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 2016/046

Pacific Region

Equivalency metrics for the determination of offset requirements for the Fisheries Protection Program

M.J. Bradford[1], K.E. Smokorowski[2], K.D. Clarke[3], B.E. Keatley[4], M.C. Wong[5]

Fisheries and Oceans Canada

[1]School of Resource and Environmental Management
Simon Fraser University, Burnaby, B.C. V5A 1S6

[2]1219 Queen St. E., Sault Ste. Marie, ON P6A 2E5

[3]PO Box 5667, St. John's NL, A1C 5X1

[4]200 Kent St., Ottawa, ON, K1A 0E6

[5]1 Challenger Dr., Dartmouth, NS, B2Y 4A2

May 2016

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.

Research documents are produced in the official language in which they are provided to the Secretariat.

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

ISSN 1919-5044

Correct citation for this publication:

Bradford, M.J., Smokorowski, K.E. Clarke, K.D., Keatley, B.E. and Wong, M.C. 2016. Equivalency metrics for the determination of offset requirements for the Fisheries Protection Program. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/046. vi+32 p.

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ABSTRACT

In November 2013, the Fisheries Protection Provisions (FPP) of the Fisheries Act came into force. The provisions and related policies specify that development projects that cause unavoidable serious harm to fish must provide offsets, such that the benefits from offsetting measures should balance project effects. Equivalency metrics are common currencies used to describe both offset benefits and project effects and are used in equivalency analysis to determine the amount of offsetting required to counterbalance the serious harm.

This paper describes equivalency metrics appropriate for offset determinations under the FPP. The simplest are based on habitat area and can be used when the offset is similar in nature and location to the serious harm. More complex metrics estimate that are often surrogates for fisheries productivity may be needed when offset benefits are different in nature to the serious harm. Metrics range in complexity and in their assumptions and uncertainties. The choice of metric will depend on the extent and nature of the serious harm and the proposed offsetting measures.

Paramètres d'équivalence visant l'établissement d'exigences de compensation aux fins du Programme de protection des pêches

RESUMÉ

Les dispositions de la *Loi sur les pêches* visant la protection des pêches sont entrées en vigueur en novembre 2013. Les dispositions et les politiques connexes précisent que les projets de développement qui provoquent des dommages graves inévitables aux poissons doivent offrir des compensations, afin que les avantages des mesures de compensation contrebalancent les effets du projet. Les paramètres d'équivalence sont des monnaies d'échange courantes qui servent à décrire les mesures de compensation et les effets d'un projet et sont utilisés pour les analyses de l'équivalence afin de déterminer l'ampleur des compensations nécessaires pour contrebalancer les dommages graves.

Ce document décrit les paramètres d'équivalence adaptés à la détermination des compensations aux termes du Programme de protection des pêches (PPP). Les plus simples s'appuient sur la zone d'habitat et peuvent servir quand la compensation présente une nature et un environnement similaires à ceux qui sont touchés par les dommages graves. Des estimations de paramètres plus complexes qui sont plutôt des mesures substitutives de la productivité des pêches peuvent être nécessaires quand la compensation présente une nature différente par rapport aux dommages graves. Les paramètres varient en complexité et en fonction des hypothèses et des incertitudes. Le choix des paramètres dépend de l'ampleur et de la nature des dommages graves et des mesures de compensation proposées.

INTRODUCTION

CONTEXT

In November 2013, the Fisheries Protection Provisions (FPP) of the *Fisheries Act* came into force. These provisions include Section 6 (s.6), which identifies four factors that the Minister must consider before authorizing a project that has the potential to cause serious harm to fish. Specifically, the Minister must consider the measures and standards to avoid, mitigate or offset serious harm to fish that are part of or support a commercial, recreational or Aboriginal fishery. In addition, as set out in the *Applications for Authorization under Paragraph 35(2)(b) of the Fisheries Act Regulations*, the proponent must include an offsetting plan to offset residual serious harm to fish as regulatory requirement when submitting an application for authorization. This offsetting plan must include a description of the measures to offset serious harm to fish, supported by an analysis that should use scientifically defensible methods describing how measures will meet the offsetting objective. The offsetting plan must also outline a monitoring plan that assesses the effectiveness of the offsetting measures as well as a contingency plan should the measures not meet the objective of offsetting.

The *Fisheries Productivity Investment Policy: A Proponent's Guide to Offsetting* (FPIP or the Offsetting Policy), was also published in November 2013. The Offsetting Policy offers flexibility in choosing offset methods provided that increases in fisheries productivity are achieved and that the four key principles outlined in the policy are met.

The second principle in the Offsetting Policy states that "*benefits from offsetting measures must balance project impacts."* This principle is meant to capture the idea of equivalency between impact and offset, in relation to fisheries productivity. While the Offsetting Policy notes that achieving such equivalency may be easier to demonstrate when offsets are designed to provide similar function to the affected habitat, it does not prescribe acceptable methods for calculating losses and gains.

"In-kind" offsetting refers to situations in which the habitat that is destroyed or permanently altered is replaced by the same quantity, quality and type of habitat, potentially with additional habitat offsetting required to account for uncertainty and time lags. The benefits of in-kind offsetting will, by definition, accrue to the fish populations affected by the project. In these situations, balancing the losses to fish and fish habitat caused by a project with the benefits that result from offsetting measures is a straight-forward calculation because the impacts and the offsets are directly comparable both in terms of the metrics used to describe them and the fish populations that will be beneficiaries of the offset.

With an "out-of-kind" approach to offsetting, offsetting measures target factors limiting productivity in a given area by means other than replacing what has been lost. It can be more complicated to measure and compare losses caused by the project with offsetting gains when out-of-kind measures are used, but in some cases greater success in obtaining productivity gains may be achieved with this approach. Out-of-kind offsetting measures may include the restoration or creation of habitat types that are different from the habitat type that was lost, or may involve other types of measures (Loughlin and Clarke 2014, DFO 2014b).

The other 3 principles of the Offsetting Policy provide additional guidance for the determination of out-of-kind offsetting requirements. These are listed here but more information is available in the Policy:

Principle 1: Offsetting measures must support fisheries management objectives or local restoration priorities.

Principle 3: Offsetting measures must provide additional benefits to the fishery.

Principle 4: Offsetting measures must generate self-sustaining benefits over the long term.

STEPS IN DEVELOPING AN OFFSET PLAN

The Offset Policy provides a general overview of the components of an offset plan and provides some general guidance on each step. These are summarized below:

1) Characterize serious harm

The residual serious harm to fish is determined after all avoidance and mitigation measures have been applied. By understanding the nature of the residual serious harm to fish, it is possible to estimate the consequences on fisheries productivity and, in turn, to characterize the contribution of relevant fish to the ongoing productivity of commercial, recreational or Aboriginal fisheries (Paragraph 6(a) of the Fisheries Act). The residual *serious harm to fish* is the loss that must be counterbalanced by the proposed offsetting measures.

The residual *serious harm to fish* should be determined and quantified for each impact type in relation to each phase of a proposed work, undertaking and activity. This may include determining the extent, duration and magnitude of the impacts on fish and fish habitat in terms of the number of fish killed, area of habitat destroyed, area of habitat permanently altered and degree of alteration. Guidance on the determination of impacts is provided elsewhere (DFO 2013b, 2014a; Bradford et al. 2014; Koops et al. unpubl. manus.[1])

2) Select offset measures

The objective and details of proposed offsetting measures should be included in the offsetting plan. The objective of offsetting measures is guided by the extent, duration and magnitude of the residual serious harm to fish and must meet the 4 principles outlined above. The offsetting plan should also include clearly articulated measures of success that are linked to the objective of the offsets as well as benchmarks for measuring progress.

3) Determine amount of offset

The Policy provides the following considerations to guide the determination of amount of offsetting required. Offset measures:

a) should provide benefits that are proportional to the loss caused by the project;

b) may need to be increased in order to manage uncertainty associated with the proposed offset; and

c) may need to be increased when there is a time lag between the impact and the time it takes for the offsetting measure to become functional.

4) Monitoring and reporting of effectiveness

Monitoring and reporting conditions should be described in the offsetting plan as they will be included as conditions of the authorization. Contingency measures should also be identified if the offsetting measures do not meet expectations.

[1] Koops, M.A., Bradford, M.J., Clarke, K.D., Doka, S.E., Enders, E.C., Randall, R.G., Smokorowski, K.E., and Watkinson, D.A. (Unpubl. manus.) A review of scientific evidence supporting generic productivity-state response curves. DFO Can. Sci. Advis. Sec. Res. Doc. (in preparation – # 8187).

EQUIVALENCY

In the context of offsets, "*equivalency*" refers to the process to determine the amount and nature of offsets required to achieve a fair exchange between project impacts and gains associated with offset measures. An equivalency metric (or currency) is the unit of loss or gain that is used to determine how much offsetting is needed to counterbalance unavoidable losses.

The use of equivalency calculations in biodiversity offset programs was reviewed by Clarke and Bradford (2014), and is an active area of academic research and program development (e.g., Bull et al. 2013; Pilgrim and Eckstrom 2014). Equivalency metrics measure gains and losses in accordance with the goals of the biodiversity program (Quétier and Lavoral 2011). Here biodiversity is a general term that could refer to any aspect of the biotic environment, from individual species to complete communities. In the context of FPP, "fisheries productivity" has been defined as the primary objective of offsetting activities under the *Fisheries Act* and will be considered analogous to biodiversity in this discussion. Irrespective on their objectives all biodiversity offsetting programs have a need for a currency for the calculation of offset requirements, whether in the form of "like for like" (in-kind) exchanges, or more complex transactions that involve the trading of one type of impact for a biodiversity gain of a different type. Although in-kind transactions are the simplest to manage, out-of-kind offsets can provide conservation benefits to key habitats or species, or benefits within landscape-level conservation planning. More generalized currencies are required to facilitate these types of exchanges (McKenney and Kiesecker 2010; Habib et al. 2013).

The essence of equivalency calculations can be expressed by the simplified equation (Levrel et al. 2012) that ignores the time dimension and adjustments for uncertainty:

$$A_p V_p I = A_o V_o R \qquad (1)$$

where A is the area of the project (A_p) impacts or offset (A_o) measures, V is the value of the fisheries or ecological resources at each site, I is the intensity of impact (i.e., the proportional reduction in services), and R is the increment of value associated with the offset measure (the increase in services). The parameter I takes a value of unity when habitat is destroyed, and R = 1 for fully successful habitat creation. For the case of successful in-kind habitat replacement (i.e., $V_p = V_o$) $A_p = A_o$, which is the expected in-kind exchange. For other situations equivalency is based on the balance of area, value, and the incremental change caused by project and offset measures. To calculate offset area the equation can be rearranged as

$$A_o = \frac{A_p V_p I}{V_o R} \qquad (2)$$

The use of equation (2) or its more detailed variants requires V_p and V_o be expressed in a common currency; this is the equivalency metric that is the subject of this report.

Principle 3 of the offsetting policy notes that offset requirements may be increased to account for uncertainty and time lags. Uncertainty is usually managed through the use of offset ratios where the offset requirement is increased to decrease the risk of failing to reach policy goals (Moilanen et al. 2009; Bull et al. 2013). Ratios can also be used to account for time differences between project impacts and the benefits, but discount rates are proposed as a useful framework for the consistent inclusion of time in equivalency analyses (Minns 2006; Moilanen et al. 2009; Clarke and Bradford 2014). Detailed analysis of time lags and uncertainties in the context of the Fisheries Protection Program is an area of ongoing research.

SCOPE OF PAPER

The goal of this paper is to review methods and approaches for determining offset requirements that will satisfy the principles of the Offset Policy. The focus is on equivalency currencies or metrics, their applications, merits and limitations. In the U.S., the National Oceanic and Atmospheric Administration (NOAA) has developed a hierarchy of metrics based on habitat, resource (e.g., fish) and value equivalency analyses (Allen et al. 2005; Clarke and Bradford 2014) and we adapt that scheme for application by FPP. We note the application of metrics in the FPP context will be guided by preferences identified in the Offset Policy.

EQUIVALENCY METRICS

Table 1 lists the major categories of equivalency metrics that could be used for the development of offsetting plans for the Fisheries Protection Program. These follow the NOAA sequence from habitat to value-based approaches. Detailed descriptions of each entry are found in subsequent sections.

Table 1. Listing of equivalency metrics that have potential to be used to determine offset requirements.

Metric class	Example application	Example metric
Habitat	In-kind habitat replacement.	Area and type.
Habitat or ecosystem function	Replacement of lost function (i.e., food production) potentially using out-of-kind offset measures.	Habitat function metrics (e.g., cover, substrate) Secondary production.
Habitat suitability or capacity for select species	Reductions in habitat quality or quantity offset by out-of-kind improvements to habitat quality.	Weighted useable area, habitat suitability indices.
Fish abundance	Creation of new habitats with similar expected fish communities. Created habitat may be unlike affected ones.	Biomass, density, smolt production either observed (baseline) or predicted (offset). Regional fish density reference data may be used.
Fish production	Habitat loss or ecosystem transformation requiring out-of-kind offsets. Can be used when new fish community is unlike the one affected by the serious harm.	Fish production lost/gained; direct measurements or regional standards, empirical predictors (P:B ratios).
Yield/fishery benefits	Ecosystem-scale transformations that cause changes to the fish community and the fishery.	Predicted benefits to fishery (catch, angler satisfaction, participation) of the offset relative to losses to the fishery caused by the project. Observed fishery statistics or predictions based on fishery models.
Monetary or other valuation	Scaling out-of-kind offsetting using the cost of replacement of lost habitat or fish.	Replacement cost.

IN-KIND HABITAT REPLACEMENT (AREA/TYPE)

Under the former Policy for the Management of Fish Habitat (DFO 1986) when a harmful alteration, disruption or destruction of fish habitat could not be avoided through mitigation or redesign, compensation was required to achieve the goal of No Net Loss. The first and most preferred option in the Hierarchy of Compensation options was to create or increase productive capacity of in-kind habitat in the same ecological unit, striving to create or enhance habitat that has the same natural integrity, structure, and function of the habitat that was adversely affected, with a preference to working on site before moving off-site (DFO 1986; 2002). Under the new Offset Policy the preferred option for habitat offsets is still to balance losses by benefitting the specific fish populations in the geographic areas that are affected by the proposed development (DFO 2013a). By keeping losses and gains comparable in type and area, it is believed that the productivity and biodiversity of the ecosystem is most likely to be maintained; this view is shared among offsetting programs across a number of jurisdictions globally (Lipton et al. 2008, Moilanen et al. 2009, McKenney and Kiesecker 2010).

Application

Replacing in-kind habitat is the simplest offset method to employ since establishing equivalency is relatively straightforward. Replacing in-kind habitat is best suited for smaller habitat losses that affect the quantity of habitat (e.g., category 1 in DFO 2013b) but do not include habitat conversion (e.g., river to reservoir). An example of this type of offset is the creation of a spawning bed in the vicinity of a lost spawning bed from, for example, a road crossing, or the creation of a wetland in the same waterbody as where a development is destroying a natural wetland (e.g., US wetland policies summarized in Votteler and Muir 2002). The biggest advantage of this currency is the ease of establishing equivalency and for implementing offset ratios as it is a simple summation of equivalent units. This simplicity allows for a streamlined assessment approach and repeatability of application. The biggest drawback of this currency is the assumption that the replacement habitat and the associated fisheries productivity will be equivalent to that lost. The use of surface area may not be appropriate if multiple habitat types are affected by the project.

Data and Methods

The metric for in-kind habitat replacement is the surface area (e.g., m^2) of habitat and since the units are the same for 'in-kind' offsets, establishing equivalency by assessing productivity is not necessary. Calculation of habitat lost is the relatively straightforward method of measuring area lost by habitat type. Establishing compliance with the authorization is simply a matter of measuring the area of the created habitat relative as required by the terms of the authorization. The data streams required include:

a) the project area (in m^2) baseline of existing habitat that will be lost, by habitat type if more than one is being destroyed;

b) The prediction of the value of offset once the habitat is created could be as simple as confirming that a required area of habitat will be created and/or enhanced equivalent to the area lost, plus some multiplier for uncertainty and time lags. Typically, since this type of offset is employed for smaller projects and the replacement habitat is often created, the value of the offset baseline is not considered in the calculation. Typically the fishery value of the offset habitat is not modelled with the in-kind equivalency assumption, although the effectiveness of the created habitat may need to be assessed as part of a monitoring plan (DFO 2012, Smokorowski et al. 2015).

Choices and Assumptions

Technical choices: In-kind habitat replacement assumes that habitat variables (e.g., macrophytes, depth, substrate, nutrients, temperature etc.) can be considered surrogates of productivity where the link to productivity was previously established by empirical research

(Minns 1997). It also assumes that in time created habitat will be functionally the same as naturally established habitat, which may not always be the case (Bull et al. 2013).

Policy choices: There are few policy choices because replacing in-kind habitat theoretically will not alter the habitat available to local fishes (given that the new location will be proximate to the lost habitat) and thus will not (theoretically) change fish community dynamics. The choice of specifically where to create the replacement habitat may have an influence on its ultimate productivity, and since the new habitat may affect existing habitat, the replacement site would need to be deemed of little of no value to the fishery in its existing form.

Key Uncertainties

Replacing in-kind habitat that solely requires measurement of habitat features (area based measurement) has little uncertainty in the measurement of baseline habitat to be lost, or in the prediction of benefits, assuming that the replacement habitat can be successfully constructed. The validity of this assumption is more certain for habitat types which have often been created and which have been the subject of much research (e.g., spawning shoals, Fitzsimons 2014). Uncertainty increases for habitat types less often studied, or in cases where the long-term structural integrity of the habitat is in question (e.g., washing out of a gravel spawning bed in a stream environment or other failures, which can occur in a high percentage of created habitats and is detected when monitored, Smokorowski et al. 1998). Long-term efficacy of in-kind created habitat should be part of the monitoring program; contingency measures may need to be implemented in case of failure.

HABITAT CHARACTERISTICS AND FUNCTION

The second tier in complexity of habitat offsetting metrics are for cases where offsets are intended to replace lost habitat characteristics that are directly linked to one or more ecosystem functions. These fall into the category of service-to-service equivalency analysis, and have been well studied in the literature (e.g., NOAA 2006; Lipton et al. 2008; French McCay et al. 2003a). Ideally, these functions would be directly or indirectly related to CRA fisheries production, and could include habitat features such as structure, cover, substrate type, or be integrated by measures such as secondary production. Habitat function offsets may or may not be applied using the same habitat type as that lost, which could fall under the category of lost habitat quantity or quality (categories 1 and 2 in DFO 2013b).

DFO's Pathway of Effects diagram endpoints list habitat characteristics or functions that are subject to negative residual effects from typical projects (e.g., change in structure and cover, sediment concentrations, nutrient concentrations; see Appendix Tables 1 and 2 in Bradford et al. 2014). These diagrams provide a useful guide for identifying ecosystem functions likely to be negatively affected by the activity. A literature review of how these ecosystem functions (i.e., PoE endpoints) respond to typical human perturbations and the anticipated shape of generic productivity-state response curves is available in Koops et al. (unpubl. manus.[1]). These negative residual effects require offsetting to mitigate lost ecosystem function, and could include replacement of the same habitat features that were destroyed, or replacement by different features that provide an equivalent or different ecosystem function.

An example of an appropriate metric evaluating habitat function is secondary production (i.e., the incorporation of organic matter into body tissue of invertebrate mass per unit time and area; Cusson and Bourget 2005). This metric may be most appropriate in marine ecosystems where fish abundance is difficult to adequately measure, given fish migration among habitats and inshore/offshore migrations. Secondary production integrates over the growth, reproduction, and mortality of individuals, and also implicitly integrates across the environmental conditions secondary producers are subjected to. Thus, secondary production is a good proxy for overall ecosystem functioning, and is a more appropriate metric than standing stock, which simply indicates the amount of biomass present. Secondary production is an appropriate metric for FPP offsetting because it directly supports CRA fisheries through energy flow from primary producers to fish (McArthur and Boland 2006) and accounts for environmental change that can influence CRA fisheries productivity (Dolbeth et al. 2012; Sturdivant et al. 2014). Furthermore, secondary producers in marine ecosystems often create habitat through their body morphology and aggregation (i.e., reefs created by mussels, oysters, or other bivalves; structure created by sponges; Gutiérrez et al. 2003) that can benefit fish communities. Relationships between secondary production and CRA fisheries can be determined using productivity-state response curves.

Application

A habitat function currency is best suited for projects affecting habitat quantity or quality where the offset is designed to balance the lost habitat characteristic or function, or to provide an alternate function which may be deemed preferable in light of predominant habitat availability (or limitation). When the offset replaces the same function (i.e., the same services of the same type and quality), quantifying equivalency will be straightforward (see in-kind habitat above). The habitat function currency also has the advantage of providing some flexibility in terms of the choice of offset. If non-critical or non-limiting services are lost offsetting could be designed to provide an alternate service that is scarce or limiting.

Secondary production might be chosen as a metric to offset damages in coastal marine ecosystems where benthic production is very high, trophic connections between fishes and the benthos are strong, use of benthic habitats by juvenile fish is common, and projects will affect the benthos (e.g., infilling, dredging, oil spills; French McCay et al. 2003b). Furthermore, as described briefly above, secondary production may be particularly useful in marine ecosystems where fish are difficult to sample adequately owing to their movement patterns and the openness of many marine ecosystems. This metric is also be relevant for marine in-kind offsetting where habitat structure is not present but production from the sea bottom would be high and important for fisheries production (i.e., intertidal sand flat, shallow subtidal mud flat). In contrast, secondary production is often not quantified directly in freshwater due to the resource intensive nature of accurate quantification, and the relative simplicity of sampling the fish community directly. If the habitat function of food supply (via secondary production) needs to be quantified in freshwater, proxies are frequently used (see Smokorowski et al. 2015 for examples).

Data and Methods

Examples of common indicators associated with habitat function include measures of substrate type and characteristics, densities or riparian or aquatic macrophytes or large woody debris.

Continuing with our secondary production example, the data required for this metric include the density and biomass of secondary producers (entire community) at some level of taxonomic separation (Dolbeth et al. 2005). Broad taxonomic categories (class or even phylum) may suffice. Relatively simple empirical methods developed to estimate secondary production include:

i. Models that relate Production to Biomass (P:B) with body mass using metabolic principles (e.g., Schwinghamer et al. 1986),

ii. Models that use multiple regression equations that relate P:B to population characteristics (e.g., life span, mean biomass, etc.) and environmental characteristics (e.g., temperature, depth, habitat) (Tumbiolo and Downing 1994, Brey 2001)

iii. Multiplying biomass by P:B available in the literature (Wong et al. 2011).

Use of P:B ratios from the literature is the simplest method for calculating secondary production, and can also be used to estimate fisheries productivity (as per Randall and Minns 2000 and productivity section below). A concern is that regionally appropriate P:B ratios for secondary producers in temperate marine ecosystems in Canada are scarce, and caution is advised in using values from other regions. However, P:B ratios can be easily calculated by using the Brey (2001) empirical model, because this model generates P:B ratios based on water temperature, habitat, and body size and then uses this ratio to determine estimated production. In fact, several empirical models calculate secondary production in this way, by determining P:B through relations with various biological and physical measures, and then simply multiplying biomass data by this ratio. The advantage of the Brey (2001) model is that a spreadsheet is publically available to implement this model. It has been shown to produce accurate and precise estimates of secondary production when compared to more classical methods (Cusson and Bourget 2005; Dolbeth et al. 2005) and is widely used (e.g., Wong et al. 2011) and cited. Data inputs required include biomass and abundance data of broad taxonomic groups (e.g., polychaetes, bivalves, gastropods) and basic environmental measures (i.e, water temperature, habitat type). Model outputs would include both P:B ratios and estimated production.

Computation steps and data requirements for managers and proponents could be reduced if a database of appropriate P:B ratios for secondary producers in various coastal marine habitats was available. Availability of P:B ratios would mean that only biomass data for broad taxonomic groups would be required (as opposed to both biomass and abundance data for the Brey model implementation). Then, biomass data would simply be multiplied by the P:B ratio to produce estimates of secondary production. For Atlantic Canada, data from eelgrass beds and bare soft sediment bottoms are available to begin creation of such a database (data from M. Wong, DFO, Bedford Institute of Oceanography). Inclusion of other important habitats and regions could be identified as a required data need for the FPP.

Choices and Assumptions

Technical choices: Service-to-service equivalency calculations assume that providing an equivalent quality and quantity of habitat service will translate into equivalent fish productivity. When the same service is provided by the offset then it is assumed that the support function provided by the ecosystem service will not alter the fish community dynamics, although it may be influenced by ecosystem context. Out-of-kind service offsets will need additional baseline data, regional benchmarks, or models to both quantify the value of habitat lost, and to predict the value of service gains provided by the offset measures. Technical choices related to habitat-based (e.g. structure/cover/substrate) offsets are described above in the in-kind section and Smokorowski et al. (2015). For an out of kind offsetting metric such as secondary production, important technical choices include:

i. Data collection vs. data compilations: data collected directly from the field site should be more precise than those from data compilations, but where logistics preclude sufficient data collection compilations or regional benchmarks may still remain valuable.

ii. Model inputs: level of taxonomic separation at which secondary production is calculated and availability of environmental measures can influence the level of precision in estimates of secondary production.

iii. Model choice: Choice between using P:B ratios from databases vs those from empirical models will influence the precision of estimates. While multiplying biomass by P:B may be easier to implement, it may be less precise given the P:B used doesn't necessarily come from the field site. Model choice should include consideration of the data used in their development. For example, models developed from data of marine ecosystems would be most appropriate for marine data, whereas models based on freshwater data would not be appropriate for a marine application.

Policy choices: Out-of-kind service offsets could change ecosystem dynamics and favour alternate fishery species relative to what is being harmed by the residual effects. Choices of habitat service offsets should be informed by the principles of the Offset Policy.

Key Uncertainties

The magnitude of uncertainty depends on the metric chosen. In the case of a service-to-service offset where area of habitat of similar function is the metric, measurement uncertainty will be minimal. In this case, the greatest uncertainty will come from the effectiveness of the proposed offset, which depends on the type of habitat service provided. For out-of-kind offsets the uncertainty increases commensurate with the complexity of the metric selected, lack of regional data, difficulty in collecting site-specific field data, and model available for predicting future value of the offset habitat. While specific relationships between certain metrics and aspects of CRA fisheries species may not be available, the scientific literature, expert knowledge, productivity-state curves, and pathways of effects models can all be used to infer linkages. Significant uncertainty may be associated with these inferences.

HABITAT SUITABILITY/CAPACITY FOR SELECT SPECIES

This category includes habitat-based models that make inferences about the biological use of physical habitat features to make quantitative assessments of habitat suitability for select species. Fish are known to prefer specific habitat features including depth, velocity, substrate type and vegetation cover, and species and life-stage specific associations with habitat features can be modelled to provide a quantitative value of the habitat lost due to residual effects, and to model the habitat to be gained from the proposed offset. For example, stream-based habitat models integrate a hydraulic model with a biological model (habitat suitability criteria, HSC), to calculate the variation of a habitat index called weighted usable area (WUA) as a function of discharge and river morphology (e.g., PHABSIM, Stalnaker et al. 1995; River2D, Katopodis 2003; MesoHABSIM, Parasiewicz 2001, Parasiewicz and Walker 2007). To assess the net gain or loss of productivity of lacustrine habitat due to development and offsets, a scientifically defensible quantitative model was developed by Minns (1995, 1997) based on the concept that suitability assessment of habitat types can be used as surrogates of productivity, and that the greater utilization of habitat types by different species and life stages are important for sustaining fish community productivity. The Defensible Methods approach (operationally known as the Habitat Alteration Assessment Tool, HAAT, for use within DFO, MacNeil et al. 2008) uses a concept similar to stream-based habitat models of assigning relative suitabilities of habitat patches and computing weighted useable areas for all fish species that may be present in the area being assessed (Minns 1997).

Application

Quantitative habitat models can be used for projects affecting both habitat quantity (e.g., infilling a lake shoreline using HAAT), or habitat quality (e.g., where instream flows are being changed due to a hydropower development, diversion, or water withdrawal). In most cases the offset method used will be to improve habitat elsewhere or to provide unlike, out-of-kind habitat improvement since, for example, a reduction in streamflow volume altering the WUA fundamentally cannot be offset by in-kind measures. Habitat improvements used to account for reduced habitat suitability can include provision of new spawning substrates, improvements in complexity of habitat for cover, refuge or food-production services, or can include creation of new stream channels (e.g., Jones et al. 2003, Enders et al. 2007).

Quantitative models do provide a more defensible basis for determining offset requirements, can provide an objective basis for negotiations between the proponent and regulators, and can provide greater protection for ecosystems by requiring more offset habitat than if the model was not used (e.g., case study analysis using HAAT, MacNeil et al. 2008). In some cases, however, model predictions may not always be realized biologically. For example, reducing velocity (via water extraction) in a high gradient, high velocity system often results in greater modelled habitat suitability for life stages or species which prefer low velocity habitats, but the modelled suitabilities may not translate into a corresponding change in fish abundance (e.g., Bradford et al. 2011). The transferability of habitat suitability criteria has also been questioned and site-specific data may be necessary, raising the costs and time required to conduct such assessments (Freeman et al. 1997, Williams et al. 1999).

Data and Methods

The metric is a habitat suitability index which when multiplied by area generates weighted useable area (WUA) by species and life stage. Lacustrine habitat patches have characteristics of depth range, substrate type and cover type and the model generates suitability values according to species requirements grouped by life stage, trophic level and thermal preference (Minns et al. 2001). Scenarios are produced for pre- and post-development conditions at a proposed site that provide an assessment of the net gain or loss of suitable habitat, and a module is available to add in the post-development conditions with compensation included (Minns et al. 2001).

Lotic habitat patches have characteristics of depth, velocity, substrate type and cover, and stage-discharge and velocity-discharge relationships are required for the reach in question; these can require extensive field data collection program. Detailed data collection protocols are available for IFIM and PHABSIM (Stalnaker et al. 1995; Bovee et al. 1998), for River 2D (Ghanem et al. 1996, Steffler and Blackburn 2002, Katopodis 2003), for MesoHABSIM (Parasiewicz 2001, Parasiewicz and Walker 2007). Once the model is developed it is possible to assess weighted useable area by species and life stage at different discharges or habitat configurations and net loss or gain can be calculated. The magnitude of the loss will at least partially dictate the offset requirement, and the type of offset proposed will dictate the method used for equivalency calculation.

Choices and Assumptions

Technical choices: Model developers advocate the use of their model over others, and criticisms can be found for all models in the literature. IFIM and PHABSIM have been criticised as being data intensive yet resulting in unrealistic one-dimensional characterizations of habitat conditions. Weighted useable areas are highly dependent on habitat suitability criteria (HSC) which are assumed to be static, and are often assumed to be transferrable to avoid site-specific data collection. However, the importance of accurate HSC depends on model outcomes and their influence on management decisions. If there is little difference in the ultimate recommendation of either the instream flow prescription or offset required regardless of the habitat suitability criteria used (based on a sensitivity analysis using confidence limits surrounding the HSC), then the specific HSC used are less critical since ultimately these models are tools to facilitate decision making (Williams et al. 1999). Two-dimensional models are considered more realistic hydrologically but the biological caveats still apply. Meso-habitat based modellers argue that the larger scale (10^3-10^5 m^2) is more applicable to management decisions, and is more biologically meaningful since it roughly corresponds with the concept of 'functional habitat', and that observations of habitat-associations at the meso-scale are less affected by coincidence than the micro-scale (Parasiewicz 2007). All models require significant amounts of field data to validate them, although the least data-intensive model is MesoHABSIM for both its development and validation (Parasiewicz 2007).

Policy choices: Except for the HAAT model which was developed to include the entire fish community in the vicinity of the development, in most cases the species and life-stages included in the modelling exercise do not include the entire fish community. Instead valued fishery species and life-stages are modelled and the choice of which species to include is a policy decision. The choice of model to use is often made by the proponent so it is essential that FPP biologists understand the limitations of the models and seek advice if uncertain about the presented model appropriateness, input parameters and outcomes.

Key Uncertainties

Complex modelling exercises carry with them large amounts of uncertainty. Uncertainty arises in the fish-habitat relationships and in the transferability of those relationships (Freeman et al. 1997, Williams et al. 1999). Three main sources of fish-habitat relationship uncertainty include: (1) the incomplete state of knowledge about such relationships; (2) high levels of natural variation in biological and environmental conditions that obscure relationships; and (3) human mistakes in inferring and (or) assessing relationships (Rose 2000, Minns and Moore 2003). Consideration should also be given to the uncertainty arising from sampling and measurement errors in the hydraulic modeling (Williams 1996), from violations of the assumptions of the hydraulic models (Ghanem et al. 1996), and from the dynamic nature of habitat preferences due to mitigating or limiting factors such as temperature, light regimes, predation, or competition, among others (Orth 1987; Heggenes et al.1996). Unfortunately model validations are rarely conducted.

FISH ABUNDANCE METRICS

This currency uses direct measures of the abundance of the affected species/community to determine offset requirements. The use of abundance metrics as a currency is the example most often referred to when discussing resource-to-resource equivalency analysis (REA; Lipton et al. 2008; Clarke and Bradford 2014). The REA approach is well suited to calculating equivalency when the impact to the fishery is the result of lethal or sub-lethal impacts that are not habitat related (Lipton et al. 2008). As the "resource-to-resource" name implies the species or community being impacted should be similar to that being targeted in the offset. Abundance metrics, especially biomass, are highly correlated to fish production (Randall et al 2013) and are often used as its proxy (Minns et al. 2011) therefore they are well aligned to the intent of the FPP with respect to protecting the sustainability and ongoing productivity of CRA fisheries.

Application

Resource-to-resource methods of equivalency will generally require more detailed information than purely habitat-based methods as there is a need to have quantitative information on species abundance for both the affected and offset sites. Information for the site affected by the project can be collected during pre-project sampling but the information for the offset will usually have to be predicted. Therefore REA requires there to be enough information about the impacted species or community and the area to allow for reasonable predictions. These predictions can be informed through a variety of methods that include past experience, expert opinion and regional benchmarks for abundance. Once completed predictions become an important component of the effectiveness monitoring (DFO 2012, Smokorowski et al. 2015).

Measures of abundance may be best suited to projects that impact a localized area that can be sampled with a reasonable amount of effort (e.g. stream reach, small lakes, embayments in large lakes or coastal areas etc.).

The primary advantage of using a direct measure of the resource is that it can provide additional options when designing offsets. Since the offset in this case is not simply trying to replace the function of the habitat that was affected, it is possible to use offsets that manipulate habitat quality or target habitats that are deemed lacking in the area of the species or community the offset is being designed to benefit. The use of direct measures of abundance is also often easier for stakeholders to understand.

While there can be technical challenges with collecting biological abundance data, this field is mature enough to have a number of good reference documents that outline both sampling and statistical methods (e.g. Bonar et al. 2009). Abundance metrics are considered primary indicators for the FPP (DFO 2014a) as they are statistically robust and it is relatively easy to interpret qualitative and quantitative changes in value. Further, many regional benchmarks exist in public and private databases for most North American freshwater fish species thus increasing the likelihood of having information for the prediction of offsetting benefits.

Method for Computation

Metrics

Density: Fish density is an absolute measure of abundance for the area being sampled at the time sampling occurs. The area can be a habitat unit or some other area that can be sampled with a reasonable effort (e.g. stream reach, small lakes, embayments, etc.). Density is reported as number of individuals per unit area (e.g., # m^{-2}, # ha^{-1}) and is one of the more common metrics reported in the habitat manipulation literature (Quigley and Harper 2006, Minns et al. 2011, de Kerckhove 2015). It is important to view density estimates in the context of life history and migration/movement behaviours, as densities can change significantly during life history transitions.

Biomass: This is also an absolute measure of abundance for the area being sampled. Biomass is most often calculated by multiplying the density of the individuals by their average weight and is reported in mass per unit of area (e.g., g m^{-2}, Kg ha^{-1}). Biomass has been shown to be highly correlated with production (Randall et al 2013) and has been a common metric when a population response to habitat manipulation has been reported in Canada (Clarke and Scruton 2002, Quigley and Harper 2006, Minns et al. 2011).

Catch per unit Effort (CPUE) or Biomass per Unit Effort (BPUE): These are relative measures of abundance; CPUE only deals with numbers while BPUE incorporates the mass of the individuals captured. These measures rely on the sampling effort being standardized to allow comparisons and are among the most common metrics reported on in the fisheries literature (Bonar et al. 2009). Commonly reported measures may include number/mass of individuals captured per electrofishing seconds or stream length or area or the number/mass of individuals per trap/net night or hour. Since these are relative measures of abundance some caution should be exercised when using these metrics in equivalency analyses, especially if they are to be used in an offset habitat that is not the same as the affected site. If the catchability of the fishes is different in the offsetting habitats than the baseline habitats comparisons may be biased.

Other considerations: Equivalency analysis based on abundance metrics will require empirical or predicted measures for the impacted area and the area proposed for offsetting (the baseline), and a prediction of abundance metrics after the project and offsetting are completed. Those predictions can be derived from regional benchmarks of abundance, or a habitat-fish model (e.g., Hanson and Leggett 1982).

Empirical estimates of fish abundance are a common metric in fisheries and thus there are several good ‘standard methods’ references that can be used when designing a sampling program. The choice of method may be affected by regional preferences (e.g. Sooley et al. 1998, Sandstrom et al. 2013) or the fish community and/or habitat to be sampled (Bonar et al. 2009). Uncertainties should be reported along with mean or median estimates.

Prediction of fish abundance metrics in the offset habitats can be informed by a number of avenues. Habitat restoration and creation has a fairly extensive history in North America and with the experience gained through these projects, especially for some species and habitats, it should be possible to characterize benefits with reasonable certainty for ‘common’ habitat manipulations (e.g., Roni et al. 2008). In situations where experience is lacking, predictions can be informed by expert panels and/or regional benchmarks. Given that abundance is among the most commonly reported fisheries metrics, regional benchmarks may be relatively common when compared to other productivity surrogates. These data sources are not mutually exclusive and a project that uses more than one should reduce the overall uncertainty with respect to the predictions. As with any prediction, a level of confidence should be reported.

It should be noted that predictions of abundance are intended to be used for waterbodies with similar biological and physical features to those of the predictive dataset. In situations where the offset consists of the creation or significant modification of habitat, there will often be time lags between the creation of the habitat and its ultimate stable state (e.g. Scrimgeour et al. 2014). In situations where fish community changes are expected abundance metrics by themselves may not be the best choice and more complicated models may be required (see below).

Choices and Assumptions

Policy choices

The use of abundance metrics in a resource-to-resource equivalency analysis assumes the resource that is being affected is also the resource that will be the subject of the offset. This type of analysis cannot easily accommodate major changes in ecosystem function and/or species composition. There may however be limited opportunities where offsets can be designed to benefit one species over the other. An example of such an offset was presented by Scruton (1996) for a small salmonid stream in Newfoundland. In that case the offset was designed with more pool habitat area and undercut banks than was available in either the impacted habitat or in the surrounding stream. This was done to create a habitat for larger brook trout which are the main fishery species in the area. The choice of offsets is a policy decision, in this case based on fishery management objectives. In the Newfoundland example there was a limited salmonid-based (Atlantic Salmon and Brook Trout) community; in situations where the fish community is more complex these type of manipulations may have high uncertainties as biodiversity and species or habitat complexity both have an impact on fisheries productivity that may not be completely understood (Randall et al. 2013).

Key Uncertainties

All projects that aim to manipulate natural systems will have uncertainties. There is a tendency to make assumptions when working in natural systems for the simple reason that nature is complex. Minns et al. (2011) provide a detailed review of assumptions and limitations for a number of productivity metrics. The key assumption made when using abundance metrics is that there is an inferred link between abundance and productivity. This leads to the assumption that abundance can be habitat limited or linked to habitat quality. These assumptions are rarely verified as there are often confounding factors in the relationship between abundance and productivity (as discussed by Minns et al. 2011). It is therefore important to have knowledge of the key production mechanisms (i.e. growth, recruitment and survival) of the habitat when using abundance as the metric to evaluate offsets. In some cases and for some species this may be relatively well known. For example, Scruton et al. (2005) showed that a high quality constructed salmonid stream with both spawning and rearing areas could provide more salmonid biomass in a smaller footprint than the low quality stream reach that was destroyed. When these mechanisms are not as well known it would be beneficial to include additional metrics in the monitoring program to facilitate learning. An excellent example of such an approach is the monitoring that has been conducted for an offset for Arctic Grayling (Jones et al. 2003, Scrimgeour et al. 2014).

FISH PRODUCTION

This currency uses either directly measured or modelled population production rates of the affected species to determine offset requirements. The use of production metrics as a currency is another example of resource-to-resource equivalency analysis (Lipton et al. 2008; Clarke and Bradford 2014). Randall and Minns (2000) consider production to be “the best indicator of the quantitative performance of a fish population in any type of habitat and it is a measure of productive capacity”. Population production is the main determinate of sustainable yield and thus is well aligned to the intent of the FPP with respect to protecting the sustainability and ongoing productivity of CRA fisheries.

Application

REA methods of equivalency generally require detailed information and production is probably the most data intensive and complicated REA. Thus production-based equivalency calculations are likely to be reserved for cases where there is extensive change in the ecosystem or where the fish species or communities benefitting from the offsetting measures are different from those affected by the project.

As with other resource-based approaches there is a need to have quantitative information on population production for both the affected and offset sites. Information for the affected site can be collected during pre-project sampling but the data requirements are high. Information for the offset habitats will have to be predicted via modelling or use of reference data. The models require detailed information on either population level and/or stage/habitat specific vital rates (i.e. reproduction, growth, survival). Predictions can be informed through a variety of methods that include past experience, expert opinion and regional benchmarks for vital rates and production but detailed productivity assessments for many species are still relatively rare in the scientific literature. Once completed these predictions become an important component of the effectiveness monitoring (Smokorowski et al. 2015).

The primary advantage of using a production-based assessment is that it can provide additional options when designing offsets. Many of the modelling options can test scenarios which can then inform offsets. Offsets in this case do not need to replace the function of the habitat that was impacted it is possible to use offsets to target habitat types processes that constrain productivity. This approach can also inform managers in situations where the ecosystem is expected to be transformed and species assemblages are expected to change.

Method for Computation

Metrics

Production: Fish production rate is the total elaboration of new body substance in a stock in a unit of time, irrespective of whether or not it survives to the end of that time (Ricker 1975). The production of a fish population depends on the amount and quality of habitat required for each life stage. The unit of time for measurement of production is often one year, and the units of production are total numbers of fish or kilograms (produced) for a specific species and area (number·yr^{-1} or kg·yr^{-1}), or in units of kilograms (or number) per hectare per year (kg·ha^{-1} ·yr^{-1} ; Randall et al. 2013).

Population structure metrics (e.g. Body size, P:B Ratios; HPI (population)):

Size structure information, especially maximum length, has been correlated with production (Randall and Minns 2000).

The P:B ratio is essentially the turnover rate of the population and is an important determinate of productivity.

Habitat Productivity Index (HPI) is used as relative measure (index) of a habitat's productive capacity. This can be calculated for both populations and communities. The population index is calculated by multiplying the P:B ratio with seasonal biomass. The community index is calculated by summing the indices of all the co-habiting species (Randall and Minns 2000).

Individual metrics (e.g. growth, survival, reproduction):

Individual metrics, also referred to as vital rates, are the building blocks of production estimates. As these are rates they have a temporal aspect (e.g. mass gained per unit of time). If they are habitat specific rates they will also have a spatial aspect (e.g. mass gained per unit of time per area).

Mechanistic Models:

There are a number of modelling approaches that can be used to investigate the mechanistic relationships between habitat and production. Two of the more common are stock-recruitment models and stage structured habitat supply models. There are several variations of each modelling approach in the literature, Ricker and Beverton-Holt models are two of the classical stock recruitment models. Examples of stage-structured models can be found in Velez-Espino and Koops (2009 a, b). Other modelling approaches that could be used include those derived from multi-linear regression or neural networks but these are not discussed further in this section.

Equivalency analysis based on fish production will require empirical or predicted measures for the affected area and the area proposed for offsetting (the baseline), and a prediction of production or its surrogates after the project and offsetting are completed. The metrics discussed here are 'population' level metrics and are not a complete list that could be used in a production assessment but are the ones most commonly found in the literature. Minns et al (2011) and de Kerckhove (2015) give details on other metrics that may be useful in a fish production context.

While the concept of productivity is central to the FPP, the actual calculation of production rate is data intensive and historically it has not been extensively used in habitat-related assessments (Smokorowski et al. 1998; Minns et al. 2011). The calculation of production may, therefore, be reserved for use in larger projects where either entire parts of the ecosystem are going to be negatively affected (e.g. whole lake destruction, the destruction or blockage of an entire tributary etc.) or the ecosystem is going to be transformed (Bradford et al. 2014). When an empirical estimate of fish production is required there are a number of methods that can be used including the instantaneous growth method (Chapman 1978) or increment summation method (e.g., Clarke and Scruton 1999) among others (see Rigler and Downing 1984; Plante and Downing 1990; Mertz and Myers 1998 for details).

Productivity assessments may combine individual metrics and/or population structure with abundance metrics to provide an assessment of fish production in a given area (Minns et al. 2011). In general a suite of metrics will be more robust than any one single metric when assessing fish productivity (DFO 2014a). There are other advantages to using more than one metric in a productivity assessment. Individual metrics can give insights into mechanistic links between habitat and production as well as being sensitive to environmental change (Pearson et al. 2005). This can provide a potential early indicator of the efficacy of the offset and can provide options for adaptive management if the offset is seen not to be working as predicted. For example, if the offset targets a spawning habitat some measure of reproduction would be logical to include, likewise if the offset targets the restoration of a nursery habitat, growth and survival estimates may be useful for effectiveness monitoring (Smokorowski et al. 2015).

Population structure metrics are often correlated with fish production (Minns et al. 2011, Randall et al. 2013) and are thus useful when combined with abundance metrics to provide a productivity assessment for a given population or when comparing populations. These metrics also have the added advantage of being derived from abundance measures and thus can be obtained efficiently. Size structure information, especially maximum length, has been shown to be a good indicator of productivity (Randall 2002, DFO 2014c, de Kerckhove 2015). This type of metric has the added advantage of being statistically robust, easy to understand is commonly reported making the development of regional benchmarks more probable (DFO 2014c).

The P:B ratio is an indicator of productivity (Randall et al. 2013, FAO 2009). The empirical calculation of P:B has the same intensive data requirements as that of production rate calculations but it can also be derived from allometric relations (Randall and Minns 2000, Randall 2002). In these situations it can be used as an operational shortcut to calculating population production (e.g., Cote 2007). With an estimate of P:B and seasonal biomass, the habitat productivity index (HPI) can be calculated (Randall and Minns 2000, 2002). The HPI was originally derived as an index of a habitat's productive capacity for the existing community (Randall and Minns 2000, 2002). It can also be calculated on a population basis as the community index is a summation of the population indices. The HPI may be particularly useful in situations where species assemblages are expected to change due to project impacts.

Stock recruitment models are more commonly used in removal fisheries than environmental impact situations because of the data requirements to develop stock recruitment relationships (Minns et al. 2011, Randall et al. 2013). Since these models use estimates of recruitment, survival, intrinsic growth rate and carrying capacity they can provide managers with information on regional and/or site specific habitat production linkages (e.g. Sharma and Hilborn 2001, Gibson 2006, Parken et al. 2006, Lobón-Cerviá 2007), particularly for the juvenile stages. Many well studied species in Canada may have existing 'regional' production estimates that have been derived from modelling (e.g. Bradford et al. 1997, 2000, Chaput et al. 1998, Parken et al. 2006) and these should be used in equivalency analysis when available.

Stage-structured habitat supply models divide the life cycle into explicit life stages which can be based on habitat. These models then use habitat/stage specific vital rates (i.e. reproduction, growth, survival) to build a trajectory for the population being modelled. While this type of model has not being extensively used in environmental impact projects in Canada, they have been used in developing the recovery strategies of endangered species (e.g. Vélez-Espino and Koops 2009 a, b). Data requirements remain high for this modelling approach but the ability to evaluate scenarios based on stage/habitat attributes may be useful when designing offsets within the equivalency analysis.

The prediction of fish production metrics in the offset habitats will rely on modelling approaches in most situations. There may be existing models or regional benchmarks for a few well studied species in Canada (see above) but these will most likely be the exception rather than the rule. Two of the more commonly used mechanistic modelling approaches are briefly described above, although others exist (see Minns et al. 2011). Since production metrics are often built on abundance metrics they will have many of the same uncertainties and assumptions. In addition, the modelling process itself will have uncertainty and it is important that this is well outlined in the equivalency analysis. Predictions of production metrics in offset habitats will be more robust when the input data (i.e. vital rates) are from waterbodies with similar biological and physical features to those of the predictive dataset.

Choices and Assumptions

Production assessments are meant to deal with population-level effects over larger spatial scales than either habitat- or abundance-based approaches and they may better inform management decisions in situations where major changes in ecosystem function and/or species composition are expected (Bradford et al. 2014). It is important to note that these types of offsets will still require policy choices about the species to be included in the calculations of impacts and offsets. In situations where the fish community is more complex these type of manipulations may have high uncertainties as biodiversity and species or habitat complexity both have an impact on fisheries productivity that may not be completely understood (Randall et al. 2013). In these instances community-based productivity metrics may be required.

Key Uncertainties

Minns et al. (2011) provide a detailed review of assumptions and limitations for a number of productivity metrics. Similar to abundance metrics the main assumption with respect to production and habitat is that there is a direct link between production rates or individual vital rates that make up production and habitat conditions. Another common assumption made in production assessments is related to sampling considerations. Since it is almost always impossible to completely sample a population in all habitats, it is often assumed that the survey locations used are representative of the habitat and ecosystem function across the entire population. With respect to offsetting directly there is often the assumption that either reproduction or recruitment of juveniles is a limiting factor in population production. This is why many offsetting programs focus on spawning habitats instead of taking a more holistic approach to habitat supply. Many of these assumptions are rarely tested; the reader can refer to the discussion of Minns et al. (2011) for further details.

FISHERY-BASED METRICS

This currency uses the potential yield of fish to fisheries or other fishery-based metrics to determine offset requirements. The use of fishery benefit as a currency is an example of value-to-value equivalency analysis (VEA; Lipton et al. 2008; Clarke and Bradford 2014). VEA defines the services of an ecosystem in terms of the value or benefits to humans, either directly (e.g., fishery yield) or in more indirect measures based on utility (e.g., the societal value of healthy ecosystem, aesthetic values, etc.). The use of fishery benefits aligns with the intent of the Fisheries Protection Provisions to provide for the sustainability and ongoing productivity of CRA fisheries.

Application

Value-to-value equivalency analyses are among the more complex of the suite of equivalency approaches and should only be employed when more direct currencies cannot be used. Their use is most likely to be restricted to large-scale projects that result in ecosystem loss or conversion, potentially affecting existing fisheries and resulting in significant offset requirements. Fisheries-based metrics are suited to “out-of-kind” offset proposals where losses associated with the project and benefits of the offset are sufficiently different that expressing the results in terms of effects on fishery metrics is more relevant or appropriate than biological currencies such as numbers or biomass of fish. Situations where fishery yield may be preferred over purely biological metrics include those where:

- The target species for offsets are significantly different (either in species composition, size, age or other attributes) from those impacted that a direct comparison of biological traits is less meaningful.
- Preference for certain fisheries could guide offsetting measures and be quantified in terms of fishery statistics. For example, offsetting measures to enhance aboriginal use may be best quantified in terms of yield or participation.
- Hatcheries or other means of artificial propagation that aim to increase fishing opportunities are proposed.
- There are meaningful differences in the regional fishery management objectives for the species impacted by the project relative to those targeted for the offsetting.
- Regional benchmarks for fisheries measures (e.g., effort or catch/effort targets, or other measures of fishing quality) exist and can be used to support offset equivalency calculations.

The primary advantage of fisheries metrics is that they are closest to the intent of FPP with respect to maintaining the productivity of CRA fisheries and will be of direct relevance to fisheries management agencies and stakeholders. It is also a currency of considerable flexibility as it can accommodate situations where the project’s effects and offsets are very different.

Technical challenges associated with this currency include the accuracy and precision of the available tools to estimate fishery metrics, especially for cases where site-specific or regional information is not available. Fisheries metrics are the result of interactions between habitat, aquatic species and human behavior, each of which contributes to variation in responses as well as uncertainty in estimates. Thus fishery metrics can be the most uncertain of the equivalency currencies.

Method for Computation

Metrics

Yield: The simplest and most commonly used metric for fisheries yield is the mass caught annually, usually scaled by unit area (e.g., $kg \cdot ha^{-1} \cdot yr^{-1}$). Salmonid fishery yields are often expressed in terms of numbers of individuals, due to their relatively uniform size.

Quality: For recreational fisheries a variety of metrics are used to describe angling quality. This includes the catch rate (e.g., $fish \cdot angler^{-1} \cdot day^{-1}$). Fish size is an important contributor to angler satisfaction. Finally, species composition can affect angling quality and participation.

Effort: In recreational and aboriginal fisheries participation rates or effort may be used to evaluate offsets as they are often a measure of angling quality in open access fisheries. Effort is usually estimated as participants-time per spatial unit (e.g., angler-days$\cdot$ha^{-1}) during creel census programs.

Data and methods

Equivalency analysis based on fisheries metrics will require empirical or predicted measures for the impacted area and the area proposed for offsetting if appropriate (these are often part of baseline studies during environmental impact assessments), and a prediction of fisheries metrics after the project and offsetting are completed.

Empirical estimates of fisheries metrics can be gathered from standard creel or catch sampling programs that estimate effort, catch, and catch composition (species, size etc.; Jones and Pollock 2013). Usually a multi-year program is required to ensure estimates are robust in the face of weather or other variables that can affect use patterns. An assessment of yield relative to a standard such as maximum sustainable yield can assist in determining the status of the existing fishery (see Fig. 2 of Lester et al. 2003).

A variety of methods and models are available to estimate fishery metrics for the predictive component of equivalency calculations. Many empirical models that predict yield using ecosystem attributes (lake size, depth, water chemistry etc.) have been developed and these provide first-order estimates (reviewed by Leach et al. 1987) using relatively easy-to-obtain covariates. For these models the yield statistics are aggregated, and do not consider the species composition. Single species models exist, for example, Baccante et al. (1996) found estimates of walleye yields are linearly related to lake size.

Empirical models are intended to be used for waterbodies similar to those of the predictive dataset. In situations where the offset consists of the creation or significant modification of habitat, there will often be time lags between the creation of the habitat and its ultimate stable state. Predictions from empirical models are based on the assumption that the final state of the offset habitat will be similar to a natural waterbody, an assumption that may not initially be true.

Predictions of changes in fishery quality or effort metrics will require the use of regionally-based models or analyses that utilize information on the characteristics of the exploited populations and the dynamics or attributes of the fishery (e.g., Shuter et al. 1998; Parkinson et al. 2004). For example, Post et al. (2008) describe a simple model that predicts fishing effort in small lakes using fish density, distance of the lake from a major urban centre, and the presence of a fishing lodge. In the absence of predictive models a regional analysis of fishery statistics may be adequate to predict the change in fishery effort or quality resulting from the offsetting. For example, average effort data for local lakes may be sufficient to estimate the potential increase in fishing activity associated with stocking a barren lake.

Unfortunately predictive models for yields in rivers are few, which limit the application of this currency in lotic habitats unless site-specific tools are developed.

Choices and Assumptions

Technical choices: Precise and accurate predictions of fishery metrics in the impact and offset areas will depend on the availability of site- and regionally-specific models and data that account for both the biological attributes of the species of interest and the dynamics of the users in the region. In the absence of such information, empirical regression models may be the only alternative although these offer considerably less precision.

Policy choices: The use of fishery metrics for equivalency computations are only likely to be employed in cases where there is a large difference in the species impacted by the project relative to those benefiting from the offset, or in cases where the offset activity can generate disproportionate benefits when viewed from a fishery perspective, rather than from strictly biological considerations. Consequently some of the most important choices for the implementation of fishery-based equivalence are the relative values of different species, fisheries and fishery attributes that will be required when weighing project impacts and offset benefits. In theory, one can establish explicit weightings for species and fisheries using utility-based approaches from the economic literature. In practice, a ranking scheme that sets priority species may be sufficient to allow proponents and regulators to settle on an appropriate approach for characterizing project impacts and offset benefits. Either way, considerable input from regional fisheries managers, Aboriginal groups and stakeholders may be required to establish a weighting or ranking scheme.

Using site-specific fishery statistics to estimate baseline conditions is based on the assumption that the existing fisheries resources are well utilized and the resulting statistics provide a reasonable measure of the “service” that is provided by that resource. Fish populations may be underutilized for a variety of reasons, including access, aesthetic values and the availability of more appealing alternatives. Criteria may be needed to determine whether empirically-derived fisheries metrics can be used to establish the baseline condition for cases where human use is minimal or restricted. In those cases it may be more appropriate for the purposes of offset calculations to estimate potential yield or use using a biologically-based predictive model.

Key Uncertainties

Empirical predictive relations between lake attributes and fish yield have considerable uncertainty associated with the predictions despite high R^2 values for many published log-log regressions. For example, Oglesby’s (1977) model prediction of yield on lake productivity has an error of at least ± 100%. Similarly, Baccante et al. (1996) found the interquartile range for area-based yield of walleye ranged from 0.5 – 2.95 $kg \cdot ha^{-1}$, a six-fold range, which illustrates the limitations associated with these simple models.

OTHER VALUE-BASED METRICS

Money is the primary currency in this category. Money is part of value-to-value equivalency analyses (VEA; Lipton et al. 2008; Clarke and Bradford 2014). Economists describe the total economic value (TEV) of a natural resource as the sum of both “use” and “non-use” values. Use values include direct use by humans (harvest, angling or other forms of recreation), and indirect services provided by the natural environment (e.g., nutrient recycling). Non-use value is the benefit derived from the maintenance or presence of the natural environment and includes existence values, which is the value to individuals of the existence of the ecosystem, and altruistic and bequest values related to the value for others or future generations (Ozdemiroglu 2008; Shaw and Wlodarz 2013). Although TEV is often expressed in monetary terms, calculations can also be done in relative units of “utility”, a unitless measure of the benefit (or loss of benefit) from components of the natural environment (see Lazo et al. 2005 for an example).

Equivalency calculations based on value may yield quite different conclusions compared to those based on habitat or resource equivalency if there is a significant difference in the use or service value attached to the lost ecosystem components or to those gained through offsetting. For example, Martin-Ortega et al. (2011) found that a protected area created as compensation for a toxic waste spill was little used by visitors relative to the impacted area, and this resulted in an imbalance in the total value of the compensation compared to loss of value caused by the spill.

Methods exist for the estimation of use and non-use values, but they require considerable effort and the expertise of economists and are usually reserved for large and complex situations (e.g., the Exxon Valdez oil spill, Carson et al. 2003).

Alternatives to a full economic analysis that are based on costs have been proposed and are variously known as pricing, or replacement or shadow project cost determination (Ozdemiroglu 2008; Strange et al. 2004). These are considered supply-side rather than demand-side approaches as they are based on the cost of replacement of the lost ecological function, rather than the total value of that function.

Early attempts to calculate replacement costs for fish losses were based on the costs of producing and stocking hatchery fish (Southwick and Loftus 2003), however, it has been recognized hatchery production is not an appropriate strategy to offset wild fish. Costing based on hatchery production has been superseded by the costs associated with enhancing natural production. Strange et al. (2004) propose a habitat replacement cost methodology; this approach monetizes impacts based on the costs of the offsets needed to replace lost ecosystem services (fish productivity in their example). The extent of the required offsetting measure is determined by habitat or resource equivalency methods described in other sections of this paper, based on the extent of the residual serious harm. The cost of implementing that amount of offset then becomes the replacement cost. This amount sets the limit for offsetting activities without prescribing the nature of those activities.

Application

Due to their complexity, equivalency calculations based on total economic value as defined above will not be considered further in this report.

Replacement cost methods are most likely to be used when unlike offsetting measures may be of particular value for achieving policy goals. Some potential applications include:

- For projects that cause impacts to common, or low value habitats, offset activities could be redirected to species or habitats of particular interest or concern. This exchange is called “trading-up” in the biodiversity literature (Quétier and Lavoral 2011) and scale of measures required could be established by the replacement cost.
- For some projects the replacement of lost habitat or service by standard offset methods may be logistically infeasible, or be known to have high levels of uncertainty. In these cases it may be more sensible to consider alternative offset measures; cost may be a suitable metric to facilitate the scaling of other types offsets.
- Offsetting for a project could consist of contributions by the proponent to large-scale restoration or enhancement activities in the region conducted, for example, by conservation authorities. In these cases replacement costs yields a monetary contribution in lieu of directed offsets by the proponent. This approach could be used to scale contributions to, or withdrawals from, out-of-kind habitat banks.

The primary advantage of replacement cost equivalency calculations is the flexibility of potential offset activities that could be employed once the monetary evaluation is completed. This should result in the maximization of the offset benefit by minimizing the requirement for offset measures that may be of uncertain value, or are known to have a limited lifespan.

Method for Computation

Metrics

Replacement cost: Replacement cost is computed as the cost of providing habitat or other attributes that replaces the services or resources impacted by the project.

Data and Methods

A habitat- or resource-based equivalency method must first be employed to identify offsetting actions that are needed to replace the lost function or service, and estimate the costs of those actions. This exercise could be site-specific and be based on actual proposed offset measures and local costs. Alternatively, it may be feasible to develop hypothetical offset requirements based on the nature of the impact and local or regional standards for offsetting and costs of implementation. Cost should include estimates for baseline and feasibility studies and should account for uncertainty and time delays (Strange et al. 2004). Many constructed offset works require periodic monitoring and maintenance, and these costs should be included in the total.

Choices and Assumptions

Technical choices: Key technical choices for implementation of the replacement cost method is method used to calculate the total impact of the project and the benefit of the offset, whether that be in terms of habitat area and quality, or the loss of production of one or more fish species. Under the HEA or REA protocols this then scales the amount of offset required to counterbalance the loss. Since many habitat-based equivalencies are likely the simplest to estimate, this should be the default approach for replacement cost computations.

Policy choices: The circumstances when it is appropriate to use replacement cost methods within the hierarchy of preferences should be established to ensure its use will meet policy goals. In some situations the value (in the full sense as described above, based on biological criteria) of affected habitats may greatly exceed generic replacement costs; it may be more appropriate to require in-kind offsets to address these losses. This situation could arise if particularly significant or rare habitats for key populations are likely to be affected, and the impacts to those populations are of greater concern than cost-based offset activities at another site.

Key Uncertainties

The performance of the habitat replacement cost method for developing offset requirements has not been evaluated. It is difficult to anticipate whether this approach will yield greater overall benefits to fisheries productivity relative to offsets that are defined by ecological or fisheries benefits.

UNCERTAINTY AND EQUIVALENCY METRICS

Table 2 summarizes the relative amounts of measurement uncertainty associated with the equivalency metrics, and the degree and significance of the assumption of a link between each metric and fisheries productivity. Uncertainties are first approximations intended to approximate interquartile ranges (capturing 50% of the uncertainty) around the median or mean. Uncertainty ranges are set at low (±10%), moderate (±10-50%) and high (>±50%), to roughly correspond to levels where uncertainty can be ignored (low), managed potentially by increasing the required amount of offsetting (medium), or require larger offset requirements or alternative risk management approaches (high). The entries in the table are based on a qualitative evaluation of the descriptions of the metrics in the previous sections and can likely be improved with a more rigorous analysis.

In general, the simpler habitat-based equivalency methods have the lowest uncertainty with respect to the calculation of the offset requirement, but the methods make no prediction of effects on fisheries productivity. Uncertainty about whether the prescribed offset will balance productivity losses is very context dependent: in simple cases where the function of the habitat is simple and well understood (e.g. shade, shelter), it is very likely that an offset can achieve the goal of replacing lost function. Habitat indices and proxies are based on the assumed relation between the index and fisheries production; there is often considerable uncertainty around this relation that generates uncertainty about whether fisheries productivity gains in the offset area will be achieved (Minns and Moore 2003).

Fish and fishery metrics make direct predictions about fish populations and their potential for use, and the calculation of offsets are based on balancing losses and gains of fish or fishery benefits. For these metrics there is greater uncertainty entering into the equivalency calculations, however they are less dependent on assumed relations between the equivalency metric and fisheries productivity. These metrics yield predictions about fisheries productivity that can form the basis for effectiveness monitoring.

Table 2. Qualitative summary of the uncertainty associated with inputs to the equivalency calculations, and uncertainty surrounding prediction of whether fisheries productivity (FP) will be counterbalanced by offset activities for each of the equivalency metrics listed in Table 1. Low uncertainty (expressed as the interquartile range as a percentage of the median) ±10%; moderate ± 10-50%; high >± 50%.

Metric class	**Uncertainty in equivalence calculation**	**Uncertainty in linkage between metric and fisheries productivity**
Habitat (like for like)	Low, based on measurements of area during project design	Assumed, but often low as losses and gains are similar and in the same area.
Habitat or ecosystem function	Moderate. Physical measurements of habitat, and secondary production or other indicator.	Assumed; uncertainty could be moderate or large if unlike habitat alterations are used.
Habitat suitability or capacity for select species	Moderate, dependent on quality of field program and specificity of species-habitat relations.	Relation between habitat index and FP assumed or known. Uncertainty could be high. Uncertainty high for non-target species.
Fish abundance	Moderate, if based on intensive site-specific surveys. Predictive models of abundance or biomass have moderate to high uncertainty.	Direct prediction of biomass or density losses and gains that are close surrogates of FP. Uncertainty dependent on methods and models, moderate to high.
Fish production	High if production estimates are based on inferred P:B ratios or correlates of P. Moderate if P estimated directly.	Direct estimation and prediction of fisheries productivity. Uncertainty high for predictive models (e.g., P:B), moderate for direct estimates.
Yield/fishery benefits	Moderate. Requires direct sampling or fishery models. Can be high if empirical predictive models are used.	Direct prediction of fishery metrics.
Monetary or other valuation	Low-moderate. Scaling unlike offsetting using the cost of replacement of lost habitat.	Relationship between dollars expended and fishery productivity uncertain because offsets not defined by losses.

MONITORING

As stated in the Offset Policy (DFO 2013a), proponents are responsible for implementing offsetting plans and monitoring and reporting on their effectiveness. In December, 2011, DFO held a science advisory process (DFO 2012) to design a standardized monitoring approach to determine the effectiveness of habitat compensation in achieving the fish habitat conservation goal of DFO's Policy for the Management of Fish Habitat (DFO 1986). One of the conclusions of the workshop was that a comprehensive program needed to be developed to help guide practitioners and proponents when monitoring the effectiveness of compensation habitat was required under the terms of an authorization. The new Fisheries Protection Provisions of the *Fisheries Act* requires that the focus of this monitoring program now be adjusted. To that end, based on the advice outlined in DFO (2012), a DFO Canadian Technical Report of Fisheries and Aquatic Sciences has been produced delineating components (and associated metrics) of a standardized effectiveness monitoring and reporting program, which if executed properly, could lead to the long-term improvement of offsetting plans and monitoring design for the ultimate benefit of fisheries productivity in Canada (Smokorowski et al. 2015).

DISCUSSION

In this report we adapted the framework for calculating offset requirements used in international damage assessment and biodiversity offset programs to the needs of the Fisheries Protection Program, and describe a suite of exchange or equivalency metrics that could be used for equivalency calculations.

The equivalency framework is a tool for scaling the magnitude of offsetting measures in the context of Principle 2 of the offsetting policy, however, that scaling is only part of the considerations when developing an offsetting plan. Maintaining the ongoing productivity of fisheries resources requires an appropriate configuration of habitats and environmental conditions for all life stages that will generate vital rates sufficient to permit sustainable use (DFO 2013b; 2014; Bradford et al. 2014). The choice, location, and configuration of offsetting measures must incorporate knowledge of species life history and the supporting role of the ecosystem in that productivity. Equivalency analysis is not designed to account for these factors; to be successful the benefits of the measures to fisheries productivity have to be evaluated in the species and ecosystem context. That evaluation is relatively straightforward in in-kind localized habitat replacement but becomes more complex and important for out-of-kind offsetting.

Equivalency methods and metrics are generally area-based and do not take into account the role of specialized functions or biological processes that could have disproportionate effects on productivity. For example, specialized nursery habitats for larval or juvenile fish may only be occupied for a few months of the year, and the abundance of fish at these sites at other times may be low. An ill-timed sampling trip could lead to a significant underestimate of the importance of the area to fish productivity, and an underestimate of offset requirements. Similarly the significance of nursery areas for adult production may be underestimated by metrics such as biomass or production and it may be more appropriate to use equivalent adult approaches (DFO 2015) to accurately predict the implications of habitat impacts on fisheries production.

Similarly, offset measures that are potentially ineffective can by justified in equivalency analysis through the use of larger amount of offsetting ratios, however, these outcomes should be reviewed critically. Large amounts of poor habitat are unlikely to be a viable substitute for important habitat; this risk can be minimized by avoiding “trading down” as much as possible.

There are no measures of species diversity among the equivalency metrics as the goal of the offset program is the maintenance of fisheries productivity. However, biodiversity and productivity are often related (Randall et al. 2013) and the provision of measures to maintain biodiversity should be an important consideration in offset design.

In summary, equivalency methods are a useful tool for scaling of offset requirements but their implementation should be viewed within the broader perspective of the ecological context of the species affected and the activities that are being proposed. Ongoing monitoring and program evaluation is needed to refine the use of equivalency calculations in the future.

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