management decisions.

3.1. QUALITATIVE NETWORK MODELS (QNM)

Qualitative network model (QNM) (Levins 1974; Melbourne-Thomas et al. 2012) is an approach that can be used in data-limited situations where species interactions cannot be fully specified

or measured but where there is some information on the key ecosystem components such as the species present, and the direction and sign of the interactions between them. In this approach, the ecological community is represented as a network of signed directed graphs where species or system variables are represented as nodes and interactions between these variables are represented as linkages between nodes (i.e., sign specified directed arrows). This network is then translated into a community matrix where the elements (i.e., interaction strength between species) are populated with simulated values. Matrix algebra is applied to the community matrix to solve for the long-term direction of response (positive or negative) of the species populations when a sustained perturbation (referred to as a press perturbation) is applied to one or more nodes of the community network. This process is repeated for many simulated matrices, based on the uncertainties, and the results are summarized to generate the probability of positive and negative outcomes for each node under the specified perturbation. More detailed, mathematical descriptions can be found in Levin (1974), Yodzis (1988); Dambacher et al. (2002), (2003), Hosack et al. (2008), and Melbourne-Thomas et al. (2012). The R package *QPress* (Melbourne-Thomas et al. 2012) is a user-friendly QNM software available for free.

As a modelling approach, QNM has certain limitations. First, the analysis assumes that systems are stable. This may be a reasonable assumption since it is thought that non-stable systems do not persist in nature, but there might be cases where systems are in transition or are in an unstable state. Second, QNM does not take into account non-linear effects and thus may underestimate density-dependent compensatory or depensatory mechanisms. Third, similar to the assumptions for a sensitivity analysis, the press perturbations are assumed to be a small, constant pressure. This limits the model's applicability for rare, catastrophic perturbations which might push the system to an alternate state. Finally, due to randomly generated, simulated parameter values, QNM outputs can only generate a probability of positive (or negative) outcomes under a perturbation and does not provide any information on the magnitude of the responses. A low probability of a negative response does not necessarily equate to a low risk since the magnitude of that negative response is unknown. To obtain estimates of magnitude, the use of real data to parameterize the community matrix would be required.

Despite these issues, QNM can be a useful tool for risk assessment and for answering specific questions about an ecosystem. The simplifying assumptions and the use of simulated parameter values allow QNM to be applied to systems lacking detailed knowledge. The model's simplicity allows the user to easily perform scenario testing and determine whether and how the qualitative response of a community is sensitive to a parameter value or the presence of an interaction. Thus, the QNM modelling approach is useful for comparing alternative hypothesized community structures and evaluating which ecological components have relatively strong effects on the rest of the ecosystem if perturbed. It is also easy to integrate non-biological variables (e.g., management decisions, economic factors, etc.) into QNM's modelling framework. Given its flexibility and ability to incorporate socio-economic considerations, qualitative modelling has been used in a variety of situations such as examining marine survival in Pacific salmon (Sobocinski et al. 2017), predicting the community effect of increasing bivalve aquaculture in Puget Sound (Reum et al. 2015), evaluating a pest eradication program on Macquarie Island (Raymond et al. 2011), and investigating the impact of gold mining on reef fish community in Papua New Guinea (Dambacher et al. 2007).

3.2. BAYESIAN BELIEF NETWORKS (BBN)

Bayesian belief networks (BBN) are developed to address management decisions under uncertain conditions (Varis 1997). The uncertainty is incorporated using Bayesian probability theory which can predict the probability of an outcome based on the state of other variables in the model. BBNs are network models where environmental variables, species variables, and management decisions are represented by nodes and the influence of one variable on another is represented by linkages made up of conditional probabilities. These probabilities cascade down the network to compute probabilities for possible states of some ecological variable of interest. BBNs are easy to understand conceptually and can be used to either fit model parameters or generate projections. There are a number of commercially available BBN software such as [Netica](#) and [Hugin Expert](#). Free software such as the R package *bnlearn* (Scutari 2010) are also available.

One of the advantages of BBN is that it depicts ecological relationships and interactions in an intuitive manner and allows its user to effectively analyze the impacts of alternative management decisions (McCann et al. 2006). Managers could rank management options based on the probabilities of desirable potential outcomes. Due to its computationally non-intensive nature, sensitivity analysis of the outcomes to varying parameter values or model structure can be performed quickly and easily relative to other community models. BBNs could be useful for both data-rich and data-limited systems. While it is possible to produce precise projections given large amounts of data, for data-limited systems it would function more as a tool for scenario testing and generating hypotheses. Examples of BBNs include examining the impact of climate change and barrier removal on trout riverine habitat (Peterson et al. 2013), habitat evaluation for caribou conservation (McNay et al. 2006), and assessing the impact of mountain pine beetle outbreak management strategies on American martens (Steventon and Daust 2009).

Bayesian belief networks also have limitations. This method is incapable of handling systems with feedback loops (i.e., variable A affects variable B, B affects C, C affects A), i.e. they are directed acyclic graphs. In addition, each of the variables in the network must be expressed as discrete values. Continuous variables can be accommodated via transformation into categorical variable with non-overlapping ranges of the original continuous values. Finally, the linkages (i.e., the relationships) between variables need to be expressed as conditional probabilities. These relationships are usually expressed as equations in the literature and the user will need to transform these equations into conditional probabilities. The conditional probability tables of variables, influenced by multiple other variables, can become very complicated and users often may need to rely heavily on expert opinions to assign probabilities.

3.3. INDIVIDUAL-BASED MODELS (IBM)

Individual-based models (IBM), also called agent-based models, track the actions and behaviours of individuals throughout their life under the assumption that once the data are aggregated at the end of a simulation, the results can help inform an understanding of the population dynamics. The agents in these models could represent actual individual animals, be a group acting in a coordinated manner (e.g., a school of fish), or represent higher level groups of individuals such as cohorts or entire populations. The important aspect of an IBM is that each agent exhibits attributes (e.g., sex, age, size for fish individuals) which can distinguish them from other agents. Agents interact with the environment and each other according to a number of rules and can often change and adapt over the course of the simulation. Since agent behaviours and interactions are often regulated by movement and spatial co-occurrence, IBMs tend to be spatially explicit. One example of an IBM that has been used in ecology is OSMOSE (Object-oriented Simulator of Marine ecOSystem Exploitation) (Shin and Cury 2001). It is a

spatially explicit model that assumes opportunistic predation based on fish size. *OSMOSE* is available as an R package. More general programming platforms include [NetLogo](#) and [MASON](#).

One of the main strengths of an IBM is its explanatory power and ability to produce complex emergent behaviour at the aggregate level based on simple rules at the individual level. Spatial considerations are easy to incorporate in IBMs. Many IBM platforms assume the presence of a 2- or 3-D grid on which the simulation is carried out. It is relatively easy to overlay the physical environment which the user wishes to model over this grid. A third strength of IBM is its ability to incorporate stochasticity. The probability of a behaviour could be determined for the population as a whole and easily translated into rules for each individual agent. Finally, IBM has a modular structure. The rules for agent behaviours are the core of an IBM and the rules could be modified, or new rules added without needing to modify the entire model.

Challenges for IBMs mainly come from the analysis of model results. It is difficult to apply a systematic analysis on how changes in the model or the model's parameters affect model results in contrast to how one could approach these analyses for an analytical model (e.g., a sensitivity analysis). The user could attempt to get around this problem by treating the IBM as an experiment by performing a large number of simulations and applying statistical analysis to the data set. Such brute force methods require significant computing power. Another practical consideration is that a modeller who wishes to utilize IBM will also likely require advanced programming skills. A more in-depth discussion of the characteristics of IBMs can be found in An et al. (2009).

3.4. SIZE-SPECTRA MODELS

The size spectrum is the distribution of organismal biomass or abundance as a function of body mass or size on a log-log scale (Guiet et al. 2015; Sprules and Barth 2016). When body size is binned into logarithmically equal size intervals, the size spectrum produces a predictable relationship between total biomass (or abundance) and body size. This relationship is an emergent property of individual-based and community-level energetic processes with a theoretical and empirical basis in the role that size has as a major structuring trait of aquatic communities and ecosystems (Andersen et al. 2016; Sprules and Barth 2016). Generally, aquatic ecosystems are structured with rarer, larger organisms preying on more abundant, smaller organisms. When size spectra are plotted, there are two main parameters of interest: the slope and the intercept (Guiet et al. 2015; Sprules and Barth 2016). The slope is related to energy conversion efficiency and tends to be invariant with deviations providing a measure of perturbations, such as from anthropogenic development, fishing, and non-native species (Guiet et al. 2015; Chu et al. 2016; Sprules and Barth 2016). The intercept (or alternatively, the midpoint height) is related to system production and can provide a strong indicator of the impact of the environment.

Aggregating data based on size provides a simplification of the complexity of aquatic ecosystems, with a focus on ecosystem-level emergent properties. This can be useful when considering impacts and function at the level of whole ecosystems, to assess current state of a community, and when faced with data limitations. However, since size-spectra models ignore taxonomic diversity, they are not appropriate for species-specific dynamics (but see dynamic multispecies size spectrum models) and can be challenging to apply when local imports and exports are important.

3.5. DYNAMIC MULTISPECIES SIZE SPECTRUM MODELS

To simulate the potential responses of aquatic species and communities, dynamic multispecies size spectrum models build on the size-based theory that serves as the foundation for size

spectra models (Andersen et al. 2016; Andersen 2019, 2020). Dynamic multispecies size spectrum models can be parameterized to represent generalized communities where individuals are only characterized by their size, they can be trait-based where species' life histories are specified based on asymptotic size, or they can be multispecies models with individual species characterized by distinct life histories (Andersen et al. 2016). Often the fish community may be resolved at the species-level with lower trophic levels represented by the resource spectrum based on an expected size-spectrum. Mizer provides an R package to run dynamic multispecies size-spectrum models of fish communities (Scott et al. 2014). Through simulations, dynamic multispecies size-spectrum models take advantage of the theoretical and empirical basis of size-spectra to over-come some of the limitations of size-spectra models and provide a basis for exploring the responses of species and communities to perturbations. To date, however, their application has been focused on fisheries (e.g., Blanchard et al. 2014; Andersen et al. 2016).

3.6. MINIMUM REALISTIC MODELS

A full ecosystem model is often not needed to address all questions. Minimum realistic models (MRM) are based on the concept of restricting model complexity to just those species most likely to have important interactions with the species of interest. The objective is to balance model realism with uncertainty by including just sufficient detail to account for components needed to capture key dynamics. In many ecosystems, this focuses the model on species that have been monitored and for which sufficient data exist. While originally employed to consider a specific ecosystem (Punt and Butterworth 1995), the MRM classification has been used to group together a number of different modelling approaches, including multispecies virtual population analysis (MSVPA), the globally applicable area-disaggregated general ecosystem toolbox (GADGET), extended single-species assessment models (ESAMs), and multispecies statistical catch-at-age models (Plaganyi 2007). The MRM term has been used almost exclusively in the fisheries science literature, however, the concept underlying MRMs is not unique to fisheries models.

One of the main challenges when applying MRMs is choosing an appropriate model complexity for the ecosystem and the question being addressed. Yodzis (1998) proposed an approach to assessing how much model complexity is needed. When applied to the Benguela ecosystem modelled by Punt and Butterworth (1995), Yodzis found that losing interactions that represented less than 10% of total consumption did not noticeably affect model predictions. A further consideration about model complexity is that the approach to removing weak links (e.g., by aggregating vs. chopping) can have implications for model predictions (e.g., Pinnegar et al. 2005). Finally, while an MRM is less complex than a full ecosystem model, the components included in the model are not necessarily data-limited and may be represented by complex dynamic models with significant data requirements.

3.7. WHOLE ECOSYSTEM MODELS

There is a growing interest in whole ecosystem models to provide information for ecosystem-based management for aquatic ecosystems. Ecosystem models have been used to better understand the impacts that arise from human pressures. They are also used to explore the trade-offs among different management objectives that aim to protect and conserve different ecosystem components (Collie et al. 2016).

Whole ecosystems models can take different forms (e.g., linear inverse modelling (LIM), Ecopath with Ecosim (EwE), ATLANTIS), but typically incorporate data from many ecosystem components and the interactions between these components, and some allow for simulation of ecosystem dynamics. Freshwater applications span all trophic levels with some resolution within trophic levels including individual species or functional groups of fish and invertebrates and

multiple basal resources pathways (i.e., primary production and detritus). LIM uses linear functions to estimate model parameters from an incomplete set of observed data that are used to constrain outcomes (e.g., mass balance). An application of this approach is the estimation of the strength of linkages or flows of the food web and is completed by solving a set of matrix equations. EwE is a software application based on a mass balance model called Ecopath, where biomass pools of components are linked by trophic interactions, and includes EcoSim with dynamic simulation capabilities parameterized using the [Ecopath model](#). ATLANTIS is a spatially explicit modelling framework that contains submodels for physical and biogeochemical processes, ecology, human uses and management (Audzijonyte et al. 2019). Atlantis differs from EwE as organisms can be represented as biomass pools and age-structured groups. It has been applied in many contexts including comparing management strategies, historical fisheries impact, and impact of global change to name a few.

For freshwater applications, data requirements are one of the challenges of developing and parameterizing whole ecosystem models. Often long-term time series data are not available for the fish community and other ecosystem components where management concerns arise. Another challenge that occurs is that parameter uncertainty increases with increasing model complexity. That said, simple models have greater uncertainty in model structure, and model structure can be very important for understanding and managing ecosystems that are undergoing significant change. Further, any whole ecosystem model is a simplification of a real ecosystem, however, the opportunity to learn from the process of building and parameterizing a whole ecosystem model in itself can provide useful advice to decision makers.

3.8. STRUCTURAL EQUATION MODELS

Structural equation modelling (SEM) is a powerful framework used to investigate both direct and indirect relationships of networks in ecosystems. Studies that involve investigating causal pathways and complex interactions among many variables are suited for SEM. Interest in community and ecosystem effects, in particular understanding the pathways of undesirable effects and teasing apart multiple factors that may lead to fish mortality may be an important application of SEM in fish and fish habitat management.

SEM is a combination of two kinds of statistical analysis (1) path analysis and (2) confirmatory factor analysis. Path analysis is used to quantify the relationships between multiple interacting variables. It is a way to address causal relationships using multiple regression techniques. One of the functions of path analysis is understanding mediation, which is when a variable can affect an outcome both directly and indirectly through another variable. In this way path analysis enables separation of direct and indirect effects and expands interpretation of causal relationships by considering potential mediating factors. Confirmatory factor analysis in SEM is used to measure latent variables. Latent variables are described as derived common factors of other measured variables or an unmeasured variable that is hypothesized to exist. An example of a latent variable that could be used in freshwater research might be climate which could reflect several observed measurements like air temperature, precipitation, thermocline depth, wind speed etc.

SEM is gaining popularity as a framework but is not without limitations. Despite the description as a method to investigate causal pathways, it is similarly problematic to claim causality with SEM as it is with linear regression. These models are also limited to linear relationships. With increasing interest in the use of SEM in several scientific disciplines there are a number of books (e.g., Grace 2006) and software options (e.g., LISREL) and R packages available (e.g., OpenMx and lavaan). Such application will allow users to implement the process associated with developing an SEM model including model specification, identification, parameter estimation, model evaluation, and model modification (Fa et al. 2016).

3.9. MULTI-SPECIES BIOENERGETICS MODELS

Multi-species bioenergetics models incorporate bioenergetics and allometry to develop plausible models with some key biological constraints. These models were developed to cover intermediate ground between models based on detailed parameterization of a particular population and more weakly constrained strategic models. This approach has been used to address management issues where species interactions are considered important, some food web and population data are available but extensive data like time series for many species are not. Note that multi-species bioenergetics models (*sensu* Yodzis and Innes 1992; Koen-Alonso and Yodzis 2005) are not the same as bioenergetics modelling tools often used to estimate growth or food consumption for fish populations (e.g., bioenergetics 4.0). Although the idea of using bioenergetic constraints to build better models with the information available is common among these approaches.

Multi-species bioenergetics models typically represent two or more species in a system of ordinary differential equations that represent ecosystem components like species as biomass pools. Biomass gains and losses are the result of predator-prey interactions and parameters that reflect population vital rates. In some of the earlier formulations, bioenergetics are incorporated by modelling vital rates like growth by physiological capacity, coefficient of metabolic type (some models combined mammals, fish and invertebrates) and body size.

Multi-species bioenergetics models have been used to assess fisheries using models of different complexity including consumer resource, intermediate complexity food web, and allometric trophic networks (ATN). In some investigations environmental factors (e.g., temperature) have been included and are linked to organism growth through key vital rates and allometric properties (e.g., body size) .

The models listed here are only a sample of available modelling techniques that can be used to investigate the impacts of fish mortality from a community perspective. Selecting a model to apply will depend on the questions to be answered and the data available. These models could be grouped roughly according to their data requirements. Models such as the QNM, BBN, and IBM are relatively light in terms of general data requirements but may require certain specific types of data to parameterize the species interactions. However, these models can be limited in their ability to produce quantitative predictions and are more suitable for scenario testing or as a diagnostic tool.

Systems where there is a moderate amount of knowledge could make use of size-spectra models as a diagnostic tool for community status. The dynamic multispecies size spectrum model builds on size spectrum theory with simulations to explore potential community responses. Other modelling techniques such as MRM, SEM, and multi-species bioenergetics models could also be used in such data moderate systems to generate more concrete predictions and perform impact assessments. Data rich ecosystems may warrant full use of whole ecosystem models to generate detailed projections of human impacts on all trophic levels in the community.

3.10. COMMUNITY MODEL EXAMPLE USING QUALITATIVE NETWORK MODEL

To illustrate how one might approach investigating the consequences of fish mortality at a community level, the Qualitative Network Model (QNM) approach was applied to a number of simple communities to seek general principles for application in decision making. QNM was used to examine the impact of fish mortality on communities by assembling community matrices that capture species interactions, adding negative effects to targeted species (simulating mortality) and then tracking qualitative responses (positive, negative, neutral) of all community members. The sensitivity of qualitative responses to changes in species interactions was of

particular interest. How this sensitivity was affected by the number of species directly targeted with a negative effect that reflected fish mortality was also examined.

The QNM approach was applied to twelve simple community networks (shown on Table 3.2); six were the simple community modules described in Gilman et al. (2010) and six were slightly more complex networks made by combining Gilman modules. For each community network, simulated matrices were created until 10,000 stable networks were found. Negative press perturbations representing increased fish mortality were applied to all combinations of nodes (each node represents a single species) in each of the networks and the probability of negative outcomes was recorded for all nodes. Simulations and analysis of qualitative networks were performed in R 3.6.3 (R Core Team 2020) and using the package *QPress* (Melbourne-Thomas et al. 2012).

Simulation results are shown in Figures 3.2 to 3.4. The number of nodes, the number of links and the maximum number of nodes perturbed are closely related and due to the limited number of networks examined, it is not possible to separate out these confounding influences for the models studied here. However, it is still possible to observe some general patterns. As the number of nodes and links in a network increase, the number of mixed results where a positive or negative response depends on specific parameter values also increases (Figure 3.2 and 3.3). This implies that as communities become larger and more complicated, it becomes more difficult to predict the direction of response based solely on the pattern of link structure and predicting the qualitative outcome requires greater in-depth knowledge of interaction strengths.

The probability of a negative response also increases when the number of pressed nodes increases (Figure 3.4). In the case when all species within a network are impacted, the probability of negative responses for each individual species is greater than ~50% with one exception at ~12%. Fish mortality imposed on many species will have a higher probability of leading to negative outcomes than mortality imposed on a few or a single species.

The scenario where mortality is applied to all species in a community network is also shown in a heat map of node responses for each of the twelve networks (Figure 3.5). Red regions, which indicate a probability of negative response greater than 70%, predominate. Figure 3.5 also shows the general tendency for top trophic nodes to exhibit a high probability of negative response. This can also be seen in Figures 3.2 to 3.4 where the probability of negative outcomes for top trophic nodes are always higher than those of other trophic levels. These results imply that top predators will tend to be negatively impacted even when fish mortality is applied to species at lower trophic levels.

Patterns specific to individual networks are tabulated in Table 3.2. An observation worth highlighting is that food chains with omnivorous links tend to have more mixed results compared to food chains without omnivory; hence, accurate predictions will require more detailed knowledge for communities with larger numbers of omnivorous interactions than for those without.

The goal and scale of this study is very limited. However, even under such narrow scope, it is possible to demonstrate the need for a community approach when informing management decisions. The general conclusions previously listed examined patterns on a very coarse scale. When managing for a specific system, it is important to have detailed knowledge about that system. The presence or absence of specific links in a network can have a strong impact on possible outcomes. Population response for a species of interest may also depend on parameter values of species not directly connected to the species of interest. However, for systems where there is a high degree of uncertainty, qualitative modelling provides a powerful method to generate predictions about the direction and consistency of community responses. It also provides a tool to clarify which species interactions have strong influence on community

responses and which interactions have little impact to help direct scientific investigations towards important interactions.

Table 3.1. A comparison of several community models which could be useful for examining WUA-related fish mortality scenarios.

Modelling Method	Description	Data Requirement	Main Indicator(s)	Difficulty of Application	Utility of Results	Type of Advice	Key References
Qualitative Network Model	<ul style="list-style-type: none"> • A network of variables and interactions where the strength of interactions is simulated to generate qualitative predictions on the direction of response 	<ul style="list-style-type: none"> • Direction and sign of interactions between variables 	<ul style="list-style-type: none"> • The probability of increase and decrease for a variable or population 	<ul style="list-style-type: none"> • Easy, require some basic programming skills • R package <i>QPress</i> 	<ul style="list-style-type: none"> • Useful for scenario testing and pointing out which interactions in the system has the strongest influence 	<ul style="list-style-type: none"> • Qualitative, strategic recommendations only 	<ul style="list-style-type: none"> • Levins 1974 • Melbourne-Thomas et al. 2012
Bayesian Belief Network	<ul style="list-style-type: none"> • A network of variables and influence pathways where the pathways are parameterized using conditional probabilities 	<ul style="list-style-type: none"> • Possible range of values for each variable • Influence pathways between variables expressed in terms of conditional probabilities 	<ul style="list-style-type: none"> • The probability of each outcome state for a response variable 	<ul style="list-style-type: none"> • Easy, require some basic programming skills • R package <i>bnlearn</i> • Various commercial software (e.g., <i>Netica</i>) 	<ul style="list-style-type: none"> • Risk assessment for management decisions 	<ul style="list-style-type: none"> • Usually strategic advice only unless extensive data is available to provide tactical advice 	<ul style="list-style-type: none"> • Marcot et al. 2006 • Jensen 1996
Size Spectra Model	<ul style="list-style-type: none"> • The size spectrum represents the abundance or biomass of organisms as a function of their size 	<ul style="list-style-type: none"> • Biomass (or abundance) and body size (e.g., weight) for all species in a food web 	<ul style="list-style-type: none"> • The slope and (or) elevation of the size spectrum 	<ul style="list-style-type: none"> • Require analytical and programming skills 	<ul style="list-style-type: none"> • Diagnostic tool 	<ul style="list-style-type: none"> • Strategic advice and community status diagnostic 	<ul style="list-style-type: none"> • Guet et al. 2016 • Anderson et al. 2016 • Sprules and Barth 2016
Dynamic Multispecies Size Spectrum Models (e.g., Mizer)	<ul style="list-style-type: none"> • A dynamic size spectrum ecological model of an entire aquatic community 	<ul style="list-style-type: none"> • Basic estimates of size, reproduction, and feeding preferences 	<ul style="list-style-type: none"> • Biomass of species or functional groups 	<ul style="list-style-type: none"> • Requires programming skills • R package <i>mizer</i> 	<ul style="list-style-type: none"> • Prediction of potential outcomes from alternative scenarios 	<ul style="list-style-type: none"> • Strategic advice and some ability for tactical advice 	<ul style="list-style-type: none"> • Scott et al. 2014
Minimum Realistic Models (e.g., MSVPA)	<ul style="list-style-type: none"> • Models which focus on a selected group of species which are likely to have important interactions with the species of interest 	<ul style="list-style-type: none"> • Time series data of biomass estimates, fishery catches • Age-size composition data 	<ul style="list-style-type: none"> • Biomass estimates with confidence intervals 	<ul style="list-style-type: none"> • Moderate to difficult, lacks general framework and model needs to be tailored to 	<ul style="list-style-type: none"> • Provide tactical fisheries management advice (e.g., total allowable catch) 	<ul style="list-style-type: none"> • Strategic or tactical advice 	<ul style="list-style-type: none"> • Punt and Butterworth 1995

Modelling Method	Description	Data Requirement	Main Indicator(s)	Difficulty of Application	Utility of Results	Type of Advice	Key References
	<ul style="list-style-type: none"> Has a focus on parameter and uncertainty estimation 			answer specific questions			
Whole Ecosystem Models (e.g., Ecopath with Ecosim; Linear Inverse Modelling (LIM); ATLANTIS)	<ul style="list-style-type: none"> Models which attempts to take into account all trophic levels in the ecosystem Predator-prey interactions often modelled using Lotka-Volterra equations 	<ul style="list-style-type: none"> Time series data of biomass estimates, life-history parameters, stock-recruitment relationships, total mortality, consumption, diet composition, fishery catches 	<ul style="list-style-type: none"> Stock and catch estimates under various scenarios 	<ul style="list-style-type: none"> Extremely difficult if trying to create from scratch Moderate if using premade software with support (e.g., <i>Ecopath with Ecosim</i>) 	<ul style="list-style-type: none"> Stock assessment Scenario investigation Theory development 	<ul style="list-style-type: none"> Strategic management advice 	<ul style="list-style-type: none"> Christensen and Walters 2004 Audzijonyte et al. 2019 https://ecopath.org
Individual-Based Models (e.g., OSMOSE)	<ul style="list-style-type: none"> Models which simulates the behaviour of each individual in a species 	<ul style="list-style-type: none"> Life-history parameters (e.g., growth, survival, reproduction, migration, etc.) for each species Behavioural rules for agent interactions Spatial data of system to be modelled and data of species distribution within the system 	<ul style="list-style-type: none"> A variety of ecological indicators (e.g., size structure, biomass, diversity indices) could be calculated by aggregating the data at different levels 	<ul style="list-style-type: none"> Difficult, requires advance programming skills General platforms (e.g., <i>NetLogo</i>, <i>MASON</i>) R package <i>OSMOSE</i> 	<ul style="list-style-type: none"> Implications for spatial management Analysis of emergent properties (e.g., stock-recruitment relationship, predator selectivity, etc.) 	<ul style="list-style-type: none"> Usually strategic advice only unless extensive data is available and the model is designed in such a way to provide tactical advice 	<ul style="list-style-type: none"> DeAngelis and Gross 1992 Shin and Cury 2001 An et al. 2009
Bioenergetic Multispecies Models	<ul style="list-style-type: none"> Models which use energy as a common currency to describe species biomass and how it is transferred between those species via differential equations 	<ul style="list-style-type: none"> Time series data of biomass, mortality, fishery catches Life-history and allometric parameters Diet information 	<ul style="list-style-type: none"> Biomass of populations 	<ul style="list-style-type: none"> Difficult, requires advance programming skills Lack premade packages 	<ul style="list-style-type: none"> Scenario testing 	<ul style="list-style-type: none"> Strategic management advice 	<ul style="list-style-type: none"> Yodzis and Innes 1992 Koen-Alonso and Yodzis 2005; Gilbert et al 2014
Structural Equation Models	<ul style="list-style-type: none"> SEM is an approach that uses observed correlations in order to evaluate complex casual relationships. It is 	<ul style="list-style-type: none"> SEM is suited to large scale observational community of population data sets. 	<ul style="list-style-type: none"> SEM is typically used to test and compare a priori hypothesized models. Also used in exploratory analysis 	<ul style="list-style-type: none"> A number of software options (e.g., LISREL) and R packages available (e.g., 	<ul style="list-style-type: none"> Useful to determine direct and indirect pathways in the structure that link 	<ul style="list-style-type: none"> Strategic Management advice 	<ul style="list-style-type: none"> Fa et al. 2016. Grace 2006

Modelling Method	Description	Data Requirement	Main Indicator(s)	Difficulty of Application	Utility of Results	Type of Advice	Key References
	described as an extension of path analysis			OpenMx and lavaan)	ecosystem components • incorporate “latent” variables		• Grace et al. 2010.

Table 3.2. Community networks used for QNM simulations and observations on the results when either single or multiple nodes were negatively pressed. The first six networks are simple community modules from Gilman et al. (2010) and the next six are more complex networks constructed from combinations of Gilman modules. For the network diagrams, the circles represent nodes and the lines represent links between nodes. An arrowhead on the line represents a positive impact in the direction of the arrow while a dot represents a negative impact. The perturbation columns list summarized simulation results for all combinations of single-node and multi-node press perturbation scenarios. Simulations were based on 10,000 community matrices with randomly generated parameter values.

Community Network	Network Structure	Number of Nodes	Number of Links	Single-node perturbations	Multiple-node perturbations
Specialist Enemy-Victim		2	1	<ul style="list-style-type: none"> • Top node always negative if either node is pressed • Bottom node positive when top node pressed 	<ul style="list-style-type: none"> • Top node always negative • Bottom node 50% chance positive
Mutualism		2	1	<ul style="list-style-type: none"> • Both nodes always exhibit negative response when one of the nodes is pressed 	<ul style="list-style-type: none"> • Both nodes always negative
Exploitative Competition		3	2	<ul style="list-style-type: none"> • The other competitor exhibits positive response when a competitor is pressed • Both competitors negative when shared resource is pressed 	<ul style="list-style-type: none"> • Competitor nodes may be positive depending on parameter values and which nodes were pressed • Shared resource node always has >50% chance of positive response
Apparent Competition		3	2	<ul style="list-style-type: none"> • Both competitor nodes positive when shared predator is pressed • The other competitor exhibits positive response when a competitor is pressed, the shared predator always exhibits negative response 	<ul style="list-style-type: none"> • Competitor nodes may be positive depending on parameter values and which nodes were pressed • Shared predator node always negative

Community Network	Network Structure	Number of Nodes	Number of Links	Single-node perturbations	Multiple-node perturbations
Food Chain		3	2	<ul style="list-style-type: none"> ● Pressed node and any nodes above it on the food chain always exhibits negative responses, hence the top node is always negative ● Node directly below pressed node exhibit positive response 	<ul style="list-style-type: none"> ● Top node always negative ● Middle and bottom node may be positive depending on parameter values and which nodes were pressed
Keystone Predation		4	4	<ul style="list-style-type: none"> ● If the top node is pressed, middle nodes exhibit high (>70%) chance of positive response while the top and bottom node is always negative ● If one of the middle nodes is pressed, the other middle node is positive, the top node has a high (>70%) chance of negative and the bottom node has a high (>70%) chance of positive ● If the bottom node is pressed, the top and bottom node is negative while middle nodes have low (<30%) chance of positive 	<ul style="list-style-type: none"> ● Outcome highly variable depending on parameter values and nodes pressed

Community Network	Network Structure	Number of Nodes	Number of Links	Single-node perturbations	Multiple-node perturbations
Omnivory Food Chain		3	3	<ul style="list-style-type: none"> • Nodes above pressed node has a medium (30%-70%) chance of positive response depending on parameter values 	<ul style="list-style-type: none"> • Outcome highly variable depending on parameter values and nodes pressed
Long Food Chain		5	4	<ul style="list-style-type: none"> • Pressed node and any nodes above it on the food chain always exhibits negative response, hence the top node is always negative • Node directly below pressed node exhibit positive response, the next node underneath is negative and the pattern of positive and negative response alternates down the food chain (i.e., a trophic cascade) 	<ul style="list-style-type: none"> • Outcome highly variable depending on parameter values and nodes pressed
Omnivory Long Food Chain		5	7	<ul style="list-style-type: none"> • Outcome highly variable depending on parameter values and nodes pressed • More mixed results compared to food chain without omnivory 	<ul style="list-style-type: none"> • Outcome highly variable depending on parameter values and nodes pressed

Community Network	Network Structure	Number of Nodes	Number of Links	Single-node perturbations	Multiple-node perturbations
Wasp Waist		5	4	<ul style="list-style-type: none"> ● If a top node is pressed, its competitor and the middle nodes are positive while the bottom and pressed nodes are negative ● If the middle node is pressed, both bottom nodes are positive and all other nodes are negative ● If a bottom node is pressed, its competitor is positive and all other nodes are negative 	<ul style="list-style-type: none"> ● Outcome highly variable depending on parameter values and nodes pressed but the middle node is always the node with the highest or among the nodes with the highest chance of a positive response
Wide Base Food Chain		4	3	<ul style="list-style-type: none"> ● If the top or middle node is pressed, the node(s) directly below it is positive while all others are negative ● If one of the bottom nodes is pressed, its competitor is positive while all other nodes are negative 	<ul style="list-style-type: none"> ● Outcome highly variable depending on parameter values and nodes pressed
Keystone Predation with Mutualism		4	5	<ul style="list-style-type: none"> ● Outcome highly variable depending on parameter values and nodes pressed 	<ul style="list-style-type: none"> ● Outcome highly variable depending on parameter values and nodes pressed

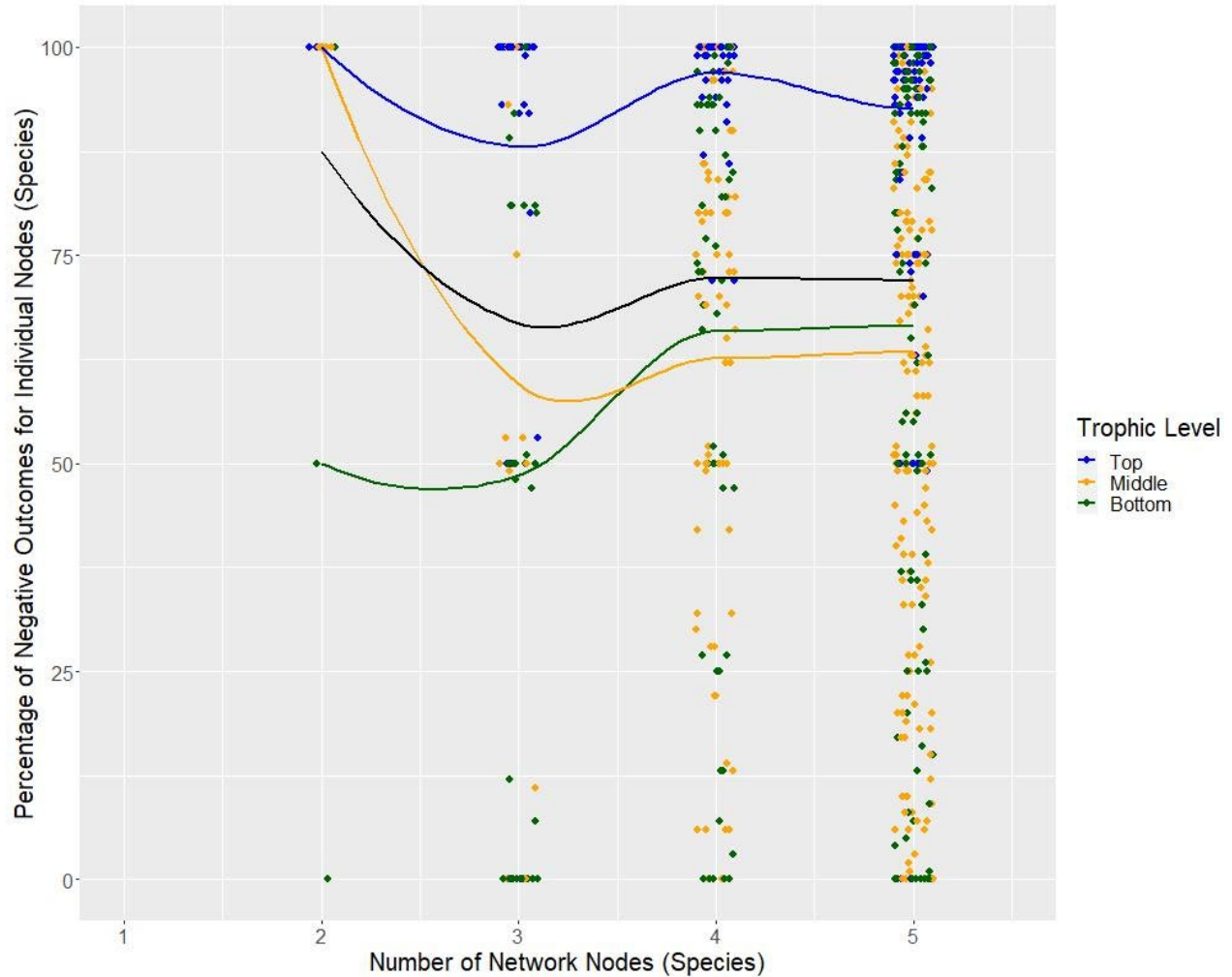


Figure 3.2. The percentage of negative outcomes in response to perturbations as a function of the numbers of nodes in a network. Each point represents the proportion of negative outcomes recorded for an individual node within a particular network across 10,000 simulations. The results are divided into three groups depending on the trophic level of the response node: Top (blue), Middle (orange), and Bottom (green). The three colored lines are loess smoothed curves corresponding to their trophic level while the black line is the curve for all data. One of the two-node networks was Mutualism and both nodes in the network were classified as belonging to the middle trophic level since neither consumes the other. This caused the middle trophic level loess curve to be pulled up on the left end. Jittering on the x-axis was added to reduce data overlay.

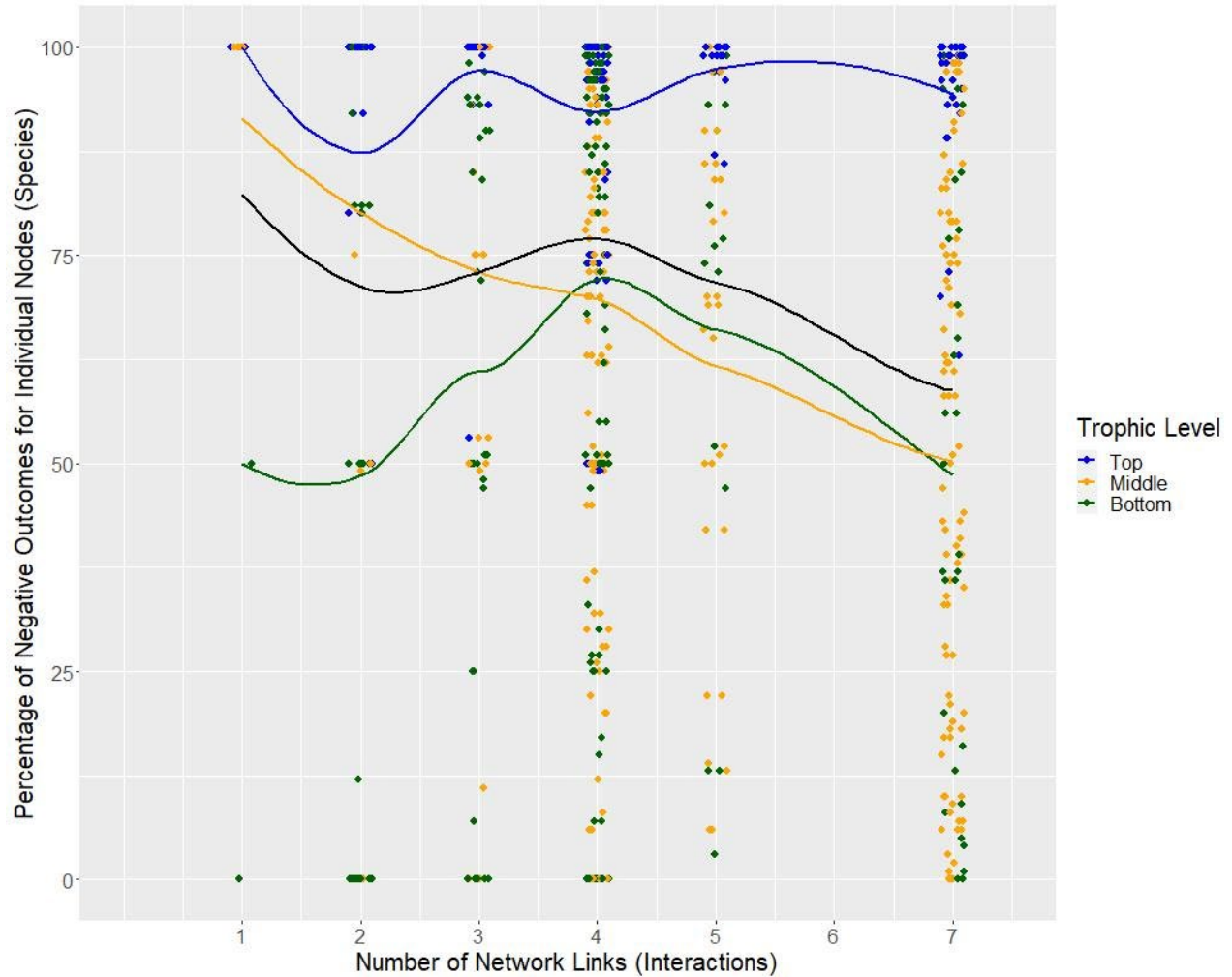


Figure 3.3. The percentage of negative outcomes in response to perturbations as a function of the number of links in a network. Each point represents the proportion of negative outcomes recorded for an individual node within a particular network across 10,000 simulations. The results are divided into three groups depending on the trophic level of the response node: Top (blue), Middle (orange), and Bottom (green). The three colored lines are loess smoothed curves corresponding to their trophic level while the black line is the curve for all data. One of the one-link networks was Mutualism and both nodes in the network were classified as belonging to the middle trophic level since neither consumes the other. This caused the middle trophic level loess curve to be pulled up on the left end. Jittering on the x-axis was added to reduce data overlay.

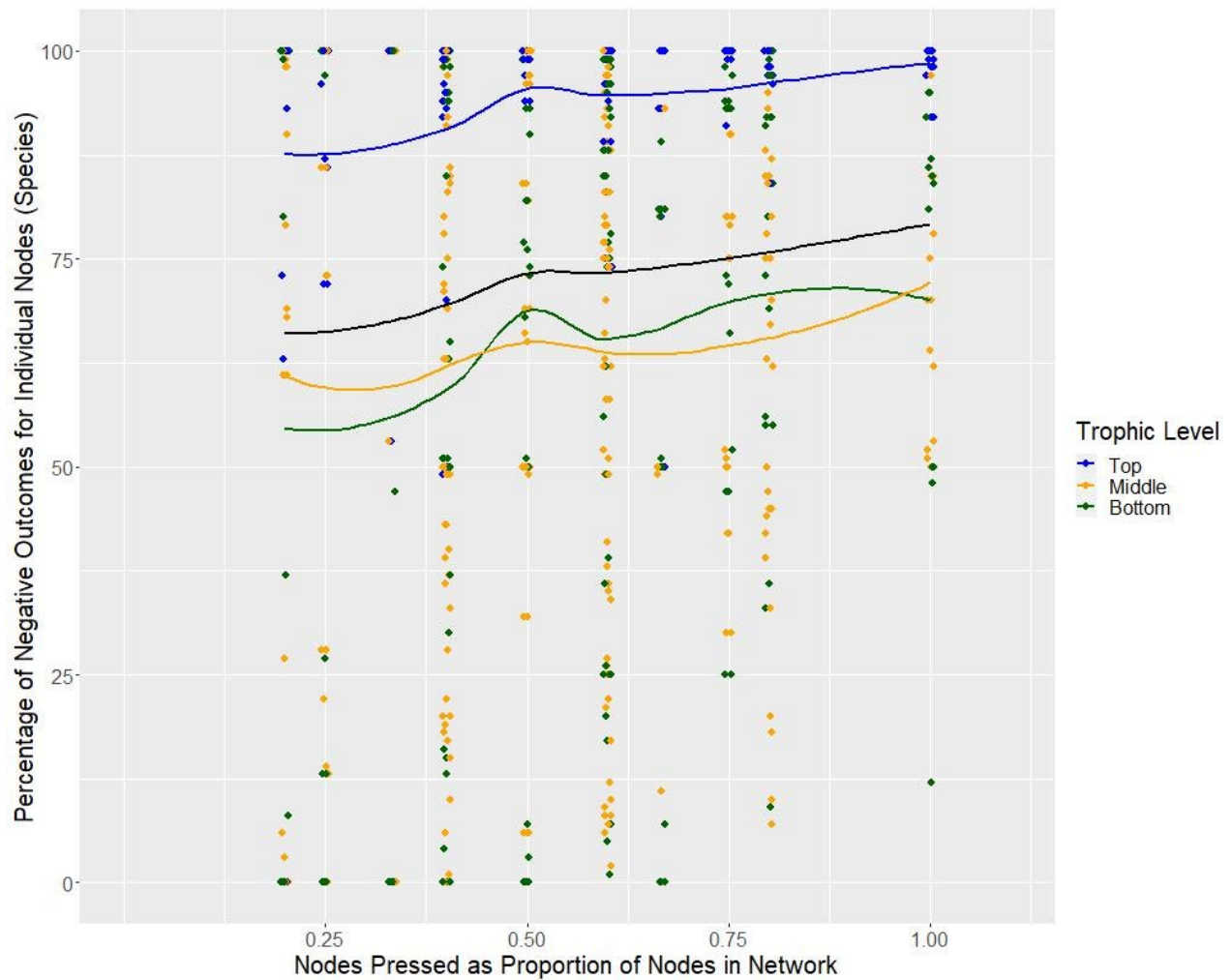


Figure 3.4. The percentage of negative outcomes in response to perturbations as a function of the proportion of nodes perturbed. Each point represents the proportion of negative outcomes recorded for an individual node within a particular network across 10,000 simulations. The results are divided into three groups depending on the trophic level of the response node: Top (blue), Middle (orange), and Bottom (green). The three colored lines are loess smoothed curves corresponding to their trophic level while the black line is the curve for all data. Jittering on the x-axis was added to reduce data overlay.

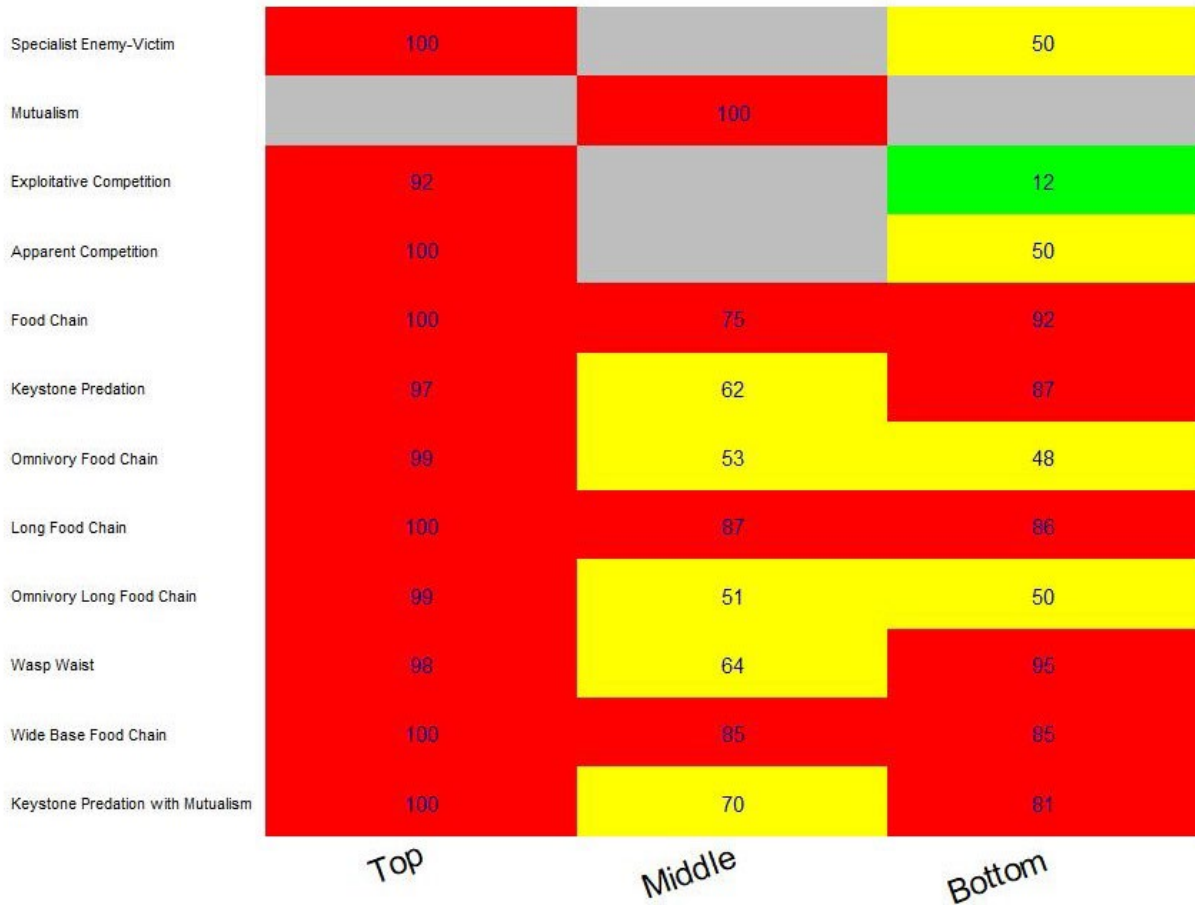


Figure 3.5. Heat map of the simulation results for the twelve community networks under the scenario where all nodes were negatively perturbed. The columns represent node(s) at the top, middle or bottom trophic levels of the network. The numbers represent the percentage of negative outcomes across all simulations. Each cell is coloured based on the percentage of negative outcomes with red being 70%-100%, yellow being 30%-70%, and green being 0%-30%.

4. CONSEQUENCES OF FISH MORTALITY AND CONSIDERATIONS FOR DECISIONS RELATED TO THE DEATH OF FISH

The response of populations or communities to fish mortality are dependent on a range of biological and ecological conditions. Consideration of the factors outlined below would inform decisions related to authorizing the death of fish under the *Fisheries Act* and in the application of a risk-based approach to implementing the fish and fish habitat protection provisions (DFO 2019a). These factors may also contribute to considerations around the effectiveness of offsetting plans, as the timing and patterns of response to fish mortality will influence the way in which offsets support population and ecosystem recovery.

4.1. AMOUNT OF FISH MORTALITY

The amount of fish mortality (e.g., number of individuals killed, number of age equivalents, etc., see Section 2 above) is a primary consideration for understanding population and ecosystem responses to fish mortality. Higher levels of fish mortality are produced when mortality events are more frequent, when mortality occurs over longer periods, and when more individuals are killed per event. Depending on the mechanism of mortality (see below) and the type of WUA, it may be difficult to estimate the magnitude of fish mortality, especially if mortality is indirect or delayed and fish disperse from the area of the WUA prior to dying. In addition, the amount of fish mortality caused by a given WUA can change over time in response to changes in population dynamics and behaviour in impacted species, and to changes in environmental conditions.

4.2. POPULATION SIZE AND TRAJECTORY

The size and trajectory of fish populations is highly important in determining the impacts of fish mortality on population abundance. Generally, populations with high abundances and those that are growing will be less sensitive to impacts from mortality. This is because larger populations have more reproductive individuals and are therefore able to more rapidly replace lost individuals. Furthermore, the impacts of mortality on population size would be partially mitigated by positive population growth rates, such that growing populations will more rapidly recover and have lower extinction risks associated with a given level of mortality compared with populations that are in decline.

Populations that have been substantially depleted by other stressors, those that are naturally small, and those that are shrinking (i.e., have negative population growth rates) will be more severely impacted by fish mortality. Small populations can take longer to replace a given number of individuals due to fewer reproductive adults. In addition, small populations are at greater risk of extinction resulting from reproductive depensation (see below), inbreeding and genetic bottlenecks (Meffe 1986), and demographic stochasticity (Gabriel and Burger 1992). Fish mortality that further decreases the population size of already small populations will further exacerbate these risks.

4.3. SPATIAL STRUCTURE OF POPULATIONS

Determining the size and trajectory of a population (see above) depends on defining the scope of the population under consideration. In many aquatic systems, especially larger water bodies (e.g., the Laurentian Great Lakes), fish have a patchy distribution associated with the heterogeneous distribution of habitat features. Depending on their size, groups of individuals (of the same species) that are spatially separated from other groups may be described as a local population (or 'sub-population') within a larger meta-population. However, there are no clear scientific criteria by which to define a 'local population' for considering the impacts of mortality

(Berryman 2002). Instead, determining the scope of the population for assessing the impacts of fish mortality should depend on the goals of management. For example, if the goal of management is to maintain the presence and abundance of a species within each bay of a large lake, then the impacts should be considered relative to the size and trajectory of the population in the impacted bay. In addition, while localized mortality may be relatively less impactful when management goals are defined at larger spatial scales, it is important to understand that the number of stressors acting on larger populations is likely to be greater. As a result, it is more important to consider cumulative effects, and the impacts of interactions between different mortality sources (see below), when populations are defined at large spatial scales.

An additional consideration is that connectivity between local populations (or sub-populations) can alter the impacts of fish mortality. When the rate of fish movement is high, declines in local population abundance (e.g., resulting from mortality events) can be mitigated by immigration from neighbouring areas. These 'rescue effects' can increase the population growth rate and lower the probability of extinction for sub-populations that are well connected (Schtickzelle and Quinn 2007; Bellard and Huguény 2020). Conversely, isolated populations or sub-populations have a higher probability of extinction from natural or human-induced mortality. Connectivity between sub-populations is impacted by the physical and physiological characteristics of the species involved (i.e., whether they are good or poor dispersers), the physical structure of waterbodies (including the spatial distribution of habitat patches), and anthropogenic activities that alter habitat connectivity (e.g., the construction of dams and other barriers).

4.4. LIFE HISTORY, LIFE STAGE IMPACTED, AND DENSITY DEPENDENCE

Fish species and populations vary in their life history in a manner that could modify the impact of mortality. Variation in life history is sometimes characterized along an axis of slow to fast life histories, with the former represented by species that are long-lived, reach reproductive maturity later in life, and have relatively low reproductive rates. The population growth rate of species with slow life histories is strongly driven by changes in adult survival rate (Sæther and Bakke 2000; Vélez-Espino et al. 2006; van der Lee and Koops 2016), and as such, mortality acting on adult fish in these species will cause a greater impact on population size than mortality on juvenile fish. Conversely, species with fast life histories (shorter-lived, rapidly reproducing species) have population growth rates that are less sensitive to the rate of adult survival and are relatively more sensitive to changes in fecundity and juvenile survival.

A species' habitat requirements and migratory life history may also impact their exposure to sources of mortality. When mortality sources are concentrated in a species' habitat or in habitats that must be traversed, a species may not be able to avoid interacting with those mortality sources. Alternatively, fish that are habitat generalists (or specialists for habitats that do not contain mortality sources), may be able to avoid exposure to mortality sources.

Additionally, mortality events can act as an evolutionary force that leads to shifts in life history within a population. For example, the well documented effects of size-selective fishing on early life growth rates and age of maturity (Kuparinen and Festa-Bianchet 2017; Perälä and Kuparinen 2020) are not unique to fisheries and could be caused by any size-biased mortality source. Similarly, if fish mortality differentially targets individuals according to whether they exhibit migratory behaviour, or whether they adopt alternative reproductive strategies (Gross 1996), then the distribution of such phenotypes within a population can shift. Such evolutionary responses to mortality can lead to changes in population dynamics (e.g., productivity and vital rates) through complex eco-evolutionary dynamics (Bell and Gonzalez 2011; Cameron et al. 2013) and can further exacerbate or mitigate the impacts of the original mortality.

Within fish populations, the per-capita rates of reproduction, growth and survival (i.e., vital rates) often vary with the density of individuals. Population vital rates are typically higher when population density is low due to decreased intraspecific competition. As a result, density dependence in vital rates can theoretically serve as a compensatory response that mitigates the impacts of mortality events on fish populations. However, many natural populations exhibit complex forms of density dependence. For example, decreases in per capita reproduction or survival can occur at low population densities. This so-called ‘Allee effect’ (Stephens et al. 1999), also known as ‘reproductive depensation’, can lead to extinction vortexes, such that populations at low density may not recover despite low degrees of intraspecific competition (Courchamp et al. 1999). Allee effects are expected to be common in species that gain substantial benefits from living in social groups, for example when schooling protects individuals from predators or facilitates mate finding (Gascoigne and Lipcius 2004; Gascoigne et al. 2009). In such cases, reductions in population density can have a negative effect on vital rates and population growth rates. Importantly, the shape of density-dependent relationships can vary (both among populations and among life stages within a population) and is often non-linear (Hodgson et al. 2017). The consequence of this variation is that a density-dependent compensatory response to mortality is not expected to occur across all species, life stages, and population densities

4.5. INTERACTIONS WITH OTHER SOURCES OF MORTALITY

A variety of anthropogenic and natural stressors can impact fish populations in a way that interacts with fish mortality. Stressors such as fisheries, other WUA, extreme environmental conditions, invasive species, or pollutants can co-occur with WUA-related residual mortality and may be either chronic or acute in nature. Importantly, mortality resulting from multiple stressors can combine to have effects that are different from the sum of the individual mortality effects (e.g., Côté et al. 2016). Indeed, several meta-analyses have demonstrated that fish populations often experience total mortality that is greater than (sometimes termed ‘synergistic effects’) or less than (sometimes termed ‘antagonistic effects’) the additive prediction from multiple sources of mortality (Darling and Côté 2008; Jackson et al. 2016). Importantly, some synergistic stressor combinations can dramatically diverge from additive predictions – for example, while independent exposure to a parasite and a pesticide each caused < 5% mortality in populations of *Daphnia magna*, simultaneous exposure to these two stressors caused mortality rates > 70% (Coors and DeMeester 2008).

Theory predicts that factors such as the mechanism of action of the stressors (Schäfer and Piggott 2018), the ecological context (Lenihan et al. 1999; Kroeker et al. 2013), and the form of density-dependence acting within a population (Hodgson et al. 2017), can all impact the amount of total mortality resulting from multiple stressors. However, there is currently a poor ability to predict the impacts of multiple stressors on mortality in most natural systems (Orr et al. 2020; Dey and Koops 2021).

4.6. DURATION, TIMING, AND MECHANISM OF MORTALITY

The mechanism and duration of fish mortality can alter the impacts on populations and ecosystems. When mortality occurs before major density-dependent life history stages, the impacts of the mortality may be partially mitigated by the increased survival of the remaining individuals (e.g., if mortality occurs just prior to winter for species where winter survival is inversely related to the number of individuals in the population). Conversely, mortality will have a greater impact on a population if it occurs after density-dependent life history stages. Understanding these dynamics would require detailed knowledge of how density-dependence acts on a population, which is not available for most populations and species, with general

expectations that density-dependence most commonly acts early in life still the subject of active research (e.g., Andersen et al. 2017; Lorenzen and Camp 2019).

A further consideration is whether the mechanism of mortality allows for the accurate quantification of the amount and frequency of fish mortality. Some WUA may cause substantial degrees of undetected mortality, which may be more likely when mortality is spatially separated from the WUA itself (e.g., downstream mortality from a pollutant spill), or if the killed fish are hidden (e.g., during infilling). Similarly, when interactions with WUA cause delayed mortality, then killed fish may leave the area of the WUA before dying and may be more difficult to detect.

In addition, some mechanisms of mortality may remove the biomass present in killed fish from the ecosystem. Mortality causing biomass removal could exacerbate impacts on the affected population or ecosystem, because the energy and nutrients present in killed fish will not be reincorporated into the food web via scavengers and lower trophic level organisms. As a result, mortality causing biomass removal will generally have more detrimental ecosystem effects than mortality without biomass removal. However, when fish killed by WUA are not removed from an ecosystem, it may be more difficult to accurately quantify the amount and frequency of mortality, or the identity of the species killed. Furthermore, biomass removal could lower pollutant levels within the ecosystem, especially if the pollutants are biomagnified and the killed fish are of a high trophic level (Gobas et al. 1999). Finally, s.36(1)(b) of the *Fisheries Act* prohibits fish offal from being left on the shore, beach or bank of any water, and must be considered separately from issues related to nutrient and pollutant cycling.

4.7. BIOTIC INTERACTIONS AND ECOSYSTEM IMPACTS

Fish mortality can have complex impacts on aquatic ecosystems through food webs and other biotic interactions. Fish species exist as members of aquatic communities and interact with other species through predator-prey relationships, host-parasite relationships, competition, mutualisms, and commensalisms. Changes in abundance of one species can impact other species directly through these interactions, or indirectly through species interaction networks (section 3, see also Gilman et al. 2010).

The impact that mortality acting on a single species has on aquatic ecosystems will depend on the type and strength of biotic interactions between the impacted species and other community members. These impacts can be estimated with multispecies community models (see e.g., section 3). However, there is a high degree of uncertainty surrounding the pattern and strength of biotic interactions in many aquatic systems, which poses challenges for robust predictions.

In some cases, fish mortality could also alter the patterns of biotic interactions in aquatic ecosystems. For example, fish mortality resulting from entrainment through turbines could allow downstream predators and scavengers access to food sources they otherwise may not access. In addition, WUA can indirectly lead to fish mortality by increasing the susceptibility of individuals to predation. Such an effect could occur if individuals are physically forced into exposed areas or are temporarily disoriented or disabled by interacting with a WUA. Such changes can alter food webs and have complex impacts on energy and nutrient fluxes within ecosystems.

WUA may also cause mortality on multiple species within a given community (see section 1, section 3). Research on harvest of multiple species in aquatic systems has demonstrated that the level of maximum sustainable yield within a community is less than the sum of maximum sustainable yields calculated for the component species (e.g., Steele et al. 2011; Link et al. 2012), and that individual species have lower maximum sustainable yields when harvest occurs on other community members (Walters et al. 2005). Applying this insight to death of fish by means other than fishing, suggests that aquatic ecosystems can experience severe

deterioration from fish mortality even if each species is impacted at a level below their maximum sustainable yield, and can have implications for other management objectives.

Furthermore, in many cases there may be uncertainty around which species are killed by WUA. This uncertainty can be caused by a lack of information about which species are present, and how the WUA interacts with the behaviour and life history of each of these species. In addition to contributing to difficulty in understanding the ecosystem impacts, such a case also creates challenges in developing appropriate offsets and estimating equivalence because it will not be clear which offsets are in-kind versus out-of-kind (Clarke and Bradford 2014). Similarly, uncertainty around which species are killed by WUA could increase the risk that an offset fails to achieve its goal (for example, if an offset attempts to increase production of predatory species but their supporting prey base has been greatly reduced by WUAs).

4.8. MANAGEMENT OBJECTIVES

The degree of impact of fish mortality further relates to the management objectives set for the focal population or ecosystem. Under s.34.1(1)(b) of the *Fisheries Act*, the Minister must consider relevant fisheries management objectives prior to considering an authorization for the death of fish by means other than fishing. Fish mortality may impact fisheries by decreasing the abundance or biomass of harvested stocks, and by increasing the total mortality rate on those stocks. These factors are key components of fisheries decision-making under DFO's Sustainable Fisheries Framework.

Additionally, fish mortality may impact fisheries indirectly by influencing the abundance of predators, prey, and competitors of harvested species. For example, forage fish often support dependent predators and mortality acting on forage fish is likely to impact the health of predator populations. Similarly, predator populations impact competition between lower trophic level species, and can impact species richness (Leibold 1996). Section 2.5(a) of the *Fisheries Act* allows the Minister to consider such an ecosystem approach in decision-making and could include the ecosystem impacts of WUA-related residual mortality. Such an approach could also consider the impacts of changing environmental or habitat conditions on the sensitivity of populations to mortality, as well as the long-term effectiveness of offsets.

In addition, the social acceptability of fish mortality may differ depending on the species impacted and the mechanism of mortality. Many stake holders and rights holders derive significant cultural value from fish and may be strongly opposed to fish mortality. The degree of acceptance of fish mortality may differ with factors such as (i) the speed at which death occurs to individual fish, (ii) whether the species impacted are part of traditional or subsistence fisheries, or (iii) whether the killed fish are viewed as wasted (i.e., are not available for ecosystem services or for human use). However, the specific form and amount of mortality that is accepted by a stakeholder group will also relate to social and cultural traditions and may therefore differ among systems. In addition to cultural impacts, fish mortality could have indirect economic and social consequences, for example if the presence of killed fish negatively impacts ecotourism or recreational opportunities (e.g., swimming, fishing) associated with aquatic systems. Again, these impacts may depend on the timing, mechanism of mortality, and species impacted. The consideration of these factors in the Minister's decisions is permitted under Section 2.5(g) of the *Fisheries Act*.

Furthermore, fish mortality may directly impact species at risk and hinder the ability to meet management goals for these species. Under Section 73(3)(c) of the *Species at Risk Act*, permitted impacts to species at risk must not jeopardize the survival or recovery of species at risk. Where WUA cause mortality to species at risk, or to species with strong biotic interactions with species at risk (e.g., important prey), it may be necessary to consider whether such

mortality jeopardizes the survival or recovery of the species at risk. In such cases, DFO's Recovery Potential Assessments, which typically include an analysis of allowable harm, may be a valuable source of information for considering whether fish mortality will jeopardize the species survival or recovery.

The simultaneous consideration of the management objectives described above, as well as coordination with the management activities of other institutions (e.g., other federal ministries, provincial and territorial ministries, conservation authorities, and Indigenous organizations with management authority over aspects of aquatic systems or connected terrestrial systems), requires an integrated approach to management (e.g., Randall et al. 2011). Such an approach should explicitly consider the trade-offs associated with different management options, as well as any important aspects of ecosystem management that fall in gaps between the mandates of different jurisdictions.

4.9. DOES THE EFFECT ON LOCAL FISH POPULATION OR COMMUNITIES CHANGE WITH RESPECT TO WHEN AND HOW FREQUENTLY FISH ARE KILLED?

The response of the population or ecosystem to fish mortality will change with respect to when mortality events occur. Different species may be vulnerable to mortality depending on the timing of different WUA (e.g., the specific timing of changes in downstream flow regulated by dams). This is because different species exhibit different patterns of habitat use and behaviour across the daily (and annual) cycle and are therefore more or less likely to interact with WUA at different times of day (or year).

Second, the timing of mortality events relates to the life stage(s) impacted by mortality. The distribution of age and size classes present in a given fish population varies throughout the year according to the timing of reproduction (and growth and survival rates). As such, WUA may cause mortality on different life stages depending on the time of year, which has differential impacts on population dynamics (see Life History, Life Stage Impacted, and Density Dependence, above).

Fish mortality will also have a larger impact on fish populations when it occurs after life-history events in which survival, reproduction, or growth is negatively density-dependent. Conversely, when fish mortality occurs before negatively density-dependent life history events, there should be less impact on populations. However, the timing of the action of density-dependence is unknown for many species and populations (Lorenzen and Camp 2019). Additionally, density-dependence can act on subsequent life-history stages, such that mortality occurring before one density-dependent stage may also be occurring after a previous density-dependent life history stage. As a result, it is challenging to predict the way in which timing of mortality will impact density-dependent processes occurring within fish populations.

All else being equal, more frequent mortality will cause a greater number of individuals to be killed and will therefore have a more negative impact on the population or community.

In addition, the frequency of mortality relates to the opportunity for populations to recover from previous mortality. For populations with high growth rates and strong density-dependence, decreases in abundance due to mortality will result in an increase in reproduction (and/or growth) of the surviving individuals. If mortality events are sufficiently infrequent, such a population may be able to fully recover without intervention.

4.10. WHAT IS THE IMPACT OF BEING WRONG ABOUT EITHER THE AMOUNT OF MORTALITY OR THE SENSITIVITY OF THE POPULATION OR ECOSYSTEM?

Fish mortality has the potential to cause serious harm to fish populations and aquatic ecosystems if the amount of mortality or the sensitivity of the population or ecosystem are underestimated. Such harm can lead to a variety of negative outcomes including negative impacts to fisheries, impaired ecosystem function and resilience, and a loss of ecosystem services.

Risk equivalency is a useful concept for managing uncertainty related to mortality and population or ecosystem sensitivity. In a resource management context, risk equivalency is the concept of maintaining a set level of risk that a management decision will fail to achieve an objective (Fulton et al. 2016). For considerations related to the impacts of fish mortality, a risk equivalent approach would attempt to maintain a set level of risk (e.g., 5%) that a decision will fail to meet policy objectives for the management of mortality. This level of risk would be maintained across different management decisions despite variable levels of uncertainty related to the magnitude of mortality and the ecosystem's sensitivity to mortality. Typically, approaches to achieve risk equivalency involve the use of precautionary buffers intended to reduce the marginal impact of human activities as uncertainty in assessing the risks increases (Duplisea et al. 2020).

In the context of fish mortality from WUA, such buffers could be exercised in the form of compensation ratios for offsets, in the requirements surrounding the timing of implementation of offsets (e.g., a requirement to demonstrate the performance of offsets prior to authorization of WUA-related mortality would decrease the risk of offsets failing to achieve their target benefit), or in the size and (or) trajectory of the stock at which WUA-related residual mortality is no longer authorized (i.e. the level of risk at which the application is denied). To achieve risk equivalency, precautionary buffers would seek to achieve comparably low risks of serious harm to the population when compared with similar projects with lower uncertainty.

4.11. CAN A PRECAUTIONARY FRAMEWORK BE USED TO SUPPORT MANAGEMENT OF WUA RELATED RESIDUAL MORTALITY?

A precautionary, risk-based approach to management could support understanding the consequences of population collapse and ecosystem harm resulting from additional (WUA related) mortality. Such an approach could be adapted from tools used to manage harvest in fish populations. Precautionary frameworks for managing harvest mortality are well developed and in many cases are also internationally standardized. DFO's Sustainable Fishing Framework (2009) is one such framework which contains guidance for setting harvest rates according to the status of fish stocks including the suggestion that: (i) harvest rates be kept at an absolute minimum when stocks are in the critical zone, (ii) that harvest rates be progressively decreased to promote stock rebuilding when stocks are in the cautious zone, and (iii) that harvest rates not exceed established maximums when stocks are in the healthy zone. The use of a common framework by fisheries and habitat managers for managing mortality would allow for the leveraging of data and information across different decision-making contexts.

Below, we demonstrate how an adaptation of the International Council for Exploration of Seas (ICES) Precautionary Approach to Fisheries Management (ICES 1998, 2002a) could be used to help inform decisions about WUA-related residual mortality for population-level changes in abundance. This framework is used to generate science advice on harvest according to the status of the stock and the current harvest rate, has a large body of research and guidance to support its usage (e.g., ICES 2002b, 2017; Lassen et al. 2014; Magnusson et al. 2018), and

conceptually aligns with DFO's Precautionary Harvest Strategy and Sustainable Fisheries Framework (DFO 2006, 2009).

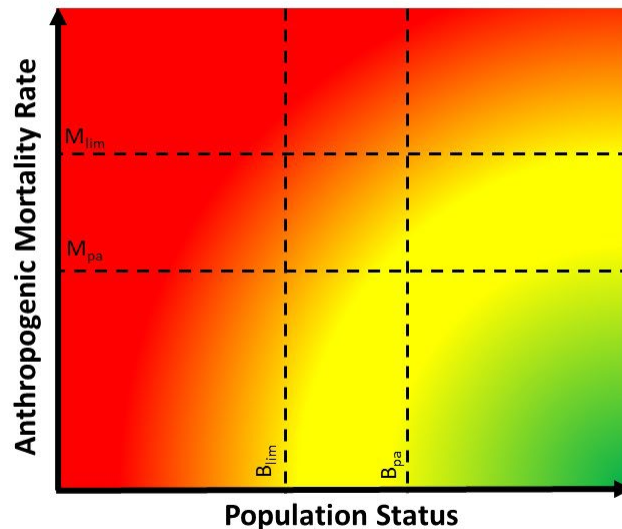


Figure 4.1. An adaptation of the ICES precautionary approach for supporting decision-making surrounding WUA-related residual mortality showing the four key reference points B_{lim} , the limit population biomass; B_{pa} , the precautionary population biomass; M_{lim} , the limit mortality level; and M_{pa} , the precautionary mortality level. Background colouration of the figure shows the relative risk of population decline and ecosystem harm, with red indicating a greater risk and green indicating a lower risk .

Briefly, the framework is defined by the location of four reference values (B_{lim} , the limit population biomass; B_{pa} , the precautionary population biomass; M_{lim} , the limit mortality level; and M_{pa} , the precautionary mortality level), along two primary axes (the total instantaneous rate of anthropogenic mortality including fishing mortality (F) and other anthropogenic mortality (A), and the biomass (status) of the population). Note that the ICES framework and all associated documentation uses mortality references F_{lim} and F_{pa} (rather than M_{lim} and M_{pa}) to indicate the values are specific to fishing mortality, but this framework has been adapted in the current document to also include other anthropogenic mortality such as that caused by WUA.

Conceptually, B_{lim} represents the threshold for population abundance (status) that should be exceeded to have a high likelihood of avoiding further population decline and ecosystem harm. B_{lim} is usually set as the population biomass below which recruitment declines with further declines in biomass, and is typically identified using a segmented regression between population biomass and recruitment (ICES 2002b, 2017). M_{lim} then, is the mortality rate that if exceeded, will lead to a population status below B_{lim} , and is set as the mortality rate that in stochastic equilibrium will result in a median population biomass equivalent to B_{lim} . Further guidance for setting B_{lim} and M_{lim} values can be found in (ICES 2017). The precautionary values (B_{pa} and M_{pa}) represent targets for mortality and the status of the stock that would ensure a high likelihood of maintaining the population's health, after accounting for uncertainty in the estimation of the true mortality rate and population status. Therefore, the location of B_{pa} and M_{pa} relative to B_{lim} and M_{lim} , are the primary mechanism through which risk due to uncertainty (i.e.

risk equivalency) can be accounted for in this framework. When uncertainty related to the true population status, or the amount and impact of mortality is high, the precautionary limits should be set further from B_{lim} and M_{lim} . Conversely, when uncertainty is lower, precautionary limits may be set closer to B_{lim} and M_{lim} . ICES guidance suggests a default value of B_{pa} as 1.4 times the B_{lim} and a default value of M_{pa} as 0.71 times the M_{lim} in the absence of population-specific data (ICES 2017).

In the context of use as a WUA-related mortality decision support tool, this framework would next require estimates of the population status and total instantaneous rate of anthropogenic mortality with and without the inclusion of WUA-related residual mortality and proposed offsets (Figure 4.2). In cases where WUA cause a one-time mortality event, the initial location of the population on the framework would be shifted to the left, as the mortality event will degrade the population status (Figure 4.2a). Where WUA result in ongoing mortality to a population, WUA will cause both a degradation in the population status and an increase in the total anthropogenic mortality rate, indicated by a shift to the upper-left of the framework (Figure 4.2b). Offsets would typically move the population to the right of the framework, by improving the status of the population without impacting the mortality rate (Figure 4.2a,b). The estimated position of the population would then be compared to the background colouration of the framework, to determine the level of risk for harm (red = high risk of population decline and ecosystem harm, green = low risk of population decline and ecosystem harm, Figure 4.2).

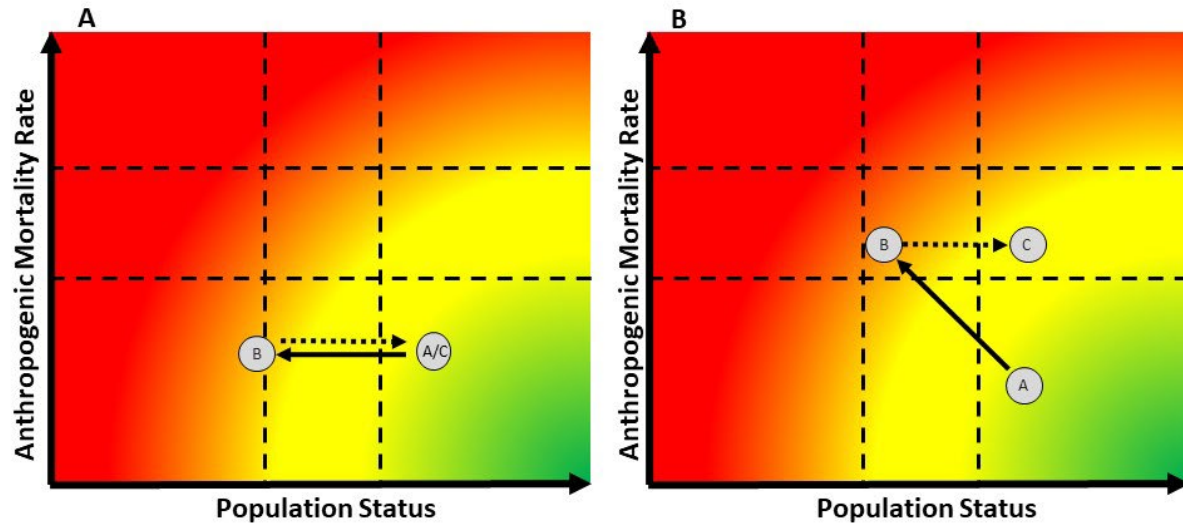


Figure 4.2. Example of the application of the adapted ICES precautionary approach framework to support population-level decisions related to WUA-related residual mortality, for a one-off mortality event (panel A) and recurring or continuous mortality (panel B). In both panels, point A indicates the initial (baseline) population status and mortality rate, whereas the solid line from point A to point B indicates the impacts of WUA-related residual mortality. The dashed line from point B to point C indicates the effect of offsetting. A numerical example of the situation described in 4.2B (ongoing mortality) is provide in Table 4.1.

Table 4.1. Numerical example demonstrating the calculations used to determine population status and anthropogenic mortality rates for a situation involving ongoing mortality and offsetting, such as demonstrated in Figure 4.2B. Note that population status, which is typically expressed in biomass, is estimated using abundance in this example.

Initial conditions (point A)		
Population status (N_A)	100000	Population size at point A
Anthropogenic fish deaths (D_A)	4877	Ongoing finite mortality rate (per unit time) at point A
Anthropogenic mortality rate (M_A)	0.05	$M_A = -\ln(1 - (D_A / N_A))$
Conditions with additional mortality from WUA (point B)		
WUA related fish deaths ($D_{A \rightarrow B}$)	10000	Hypothetical ongoing mortality (per unit time) from WUA
Population status (N_B)	90000	$N_B = N_A - D_{A \rightarrow B}$
Total anthropogenic fish deaths (D_B)	14877	$D_B = D_A + D_{A \rightarrow B}$
Anthropogenic mortality rate (M_B)	0.16	$M_B = -\ln(1 - (D_B / N_A))$
Conditions with additional mortality from WUA and offset (point C)		
Offset (O_C)	10000	Ongoing gains in abundance from hypothetical offset (per unit time)
Population status (N_C)	100000	Population size after accounting for offset $N_C = N_B + O_C$
Anthropogenic mortality rate (M_C)	0.16	$M_C = -\ln(1 - (D_B / N_A))$

This framework would allow for the incorporation of the various considerations related to the consequences of fish mortality that are described above, when data are available. For example, the anthropogenic mortality rate, which is represented by the position along the y-axis in the framework, would be influenced by the amount of WUA-related mortality and by additional (multiple) stressors that cause mortality. Population status, represented by the position along the x-axis, would be impacted by considerations of the spatial structure of populations, by population size and trajectory, and by WUA-related and other forms of mortality. The biological reference limits (M_{lim} and B_{lim}), which are set based on biological characteristics, would include considerations of species' life history, the life stage(s) impacted, density-dependence, population connectivity, and the impact of WUA on biotic interactions. Finally, the setting of the precautionary limits (B_{pa} and M_{pa}) could include consideration of the management objectives and of the mechanism of mortality.

A quantitative application of the framework described above requires an estimate of the stock status (population size) and the instantaneous mortality rate with and without additional WUA-related mortality. In addition, data on population dynamics (i.e., stock recruitment curves) is required to set the reference values (B_{lim} , B_{pa} , M_{lim} , M_{pa}). The leveraging of data collected for fisheries management would help to enable a quantitative application of the framework for management WUA related mortality. However, considerable guidance has also been developed for applying the ICES fisheries management framework when stocks differ in the amount of data available (ICES 2012) and analogous guidance for the qualitative and semi-quantitative application of the framework described above could be developed. This framework could also allow for the incorporation of an ecosystem approach to managing mortality in a manner analogous to that used in an ecosystem approach to fisheries management (EAFM; Morishita 2008). An EAFM primarily retains a focus on individual stocks while incorporating ecosystem factors (e.g., environmental conditions, condition of other co-occurring fish populations, and habitat conditions) to better inform decision-making. Furthermore, such an approach acknowledges that the environmental and ecological conditions underlying productivity in fish stocks are not stationary (i.e., they are spatially and temporally variable), and may be shifting over time (e.g., changes in global climate). In the context of WUA-related mortality, ecosystem factors could be considered in setting the four reference values (B_{lim} , B_{pa} , M_{lim} , M_{pa}), as well as to help understand the future performance of offsets (i.e., the B->C arrow in Figure 4.2, see also Section 5). Such an approach is already implemented in many of DFO's Integrated Fisheries Management Plans.

Future work on managing fish mortality should work towards an ecosystem-based management (EBM) approach, which is an integrated, holistic and synthetic approach that coordinates and accounts for all factors influencing ecosystem structure and function (Link 2010). While there is widespread support for EBM in theory, the operational basis for EBM has been hindered by challenges in defining meaningful reference values for desirable and undesirable states of the ecosystem, and by a dearth in understanding of how different management actions impact ecosystem state. Concepts such as viability kernels (e.g., Cury et al. 2005) and safe operating spaces (e.g., Carpenter et al. 2017) can help to define both desirable ecosystem states, as well as the management actions that can help to maintain ecosystems in those states. Such approaches could provide for an operational EBM framework within DFO (e.g., Duplisea et al. 2020).

4.11.1. Summary

Decisions for authorizing WUA-related residual mortality should consider the biological and ecological factors that determine the response of populations and ecosystems to fish mortality. In addition, such decisions should consider the uncertainty surrounding estimates of the status of fish populations, the impact that mortality will have on a population, and the performance of offsets. Management of fish mortality, at least at the population level, could be supported by similar frameworks to those being used in fisheries management and could utilize precautionary buffers to achieve risk equivalency across different scenarios.

5. OFFSETTING MORTALITY

5.1. WHAT ARE CURRENT OFFSETTING PRACTICES FOR RESIDUAL MORTALITY?

The following section covers methods commonly used to offset residual mortality. Offsetting measures and targets in cases of fish mortality differ from traditional offsetting due to the negative impacts occurring after the completion of a development project (DFO 2019). Offsetting for fish mortality encompasses all activities implemented by the proponent to offset fish mortality occurring after the development project has been finished. Offsetting measures aim for equivalency in terms of fish injury and mortality to meet official offsetting requirements (DFO 2019). Measures to offset fish mortality fall into three primary categories (i) habitat creation, (ii) habitat restoration and enhancement, and (iii) biological and chemical manipulation (Table 5.1). Results in this section are based on a literature review with both quantitative and qualitative studies (Appendix B, Table B.1). Studies will be used to describe the current offsetting practices for fish mortality. Quantitative studies were used in a meta-analysis (Appendix B, Table B.2) to calculate effect size for the different practices in relation to project goals and offsetting requirements.

Table 5.1. Types of offsets used in cases of fish mortality in aquatic ecosystems. Results and classifications are based on a literature review and meta-analysis (Appendix B, Tables B.1 – B.3).

Type	Subtype	Measure	Associated benefits/ goals		
Habitat creation	Off channel habitat creation	Side channel creation	Spawning habitat provision, rearing habitat provision, overwintering habitat		
		Overwintering pond creation			
Habitat restoration and enhancement	R-Restoration	(Riparian) restoration, Rehabilitation	Buffer zone creation, reduction of environmental impacts, food availability, habitat coupling		
		Structure and Cover		Bank stabilization	In-stream habitat provision (shelter, food availability), flow regime
				Riparian heterogeneity	
				Large Woody Debris & Logjams	
	Boulders				
Connectivity	Dam and barrier removal	Lateral & longitudinal habitat connection, migration corridors, nutrient and sediment exchange and transport, flow regime			
Fish Passage					

Type	Subtype	Measure	Associated benefits/ goals
		Reconnection (Floodplain...)	
	Substrate	Channel Dugouts Substrate addition Substrate removal	Spawning substrate provision, Channel morphology changes, Temperature reduction, climate refuges
Bio and Chemical Manipulation	Stocking	Stocking	Direct addition of individuals and biomass, Potential increase in Productivity
		(Re)Introduction	
		Translocation	
	Nutrients	Nutrient enrichment	Productivity boost for biotic production, compensation for nutrient loss through lack of anadromous fish/ carcasses

5.1.1. Habitat creation

Habitat creation refers to the practice of creating entirely new habitat to offset fish mortality by increasing productivity, abundance, density, and fish survival. Projects applying created habitat to offset fish mortality used off channel habitat construction to provide essential life-history components, mostly for salmonid species. Off-channel habitat can take the form of side channels, sloughs, ponds, floodplains, and wetlands (Rosenfeld et al. 2008).

5.1.1.1. Side channel construction

Constructed side channels are normally excavated in a current or former floodplain near the main channel and can receive further enhancements through gravel addition, bank stabilization and cover provision. Side channels are primarily fed through groundwater sources (Roni et al 2006). Projects in the Pacific Northwest (British Columbia, Oregon, and Washington State) use groundwater fed side channels to create new spawning and rearing habitat for various salmonids to offset lost productivity and increase juvenile survival (Giannico and Hinch 2003). One example case study highlights the construction of eleven groundwater fed side channels to offset habitat loss and juvenile mortality (Morley et al. 2005). While the side channels featured a higher depth and less physical cover and lower levels of habitat heterogeneity than natural references, fish densities were higher overall in the newly created habitat for both sampling seasons (1.46 ± 1.23 , 1.05 ± 0.925). Coho Salmon (*Oncorhynchus kisutch*) populations benefited from the side channels. Results from this study also underline the benefit of warmer winter temperatures in groundwater fed channels, potentially overriding the lack of structural habitat richness (Giannico and Hinch 2003). Side channel habitat is mainly used for salmonid or substrate spawning species (Roni et al. 2006).

5.1.1.2. Off channel pond creation

Overwintering pond creation often accompanies side channel construction. Off channel pond habitat is also associated with wetlands and is used as overwintering and rearing habitat for many fish species. Ponds can be newly excavated or re-purposed from logging and mining activity, e.g., gravel pits and mill ponds. Off channel ponds can also be created through impoundment or re-connection of formerly isolated habitats (Roni et al. 2006). In a case study

from the Skagit River basin in the Pacific Northwest, populations of Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*), and other Pacific salmon species declined significantly due to loss of habitat and increased juvenile mortality (Henning et al. 2006). Monitoring data for 13 years (3 to 7 years of data per basin) was evaluated with a focus on smolt density of Coho Salmon, effect of project size, and offset morphology for 30 constructed and natural reference sites. Smolt density in constructed off channel ponds approached natural reference values with 0.37 smolts/m² and an average abundance of 2,492 fish per site, indicating a successful offset in terms of natural productivity rates. Smolt production was positively correlated with total wetted area. Smolt length was significantly higher in constructed ponds compared to natural or constructed channel habitats. Length differences were due to shoreline heterogeneity and cover (70%). Smolt length was negatively correlated with density (Roni et al 2006).

Table 5.2. Summary of habitat creation project (n = 9) metrics and benefits for offsets associated with fish mortality (c.f. Appendix B, Table B.3).

Offset Method	Monitoring average	Time for first benefit (years)	Cost area /m ²	Preferred species*	Commonly applied ratio	Effect size (hedge's g)	ΔB
Side channel	5.7 +-1.6	1.1 +-0.9	150 +-46	Salmonid	1:5.7	1.050	1.88
Off channel pond or floodplain	4.9 +-1.5	1.2 +-0.6	85 +-27	Salmonid	1:5.4	1.045	1.47

*Coho Salmon (*O. kisutch*), Chum Salmon (*O. keta*), Chinook Salmon (*O. tshawytscha*), Steelhead (*O. m. irideus*), Brook Trout (*S. fontinalis*)

5.1.1.3. Summary and common application regarding fish mortality

Overall, side channel creation can be an effective tool to increase fish productivity and increase juvenile survival by providing spawning and rearing habitat and thus is a suitable approach for offsetting fish mortality. Side channels are most often used in cases where harm stems from hydropower development projects leading to loss of connectivity as well as habitat degradation and juvenile mortality linked to a lack of rearing habitat (Scruton et al. 2005). Results from the case studies show the complementary nature of side channel and off channel pond creation. Both types of floodplain habitat construction increase productivity for salmonid species making newly created habitat that meet or exceed natural references, thus adhering to the no net loss (NNL) criteria (Table 5.2). The main drivers for productivity seem to be temperature, wetted area, and habitat heterogeneity. Greater depth and pond morphology produced larger smolts compared to channel-type habitat, with an average fork length difference of 13.3% (Roni et al. 2006). Benefit-cost ratios (BCR) are similar for both types. Offset size and costs increase rapidly with larger losses in biomass or productivity. Cost and space requirements may make habitat creation impractical for some mortality cases but could be used in conjunction with habitat restoration (floodplain reconnection, flow enhancement, gravel addition) or stocking. Habitat creation as offsets for fish mortality focus mainly on salmonids with density as the primary assessment and success metric. Harm and mortality mainly stem from habitat degradation and affects juvenile mortality and larval emergence. Considering the large effect sizes in the assessed projects, habitat creation should be considered for cases of direct mortality, especially for salmonids. The main factor here was project size; 5,000 to 10,000 m² seem to be the

optimum size given associated costs and target life stages (Rosenfeld et al. 2011). Commonly achieved offset to natural reference ratios were around 1:5 based on the assessed studies. Most ratios are applied to account for uncertainty in predicted gains or were derived from a 1:1 area replacement not accounting for higher productivity of the offset. Varying periodic mortality should be handled through adequate monitoring timeframes. Additional types of habitat creation and their application for offsetting mortality need to be explored for more species to inform better practices. Monitoring averages are 4 to 5 years, including pre-impact assessments, with early benefits requiring at least 1 year post construction to manifest (Table 5.2). Created off channel habitat for salmonid species provided high biomass (ΔB) benefits (1.47 to 1.88) compared to natural reference systems. Onsite and like-for-like options are more common but offsite construction and out-of-kind offsets are possible as well through newly created habitat.

5.1.2. Habitat restoration

Habitat enhancement and restoration for projects involving residual harm and (or) mortality can be divided into three categories targeting (i) structure and cover, (ii) connectivity, and (iii) substrate, with riparian restoration as a fourth mixed category.

5.1.2.1. Structure and Cover

Adding structure and cover to existing aquatic ecosystems can take many different forms, from creating riparian cover, constructing boulder weirs, adding pools and riffles, or introducing large woody debris. Structural enrichment and their beneficial effects for fish productivity have been supported by studies such as Roni et al. (2010) or Morley et al. (2005). Adding structure and cover can be a cost-efficient measure, especially in cases with smaller impacts or urbanized systems that do not offer the necessary space for habitat creation.

5.1.2.2. Connectivity

Connectivity and associated habitat restoration are common offsetting measures associated with fish mortality due to impingement and entrainment at cooling water intakes or hydropower facilities. One case study from Lake Huron is investigating the restorative effect of a dam removal to estimate the potential offset provided for fish losses at a nearby power plant. Translating the number of Lake Whitefish (*Coregonus clupeaformis*) lost at the water intake to production forgone (295.1kg lost age-1 biomass) allowed a comparison between the lost fish and the increase in ecosystem productivity through a dam removal on the Saugeen River. These models suggest that the dam removal (providing access to 94.3 ha of restored habitat) could offset the number of lost whitefish on a 10:1 ratio when combined with a short-term stocking effort of 295.1 kg of hatchery fish (Barnthouse et al. 2019). Backed by reference studies and collected data, restoration provides an efficient approach to offset fish mortality. Another restoration case study from New Jersey found that restoration and enhancement of a degraded salt marsh through re-connection was effective in offsetting losses of River Herring (*Alosa pseudoharengus*) due to entrainment and impingement at power generation stations in conjunction with commonly accepted impingement mitigation measures like deterrent systems, water intake regulation, and upgraded fish protection technology (Baletto and Teal 2011). The project set a 12-year monitoring timeframe to meet final success criteria which involved environmental variable thresholds like desired plant coverage, open water percentage and species abundance. Several other studies have shown the effectiveness of restoration measures linked to barrier removal and habitat re-connection, especially for migratory species (Hogg et al. 2015). While Salmonid species are the focus of most projects, habitat restoration can also provide benefits for other species. For instance, a 6-year dam removal study in the headwater streams of Shenandoah National Park, Virginia, showed that American Eel (*Anguilla rostrata*) abundance at 15 sites increased from 1.6 eel/100 m (± 0.825) to 3.75 eel/100 m (\pm

3.15), meeting numbers from unimpeded natural reference systems. Average length decreased in headwater locations, indicating successful passage of smaller size classes (<300 mm) (Hitt et al. 2012). These results demonstrate that the removal of a key bottleneck dam can offset negative effects on American Eel productivity and abundance for populations up to 150 km distance and on a landscape scale.

5.1.2.3. Substrate

Substrate changes have occurred in a wide range of aquatic system due to development projects like flow regulation, forest harvesting, and logging operations. Adverse effects on substrate spawners can be severe and lead to fish mortality at multiple life-stages. A study from Carnation Creek, British Columbia, showed the increase of fine substrate material from 4.6 to 5.7% due to logging activities. Accumulation of fine substrate led to a decline in juvenile emergence from 29.1 to 16.4% for Coho Salmon (*Oncorhynchus kisutch*) and 22.2 to 11.5% for Chum Salmon (*Oncorhynchus keta*) (Scrivener and Brownlee 1989). Substrate composition explained between 60 to 73% of the variability in emergence survival. Substrate addition and (or) removal can have a beneficial effect for many gravel spawning species and is well supported in the literature. For instance, a systematic review of 75 studies from 64 articles conducted by Taylor et al. (2019) found that a lack of spawning substrate combined with a lack of access to suitable spawning habitat can be the main drivers for population collapse, especially for salmonid species. Spawning habitat tailored towards species-specific niche requirements can be effective in offsetting negative impacts from human development (Taylor et al. 2019). Substrate addition has also been used to offset fish mortality, mainly in combination with the creation of new habitat in the form of side channel creation described in the previous section. A study from British Columbia summarizing data from over 30 studies confirms the benefits of spawning gravel and linked spawning habitat for both anadromous salmonids (Coho, Chinook, and Steelhead) and non-anadromous salmonids (Brook Trout, Brown Trout, Cutthroat Trout, and Rainbow Trout). An 8-fold increase in gravel area led to an 88% increase in production per m² for anadromous species and 25 to 73% increase for non-anadromous resident species (Keeley et al. 1996). Results from these studies highlight the beneficial effect of adding or providing spawning substrate to increase productivity and juvenile survival rates. The high variability in productivity and survival benefits demonstrates that outcomes vary on a case-by-case basis and the expected benefit rates and ratios may not be realized. Conversely, substrate removal can be implemented to change channel morphology and increase average channel depth to offset fish mortality linked to thermal stress (May and Lee 2004, Auer et al. 2017).

Table 5.3. Summary of habitat restoration and enhancement project (n = 14) metrics and benefits for offsets associated with fish mortality (c.f. Appendix B, Table B.3).

Offset Method	Monitoring average	Time for first benefit (years)	Cost area /m ² /m	Preferred species*	Commonly applied ratio	Effect size (hedge's g)	ΔB
Riparian Restoration	1.8+-1.1	0.8 +- 0.4	68(m) +-26	Community	1:1.2	1.471	0.21
Structure addition	3+-0.7	1+-0.7	188(m) +-123	Salmonid, Community	1:1.6	0.794	1.62
Connectivity	4.1+-1.8	1+-0.6	84(m ²) +- 77**	Diadromous, potamodromous, Rheophilic	1:4.6	0.495	1.24
Substrate	2.3+-1.1	1.1+-0.9	11(m ²) +- 7	Salmonid, substrate spawner/ Lithophilic	1:2.1	0.694	1.12

*American eel (*A. rostrata*), Coho Salmon (*O. kisutch*), Chum Salmon (*O. keta*), Chinook Salmon (*O. tshawytscha*), Steelhead (*O. m. irideus*), Brook Trout (*S. fontinalis*), Brown Trout (*S. trutta*), Yellow Perch (*P. flavescens*), White Sucker (*C. commersonii*), Lake Whitefish (*C. clupeaformis*), Walleye (*S. vitreus*), Arctic Grayling (*T. arcticus*), Eurasian minnow (*P. phoxinus*), river herring (*A. pseudoharengus*)

**highly variable and depends on the size of connected or reconnected habitat

5.1.2.4. Summary and common application regarding fish mortality

Habitat restoration, due to its versatile nature and demonstrated effect size, can be a highly effective measure (Table 5.3) to offset fish mortality. It can be applied in various scenarios and is often used by combining different restoration and enhancement measures. Like habitat creation, most past and recent studies were applied to cases that featured indirect mortality due to habitat degradation, loss of connectivity, and juvenile mortality. However, studies like Barnthouse et al. (2019) show how habitat restoration can be utilized to offset periodic or chronic mortality events by increasing overall habitat productivity and compensating for lost fish through quantification of equivalent biomass, habitat productivity index (HPI), or age equivalents (EA). Restored habitat often benefits multiple members of an aquatic community. In the case of fish mortality, it is important to be specific about like-for-like or out-of-kind replacement. Offset ratios, benefits, and consequently sizes will differ significantly if the offset aims to meet lost biomass for a single species or for a community. Average offset ratios were often around 1:1.5 to account for uncertainties, though connectivity offsets often had higher offset ratios (1:4.6) since this type of offset, and its size, is more dependent on the environment and associated ecosystems than the measure itself (e.g., dam removal). Costs varied greatly across offsets. Riparian restoration and structure addition is mainly assessed in restored or enhanced meters while connectivity and substrate offsets are measured in area (m²). Substrate addition can be a cheap and effective measure when targeting species and spawning related aspects. Like habitat creation, early benefits are normally measurable one-year post-construction. Overall monitoring times for restoration projects range from 2 to 4 years, including pre-assessments. Mean expected biomass benefits (ΔB) are generally greater than one, except for riparian restoration measures which usually do not target productivity directly. Variability of the described measures

shows that compatible joint measures can complement each other. Evaluated case studies show the potential of habitat restoration to offset fish mortality when losses can be translated into habitat metrics. Most monitoring assessments focus on densities and rarely biomass; this means that monitoring requirements need to be adjusted accordingly to ensure that restoration activities are effective. Offsets thus require pre-assessments and regular post monitoring to evaluate the full benefits properly. Early estimates can be derived from abundant literature and restoration studies from similar systems and species.

5.1.3. Biological and Chemical Manipulation

Biological and chemical manipulation of habitats and ecosystems has been commonly used to either enhance productivity of nutrient poor systems or to control nutrient inputs and eutrophication, e.g., algal blooms (Sierp et al. 2009). It also refers to the practice of increasing fish abundance through stocking, (re)introduction, and translocation. Biological and chemical manipulation cover a wide range of aspects ranging from the simple addition of physical specimens to influencing specific trophic levels or whole food webs through nutrients.

5.1.3.1. Stocking

Stocking, (re)introduction, and transfer of fish is regularly used to mitigate losses of both recreationally and commercially important species as well as offset negative anthropogenic impacts. Stocking has been used in many instances to bolster impaired or heavily harvested populations. Most stocking applications and studies do not relate to direct mortality but rather indirect effects from habitat degradation, loss of connectivity, or reduced juvenile survival. Cases where stocking was used to offset direct mortality are rare and most often used when the main sources of harm were from entrainment and impingement in hydropower facilities, flow regulation, and stranding events. Most case studies found in the literature deemed impingement and entrainment losses to be inconsequential and rarely move beyond basic mitigation measures. Large stocking efforts aiming to offset anthropogenic factors and mortality for diadromous species along the United States Atlantic coast have shown that stocking by itself is not sufficient due to low connectivity. Only 3% of the fish were able to complete vital passages (Brown et al. 2013). Studies and reviews from New Zealand show that stocking increased population numbers for diadromous salmonids, however, not to the anticipated degree due to significantly lower survival rates than initially predicted (Holmes 2018, Unwin and Gabrielson 2018). Overall, fish stocking can increase impaired fish populations but needs to be applied in very specific ways when considered as offsets for mortality events. The previously mentioned Lake Huron case study, investigating a dam removal to offset fish losses at a nearby power plant, could also be offset by stocking. Translating the number of whitefish lost at the water intake to production forgone (295.1 kg lost age-1 biomass) could be offset through a short-term stocking effort of 295.1 kg of hatchery fish (Barnhouse et al. 2019). The feasibility of this scenario relies on the impact from fish mortality without any additional bottlenecks or habitat degradation, which should be vital considerations when considering stocking as a mortality offset.

5.1.3.2. Nutrients

Nutrient enrichment is a highly situational offset requiring two main considerations. First, it requires an aquatic ecosystem that is nutrient poor or features lower nutrient levels than naturally expected. Second, these conditions need to be present due to non-natural circumstances, as eutrophication of a natural system beyond its natural levels can lead to significant community changes (Wipfli et al. 2010). It is most applicable for habitats that are cut off from their usual nutrient sources and allow for a certain level of treatment control (Naiman et al. 2002). It targets primary and secondary productivity which consequently benefit higher

trophic levels, e.g., fish productivity. An example of nutrient enrichment as an offset is a case study from the Columbia River basin in the United States. Anthropogenic influences led to a decline in anadromous species abundance, thus reducing the organic matter (nitrogen (N), phosphorus (P), and carbon (C)) input as well as marine derived nutrients carried across the river basin by anadromous fishes. Biochemical manipulation through nutrient input (Chinook Salmon, *Oncorhynchus tshawytscha*, carcasses and marine fish bone meal) in ten streams influenced N levels measurably without significant changes in P and C concentrations (Kohler et al. 2012). Periphyton, chlorophyll-a, and ash-free dry biomass (AFDM) were between 178 and 214% higher in downstream treatment reaches; increases in macroinvertebrate densities (158%) were also observed. Fish growth in length (78% to 228%) and weight (71% to 375%) were positively influenced by the treatments. These changes were analogous to an increase in fish density (>50%). Other studies also showed average growth benefits in length (>15%) and weight (>50%).

Table 5.4. Summary of biological and chemical manipulation project (n = 7) metrics and benefits for offsets associated with fish mortality (c.f., Appendix B, Table B.3).

Offset Method	Monitoring average	Time for first benefit (years)	Cost area /m ²	Preferred species*	Common applied ratio	Effect size (hedge's g)	ΔB
Stocking	8.3+-9	1.2+-0.7	Species dependent	Salmonid, Community	1:3.1	0.331	0.84
Nutrients	4+-0.7	0.3 +-0.47	-	Different trophic levels	-	1.036	2.01

*Coho Salmon (*O. kisutch*), Chinook Salmon (*O. tshawytscha*), Sockeye Salmon (*O. nerka*), Dolly Varden (*S. malma*), Cutthroat Trout (*O. clarkii*), Alewife (*A. pseudoharengus*), Rainbow Smelt (*O. mordax*), Yellow Perch (*P. flavescens*)

5.1.3.3. Summary and common application regarding fish mortality

Nutrient enrichment can have significant short-term productivity increases in treated aquatic systems. However, most studies only suggest nutrient enrichment as an interim tool to offset nutrient deficits until natural pathways can be restored. It could be suitable for situations where nutrient pathways are blocked or disrupted, or where reliant fish populations are extirpated or significantly reduced. Potential significant community changes need to be considered. Nutrient enrichment could be a suitable method to offset fish mortality given its mean effect size based on the literature and fast response time for first benefits (immediate to 3 months) by increasing overall system productivity in systems that allow for the treatment while also being easily monitored and controlled (Table 5.4). Complex systems and communities, both in size and species richness, are rarely suitable due to the magnitude of potential interactions. The main target species for nutrient enrichment are diadromous salmonid species (Kohler et al. 2012). Many enrichment programs are already in place, which should allow for an easier implementation of nutrient enrichment for mortality offsets and falling back to recorded and established benchmarks from these studies. Nutrient enrichment can be adjusted and tailored to target life stages or important timeframes during the year. Cost per area as well as benefits are highly variable and depend on the target species and enrichment intensity (stoichiometry) but in most cases, carcasses can be readily acquired from hatcheries. Nutrient enrichment is often

jointly conducted with stocking efforts. In these cases, its main aim is to increase nursery habitat productivity and increasing fry abundance through stocking (Koenings et al. 2000). Nutrient enrichment requires extensive data on prior nutrient levels as well as system productivity. Evaluations rely on several important benchmarks to capture trophic responses and benefits of the ecosystem (primary production, secondary production, fish response). To evaluate and monitor benefits for target fish species linked to nutrient enrichment, the target level and lower trophic levels need to be monitored (Koenings et al. 2000). Enrichment effects are mostly assessed in fish growth parameters and primary and secondary production levels with monitoring timeframes around 4 years to adequately capture long-term effects and seasonal variation and population dynamics (Table 5.4). Lost fish should be translated into production forgone (biomass) that can be matched with enrichment monitoring metrics and expected productivity benefits.

Fish stocking differs from other approaches when considered as a mortality offset. Stocking does not meet the self-sustaining nature of an offset (*as per* DFO 2019b). Study results underline the inherent difficulty of using stocking as an effective offset or restoration measure. While survival differences between wild and hatchery fish can be considered by incorporating offset ratios (1.5 to 3), bottlenecks are often overlooked (Antonio Agostinho et al. 2010). For instance, stocking can rarely compensate for a lack of connectivity or degraded rearing and spawning habitat (Michaud 2000). Thus, stocking mainly seems to be an appropriate mortality offset when the mortality is direct and not linked to indirect sources or habitat loss or degradation (Figure 5.1). Examples of these types of cases include one-time fish kills, losses due to entrainment and impingement, or stranding events through flow regulation (Young et al. 2011). In these cases, lost fish can be translated into age equivalents or production forgone and stocked according to these numbers or biomass. Like-for-like and out-of-kind scenarios are possible depending on the species. Both timeframe and hatchery fish survival need to be considered. A one-time fish mortality event requires only a one-time stocking offset while periodically or regularly occurring losses, e.g., power plant water intake, need to be adjusted accordingly, e.g., on a yearly basis. Furthermore, hatchery fish could exhibit lower survival rates (Margenau 1992), which need to be included in the accounting. Some studies also suggest a higher impingement and entrainment rate for hatchery fish due to behavioural differences compared to wild individuals (Michaud 2000). These uncertainties and potential long term stocking requirements translate to common offset ratios around 1:3 and monitoring timeframes of 8 years and more (Table 5.4). Ratios coincide with commonly accepted uncertainty and time-lag related considerations (Bradford 2017). Overall, stocking could be suitable for direct mortality events that do not include a habitat component. For scenarios with harm stemming from indirect mortality, offsets based on habitat restoration and creation should be preferred. Re-introductions are only feasible after removal of the harm source (e.g., post clean-up after a spill event) and restoration of the affected habitat (Dunham and Gallo 2008). Most stocking studies measure success in survival rates for both stocked fish as well as reference populations and population impacts, e.g., stocking was implemented to offset a reduction in juvenile survival or juvenile to adult survival. Survival rates for both should be translated either into surviving equivalent adults (or other equivalent age class) or in production forgone (biomass).

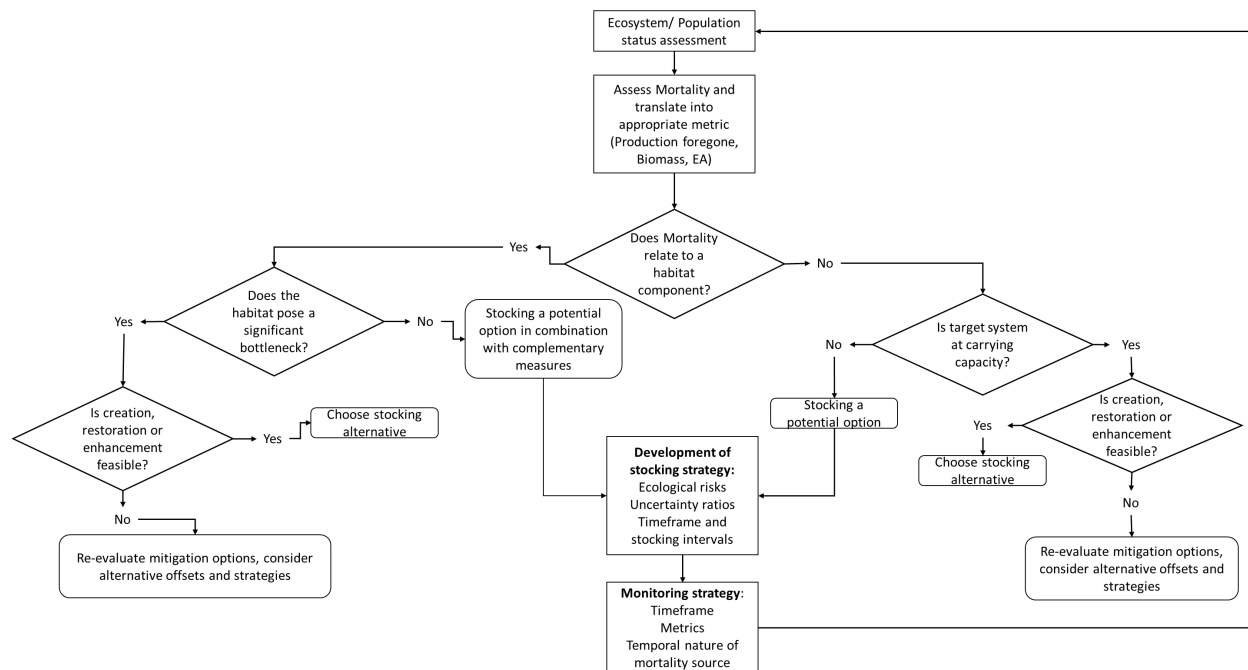


Figure 5.1. Flowchart of potential stocking application in the context of WUA related residual mortality and considerations on whether or not stocking could be viable and appropriate.

5.2. GENERAL MONITORING INDICATORS FOR OFFSETTING SUCCESS

Results from the literature review and meta-analysis show that monitoring time is related to offsetting success when broken down into three basic numerical categories (success (2), partial success (1), no success (0)) and major time increments in years (<4, 4 to 6, >6). A minimum monitoring timeframe, including pre-assessments, of more than 4 years is associated with significantly increased success. Projects with pre-impact assessment studies similarly showed higher success than projects without pre-impact assessments. The location of the offset, whether it is onsite or offsite, did not play a significant role in project outcomes (Table 5.5).

Table 5.5. Effect of monitoring timeframes and location on general offset success, as well as frequency of collection pre-assessment data.

Monitoring time (n = 30)	<4 years	4 to 6 years	>6 years
Success Score	0.86 ± 0.89	1.53 ± 0.79	1.40 ± 0.84
Pre assessment (n = 27)	Yes	No	-
Success Score	1.70 ± 0.57	0.75 ± 0.89	-
Onsite/ Offsite (n = 29)	Onsite	Offsite	Both*
Success Score	1.29 ± 0.86	1.41 ± 0.89	1.5 ± 0.71

*low sample size $n < 3$.

5.3. OFFSETTING DISCUSSION

Benefits from residual mortality offsets can be summarized into three main categories related to their temporal and target specific benefits (long-term, short-term, one time). These three categories should be related to either a habitat (or ecosystem), population, or habitat (or ecosystem) and population effect. For instance, habitat creation provides long-term benefits at both a habitat and population level; thus, habitat creation can be suitable to offset mortality events that either happen on longer temporal scales or relate back to detrimental habitat effects in addition to the residual mortality (e.g., larval mortality through flow reduction and sediment accumulation during spawning season). Restoration and enhancement measures can fall into all categories. For instance, restored connectivity will likely benefit a whole fish community long-term, while spawning gravel addition often targets a single salmonid species and deteriorates over time without maintenance. Stocking and nutrient addition on the other hand, have short term benefits with stocking having a sole population effect and nutrient enrichment targeting biochemical ecosystem processes. Both require long-term management to meet consistent benefits. Overall, besides the stated generalized benefits like biomass (Tables 5.2, 5.3, 5.4), temporal and target related benefits need to be considered in the strategic planning process for residual mortality offsets.

While benefits are often evaluated and can be derived from the literature, risks and unintentional effects are not regularly considered. Stocking for instance, generally poses risks due to the interaction of hatchery fish with wild stocks and subsequent effects such as the introduction of genetic material and food web and community shifts (Pastorino 2019). To counter these risks in the cases of stocking as an offset for residual mortality, clear objectives and a sound strategic approach are necessary (Figure 5.1). Stocking strategies in cases of residual mortality should include factors such as the source of stocked fish (hatchery information), stocking timeframe and intervals, stocking density in relation to density dependent effects and carrying capacity, and potential genetic, pathogen, community, and behavioral impacts (Cowx 1994). Following a clear pathway, as outlined in the example of Figure 5.1, will help determine if a) stocking could be an appropriate mortality offset and b) how to ensure tangible benefits while minimizing risks. Long-term monitoring will further reduce the potential bias of annual fluctuations, aid the decision-making process, and help adjust stocking levels.

Overall, all offsets that can be utilized for residual mortality hold the potential for unintended and (or) adverse effects on ecosystems or aquatic communities. Creation as well as restoration and enhancement measures can impact physical processes and structural properties of an aquatic ecosystem as well as biogeochemical characteristics (nutrient turnover) or biodiversity and community related aspects (Schirmer et al. 2014). Main concerns include the spread of invasive species through restored connectivity of waterways, shifts in community and food web structure through nutrient addition, and enhancements for a specific target species. Other challenges for restored or created habitat, similar to stocking, include density dependent effects. Created habitat for salmonids for instance can lead to an increase in fish density but at a certain point affects fish condition and ultimately reduces the biomass per fish gain. Planning strategies for residual mortality offsets, similar to planning strategies for harmful alteration, disruption or destruction (HADD) related offsets, should incorporate an assessment of potential unintended and adverse effects (Figure 5.2). The self-sustaining nature of habitat offsets also needs to be considered in the planning process. Almost all major offsets require maintenance to adhere to the in-perpetuity requirement of their benefits. Maintenance and long-term adaptive management relate directly to adverse and unintended effects which then can be compensated and adjusted for as well as a potential reduction in offset benefits. For instance, a river restoration project on the Thur River led to the gradual formation of a point bar over the course of 5 years, which in turn led to large scale bank erosion and subsequent removal of riparian

forest area (Schirmer et al. 2014). This example shows how long-term maintenance and project adjustment is often necessary to balance benefits and unintended impacts.

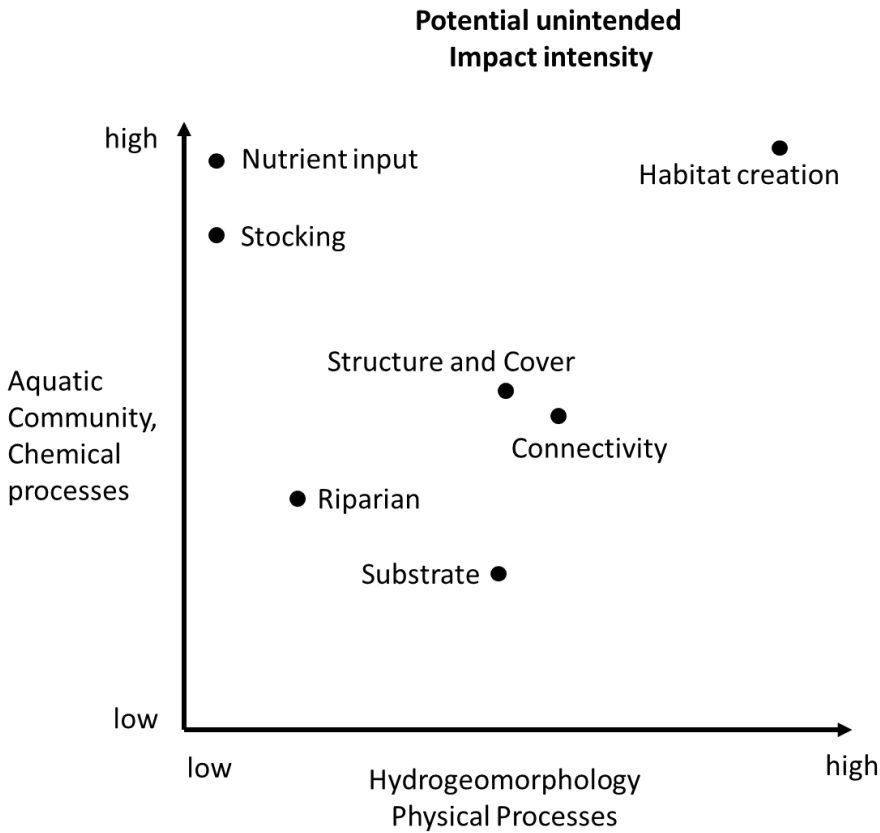


Figure 5.2. Potential of unintended impact intensity for the main offsets for WUA-related residual mortality offsets as identified in literature.

6. KEY UNCERTAINTIES

There are three main pieces of information about species and populations that are generally lacking to inform decision making about fish mortality: population abundance, population trajectory, and mortality rates. Population abundance and trajectory can be estimated from population monitoring, though they do take time, especially to estimate trajectory. Pre-impact assessments can provide these important pieces of information. Population abundance and trajectory provide information about the current status of the population. Risk is elevated when populations have low abundance and (or) are exhibiting a declining trajectory. Mortality rates are challenging to measure in natural systems. Estimates of natural mortality are needed for many of the metrics to quantify equivalence of fish mortality and offsets. Estimates of mortality rates associated with WUAs would be advantageous to better inform decisions. Both experimental studies (e.g., experimental release of fish or surrogates through turbines) and tagging studies can help to estimate mortality rates.

Mortality events typically affect multiple species in a community or food web. Given the potential for (i) unexpected outcomes as community or food web complexity increases, (ii) consequences for species that may not be directly affected by a fish mortality event, and (iii) community-wide outcomes that may exceed what is expected based on a sum of mortality across the affected species, more research on the community level outcomes of mortality events is needed. This should include various mortality combinations including pulse versus press events (i.e., events of different frequencies), selectivity of mortality events (e.g., which fish community members does the mortality affect), magnitude of mortality events, and the inclusion of offsets. Simple community matrix models have the potential to elucidate community responses and dynamics that could lead to general results to help to inform an understanding of how fish communities will respond to fish mortality events.

Mortality on multiple life stages and (or) from multiple anthropogenic sources has the potential to produce cumulative effects that can differ from expected outcomes based solely on summing mortalities. This is not a challenge that is unique to managing fish mortality. While there is active research on the topics of multiple stressors and cumulative effects, a predictive approach to expected outcomes has yet to be established. This creates uncertainties when managing in the presence of cumulative effects which need to be included when considering the risks associated with authorizing fish mortality and associated offsets. This challenge is even greater when combined with environmental change (e.g., climate change), which can affect how fish populations and communities respond to, and interact with, other anthropogenic stressors such as fish mortality.

Most WUAs that involve fish mortality also have the potential to produce sub-lethal effects with implications for population status and resilience to other stressors. Despite this, sub-lethal effects could not be dealt with here. The occurrence of sub-lethal effects can lead to additional fish mortality or have consequences for population responses. Specific science advice is needed on the integration of sub-lethal effects into management decisions around the death of fish.

Non-stationarity in environmental conditions can impact the consequences of mortality for fish populations, and the performance of offsets. Management uncertainty resulting from environmental non-stationarity is caused by (i) uncertainty in estimating the future environmental conditions themselves, and (ii) uncertainty in the relationship between offset performance and (or) population consequences of mortality, and future environmental conditions.

7. CONCLUSIONS

This document provides information relevant to management decisions about WUA-related residual mortality, including (i) metrics available for the quantification of fish mortality and potential offsets, (ii) considerations to inform decisions about the authorization of WUA-related residual mortality, and (iii) current practices that can inform decisions about potential offsets.

The various **metrics propose for quantification of fish mortality** provide equivalency for different life-history characteristics between the impact of the fish mortality and any potential offset. While seven metrics were reviewed, there are four main metrics most commonly applied, though they provide different equivalencies:

- Equivalent ages – provides equivalency in age-structure between the impact and offset;
- Reproductive value – provides equivalency in future reproduction output between the impact and offset;
- Production forgone – provides equivalency in future biomass production between the impact and offset;
- Total biomass lost – provides equivalency in standing stock and future biomass production between the impact and offset.

Maintaining standing stock levels and ensuring future production are most consistent with the conservation and protection objective (*Fisheries Act*; DFO 2019a) and the FFHPP principle that offsets should balance adverse effects (DFO 2019b). Therefore, total biomass lost is recommended as the preferred equivalency metric under most circumstances.

Full accounting of fish mortality must take into account time-lags in the delivery and functionality of an offset and uncertainty in the magnitude of harm and efficacy of proposed offsets. These aspects can be accounted for with calculation of time-lag and uncertainty compensation ratios. Time-lag compensation ratios require the selection of a time-horizon; determination of the impact and offset schedules and application of discounting. Uncertainty compensation ratios are less straight forward to determine. One option is to perform Monte Carlo simulations which require an estimate of variability in the fish mortality estimate and variability in the efficacy of the proposed offset. Using this method, the likely compensation ratio required to provide a high degree of equivalency (e.g., 90%) between the impact and offset can be estimated.

Projections of fish mortality impacts on populations can be incomplete, as harm applied to a single species can impact whole communities and may cause counter-intuitive results due to indirect effects and feedback loops. This can include species that are not directly affected by the fish mortality events. A brief review of community modelling techniques provided a sample of available approaches that can be used to investigate the **community impacts of fish mortality**. Which model to apply will depend on the questions to be answered and the data available. These models could be grouped roughly according to their data requirements. Models such as qualitative network models (QNM), Bayesian belief networks (BBN) and individual-based models (IBM) are relatively light in terms of general data requirements but may require specific types of data to parameterize species interactions. However, these models can be limited in their ability to produce quantitative predictions and are more suitable for scenario testing or as diagnostic tools. For systems where there is a moderate amount of knowledge, size-spectra models could provide a diagnostic tool for community status. The dynamic multispecies size spectrum model builds on size spectrum theory with simulations to explore potential community responses. Other modelling techniques such as minimum realistic models (MRM), structural equation modelling (SEM), and multi-species bioenergetics models could also

be used in such data moderate systems to generate more concrete predictions and perform impact assessments. Data rich ecosystems may warrant full use of whole ecosystem models to generate detailed projections of human impacts on all trophic levels in the community.

Simple community models (such as qualitative network models, QNM) can improve understanding of how fish communities may respond to fish mortality events. The QNM application presented demonstrated that, first, predicting responses based solely on community structure becomes less tractable as community complexity increases. Second, multi-species fish mortality has a higher probability of negative outcomes for the whole community. Third, top trophic level species tend to be negatively affected by mortality at lower trophic levels within the fish community. And fourth, that complex linkages among species, such as omnivory, lead to mixed outcomes suggesting the need for more detailed analyses. These simple community models demonstrated that the outcomes for fish communities from fish mortality events are likely to be more than just the sum of single species responses.

Decisions related to authorizing the death of fish should consider the biological and ecological factors that determine the sensitivity of populations to mortality. Even if offsets fully account for a given source of mortality, serious harm to fish populations can still occur when there are differences between the timing of mortality and the implementation of offsets, when populations are highly sensitive to decreases in abundance (e.g., via reproductive depensation), or by changes in the ecosystem that interact with WUA-related residual mortality in a synergistic manner.

The presented **adaptation of a precautionary fisheries management framework** provides an example of a risk management framework for decisions about fish mortality impacts and offsets. Given the similarity between the population consequences of fisheries mortality and other anthropogenic sources of mortality, and the fisheries protection objectives of the *Fisheries Act* and the Fish and Fish Habitat Protection Program, the use of a common framework by fisheries and habitat managers would allow for the leveraging of data and information across different decision-making contexts.

The presented review of the literature on **offsets for fish mortality** demonstrated that habitat creation, habitat restoration and enhancement, and biochemical manipulation can all be feasible options for offsetting fish mortality given caveats and general monitoring timeframes.

Habitat creation to offset fish mortality is mostly studied for Salmonid species and requires further study and application to other species and communities. Offset costs and size can increase rapidly in habitat creation projects, with a potential size threshold beyond which benefits become difficult to achieve. Habitat creation provides the most benefits for larval and juvenile live stages. Based on the assessed literature, applied offsetting ratios were around 1:5.

Restoration and enhancement are the most commonly used offsets in cases of fish mortality. In particular, reconnection can be an easily implemented measure to provide benefits on a large scale. Restoration measures often target whole communities and need to be carefully considered when targeting a specific species. Enhancement measures, such as spawning substrate introduction, may be more likely to ensure species-specific benefits. Habitat enhancement and restoration provides the most benefits for larval and juvenile life stages. Based on the assessed literature, applied offsetting ratios were around 1:2.5.

Stocking can be an effective replacement for lost fish, given a stable and unimpaired ecosystem and no significant bottlenecks. Hatchery fish tend to have slightly lower survival rates than wild fish and are more vulnerable to harm and mortality sources, e.g., impingement. Offset ratios (commonly between 1:1.5 and 1:3) can be applied to compensate for this uncertainty about survival. Stocks need to be monitored frequently to ensure benefits. Stocking needs to be

conducted in frequent intervals when fish mortality stems from a regular occurring harm source. Based on the reviewed literature, applied offsetting ratios were around 1:3.

All three offsetting types can be potentially detrimental when an out-of-kind replacement or a species versus community effect takes place on a magnitude that disrupts or alters community structure and food web composition.

Pre-impact assessments tend to increase offsetting success significantly and should be conducted for cases involving fish mortality if possible.

Time to achieve first benefits in most offsets required one or more years. This time lag needs to be accounted for in both implementation and monitoring.

Cases using habitat productivity metrics to quantify creation or restoration offsets should use unimpaired reference systems. References should reflect the regional average and the appropriate target species or community. Single reference sites, systems, or unsuitable literature reference values can easily distort the value of offsets.

Monitoring timeframes with a minimum of four years tend to be associated with higher chances of success in projects offsetting fish mortality.

The science request laid out the following series of questions, and this document has provided information relevant to answering these questions as follows:

1. What approaches can be used to quantify the impacts of WUA-related residual mortality, and associated offsetting requirements?

- a. What are the advantages and limitations of the different approaches?

Section 2 provides descriptions of the available metrics for quantifying impacts and offsets for fish mortality. There are differences in the data requirements and equivalency objectives among metrics. Total biomass lost is the recommended metric for most cases.

2. What determines local fish population or community responses to WUA-related residual mortality?

- a. Does the effect on local fish populations or communities change with respect to when and how frequently fish are killed?

- b. What criteria should be considered when quantifying or describing impacts from WUA-related residual mortality?

Section 3 provides an overview of available community modelling techniques. An example of a simple community model demonstrated that the outcomes for fish communities from fish mortality events are likely to be more than the sum of single species responses.

Section 4 provides an overview of the considerations that should be included in management decisions about fish mortality, providing information on the components that could be included in a risk management framework for WUA-related residual mortality.

3. What are the current domestic and international practices for offsetting the effects of WUA-related residual mortality?

- a. What are the options for offsetting WUA-related residual mortality?

- b. What is the effectiveness of the available offsetting options?

- c. What are the rationales for selecting certain offsetting options?

Section 5 provides a review of current practices for offsetting fish mortality events. There is evidence that habitat creation, habitat restoration and enhancement, and biological and

chemical manipulation can all provide approaches to offsetting fish mortality, but consideration of the specifics of the offset application and sufficient monitoring, including pre-impact assessment, are needed to effectively achieve an offset for fish mortality.

8. REFERENCES CITED

- An, G., Mi, Q., Dutta-Moscato, J., and Vodovotz, Y. 2009. Agent-based models in translational systems biology. *Wiley Interdiscip. Rev. Syst. Biol. Med.* **1**(2): 159-171.
- Andersen, K.H., Jacobsen, N.S., Jansen, T., and Beyer, J.E. 2017. When in life does density dependence occur in fish populations? *Fish and Fisheries* **18**: 656-667.
- Andersen, K.H. 2019. *Fish Ecology, Evolution, and Exploitation: A New Theoretical Synthesis*. Princeton University Press, Princeton, NJ.
- Andersen, K.H. 2020. Size-based theory for fisheries advice. *ICES J. Mar. Sci.* **77**: 2445-2455.
- Andersen, K.H., Jacobsen, N.S., and Farnsworth, K.D. 2016. The theoretical foundations for size spectrum models of communities. *Can. J. Fish. Aquat. Sci.* **73**: 575-588.
- Anderson, C.N.K, Hsieh, C., Sandin, S.A., Hewitt, R., Hollowed, A., Beddington, J., May, R.M., and Sugihara, G. 2008. Why fishing magnifies fluctuations in fish abundance. *Nature* **452**: 835-839.
- Antonio Agostinho, A., Mayer Pelicice, F., Carlos Gomes, L., and Ferreira Júlio Junior, H. 2010. Reservoir fish stocking: When one plus one may be less than two. *Natureza & Conservação* **08**(02): 103-111. doi:10.4322/natcon.00802001.
- Audzijonyte, A., Pethybridge, H., Porobic, J., Gorton, R., Kaplan, I., and Fulton, E.A. 2019. Atlantis: A spatially explicit end-to-end marine ecosystem model with dynamically integrated physics, ecology and socio-economic modules. *Methods Ecol. Evol.* **10**(10): 1814– 1819.
- Auer, S., Zeiringer, B., Führer, S., Tonolla, D., and Schmutz, S. 2017. Effects of riverbank heterogeneity and time of day on drift and stranding of juvenile European grayling (*Thymallus thymallus L.*) caused by hydropeaking. *Sci. Total Environ.* **575**: 1515-1521. doi:10.1016/j.scitotenv.2016.10.029.
- Balletto, J.H., and Teal, J.M. 2011. PSEG's Estuary enhancement program: An innovative solution to an industry problem. pp. 229-253 *In*: J. Burgers (ed.) *Stakeholders and Scientists*. Springer, New York, NY. doi:10.1007/978-1-4419-8813-3_10.
- Barnthouse, L.W., Fietsch, C.L., and Snider, D. 2019. Quantifying restoration offset at a nuclear power plant in Canada. *Environ. Manage.* **64**: 593-607.
- Bell, G., and Gonzalez, A. 2011. Adaptation and evolutionary rescue in metapopulations experiencing environmental deterioration. *Science* **332**(6035): 1327–1330. doi:10.1126/science.1203105.
- Bellard, C., and Hugueny, B. 2020. Importance of metapopulation dynamics to explain fish persistence in a river system. *Freshw. Biol.* **65**(11): 1858–1869. doi:10.1111/fwb.13571.
- Berryman, A.A. 2002. Population: a central concept for ecology? *Oikos* **97**(3): 439-442.
- Blanchard, J.L., Andersen, K.H., Scott, F., Hintzen, N.T., Piet, G., and Jennings, S. 2014. Evaluating targets and trade-offs among fisheries and conservation objectives using a multispecies size spectrum model. *J. Appl. Ecol.* **51**: 612-622.
- Bradford, M.J. 2017. Accounting for uncertainty and time lags in equivalency calculations for offsetting in aquatic resources management programs. *Environ. Manage.* **60**: 588-597.
- Boreman, J. 1997. Methods for comparing the impacts of pollution and fishing on fish population. *T. Am. Fish. Soc.* **126**: 506-513.

-
- Brown, J.J., Limburg, K.E., Waldman, J.R., Stephenson, K., Glenn, E. P., Juanes, F., and Jordaan, A. 2013. Fish and hydropower on the U.S. Atlantic coast: failed fisheries policies from half-way technologies. *Conserv. Letters* **6**(4): 280-286. doi:10.1111/conl.12000.
- Brownscombe, J.W., Smokorowski, K.E. 2021. Review of Pathways of Effects (PoE) diagrams in support of FFHPP risk assessment. DFO Can. Sci. Advis. Sec. Res. Doc. 2021/079. iv + 55 p.
- Bull, J.W., Lloyd, S.P., and Strange, N. 2017. Implementation gap between theory and practice of biodiversity offset multipliers. *Conserv. Letters* **10**: 656-669.
- Butterworth, D.S., and Plaganyi, E.E. 2004. A brief introduction to some approaches to multispecies/ecosystem modelling in the context of their possible application in the management of South African fisheries. *Afric. J. Mar. Sci.* **26**: 53-61.
- Cameron, T.C., O'Sullivan, D., Reynolds, A., Pieterney, S.B., and Benton, T.G. 2013. Eco-evolutionary dynamics in response to selection on life-history. *Ecol. Letters* **16**(6): 754–763. doi:10.1111/ele.12107.
- Carpenter, S.H., Brock, W.A., Hansen, G.J.A., Hansen, J.F., Hennessy, J.M., Isermann, D.A., Pedersen, E.J., Martin Perales, K., Rypel, A.L., Sass, G.G., Tunney, T.D., and Vander Zanden, M.J. 2017. Defining a safe operating space for inland recreational fisheries. *Fish and Fisheries* **18**(6): 1150-1160
- Caswell, H. 2001. Matrix population models: construction, analysis, and interpretation. Sinauer Associates, Sunderland, MA. 722 p.
- Christensen, V., and Walters, C.J. 2004. Ecopath with Ecosim: methods, capabilities and limitations. *Ecol. Model.* **172**(2-4): 109-139.
- Chu, C., Lester, N.P., Giacomini, H.C., Shuter, B.J., and Jackson, D.A. 2016. Catch-per-unit-effort and size spectra of lake fish assemblages reflect underlying patterns in ecological conditions and anthropogenic activities across regional and local scales. *Can. J. Fish. Aquat. Sci.* **73**: 535-546.
- Clarke, K.D. and Bradford, M.J. 2014. A Review of Equivalency in Offsetting Policies. DFO Can. Sci. Advis. Sec. Res. Doc. 2014/109. v + 18 p.
- Coors, A., and DeMeester, L. 2008. Synergistic, antagonistic and additive effects of multiple stressors: predation threat, parasitism and pesticide exposure in *Daphnia magna*. *J. Appl. Ecol.* **45**(5): 1820–1828. doi:10.1111/j.1365-2664.2009.01727-1.x.
- Côté, I.M., Darling, E.S., and Brown, C.J. 2016. Interactions among ecosystem stressors and their importance in conservation. *Proc. R Soc. B* **283**(1824): 1–9. doi:10.1098/rspb.2015.2592.
- Courchamp, F., Clutton-Brock, T., and Grenfell, B. 1999. Inverse density dependence and the Allee effect. *Trends Ecol. Evol.* **14**(10): 405–410. doi:10.1016/S0169-5347(99)01683-3.
- Cowx, I. 1994. Stocking strategies. *Fish. Manage. Ecol.* **1**(1): 15-30.
- Cury, P. M., Mullon, C., Garcia, S. M., and Shannon, L. J. 2005. Viability theory for an ecosystem approach to fisheries. *ICES J. Mar. Sci.* **62**(3): 577-584.
- Dambacher, J.M., Li, H.W., and Rossignol, P.A. 2002. Relevancy of community structure in assessing indeterminacy of ecological predictions. *Ecology* **85**(5): 1372-1385.
- Dambacher, J.M., Li, H.W., and Rossignol, P.A. 2003. Qualitative predictions in model ecosystems. *Ecol. Model.* **161**: 79-93.

-
- Dambacher, J.M., Brewer, D.T., Dennis, D.M., MacIntyre, M., and Faole, S. 2007. Qualitative modelling of gold mine impacts on Lihir Island's socioeconomic system and reef-edge fish community. *Environ. Sci. Tech.* **41**: 555-562.
- Darling, E.S., and Côté, I.M. 2008. Quantifying the evidence for ecological synergies. *Ecol. Letters* **11**(12): 1278–1286. doi:10.1111/j.1461-0248.2008.01243.x.
- DeAngelis, D.L. and Gross, M.J. 1992. Individual-based models and approaches in ecology: populations, communities and ecosystems. Chapman and Hall, New York.
- Dey, W. (2002) Use of equivalent loss models under section 316(b) of the Clean Water Act. In *Defining and Assessing Adverse Environmental Impact Symposium 2001*. The Sci. World J. **2**(S1): 254–270.
- Davic, R.D. 2003. Linking keystone species and functional groups: A new operational definition of the keystone species concept. *Ecol. Society* **7**(1): r11. doi:10.5751/es-00502-0701r11.
- Dey, C.J., and Koops, M.A. 2021. The consequences of null model selection for predicting mortality from multiple stressors. *Proc. R. Soc. B.* 20203126. doi:10.1098/rspb.2020.3126.
- DFO. 2006. A Harvest Strategy Compliant With the Precautionary Approach. *Can. Sci. Advis. Sec. Sci. Advis. Rep.* 2006/023.
- DFO. 2008. National Workshop on Modelling Tools for Ecosystem Approaches to Management; 22-25 October 2007. *DFO Can. Sci. Advis. Sec. Proceed. Ser.* 2008/007.
- DFO. 2009. A fishery decision-making framework incorporating the precautionary approach.
- DFO. 2014. A science-based framework for assessing the response of fisheries productivity to state of species or habitats. *DFO Can. Sci. Advis. Sec. Sci. Advis. Rep.* 2013/067.
- DFO. 2019a. Fish and Fish Habitat Protection Policy Statement, August 2019. Fisheries and Oceans Canada.
- DFO. 2019b. Policy for Applying Measures to Offset Adverse Effects to Fish and Fish Habitat under the *Fisheries Act*, December 2019. Fisheries and Oceans Canada.
- Dunham, J.B., and Gallo, K. 2008. Assessing the feasibility of native fish reintroductions: a framework and example applied to bull trout in the Clackamas River, Oregon. U.S. Geological Survey, Open-File Report 2008-1007. doi:10.3133/ofr20081007.
- Duplisea, D.E., Roux, M.-J., Hunter, K.L., and Rice, J. 2020. Resource management under climate change: a risk-based strategy to develop climate-informed science advice. *DFO Can. Sci. Advis. Sec. Res. Doc.* 2019/044.
- Fa, Y., Chen, J., Shirkey, G., John, R., Wu, S.R., Park, H., and Shao, C. 2016. Applications of structural equation modeling (SEM) in ecological studies: an updated review. *Ecol. Processes* **5**: 19. doi:10.1186/s13717-016-0063-3.
- Favaro, B., and Olszynski, M. 2017. Authorized net losses of fish habitat demonstrate need for improved habitat protection in Canada. *Can J. Fish. Aquat. Sci.* **74**: 285-291.
- Fulton, E.A., Punt, A.E., Dichmont, C.M., Gorton, R., Sporcic, M., Dowling, N., Little, L.R., Haddon, M., Klaer, N., and Smith, D.C. 2016. Developing risk equivalent data-rich and data-limited harvest strategies. *Fish. Res.* **183**: 574–587. doi:10.1016/j.fishres.2016.07.004.
- Gabriel, W., and Bürger, R. 1992. Survival of small populations under demographic stochasticity. *Theor. Pop. Biol.* **41**(1): 44–71. doi:10.1016/0040-5809(92)90049-Y.
-

-
- Gascoigne, J., Berec, L., Gregory, S., and Courchamp, F. 2009. Dangerously few liaisons: A review of mate-finding Allee effects. *Pop. Ecol.* **51**(3): 355–372. doi:10.1007/s10144-009-0146-4.
- Gascoigne, J.C., and Lipcius, R.N. 2004. Allee effects driven by predation. *J. Appl. Ecol.* **41**(5): 801–810. doi:10.1111/j.0021-8901.2004.00944.x.
- Giannico, G.R., and Hinch, S.G. 2003. The effect of wood and temperature on juvenile coho salmon winter movement, growth, density and survival in side-channels. *River Res. Appl.* **19**(3): 219-231. doi:10.1002/rra.723.
- Gibeau, P., Bradford, M.J., and Palen, W.J. 2020. Can creation of new freshwater habitat demographically offset losses of Pacific Salmon from chronic anthropogenic mortality? *PLoS ONE* **15**(12): e0237052. doi:10.1371/journal.pone.0237052.
- Gilbert, B., Tunney, T.D., McCann, K.S., DeLong, J.P., Vasseur, D.A., Savage, V., Shurin, J.B., Dell, A.I., Barton, B.T., Harley, C.D.G., Kharouba, H.M., Kratina, P., Blanchard, J.L., Clements, C., Winder, M., Greig, H.S., and O'Connor, M.I. 2014. A bioenergetic framework for the temperature dependence of trophic interactions. *Ecol. Letters* **17**(8): 902-914. doi:10.1111/ele.12307.
- Gilman, S.E., Urban, M.C., Tewksbury, J., Gilchrist, G.W., and Holt, R.D. 2010. A framework for community interactions under climate change. *Trends Ecol. Evol.* **25**(6): 325-331.
- Gobas, F.A.P.C., Wilcockson, J.B., Russell, R.W., and Haffner, G.D. 1999. Mechanism of biomagnification in fish under laboratory and field conditions. *Environ. Sci. Tech.* **33**(1): 133–141. doi:10.1021/es980681m.
- Goodyear, C.P. 1988. Implications of power plant mortality for management of the Hudson River striped bass fishery. Pages 245-254 in L.W. Barnhouse, R.J. Klauda, D.S. Vaughan, and R.L. Kendall, editors. *Science, law, and Hudson River power plants: a case study in environmental impact assessment*. American Fisheries Society, Monograph 4. Bethesda, Maryland.
- Grace, J.B. 2006. *Structural equation modeling and natural systems*. Cambridge University Press.
- Grace, J.B., Anderson, T.M., Olf, H., and Scheiner, S.M. 2010. On the specification of structural equation models for ecological systems. *Ecol. Monogr.* **80**(1): 67–87. doi:10.1890/09-0464.1.
- Gross, M.R. 1996. Alternative reproductive strategies and tactics: diversity within sexes. *Trends Ecol. Evol.* **2**(2): 92–98.
- Guiet, J., Poggiale, J.-C., and Maury, O. 2016. Modelling the community size-spectrum: recent developments and new directions. *Ecol. Model.* **337**: 4-14.
- Henning, J.A., Gresswell, R.E., and Fleming, I.A. 2006. Juvenile salmonid use of freshwater emergent wetlands in the floodplain and its implications for conservation management. *N. Am. J. Fish. Manage.* **26**(2): 367-376. doi:10.1577/m05-057.1.
- Hitt, N.P., Eyler, S., and Wofford, J.E. 2012. Dam removal increases American eel abundance in distant headwater streams. *Trans. Am. Fish. Soc.* **141**(5): 1171-1179. doi:10.1080/00028487.2012.675918.
- Hodgson, E.E., Essington, T.E., and Halpern, B.S. 2017. Density dependence governs when population responses to multiple stressors are magnified or mitigated. *Ecology* **98**(10): 2673–2683. doi:10.1002/ecy.1961.
-

-
- Hogg, R.S., Coghlan, S.M., Zydlewski, J., and Gardner, C. 2015. Fish community response to a small-stream dam removal in a Maine coastal river tributary. *Trans. Am. Fish. Soc.* **144**(3): 467-479. doi:10.1080/00028487.2015.1007164.
- Holmes, R. 2018. [A review of contemporary salmonid stocking practices](#). Cawthron Institute.
- Horst T.J. 1975. The assessment of impact due to entrainment of ichthyoplankton. In: Saila S (ed.) *Fisheries and energy production: a symposium*. D. C. Heath, Lexington, MA, p 107–118.
- Hosack, G.R., Hayes, K.R., and Dambacher, J.M. 2008. Assessing model structure uncertainty through an analysis of system feedback and Bayesian networks. *Ecol. Appl.* **18**(4): 1070-1082.
- ICES. 1998. Report of the study group on the precautionary approach to fisheries management. *In* ICES Document CM 1998/ACFM:10. Copenhagen.
- ICES. 2002a. Report of the study group on the further development of the precautionary approach to fishery management. *In* ICES Document CM 2002/ACFM: 10.
- ICES. 2002b. The form of ICES Advice. *In* Report of the ICES Advisory Committee on Fishery Management, 2002. ICES Cooperative Research Report No. 255, Copenhagen.
- ICES. 2012. ICES implementation of advice for data-limited stocks in 2012 in its 2012 Advice. ICES CM 2012/ACOM 68
- ICES. 2017. ICES fisheries management reference points for category 1 and 2 stocks. ICES Advice Technical Guidelines doi:10.17895/ices.pub.3036.
- Jackson, M.C., Loewen, C.J.G., Vinebrooke, R.D., and Chimimba, C.T. 2016. Net effects of multiple stressors in freshwater ecosystems: A meta-analysis. *Global Change Biol.* **22**(1): 180–189. doi:10.1111/gcb.13028.
- Jensen, F.V. 1996. *An Introduction to Bayesian Networks*. Springer, New York.
- Juan-Jordá, M.J., Mosqueira, I., Freire, J., and Dulvy, N.K. 2015. Population declines of tuna and relatives depend on their speed of life. *Proc. R. Soc. B.* **282**(1811). doi:10.1098/rspb.2015.0322.
- Keeley, E.R., Slaney, P.A., and Zaldokas D. 1996. [Estimates Of Production Benefits For Salmonid Fishes From Stream Restoration Initiatives \(No. 4\)](#). Watershed Restoration Program Ministry of Environment, Lands and Parks and Ministry of Forests.
- Kenchington, T.J. 2014. Natural mortality estimators for information-limited fisheries. *Fish and Fisheries* **15**: 533-562.
- Koen-Alonso, M., and Yodzis, P. 2005. Multispecies modelling of some components of the marine community of northern and central Patagonia, Argentina. *Can. J. Fish. Aquat. Sci.* **62**(7): 1490-1512.
- Koenings, J.P., Burkett, R.D., Haddix, M., Kyle, G.B., and Barto, D.L. 2000. [Experimental manipulation of lakes for sockeye salmon \(*Oncorhynchus nerka*\) rehabilitation and enhancement](#). Alaska Department of Fish and Game Division of Fisheries Rehabilitation, Enhancement and Development.
- Kohler, A.E., Pearsons, T.N., Zandt, J.S., Mesa, M.G., Johnson, C.L., and Connolly, P.J. 2012. Nutrient enrichment with salmon carcass analogs in the Columbia River Basin, USA: A stream food web analysis. *Trans. Am. Fish. Soc.* **141**(3): 802-824. doi:10.1080/00028487.2012.676380.
-

-
- Kroeker, K.J., Micheli, F., and Gambi, M.C. 2013. Ocean acidification causes ecosystem shifts via altered competitive interactions. *Nature Climate Change* **3**(2): 156–159. doi:10.1038/nclimate1680.
- Kuparinen, A., and Festa-Bianchet, M. 2017. Harvest-induced evolution: Insights from aquatic and terrestrial systems. *Phil. Trans. R. Soc. B* **372**(1712). doi:10.1098/rstb.2016.0036.
- Lassen, H., Kelly, C., and Sissenwine, M. 2014. ICES advisory framework 1977-2012: from Fmax to precautionary approach and beyond. *ICES J. Mar. Sci.* **71**(2): 166–172.
- Leibold, M.A. 1996. A graphical model of keystone predators in food webs: trophic regulation of abundance, incidence, and diversity patterns in communities. *Am. Nat.* **147**: 784-812.
- Lenihan, H.S., Micheli, F., Shelton, S.W., and Peterson, C.H. 1999. The influence of multiple environmental stressors on susceptibility to parasites: An experimental determination with oysters. *Limnol. Oceanogr.* **44**(3 II): 910–924. doi:10.4319/lo.1999.44.3_part_2.0910.
- Levins, R. 1974. The qualitative analysis of partially specified systems. *Ann. New York Acad. Sci.* **231**: 123-138.
- Link, J. 2010. *Ecosystem-based fisheries management: confronting tradeoffs*. Cambridge University Press.
- Link, J.S., Gaichas, S., Miller, T.J., Essington, T., Bundy, A., Boldt, J., Drinkwater, K.F., and Moksness, E. 2012. Synthesizing lessons learned from comparing fisheries production in 13 northern hemisphere ecosystems: Emergent fundamental features. *Mar. Ecol. Prog. Ser.* **459**: 293–302. doi:10.3354/meps09829.
- Lorenzen, K., and Camp, E.V. 2019. Density-dependence in the life history of fishes: when is a fish recruited? *Fish. Res.* **217**: 5-10.
- Loughlin, K.G., and Clarke, K.D. 2014. A review of methods used to offset residual impacts of development projects on fisheries productivity. *DFO Can. Sci. Advis. Sec. Res. Doc.* 2013/097. vi + 72 p.
- Magnusson, A., Millar, C., and Cooper, A. 2018. Package “icesAdvice.”
- Marcot, B.G., Steventon, J.D., Sutherland, G.D., and McCann, R.K. 2006. Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation. *Can. J. For. Res.* **36**(12): 3063-3074.
- Margenau, T.L. 1992. Survival and cost-effectiveness of stocked fall fingerling and spring yearling muskellunge in Wisconsin. *N. Am. J. Fish. Manage.* **12**(3): 484-493. doi:10.1577/1548-8675(1992)012<0484:SACEOS>2.3.CO;2.
- May, C.L., and Lee, D.C. 2004. The relationships among in-channel sediment storage, pool depth, and summer survival of juvenile salmonids in Oregon Coast Range streams. *N. Am. J. Fish. Manage.* **24**(3): 761-774. doi:10.1577/m03-073.1.
- McCann, R.K., Marcot, B.G., and Ellis, R. 2006. Bayesian belief networks: applications in ecology and natural resource management. *Can. J. For. Res.* **36**(12): 3053-3062.
- McNay, R.S., Marcot, B.G., Brumovsky, V., and Ellis, R. 2006. A Bayesian approach to evaluating habitat for woodland caribou in north-central British Columbia. *Can. J. For. Res.* **36**(12): 3117-3133.
- Meffe, G.K. 1986. Conservation Genetics and the Management of Endangered Fishes. *Fisheries* **11**(1): 14–23. doi:10.1577/1548-8446(1986)011<0014:cgatmo>2.0.co;2.
-

-
- Melbourne-Thomas, J., Wotherspoon, S., Raymond, B., and Constable, A. 2012. Comprehensive evaluation of model uncertainty in qualitative network analyses. *Ecol. Monogr.* **82**(4): 505-519.
- Minns, C.K. 2006. Compensation ratios needed to offset timing effects of losses and gains and achieve no net loss of productive capacity of fish habitat. *Can. J. Fish. Aquat. Sci.* **63**: 1172-1182.
- Moilanen, A., van Teekelen, A.J.A., Ben-Haim, Y., and Ferrier, S. 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratio for impacted habitat. *Restoration Ecol.* **17**: 470-478.
- Morishita, J. 2008. What is the ecosystem approach for fisheries management? *Marine Policy* **32**: 19-26.
- Morley, S.A., Garcia, P.S., Bennett, T.R., and Roni, P. 2005. Juvenile salmonid (*Oncorhynchus* spp.) use of constructed and natural side channels in Pacific Northwest rivers. *Can. J. Fish. Aquat. Sci.* **62**(12): 2811-2821. doi:10.1139/f05-185.
- Naiman, R.J., Bilby, R.E., Schindler, D.E., and Helfield, J.M. 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. *Ecosystems* **5**(4): 399-417. doi:10.1007/s10021-001-0083-3.
- NOAA (National Oceanic and Atmospheric Administration). 1999. Discounting and the Treatment of Uncertainty in Natural Resource Damage Assessment: Technical Paper 99-1. Silver Spring, MD
- Orr, J.A., Vinebrooke, R.D., Jackson, M.C., Kroeker, K.J., Kordas, R.L., Mantyka-Pringle, C., Van den Brink, P.J., De Laender, F., Stoks, R., Holmstrup, M., Matthaei, C.D., Monk, W.A., Penk, M.R., Leuzinger, S., Schäfer, R.B., and Piggott, J.J. 2020. Towards a unified study of multiple stressors: divisions and common goals across research disciplines. *Proc. R. Soc. B* **287**(1926): 20200421. doi:10.1098/rspb.2020.0421.
- Pastorino, P., Pizzul, E., Menconi, V., Bertoli, M., Mugetti, D., Mignone, W., and Prearo, M. 2019. Fish stocking and health risk: A neglected threat for aquatic biodiversity? *Frontiers Mar. Sci.* **6**. doi:10.3389/conf.fmars.2019.07.00148.
- Perälä, T., and Kuparinen, A. 2020. Eco-evolutionary dynamics driven by fishing: From single species models to dynamic evolution within complex food webs. *Evol. Appl.* **13**(10): 2507–2520. doi:10.1111/eva.13058.
- Peterson, D.P., Wenger, S.J., Rieman, B.E., and Isaak, D.J. 2013. Linking climate change and fish conservation efforts using spatially explicit decision support tools. *Fisheries* **38**(3): 112-127.
- Pinnegar, J.K., Blanchard, J.L., Mackinson, S., Scott, R.D., and Duplisea, D.E. 2005. Aggregation and removal of weak-links in food-web models: system stability and recovery from disturbance. *Ecol. Model.* **184**: 229-248.
- Plaganyi, E.E. 2007. Models for an ecosystem approach to fisheries. *FAO Fisheries Technical Paper 477*. FAO, Rome, pp.108.
- Pope, J.G., Rice, J.C., Daan, N., Jennings, S., and Gislason, H. 2006. Modelling an exploited marine fish community with 15 parameters – results from a simple size-based model. *ICES J. Mar. Sci.* **63**: 1029-1044.

-
- Punt, A.E., and Butterworth, D.S. 1995. The effects of future consumption by the Cape fur seal on catches and catch rates of the Cape hakes. 4. Modelling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. *S. Afr. J. Mar. Sci.* **16**(1): 255-285.
- Quigley, J.T., and Harper, D.J. 2006. Effectiveness of fish habitat compensation in Canada in Achieving no net loss. *Environ. Manag.* **37**(3): 351-366.
- R Core Team, 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Rago, P.J. 1984. Production forgone: an alternative method for assessing the consequences of fish entrainment and impingement losses at power plants and other water intakes. *Ecol. Model.* **24**: 79-111.
- Randall, R.G., and Minns, C.K. 2000. Use of fish production per unit biomass ratios for measuring the productive capacity of fish habitats. *Can. J. Fish. Aquat. Sci.* **57**: 1657-1667.
- Randall, R.G., and Minns, C.K. 2002. Comparison of a habitat productivity index (HPI) and an index of biotic integrity (IBI) for measuring the production capacity of fish habitat in nearshore areas of the Great Lakes. *J. Gr. Lakes Res.* **28**: 240-255.
- Randall, R.G., Minns, C.K., and Kelso, J.R.M. 1995. Fish production in freshwaters: are rivers more productive than lakes? *Can. J. Fish. Aquat. Sci.* **52**: 631-643.
- Randall, R.G., Koops, M.A., and Minns, C.K. 2011. A comparison of approaches for integrated management in coastal marine areas of Canada with the historical approach used in the Great Lakes (Bay of Quinte). *Aquat. Ecosyst. Health Manage.* **14**(1), 104-113.
- Raymond, B., McInnes, J., Dambacher, J.M., Way, S., and Bergstrom, D.M. 2011. Qualitative modelling of invasive species eradication on subantarctic Macquarie Island. *J. Appl. Ecol.* **48**: 181-191.
- Reum, J.C.P, MacDonald, P.S., Ferriss, B.E., Farrell, D.M., Harvey, C.J., and Levin, P.S. 2015. Qualitative network models in support of ecosystem approaches to bivalve aquaculture. *ICES J. Mar. Sci.* **72**(8): 2278-2288.
- Roni, P., Bennett, T., Morley, S., Pess, G. R., Hanson, K., Slyke, D. V., and Olmstead, P. 2006. Rehabilitation of bedrock stream channels: the effects of boulder weir placement on aquatic habitat and biota. *River Res. Appl.* **22**(9): 967-980. doi:10.1002/rra.954.
- Roni, P., Pess, G., Beechie, T., and Morley, S. 2010. Estimating changes in coho salmon and steelhead abundance from watershed restoration: how much restoration is needed to measurably increase smolt production? *N. Am. J. Fish. Manage.* **30**(6): 1469-1484. doi:10.1577/m09-162.1.
- Rose, K.A., Tyler, J.A., Chambers, R.C., Klein-McPhee, G., and Danila, D.J. 1996. Simulating winter flounder population dynamics using coupled individual-based young-of-the-year and age-structured adult models. *Can. J. Fish. Aquat. Sci.* **53**: 1071-1091.
- Rosenfeld, J.S., Raeburn, E., Carrier, P.C., and Johnson, R. 2008. Effects of side channel structure on productivity of floodplain habitats for juvenile coho salmon. *N. Am. J. Fish. Manage.* **28**(4): 1108-1119. doi:10.1577/m07-027.1.
- Sæther, B.E., and Bakke, Ø. 2000. Avian life history variation and contribution of demographic traits to the population growth rate. *Ecology* **81**(3): 642-653. doi:10.1890/0012-9658(2000)081[0642:alhvac]2.0.co;2.
-

-
- Schäfer, R.B., and Piggott, J.J. 2018. Advancing understanding and prediction in multiple stressor research through a mechanistic basis for null models. *Global Change Biol.* **24**(5): 1817–1826. doi:10.1111/gcb.14073.
- Schirmer, M., Luster, J., Linde, N., Perona, P., Mitchell, E.A., Barry, D.A., Hollender, J., Cirpka, O.A., Schneider, P., Vogt, T., Radny, D., and Durisch-Kaiser, E. 2014. Morphological, hydrological, biogeochemical and ecological changes and challenges in river restoration – the Thur river case study. *Hydrology and Earth System Sciences* **18**(6), 2449-2462. doi:10.5194/hess-18-2449-2014.
- Schtickzelle, N., and Quinn, T.P. 2007. A metapopulation perspective for salmon and other anadromous fish. *Fish and Fisheries* **8**(4): 297–314. doi:10.1111/j.1467-2979.2007.00256.x.
- Scott, F., Blanchard, J.L., Andersen, K.H. 2014. mizer: an R package for multispecies, trait-based and community size spectrum ecological modelling. *Methods Ecol. Evol.* **5**: 1121-1125.
- Scrivener, J.C., and Brownlee, M.J. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (*Oncorhynchus keta*) and coho salmon (*O. kisutch*) in Carnation Creek, British Columbia. *Can. J. Fish. Aquat. Sci.* **46**(4): 681-696. doi:10.1139/f89-087.
- Scruton, D.A., Pennell, C.J., Robertson, M.J., Ollerhead, L.M., Clarke, K.D., Alfredsen, K., Harby, A., and McKinley, R.S. 2005. Seasonal response of juvenile Atlantic salmon to experimental hydropeaking power generation in Newfoundland, Canada. *N. Am. J. Fish. Manage.* **25**(3): 964-974. doi:10.1577/m04-133.1.
- Scutari M. 2010. Learning Bayesian Networks with the bnlearn R Package. *J. Stat. Softw.* **35**(3): 1–22.
- Shin, Y.J., and Cury, P. 2004. Using an individual-based model of fish assemblages to study the response of size spectra to changes in fishing. *Can. J. Fish. Aquat. Sci.* **61**(3): 414-431.
- Sierp, M.T., Qin, J.G., and Recknagel, F. 2008. Biomanipulation: A review of biological control measures in eutrophic waters and the potential for Murray cod *Maccullochella peelii peelii* to promote water quality in temperate Australia. *Rev. Fish Biol. Fish.* **19**(2): 143-165. doi:10.1007/s11160-008-9094-x.
- Sobocinski, K.L., Greene, C.M., and Schmidt, M.W. 2017. Using a qualitative model to explore the impacts of ecosystem and anthropogenic drivers upon declining marine survival in Pacific Salmon. *Environ. Conserv.* **45**: 278-290.
- Sprules, W.G., and Barth, L.E. 2016. Surfing the biomass size spectrum: some remarks on history, theory, and application. *Can. J. Fish. Aquat. Sci.* **73**: 477-495.
- Steele, J.H., Gifford, D.J., and Collie, J.S. 2011. Comparing species and ecosystem-based estimates of fisheries yields. *Fish. Res.* **111**(3): 139–144. doi:10.1016/j.fishres.2011.07.009.
- Stephens, P.A., Sutherland, W.J., and Freckleton, R.P. 1999. What is the Allee effect? *Oikos* **87**(1): 185–190.
- Steventon, J.D., and Daust, D.K. 2009. Management strategies for a large-scale mountain pine beetle outbreak: Modelling impacts on American martens. *Forest Ecol. Manag.* **257**(9): 1976-1985.
- Taylor, J.J., Rytwinski, T., Bennett, J.R., Smokorowski, K.E., Lapointe, N.W., Janusz, R., Clarke, K., Tonn, B., Walsh, J.C., and Cooke, S.J. 2019. The effectiveness of spawning habitat creation or enhancement for substrate-spawning temperate fish: a systematic review. *Environ. Evidence* **8**: 19. doi:10.1186/s13750-019-0162-6.

-
- Then, A.Y., Hoenig, J.M., Hall, N.G., and Hewitt, D.A. 2015. Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. *ICES J. Mar. Sci.* **72**: 82-92.
- Unwin M., and Gabrielson R. 2018. [Review of historical hatchery releases of Chinook salmon in New Zealand](#). Cawthron Institute.
- van der Lee, A.S., and Koops, M.A. 2016. Are small fishes more sensitive to habitat loss? A generic size-based model. *Can. J. Fish. Aquat. Sci.* **73**: 716-726.
- Varis, O. 1997. Bayesian decision analysis for environmental and resource management. *Environ. Model. Softw.* **12**(2-3): 177-185.
- Vélez-Espino, L.A., and Koops, M.A. 2012. Capacity for increase, compensatory reserves, and catastrophes as determinants of minimum viable population in freshwater fishes. *Ecol. Model.* **247**: 319-326.
- Vélez-Espino, L.A., Fox, M.G., and McLaughlin, R.L. 2006. Characterization of elasticity patterns of North American freshwater fishes. *Can. J. Fish. Aquat. Sci.* **63**: 2050-2066.
- Walters, C.J., Christensen, V., Martell, S.J., and Kitchell, J.F. 2005. Possible ecosystem impacts of applying MSY policies from single-species assessment. *ICES J. Mar. Sci.* **62**(3): 558–568. doi:10.1016/j.icesjms.2004.12.005.
- Wipfli, M.S., Hudson, J.P., Caouette, J.P., and Chaloner, D.T. 2003. Marine subsidies in freshwater ecosystems: salmon carcasses increase the growth rates of stream-resident salmonids. *Trans. Am. Fish. Soc.* **132**(2): 371-381. doi:10.1577/1548-8659(2003)1322.0.co;2.
- Yodzis, P. 1988. The indeterminacy of ecological interactions as perceived through perturbation experiments. *Ecology* **69**(2): 508-515.
- Yodzis, P. 1998. Local trophodynamics and the interaction of marine mammals and fisheries in the Benguela ecosystem. *J. Anim. Ecol.* **67**: 635-658.
- Yodzis, P. and Innes, S. 1992. Body size and consumer-resource dynamics. *Am. Nat.* **139**(6): 1151-1175.
- Young, P.S., Cech, J.J., and Thompson, L.C. 2011. Hydropower-related pulsed-flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs. *Rev. Fish Biol. Fish.* **21**(4): 713-731. doi:10.1007/s11160-011-9211-0.

APPENDIX A. EQUIVALENCY CALCULATIONS

Summary of methods used to estimate equivalency metrics (Table 2.4). The example is hypothetical and does not reflect real data from any specific power plant nor do the parameters used in the calculations reflect recommended values for the species considered. All analyses were conducted in R 4.0.3 (R core team 2020).

Life-history data for each species (Table A.1) were sourced from Coker et al. (2001) and Fishbase (Froese and Pauly 2020). The data were used to build life tables for each species (Tables A.2, A.3, and A.4). Instantaneous mortality was estimated using a predictive relationship based on VBGF coefficients (Then et al. 2015) where:

$$M = 4.118k^{0.73}L_{\infty}^{-0.33}. \quad (\text{A.1})$$

Adult annual survival rate was assumed to constant. Juvenile survival rate was estimated from adult mortality by assuming mortality is an inverse function of length (Lorenzen 2000). Assuming growth follows the VBGF survival between two lengths or ages can be calculated from (van der Lee and Koops 2016):

$$s_t = \left(\frac{L_{t+\Delta t}}{L_t} e^{-k\Delta t} \right)^{\frac{ML_{mat}}{kL_{\infty}}}. \quad (\text{A.2})$$

Where Δt is the length of time survival is estimated over and L_{mat} is length at maturity (the reference length). Equation A.2 was applied between 6 months of age and T_{mat} . Before 6 months, mortality was expected to be higher than the predicted values from Equation A.2. Egg mortality was assumed to be 90% (Jung et al. 2009). Larval mortality (hatch to age-0.5) was solved for such that the intrinsic rate of population increase, r , was 0 (stability). Length-, weight-, and fecundity-at-age were estimated with the species' relationships.

Stage-specific fish mortality counts were assigned to age-classes (Table A.5). 75% of the YOY stage mortalities were assigned to the larval stage with the remaining 25% assigned to age-0.5. Fish mortality counts in the juvenile and adult stages were assigned using the within stage stable-age-distributions; assuming the rate of impingement and entrainment reflects the relative proportion of age-classes in the environment.

Mortality was assumed to be continuous throughout the year. To simplify calculations of the metrics, all ages of fish age-1+ were set to $\frac{1}{2}$ year values; typically, however, survival rates should be scaled to when mortality occurred during the year.

Equivalent ages was calculated using age-1 as the AOE (Tables A.6, A.7, A.8). EA-1 values were generated as counts and biomass. Biomass as EA-1 was estimated by multiplying EA-1 counts by expected mean weight and an Age-1 individual estimated from the VBGFs and length-weight relationships (Table A.1). Equivalent age estimates are constant within stages because the individuals were distributed among ages within stages using the stable age distribution (i.e., survival schedule).

Reproductive potential values were generated as the total reproductive potential forgone and the age-1 equivalent reproductive potential (EA-1 RP) counts and biomass (Table A.9, A.10, A.11). Survival rates were estimated at $\frac{1}{2}$ year increments to account for the time between mean mortality during the year and spawning. EA-1 RP was calculated by dividing age-specific RP estimates by age-1 RP and multiplying by the age-specific fish mortalities.

Production forgone values were estimated as lifetime production forgone and annual production forgone (Tables A.12, A.13, A.14). The annual PF estimate represents the biomass production lost from just a single year and does not account for the ongoing production from the remainder of the fish's life. Annual PF was estimated for comparison to HPI estimates. HPI estimates were

the total annual biomass of species-specific mortality times the estimate of P/B estimated using expected weight-at-maturity.

REFERENCES

- Coker, G.A, Portt, C.B., and Minns, C.K. 2001. Morphological and Ecological Characteristics of Canadian Freshwater Fishes. Can. MS Rpt. Fish. Aquat. Sci. 2554: iv+89p.
- Froese, R. and D. Pauly. Editors. 2020. FishBase. World Wide Web electronic publication. www.fishbase.org, version (12/2020).
- Jung, S., Choi, I., Jin, H., Lee, D., Cha, H, Kim, Y., and Lee, J. 2009. Size-dependent mortality formulation for isochronal fish species based on their fecundity: an example of Pacific Cod (*Gadus microcephalus*) in the eastern coastal areas of Korea. Fish. Res. **97**: 77-85.
- Lorenzen, K. 2000. Allometry of natural mortality as a basis for assessing optimal release size in fish-stocking programmes. Can. J. Fish. Aquat. Sci. **57**: 2374-2381.
- R Core Team, 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Then, A.Y., Hoenig, J.M., Hall, N.G., and Hewitt, D.A. 2015. Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. ICES J. Mar. Sci. **72**: 82-92.
- van der Lee, A.S., and Koops, M.A. 2016. Are small fishes more sensitive to habitat loss? A generic size-bases model. Can. J. Fish. Aquat. Sci. **73**: 716-726.

TABLES

Table A.1. Life-history data used to parameterize equivalency metrics for species in a hypothetical example. Data were sourced from Coker et al. (2001) and Fishbase (Froese and Pauly 2020).

Parameter	Alewife	White Sucker	Gizzard Shad
Longevity	6	15	6
Age-at-Maturity	3	4	2
VBGF	$21.4(1 - e^{-0.405t})$	$51.7(1 - e^{-0.152(t+0.108)})$	$43.7(1 - e^{-0.48t})$
L-W relationship	$0.012L(cm)^{2.88}$	$0.0153L(cm)^{2.94}$	$0.0129L(cm)^{3.04}$
Fecundity	$-101,534 + 914.5L(mm)$	$0.521L(mm)^{1.754}$	300,000
Instantaneous Mortality	0.775	0.283	0.693

Table A.2. Life table for Alewife.

Stage	Age	Length (cm)	Weight (g)	Maturity	Fecundity	Survival
Egg	0.0	0.00	0.00	0	0	0.1000
YOY	Larva	0.50	0.00	0	0	0.0028
	0.5	3.92	0.61	0	0	0.3411
Juvenile	1.0	7.13	3.43	0	0	0.2917
	2.0	11.88	14.95	0	0	0.4219
Adult	3.0	15.05	29.55	1	36,553	0.4609
	4.0	17.16	43.15	1	55,955	0.4609
	5.0	18.58	54.16	1	68,895	0.4609
	6.0	19.52	62.45	1	77,525	0.0000

Table A.3. Life table for White Sucker.

Stage	Age	Length (cm)	Weight (g)	Maturity	Fecundity	Survival
Egg	0.0	0.00	0.00	0	0	0.1000
YOY	Larva	0.50	0.00	0	0	0.0013
	0.5	4.56	1.33	0	0	0.5754
Juvenile	1.0	8.01	6.95	0	0	0.5354
	2.0	14.17	37.16	0	0	0.6664
	3.0	19.47	94.43	0	0	0.7312
Adult	4.0	24.01	175.02	1	7,810	0.7534
	5.0	27.92	272.56	1	10,173	0.7534
	6.0	31.27	380.48	1	12,413	0.7534
	7.0	34.15	493.02	1	14,489	0.7534
	8.0	36.62	605.61	1	16,380	0.7534
	9.0	38.75	714.88	1	18,084	0.7534
	10.0	40.58	818.51	1	19,606	0.7534
	11.0	42.15	915.06	1	20,954	0.7534
	12.0	43.49	1,003.76	1	22,144	0.7534
	13.0	44.65	1,084.34	1	23,188	0.7534
	14.0	45.64	1,156.86	1	24,101	0.7534
15.0	46.50	1,221.65	1	24,897	0.0000	

Table A.4. Life table for Gizzard Shad.

Stage	Age	Length (cm)	Weight (g)	Maturity	Fecundity	Survival
Egg	0.0	0.00	0.00	0	0	0.1000
YOY	Larva	0.50	0.00	0	0	0.0001
	0.5	9.32	11.43	0	0	0.4816
Juvenile	1.0	16.66	66.74	0	0	0.4246
Adult	2.0	26.97	288.64	1	300,000	0.5002
	3.0	33.35	550.37	1	300,000	0.5002
	4.0	37.29	773.30	1	300,000	0.5002
	5.0	39.74	937.77	1	300,000	0.5002
	6.0	41.25	1,050.46	1	300,000	0.0000

Table A.5. Death of fish data assigned to age-classes.

Species	Stage	Age	Proportion of harm by stage	Mortalities
Alewife	YOY	0.1	0.75	140,625.00
		0.5	0.25	46,875.00
	Juvenile	1.0	0.70	35,164.10
		2.0	0.30	14,835.90
	Adult	3.0	0.60	7,470.34
		4.0	0.28	3,442.90
		5.0	0.13	1,586.75
		6.0	0.00	0.00
White Sucker	YOY	0.1	0.75	937.50
		0.5	0.25	312.50
	Juvenile	1.0	0.46	406.30
		2.0	0.31	270.74
	Adult	3.0	0.23	197.97
		4.0	0.26	96.76
		5.0	0.19	72.90
		6.0	0.15	54.93
		7.0	0.11	41.38
		8.0	0.08	31.18
		9.0	0.06	23.49
		10.0	0.05	17.70
		11.0	0.04	13.33
		12.0	0.03	10.05
		13.0	0.02	7.57
		14.0	0.02	5.70
15.0	0.00	0.00		
Gizzard Shad	YOY	0.1	0.75	16,875.00
		0.5	0.25	5,625.00
	Juvenile	1.0	1.00	6,000.00
	Adult	2.0	0.53	799.82
		3.0	0.27	400.03
		4.0	0.13	200.08
		5.0	0.07	100.07

Table A.6. Age-1 equivalents age-specific estimates for Alewife.

Stage	Age	Survival	Survival to/from Age-1	Harm	EA1-Count	EA1-Biomass (kg)
YOY	0.1	0.0028	0.0010	140,625.00	135.14	0.46
	0.5	0.3411	0.3411	46,875.00	15,990.84	54.87
NA	1.0	1.0000	1.0000	NA	NA	NA
Juvenile	1.5	0.3694	0.3694	35,164.10	95,193.39	326.67
	2.5	0.4589	0.1695	14,835.90	87,509.63	300.30
Adult	3.5	0.4609	0.0781	7,470.34	95,608.92	328.09
	4.5	0.4609	0.0360	3,442.90	95,608.92	328.09
	5.5	0.4609	0.0166	1,586.75	95,608.92	328.09

Table A.7. Age-1 equivalents age-specific estimates for White Sucker.

Stage	Age	Survival	Survival to/from Age-1	Harm	EA1-Count	EA1-Biomass (kg)
YOY	0.1	0.0013	0.0007	937.50	0.70	0.00
	0.5	0.5754	0.5754	312.50	179.80	1.25
NA	1.0	1.0000	1.0000	NA	NA	NA
Juvenile	1.5	0.6141	0.6141	406.30	661.61	4.60
	2.5	0.7035	0.4320	270.74	626.67	4.35
	3.5	0.7526	0.3252	197.97	608.84	4.23
Adult	4.5	0.7534	0.2450	96.76	394.98	2.74
	5.5	0.7534	0.1846	72.90	394.98	2.74
	6.5	0.7534	0.1391	54.93	394.98	2.74
	7.5	0.7534	0.1048	41.38	394.98	2.74
	8.5	0.7534	0.0789	31.18	394.98	2.74
	9.5	0.7534	0.0595	23.49	394.98	2.74
	10.5	0.7534	0.0448	17.70	394.98	2.74
	11.5	0.7534	0.0338	13.33	394.98	2.74
	12.5	0.7534	0.0254	10.05	394.98	2.74
	13.5	0.7534	0.0192	7.57	394.98	2.74
	14.5	0.7534	0.0144	5.70	394.98	2.74

Table A.8. Age-1 equivalents age-specific estimates for Gizzard Shad.

Stage	Age	Survival	Survival to/from Age-1	Harm	EA1-Count	EA1-Biomass (kg)
YOY	0.1	0.0001	0.0000	16,875.00	0.68	0.05
	0.5	0.4816	0.4816	5,625.00	2,708.83	180.80
NA	1.0	1.0000	1.0000	NA	NA	NA
Juvenile	1.5	0.4954	0.4954	6,000.00	12,112.20	808.42
Adult	2.5	0.5002	0.2478	799.82	3,228.20	215.46
	3.5	0.5002	0.1239	400.03	3,228.20	215.46
	4.5	0.5002	0.0620	200.08	3,228.20	215.46
	5.5	0.5002	0.0310	100.07	3,228.20	215.46

Table A.9. Reproductive potential age-specific estimates for Alewife. RP values were converted into age-1 equivalents with the EA-1 ratio (RP_t/RP_1).

Stage	Age	Survival	Fecundity (x1000)	Harm (x1000)	RP per age (x1000)	RP Total (x1000)	EA-1 Ratio	EA-1 Count	EA-1 Biomass (kg)
YOY	0.1	0.0028	0.00	140.63	0.004	497.6	0.001	135.14	0.46
	0.5	0.3411	0.00	46.88	1.26	58,879.56	0.34	15,990.84	54.87
Juv.	1.0	0.5000	0.00	0.00	3.68	0.00	1.000	0.00	0.00
	1.5	0.5833	0.00	35.16	7.36	258,928.9	2	70,321.36	241.32
	2.0	0.6333	0.00	0.00	12.62	0.00	3.43	0.00	0.00
	2.5	0.6662	0.00	14.84	19.93	295,734.5	5.41	80,317.22	275.62
Adult	3.0	0.6789	36.55	0.00	29.92	0.00	8.12	0.00	0.00
	3.5	0.6789	0.00	7.47	25.80	192,738.2	7.01	52,344.91	179.63
	4.0	0.6789	55.95	0.00	38.00	0.00	10.32	0.00	0.00
	4.5	0.6789	0.00	3.44	28.00	96,415.1	7.61	26,184.94	89.86
	5.0	0.6789	68.89	0.00	41.25	0.00	11.20	0.00	0.00
	5.5	0.6789	0.00	1.59	26.32	41,755.7	7.15	11,340.24	38.92
	6.0	1.0000	77.53	0.00	38.76	0.00	10.53	0.00	0.00

Table A.10. Reproductive potential age-specific estimates for White Sucker. RP values were converted into age-1 equivalents with the EA-1 ratio (RP_t/RP_1).

Stage	Age	Survival	Fecundity (x1000)	Harm	RP per age (x1000)	RP Total (x1000)	EA-1 Ratio	EA-1 Count	EA-1 Biomass (kg)
YOY	0.1	0.0013	0	937.50	0.004	4.1	0.001	0.70	0.00
	0.5	0.5754	0	312.50	3.37	1,051.7	0.58	179.80	1.25
Juvenile	1.0	0.7003	0	0.00	5.85	0.00	1.00	0.00	0.00
	1.5	0.7644	0	406.30	8.35	3,393.4	1.43	580.14	4.03
	2.0	0.8034	0	0.00	10.93	0.00	1.87	0.00	0.00
	2.5	0.8295	0	270.74	13.60	3,682.1	2.33	629.5	4.37
	3.0	0.8481	0	0.00	16.40	0.00	2.80	0.00	0.00
	3.5	0.8621	0	197.97	19.33	3,827.2	3.31	654.30	4.55
Adult	4.0	0.8680	7.81	0.00	22.42	0.00	3.83	0.00	0.00
	4.5	0.8680	0.00	96.76	21.93	2,121.9	3.75	362.76	2.52
	5.0	0.8680	10.17	0.00	25.26	0.00	4.32	0.00	0.00
	5.5	0.8680	0.00	72.90	24.02	1,751.0	4.11	299.36	2.08
	6.0	0.8680	12.41	0.00	27.67	0.00	4.73	0.00	0.00
	6.5	0.8680	0.00	54.93	25.67	1,410.1	4.39	241.08	1.68
	7.0	0.8680	14.49	0.00	29.58	0.00	5.06	0.00	0.00
	7.5	0.8680	0.00	41.38	26.83	1,110.3	4.59	189.82	1.32
	8.0	0.8680	16.38	0.00	30.91	0.00	5.29	0.00	0.00
	8.5	0.8680	0.00	31.18	27.42	855.0	4.69	146.16	1.02
	9.0	0.8680	18.08	0.00	31.59	0.00	5.40	0.00	0.00
	9.5	0.8680	0.00	23.49	27.35	642.6	4.68	109.85	0.76
	10.0	0.8680	19.61	0.00	31.51	0.00	5.39	0.00	0.00
	10.5	0.8680	0.00	17.70	26.50	469.1	4.53	80.19	0.56
	11.0	0.8680	20.95	0.00	30.53	0.00	5.22	0.00	0.00
	11.5	0.8680	0.00	13.33	24.70	329.3	4.22	56.30	0.39
	12.0	0.8680	22.14	0.00	28.45	0.00	4.86	0.00	0.00
	12.5	0.8680	0.00	10.05	21.71	218.1	3.71	37.29	0.26
13.0	0.8680	23.19	0.00	25.01	0.00	4.28	0.00	0.00	
13.5	0.8680	0.00	7.57	17.22	130.3	2.95	22.28	0.15	
14.0	0.8680	24.10	0.00	19.84	0.00	3.39	0.00	0.00	
14.5	0.8680	0.00	5.70	10.81	61.6	1.85	10.54	0.07	
15.0	1.0000	24.90	0.00	12.45	0.00	2.13	0.00	0.00	

Table A.11. Reproductive potential age-specific estimates for Gizzard Shad. RP values were converted into age-1 equivalents with the EA-1 ratio (RP_t/RP_1).

Stage	Age	Survival	Fecundity (x1000)	Harm (x1000)	RP per age (x1000)	RP Total (x1000)	EA-1 Ratio	EA-1 Count	EA-1 Biomass (kg)
YOY	0.1	0.0001	0.00	16.90	0.004	60.5	0.001	0.68	0.05
	0.5	0.4816	0.00	5.62	42.60	239,621.4	0.48	2,708.83	180.80
Juvenile	1.0	0.6196	0.00	0.00	88.46	0.00	1.00	0.00	0.00
	1.5	0.6853	0.00	6.00	142.77	856,608.0	1.61	9,683.63	646.33
Adult	2.0	0.7072	300	0.00	208.34	0.00	2.36	0.00	0.00
	2.5	0.7072	0.00	0.80	144.59	115,642.5	1.63	1,307.30	87.25
	3.0	0.7072	300	0.00	204.44	0.00	2.31	0.00	0.00
	3.5	0.7072	0.00	0.40	139.08	55,637.6	1.57	628.96	41.98
	4.0	0.7072	300	0.00	196.66	0	2.22	0.00	0.00
	4.5	0.7072	0.00	0.20	128.08	25,626.0	1.45	289.69	19.34
	5.0	0.7072	300	0.00	181.11	0	2.05	0.00	0.00
	5.5	0.7072	0.00	0.10	106.08	10,615.6	1.20	120.01	8.01
6.0	1.0000	300	0.00	150.00	0	1.70	0.00	0.00	

Table A.12. Production forgone (PF) age-specific estimates for Alewife.

Stage	Age	Weight (g)	Survival	Harm	M	G	PF - Annual (kg)	PF - Lifetime (kg)
YOY	0.1	0.0016	0.0028	140,625.00	5.872	5.933	1.402	3.332
	0.5	0.6149	0.1706	46,875.00	1.769	2.620	119.115	228.412
Juvenile	1.5	8.4458	0.3694	35,164.10	0.996	0.966	282.565	480.641
	2.5	22.1864	0.4589	14,835.90	0.779	0.501	144.003	226.232
Adult	3.5	36.6179	0.4609	7,470.34	0.775	0.292	63.233	90.217
	4.5	49.0096	0.4609	3,442.90	0.775	0.180	22.785	26.985
	5.5	58.6283	0.4609	1,586.75	0.775	0.063	4.120	4.120

Table A.13. Production forgone (PF) age-specific estimates for White Sucker.

Stage	Age	Weight (g)	Survival	Harm	M	G	PF - Annual (kg)	PF - Lifetime (kg)
YOY	0.1	0.0020	0.0013	937.50	6.650	6.501	0.011	0.128
	0.5	1.3277	0.4030	312.50	0.909	2.642	2.947	29.990
Juvenile	1.5	18.6463	0.6141	406.30	0.488	1.210	13.447	87.255
	2.5	62.5447	0.7035	270.74	0.352	0.748	15.553	80.088
	3.5	132.1851	0.7526	197.97	0.284	0.519	15.305	67.078
Adult	4.5	222.0954	0.7534	96.76	0.283	0.383	8.645	33.623
	5.5	325.6126	0.7534	72.90	0.283	0.293	6.991	24.977
	6.5	436.4911	0.7534	54.93	0.283	0.230	5.379	17.986
	7.5	549.5503	0.7534	41.38	0.283	0.184	3.993	12.607
	8.5	660.8294	0.7534	31.18	0.283	0.150	2.887	8.614
	9.5	767.5088	0.7534	23.49	0.283	0.123	2.045	5.727
	10.5	867.7349	0.7534	17.70	0.283	0.102	1.425	3.683
	11.5	960.422	0.7534	13.33	0.283	0.085	0.981	2.257
	12.5	1,045.067	0.7534	10.05	0.283	0.071	0.668	1.276
	13.5	1,121.588	0.7534	7.57	0.283	0.059	0.452	0.608
14.5	1,190.197	0.7534	5.70	0.283	0.026	0.156	0.156	

Table A.14. Production forgone (PF) age-specific estimates for Gizzard Shad.

Stage	Age	Weight (g)	Survival	Harm	M	G	PF - Annual (kg)	PF - Lifetime (kg)
YOY	0.1	0.0016	0.0001	16,875.00	9.383	8.894	0.186	0.415
	0.5	11.4349	0.2984	5,625.00	1.210	2.668	388.357	907.213
Juvenile	1.5	164.8343	0.4954	6,000.00	0.703	0.938	1,046.363	1,854.827
Adult	2.5	421.2084	0.5002	799.82	0.693	0.463	139.158	217.557
	3.5	668.8747	0.5002	400.03	0.693	0.255	55.135	78.400
	4.5	862.7316	0.5002	200.08	0.693	0.147	19.617	23.265
	5.5	999.7871	0.5002	100.07	0.693	0.049	3.648	3.648

Table A.15. Example of accounting for time lag.

Year	Discount rate (d_t)	Impact			Offset		
		Value (kg) (X)	Schedule (p_t)	Discounted value (kg) ($Xp_t d_t$)	Value (kg) (X)	Schedule (p_t)	Discounted value (kg) ($Xp_t d_t$)
1	1.00	7,651.65	1	7,651.65	11,197.36	0	0.00
2	0.97	7,651.65	1	7,428.79	11,197.36	0	0.00
3	0.94	7,651.65	1	7,212.41	11,197.36	0	0.00
4	0.92	7,651.65	1	7,002.34	11,197.36	0	0.00
5	0.89	7,651.65	1	6,798.39	11,197.36	0	0.00
6	0.86	7,651.65	1	6,600.38	11,197.36	0.2	1,931.79
7	0.84	7,651.65	1	6,408.14	11,197.36	0.4	3,751.05
8	0.81	7,651.65	1	6,221.49	11,197.36	0.6	5,462.69
9	0.79	7,651.65	1	6,040.28	11,197.36	0.8	7,071.44
10	0.77	7,651.65	1	5,864.35	11,197.36	1	8,581.84
11	0.74	7,651.65	1	5,693.55	11,197.36	1	8,331.89
12	0.72	7,651.65	1	5,527.71	11,197.36	1	8,089.21
13	0.70	7,651.65	1	5,366.71	11,197.36	1	7,853.60
14	0.68	7,651.65	1	5,210.40	11,197.36	1	7,624.86
15	0.66	7,651.65	1	5,058.64	11,197.36	1	7,402.77
16	0.64	7,651.65	1	4,911.30	11,197.36	1	7,187.16
17	0.62	7,651.65	1	4,768.26	11,197.36	1	6,977.83
18	0.61	7,651.65	1	4,629.37	11,197.36	1	6,774.59
19	0.59	7,651.65	1	4,494.54	11,197.36	1	6,577.27
20	0.57	7,651.65	1	4,363.63	11,197.36	1	6,385.70
21	0.55	7,651.65	1	4,236.53	11,197.36	1	6,199.71
22	0.54	7,651.65	1	4,113.14	11,197.36	1	6,019.13
23	0.52	7,651.65	1	3,993.34	11,197.36	1	5,843.82
24	0.51	7,651.65	1	3,877.03	11,197.36	1	5,673.61
25	0.49	7,651.65	1	3,764.10	11,197.36	1	5,508.36
26	0.48	7,651.65	1	3,654.47	11,197.36	1	5,347.92
27	0.46	7,651.65	1	3,548.03	11,197.36	1	5,192.16
28	0.45	7,651.65	1	3,444.69	11,197.36	1	5,040.93
29	0.44	7,651.65	1	3,344.36	11,197.36	1	4,894.11
30	0.42	7,651.65	1	3,246.95	11,197.36	1	4,751.56
Total				154,475			154,475

APPENDIX B. OFFSETTING SYSTEMATIC REVIEW

Table B.1. Critical appraisal to assess project validity based on study design and assessment bias.

Category no/ bias/ data quality feature	Specific data feature	Study design	Score	Validity
1. Selection and performance bias: study design	Design	BACI	NA	High
		BA, CI, or Incomplete BACI	NA	Medium
		BA comparison (> 3 before, > 3 after)	25	NA
	Temporal repetition	BA comparison (< 3 before, > 3 after)	20	NA
		BA comparison (> 3 before, < 3 after)	15	NA
		BA comparison (< 3 before, < 3 after)	10	NA
		Deficient BA	5	NA
		No BA	0	NA
		Spatial repetition	Site comparison/CI > 2 control and impact)	25
	Site comparison/CI < 2 control, > 2 impact)		20	NA
	Site comparison/CI > 2 control, < 2 impact)		15	NA
	Site comparison/CI < 2 control, < 2 impact)		10	NA
	Deficient CI		5	NA
	No CI		0	NA
2. Assessment bias	Measured Outcome		Quantitative	NA
		Quantitative estimate	NA	Medium
		Semi quantitative	NA	Low
	Monitoring	Frequent Mon	NA	High
		1 time Mon	NA	Medium
		No Mon	NA	Low

Temp + Spat score	30 to 50	High
	20 to < 30	Medium
	0 to < 20	Low

Table B.2. Systematic review and meta-analysis protocol.

<p>1. Search Strategy for meta-analysis PI(E)CO search PI(E)CO search criteria were used to define the important aspects (James et al. 2016)</p>
<p>Early screening articles were specifically referring to the concrete negative impacts on fish populations or the causation e.g., flow alteration. However, the key focus of the review is that residual fish mortality is present on a temporal scale, happens in certain intervals or persist after construction of the development project is done. Thus:</p>
<p>P – Fish populations I (E) - must lead to mortality or serious harm C – pre-assessment comparing offset to O – negative impact must be offset plus recorded method and outcome put in terms of productivity, abundance, condition, diversity, or biomass</p>
<p>2. Search terms Search terms were based on the results of screening and extracting keywords from several scientific and grey literature documents covering the topic of fish mortality in regard to human development projects.</p>
<p>Fish, Fisheries, Productivity, Habitat, Offset, Measures, Report, Residual, Mortality, Canada, Monitoring, No net loss, Ecosystem, Aquatic, Effort, Development, Creation, Restoration, Temporal, Nutrient, Addition, Chemical, Restoration, Alter, Increase, Policy, Net, Effective, Commercial, Recreational, Mitigation, Banking, Avoid, Practice, Negative, Mitigate, Outcome, Maintaining, Priority, Reducing, Relocation, Ocean, Unavoidable, Ongoing, Manage, Sustainable, Techniques, Stocking, Dam, Passage, Downstream, Oxygen, Discharge, Electrical, Power, Turbine, Energy, Disturbance, Salmon, Young, Juvenile, Size, Chronic, abundance, Fishway, Spawner, Shutdown, Turbine, Growth, Species, Preservation, Food, Nutrients, Insects, Invertebrates, Resources, Population, Abundance, Water, Flow, Discharge, Speed, Velocity</p>
<p>Keywords were extracted using R and the packages ‘slowraker’ ‘udpipe’ and ‘textrank’ and the ‘rake’ command which part of base R (JONES 2017).</p>
<p>3. Boolean Search String</p>
<p>Boolean search terms were formulated and used for Web of Science, Google Scholar and to some extent the web-scraper. All searches were streamlined through publish or perish software. (Fish* OR Spawn* OR Juvenile OR Salmon* OR Young OR Species) AND (Aquatic OR Stream OR River* OR Lake OR Impound* OR Pond OR Reserv* OR Ocean OR Coast* OR Eco* OR Lentic OR Lotic OR Marine OR Freshwater) AND (Mortality OR Kill* OR Harm OR Injur*) AND (Tempor* OR Residual OR Remain* OR Continu*) AND</p>

(Develop* OR Anthropo* OR Industr* OR Farm* OR Construct* OR Power OR Turbine OR Electric*)

AND

(Offset* OR Compensat* OR Mitigat* OR Reduc* OR Reverse OR NNL OR No Net Loss)

AND

(Creat* OR Restor* OR Enhanc* OR Preserv* OR Bank* OR Credit)

AND (Product* OR Biomass OR Abundan* OR Biodiversity OR Divers*)

AND (Method* OR Polic* OR Outcome OR Report* OR Manag* OR Monitor* OR Practice)

4. Search output

Google Scholar First 200 search results screened sorted by relevance August 17th

Web of Science Full search using the Boolean search terms – 181 results August 19th

Grey literature Any other listed websites, parsed through the web-scraper in combination with Boolean search terms

1. Alberta Hydro – 2654 parsed sites, 0 hits

2. US Corps of Engineers – split into regional divisions – 25199 parsed sites, 79 collected pages

3. Fisheries and Oceans Canada – Waves – 86722 parsed sites, 28 collected pages

4. NOAA – 12872 parsed sites, 321 collected pages

5. Google Scholar extended – 78798 parsed sites, 140 collected pages

5. Literature assessment and scan

Title and abstract screening

Google Scholar: 29 documents and papers saved as pdfs out of 200

Web of science: 181 – 35 documents and papers saved as pdfs

Combined other websites: 34 papers and documents saved as pdfs

Body of literature Total: 98 documents - 30 with usable data – validity assessment (Appendix Table B.1)

BARNTHOUSE ET AL.	2019	MORLEY ET AL.	2005
CLARKE ET AL.	2008	PATRICK ET AL.	2015
GIBEAU ET AL.	2020	RAYMOND	1988
GREENWOOD	2008	RONI ET AL.	2006
HADDERINGH & JAGER	2002	RONI ET AL.	2010
HANSEN ET AL.	2017	ROSENFELD ET AL.	2008
HARVEY ET AL.	1998	SCRIVENER & BROWNLEE	1988

HIGGINS & BRADFORD	1996	SCRUTON ET AL.	2005
HITT ET AL.	2012	SKALSKI ET AL.	2016
HORNE ET AL.	2004	STANTEC	2017
HUNT ET AL.	2012	STOCKNER & MACISAAC	1996
KEELEY ET AL.	1996	THOMAS ET AL.	2013
KNIGHT PIÉSOLD LTD.	2015	TONALLA ET AL.	2017
LEMLY	2010	UNWIN & GABRIELSON	2018
MAES ET AL.	2004	YOUNG ET AL.	2011

Table B.3. Project information divided into offsets, methods, common metric, and study validity based on table B.1.

ID	Offset	Method	Metric	Study Validity	Effect size	Species	Sites Reference/Treatment
1	Habitat Creation	Off channel construction	Density	High	0.76137	Coho	7/7
2	Habitat Creation	Off channel construction	Density	High	0.87095	Coho, Chum, Chinook	11/11
3	Habitat Creation	Off channel construction	Density	High	0.22857	Coho	9/10
4	Habitat Creation	Pond & F-plain	Density	Medium	1.65517	Coho, Steelhead	3/5
5	Habitat Creation	Pond & F-plain	Density	Medium	0.81937	Coho, Steelhead	10/10
6	Habitat Creation	Channel & Flow	Biomass	High	2.35467	Brook Trout	3/3
7	Habitat Creation	Off channel construction	Density	Medium	0.52446	Chum, Coho	24/24
8	Habitat Creation	Off channel construction	Survival	Medium	1.56256	Coho	24/24
9	Habitat Creation	Floodplain & Channel	Density (smolt production)	Low	0.66032	Coho	5/11
1	Restoration and Enhancement	Culvert removal	Density	Medium	0.64027	Community	6/6
2	Restoration and Enhancement	Large Woody Debris introduction	Density	Medium	0.39474	Salmonids	58/58
3	Restoration and Enhancement	Boulder Weir	Density	Medium	1.38985	Salmonids	12/12
4	Restoration and Enhancement	Logjam	Density	Medium	0.59831	Salmonids	24/24

5	Restoration and Enhancement	Substrate	Density	Medium	0.69442	Salmonids	3/13
6	Restoration and Enhancement	Dam removal	Density	Low	-0.6022	Community	3/13
7	Restoration and Enhancement	Dam removal	Biomass	Low	-0.0157	Community	3/3
8	Restoration and Enhancement	Dam removal	Density	Medium	0.59615	Eel	16/25
9	Restoration and Enhancement	Dam removal	Density	Medium	1.21918	Eel	15/15
10	Restoration and Enhancement	Dam removal	Density	Medium	1.32235	Eel	15/15
11	Restoration and Enhancement	Dam removal	Density	Medium	0.85944	Eel	15/15
12	Restoration and Enhancement	Dam removal	Richness	Medium	-0.2812	Community	15/15
13	Restoration and Enhancement	Dam removal	Richness	Medium	0.71623	Community	5/5
14	Restoration and Enhancement	Riverbank enrichment	Temperature	Low	1.47122	Small-bodied fish	3/3

1	Bio and Chemical Manipulation	Model Stocking	Survival	Medium	0.73521	Salmonids	40/40
2	Bio and Chemical Manipulation	Stock vs Wild	Survival	Medium	-0.54608	Salmonids	21/19
3	Bio and Chemical Manipulation	Stock vs Wild	Survival	Medium	-0.40901	Salmonids	21/18
4	Bio and Chemical Manipulation	Nutrient enrichment	Biomass	High	1.12581	Salmonids	10/10
5	Bio and Chemical Manipulation	Model Stocking	Survival	Medium	1.18182	Salmonids	8/8
6	Bio and Chemical Manipulation	Stock vs Wild	Survival	Low	0.69298	Salmonids	3/3
7	Bio and Chemical Manipulation	Nutrient enrichment	Biomass/ environmental variables	High	0.94533	Community	5/5
