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Accounting for Uncertainty and Time Lags in Equivalency Calculations for Offsetting in Aquatic Resources Management Programs

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Abstract Biodiversity offset programs attempt to minimize unavoidable environmental impacts of anthropogenic activities by requiring offsetting measures in sufficient quantity to counterbalance losses due to the activity. Multipliers, or offsetting ratios, have been used to increase the amount of offsets to account for uncertainty but those ratios have generally been derived from theoretical or ad-hoc considerations. I analyzed uncertainty in the offsetting process in the context of offsetting for impacts to freshwater fisheries productivity. For aquatic habitats I demonstrate that an empirical risk-based approach for evaluating prediction uncertainty is feasible, and if data are available appropriate adjustments to offset requirements can be estimated. For two data-rich examples I estimate multipliers in the range of 1.5:1 – 2.5:1 are sufficient to account for the uncertainty in the prediction of gains and losses. For aquatic habitats adjustments for time delays in the delivery of offset benefits can also be calculated and are likely smaller than those for prediction uncertainty. However, the success of a biodiversity offsetting program will also depend on the management of the other components of risk not addressed by these adjustments.

Keywords Biodiversity offsetting · Fish habitat · Offsetting multipliers · Risk · Mitigation

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Introduction

Faced with the prospect of accidental or planned anthropogenic impacts to the natural environment, a variety of regulatory schemes have been developed to manage those impacts. Many jurisdictions use the "polluter pays" principle, and require that parties causing damage be responsible for measures to remediate or restore affected resources. For planned activities, a hierarchy of preferences is often invoked; that hierarchy asks that developers of projects first seek to avoid, then minimize, and finally as a last resort compensate or offset for damages, usually with the goal of no overall change to environmental values (McKenney and Kiesecker 2010). The hierarchy (also known as the "mitigation sequence" or "mitigation hierarchy") was first introduced in 1980 regulations for the US Clean Water Act of 1972 (Hough and Robertson 2009) and has been widely adopted since. For unplanned impacts, mitigation and compensation may be required in proportion to the extent of the damage (Dunford et al. 2004). Compensation activities include the creation or restoration of habitats, or other measures to increase ecological values. A calculus known as equivalency analysis is used to determine the amount of compensation required to balance losses (Dunford et al. 2004). Factors such as the value of the affected biodiversity, the magnitude and duration of impacts, feasibility of offsets, and uncertainty can be included in equivalency analyses (Allen et al. 2005; Quétier and Lavorel 2011).

In Canada, 1977 amendments to the *Fisheries Act* included new sections that prohibited the harmful alteration or destruction of fish habitat unless permitted by the Minister of Fisheries, through the issuance of a permit known as an authorization. The subsequent Policy for Management of Fish Habitat (DFO 1986) invoked the mitigation hierarchy for development projects to support the goal of maintaining the productive capacity of fish habitats and the fisheries resources they support. If residual effects were found to result after reasonable avoidance and mitigation measures were employed, compensation may be required as a condition of the authorization.

In 2012 the habitat protection sections of Canada's *Fisheries Act* were amended and a new offsetting policy was developed (DFO 2013). The policy confirms the primacy of the avoid-mitigate-offset hierarchy, and describes offsetting requirements when unavoidable residual impacts are likely to occur. Those offsets can be "in-kind", replacing what was lost or impacted by the project, or "out-of-kind" designed to increase fisheries productivity in ways consistent with regional fisheries management plans or restoration priorities (DFO 2013). When a proponent applies for an authorization under the *Fisheries Act* an offsetting plan must be submitted; contents of the plan are specified in policy (DFO 2013) and regulation[1]. The plan includes estimates of the impact of the residual effects, specification of the amount and nature of the proposed offsets, including allowances for uncertainty and time lags such that impacts and benefits are balanced. These requirements are similar in many ways to generalized frameworks for biodiversity offset policies and programs (e.g., McKenney and Kiesecker 2010; ICMM and IUCN 2012; BBOP 2013).

There are a number of places in the regulatory process where uncertainty will affect the success of an offsetting program (Fig. 1). Uncertainty exists in the prediction of losses and gains in the during the planning and permitting phase, in compliance with permit conditions, in project implementation, and in the short- and long-term effectiveness of the offsetting measures (Pilgrim and Ekstrom 2014). Time delays in the delivery of offsetting benefits will also contribute to an imbalance between losses and gains.

Risk in biodiversity offsetting programs has been considered in theoretical analyses (Minns and Moore 2003; Moilanen et al. 2009; Gibbons et al. 2015) or implemented through the use of arbitrary adjustments, usually in the form of multipliers to increase the offsetting requirements (Levrel et al. 2012; Tallis et al. 2015). In some cases multipliers are based on the conservation value of the impacted habitats (DEFRA 2012; Saenz et al. 2013), presumably to act as a deterrent to development in areas of high value. Both Bull et al. (2013) and Tallis et al. (2015) note that there is no standardized approach to setting allowances for uncertainty and risk and both identify a need for a rigorous and repeatable approach for the determination of offset multipliers.

To advance the development of practical advice on uncertainty in offsetting I present a means to incorporate prediction uncertainty and time lags in offsetting design in the context of freshwater fish habitat and Canada's Fisheries Act. Although Canada's offsetting policy for aquatic habitats requires allowances to be made for these factors, no practical guidance for proponents or regulators is supplied. While I focus on fish habitat the approach could be adapted to similar regulatory regimes that use the mitigation hierarchy and offsetting to manage impacts to biodiversity values.

[1] Applications for Authorization under Paragraph 35(2)(b) of the Fisheries Act Regulations (SOR/2013-191)

Equivalency Analysis and Metrics

Equivalency analysis balances losses of biodiversity and gains from offsetting. A simple form of equivalency analysis that ignores allowances for uncertainty and time lags (Levrel et al. 2012) can be used to compute the amount of offsetting that will counterbalance area-based habitat impacts:

$$A_i d_i = A_o d_o \tag{1}$$

where A_i is the area of the project's residual impacts, A_o is the area of the offset; d_i is loss of biodiversity per unit area for the unavoidable impact and d_o is the gain per unit area associated with the offsetting activity. Both d_i and d_o are long-term averages. The area of offset required for equivalency is:

$$A_o = A_i \frac{d_i}{d_o} \quad or \quad A_o = A_i M \tag{2}$$

In Eq. 2 the multiplier, M, scales the size of the offset needed to achieve equivalency by the ratio of the predicted biodiversity value per unit area of the project's area of impact to that of the offset. Thus in cases where the offsetting measures are expected to have lower value than the area affected by the project, $M > 1$ and a larger offset will be required. Conversely if the offsets have greater biodiversity value than the area impacted M will be less than unity.

In application of Eq. 1, d is the equivalency metric that is a common currency to express both losses and gains in biodiversity. In the case of Canada's *Fisheries Act* direction for decision making is provided by section 6.1 of the *Act* that states the purpose "is to provide for the sustainability and ongoing productivity of commercial, recreational and Aboriginal fisheries". This purpose statement defines the goal of the offsetting program and leads to the suggestion that metrics of fisheries productivity or suitable surrogates may be appropriate as equivalency metrics.

Fisheries science has a rich tradition of developing direct or proxy measures for fish population abundance or productivity that can be used for equivalency analysis (Minns et al. 2011; Bradford et al. 2016). Most are surrogates, and are often based on habitat or fish population characteristics.

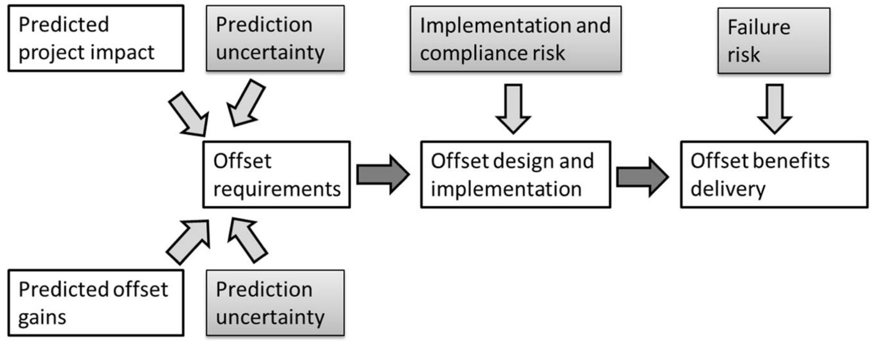

Fig. 1 Flowchart illustrating the sources of risk and uncertainty in the planning and implementation of measures to offset residual effects from development projects, based on the process for Canada's fisheries offsetting policy

The simplest is habitat area alone, and they progress to metrics that include both area and some index of habitat quality or function (Minns et al. 2011). These are similar to the habitat-hectares or functional assessment approaches used in other applications (Parkes et al. 2003; Quétier and Lavorel 2011; Tallis et al. 2015). Species-specific fish-habitat models similar to habitat suitability indices (HSI) or habitat evaluation procedures (HEP) that have been proposed for equivalency calculations for cases where key or "umbrella" species can be identified (e.g., Strange et al. 2002). The most complex metrics are those that directly estimate effects on fish populations or fish communities, their productivity, and fisheries that depend on them (Minns et al. 2011).

Adjustments for Prediction Uncertainty

Prediction uncertainty creates the risk that offset requirements resulting from equivalency calculations may not be sufficient to balance the project's actual residual impacts if project impacts are underestimated or benefits of the offset are overestimated. An increase in offset requirements can reduce this risk to a level determined to be tolerable.

Assuming that areas in Eq. 2 can be estimated accurately, prediction uncertainty results from uncertainty in d_i and d_o for an individual project and its effects on the ratio d_i/d_o. In the case of Canada's *Fisheries Act*, prediction error will often be the result of application of a fish-habitat model (Minns and Moore 2003), although the approach applies to any equivalency metric.

Since M is a ratio of two uncertain quantities (d_i/d_o), uncertainty is calculated with an approximation for the variance of a ratio (Kendall and Stuart 1963) using the mean, μ, variance, σ^2, the covariance, COV, of d_i and d_o:

$$\sigma_M^2 \approx \frac{\mu_i^2}{\mu_o^2}\left[\frac{\sigma_i^2}{\mu_o^2} + \frac{\sigma_o^2}{\mu_o^2} - 2\frac{COV(d_i, d_o)}{\mu_i \mu_o}\right] \quad (3)$$

Unfortunately a ratio estimator has complex properties such that percentiles cannot be derived from a known probability distribution (Marsaglia 2006). Instead I used Monte Carlo simulation to generate a frequency distribution of M, given uncertainty about the loss of biodiversity value resulting from the project's impacts (d_i), and the gain associated with the offsetting measures (d_o). I assumed d_i and d_o are random variables that follow a normal distribution with parameters (μ,σ); the mean represents the average or best estimate for the habitat type or activity. Covariation refers to whether the uncertainty in d_i and d_o for an individual project around their expected means are independent of each other or not. In simulations very small and negative values arising from the generation of the random variables were rejected using a fixed lower threshold of 0.02. Simulation results were compared to approximations that made use of Eq. 3.

I used a risk-based approach to evaluate the distribution of M values from the simulations. I used a risk tolerance threshold of 0.8, which means that equivalency will be achieved 80% of the time given uncertainty in both impact and offset predictions. Hence for each simulation, I found M_{80}, the 80th percentile of the distribution of M values. The choice of a risk tolerance threshold is a policy decision that will result from balancing large and potentially impractical multipliers that would be needed to achieve equivalency with very high probability against the chance of a net loss of biodiversity for an individual project.

To analyze uncertainty adjustments I first considered the special case where $\mu_i = \mu_o$ and $\sigma_i = \sigma_o$, and COV = 0, which represents the situation where impacts and offsets are similar and site-specific variation between the impact and offset sites is independent. I found an increasing relation between the coefficient of variation (CV = σ/μ) of the prediction uncertainty for d_i and d_o and M_{80} (solid dark line, Fig. 2). A useful approximation for this level of risk tolerance was $M_{80} \approx M + \sigma_M$, where σ_M is calculated from Eq. 3 (dashed line, Fig. 2). Although the approximation is presented for the case where the CV of impact and offset uncertainty were the same, it was also found to hold when they were different (for CV < 0.5). Increasing the risk tolerance to 0.9 increased the uncertainty adjustment significantly (*dotted line*, Fig. 2).

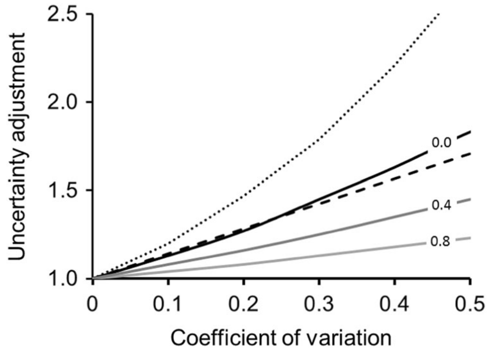

Fig. 2 Relation between the coefficient of variation for the uncertainty in estimates of project impacts or offset benefits and the proportional uncertainty adjustment required to ensure equivalency with specified probability, p. Here the area affected by the project and the offset area as assumed to have the same values for the equivalency metric (i.e, $M = 1$, in the absence of uncertainty), and the same CV. *Solid lines* are simulated results for $p = 0.80$ for three levels of covariation between impact and offset values (indicated are ρ^2 values of 0, 0.4 and 0.8). *Dashed line* is the approximation $M + \sigma_M$. Dotted line shows results for $p = 0.9$, $\rho = 0$, illustrating the increased offsetting requirement for a lower level of risk tolerance

To evaluate the effects of covariation in the biodiversity value of the impact and offset sites I first note that $COV(d_i/d_o) = \rho\ \sigma_i\sigma_o$ where ρ is the correlation coefficient between variables. If $\mu_i = \mu_o$ and $\sigma_i = \sigma_o$ as before then Eq. 3 simplifies to

$$\sigma_M^2 \approx 2CV^2(1 - \rho) \tag{4}$$

Inspection of Eq. 4 shows uncertainty in M increases with prediction uncertainty, as shown before, but decreases with increasing levels of covariance of d_o and d_i. To confirm this result I ran simulations that included the correlation between d_i and d_o values and found that for high values of ρ offset multipliers were greatly reduced (Fig. 2).

To illustrate the application of uncertainty adjustments I developed two examples for freshwater fish habitat. In the first, I assumed a hypothetical land development will result in unavoidable losses of stream side-channel that is used as rearing habitat for juvenile coho salmon (*Oncorhychus kistuch*); those juveniles will ultimately migrate to the ocean as smolts. The impact will be offset with the conversion of an unused gravel pit to a rearing pond, a technique commonly used in salmonid habitat enhancement in western North America (Roni et al. 2006).

Here d is expressed as coho salmon smolts $\cdot$ m^{-2}. The density of smolts is a useful metric of fisheries productivity because smolt production is strongly linked to habitat characteristics and can be used to predict the production of adult fish. Thus d_i is the density of smolts in the area lost to the project, and d_o is the density predicted for the offset. A database of smolt abundances from these habitats was used to estimate variability in production among sites, as well as among annual smolt estimates from individual sites (Roni et al. 2006). Data were available for 6 gravel pit ponds and 14 side channels, and there was an average of 6 years of data for each site.

I constructed five hypothetical scenarios that vary in the use of data to compute the uncertainty adjustment needed to achieve at least equivalency at $p \geq 0.8$. I assumed $\rho = 0$, which is the conservative case that simulates no similarity in relative productivity of offset and impact sites. In Scenario 1 uncertainty was ignored and M was computed as the ratio of the average productivity of habitat types (the basic equivalency calculation of Eq. 2). For Scenarios 2–5 uncertainty in the prediction of smolt abundance was introduced in different ways and in all scenarios was based on the observed variability among and within similar sites from Roni et al. (2006). I used simulation to find M_{80}, the multiplier needed to achieve equivalency at $p \geq 0.80$ and calculated the ratio of M_{80} from Scenarios 2–5 to M in Scenario 1 to compute how much more offsetting is required to adjust for uncertainty under different scenarios of data availability.

Scenario 2 assumed uncertainty in the prediction of abundance at the impact site, but used a fixed value for the offset, simulating the case where a fixed reference standard is used for the prediction of offset benefits. For Scenario 3 uncertainty at both sites was incorporated, based on estimates from Roni et al.'s data; this simulated the case where no-site specific data were collected. In Scenarios 4 and 5, I assumed uncertainty in smolt production at the impact site was estimated from 3 to 6 years of monitoring data, simulating the situation where pre-project baseline monitoring occurs. I assumed the average production from the impact site was the same as the average of the database and used the observed average interannual variability in smolt production to estimate the uncertainty that would result from the collection of 3 or 6 years of data. The prediction of benefits and associated uncertainty for the offset site were drawn from the database as in Scenario 3.

For Scenario 1, where uncertainty is ignored, the offset must be 2.2 times larger than the impacted area because, on average, gravel pit ponds produce fewer smolts than side channels. When uncertainty is included the offset area needed to ensure equivalency with 80% probability increases to 3.6–4.7 times larger than the area of lost habitat. Thus uncertainty increases the offsets by factors of 1.7 to 2.1 over Scenario 1, depending on which sources of uncertainty were introduced into the computation (Table 1). There was some benefit to obtaining site-specific data compared to using the regional database for the impact site. The savings resulting from the requirement for a smaller offset will likely be evaluated relative to the potential delays and costs associated with collection of site-specific monitoring data prior to development.

Table 1 Multipliers for offsetting requirements arising from uncertainty in prediction of losses and gains, based on data for coho salmon smolt abundance from Roni et al. (2006)

Scenario	Side channel		Ponds			
	μ	σ	μ	σ	M_{80}	UncAdj
1. No uncertainty	0.42	0	0.19	0	2.2	
2. Fixed reference level, offset	0.42	0.36[a]	0.19	0	4.0	1.8
3. Regional database	0.42	0.36	0.19	0.085[a]	4.7	2.1
4. Site-specific sampling $n=6$	0.42	0.15[b]	0.19	0.085	3.6	1.7
5. Site-specific sampling $n=3$	0.42	0.22[b]	0.19	0.085	3.9	1.8

The example assumes loss of side-channel habitat that will be replaced by offsetting in the form of constructed ponds. For each habitat type μ is the mean coho smolt density (m^{-2}), σ is the standard deviation. M_{80} is from Monte Carlo simulation results (100,000 trials) and is the ratio of the area of offset to area of impact that will achieve equivalency in smolt abundance 80% of the time, given the uncertainty about the true production in the impacted and offsetting habitats. UncAdj is the uncertainty adjustment and is the proportional increase in offset area required compared to the no uncertainty case; scenarios are fully described in the text.

[a] estimated as $(\sigma_b^2 - \sigma_w^2/n)^{½}$ where σ_b^2 is the variance of means and σ_w^2 is the average variance of estimated from an average of n stream-specific annual estimates. This correction removes the within stream error from the between stream variance

[b] estimated as $(\sigma_w^2/n)^{½}$ where σ_w^2 is the average variance of annual estimates of smolt abundance and n is the average sample size (years)

As a second example, I used a regression-based predictor of fish production. Many regional relations have been developed between fish production and variables that are indicators of physical features or biological productivity. For example, Schlesinger and Regier (1982) developed a regression model that predicts fishery yields based from estimates of lake depth, water chemistry and mean air temperature; Bérubé et al. (2005) proposed its use for the assessment of large hydroelectric developments. The model has relatively high explanatory power ($R^2 = 0.80$) and the standard error of a prediction of log$_{10}$(Yield), as kg ha^{-1} year^{-1}, is 0.30. Assuming the impact and offset sites are both at the mean of the habitat variables such that $\mu_i = \mu_o$, and assuming that deviations in productivity from the mean are independent, the CV of a prediction of log$_{10}$(Yield) is 0.35 resulting in $\sigma_{\mathrm{M}} = 0.50$ (Eq. 4, COV = 0). To achieve an 80% chance of the offset having similar or greater yield than the impacted site, the uncertainty adjustment was 1.55, assuming uncertainty in both measures. The adjustment was reduced to 1.42 if the prediction of either loss or gain uses a fixed reference value.

In the examples, I assumed the prediction errors for impact and offset sites are independent and are thus most appropriate for cases where the offset measures are of a different nature or located at sufficient distance from the impact site so that regional factors that affect productivity (e.g., water chemistry) are different. This is the worst case scenario as it results in the largest adjustments for uncertainty. The uncertainty adjustment will be smaller in cases where the offset is similar or nearby to the impact site because there is a greater likelihood that the deviation from the predictive model for the impact and offset site will be similar (i.e., COV > 0 in Eq. 4). For example, if the impact and offset both occur in particularly productive habitats, the losses due to the project's residual impacts, and the gains associated with offset measures will both be underestimated by the predictive model. Thus covariation will reduce the requirement for uncertainty adjustments; indeed relatively minor adjustments for uncertainty may be appropriate for in-kind offsetting located near the site of the project's impacts (Fig. 2).

Time Lags

Time lags can occur between losses caused by the project and the accrual of benefits from offsetting measures. Canada's Fisheries Offset Policy indicates that proponents should be expected to provide additional offsets to account for loss of productivity that result from time lags (DFO 2013). Such situations arise when offsetting measures are not implemented until after the project has started, or if the offsetting measures take time to become fully functional (Fonseca et al. 2000; Lennox et al. 2011).

A framework for including the time dimension was developed under the Natural Resource Damage Assessment equivalency framework to calculate compensation required for ecological damage or injury (NOAA 1999). Time lags are usually accounted for using net present value (NPV) calculations that incorporate time-dependent changes in both impacts and benefits from the offsetting measures. This framework was proposed for fish habitat compensation in Canada (Minns 2006; Clarke and Bradford 2014) and is conceptually identical to formulations for other biodiversity offset or damage assessment programs (e.g., Dunford et al. 2004; Overton et al. 2013; Laitila et al. 2014).

If project impacts and planned offsetting actions occur at discrete times and take full effect instantaneously, and if the service loss or gain is the equivalent, a multiplier (M_{lag}) that accounts for discounted loss of NPV due to time lags is:

$$M_{lag} = (1 + r)^{t_{lag}} \tag{5}$$

where t_{lag} is the number of years that elapse between the project impact, and the full benefit of the offset. Implementation of Eq. (5) shows that for lags less than 10 years multiplier required to achieve equivalency ranges from 1 to 1.3 for a 3% discount rate (Fig. 3). The simple approach based on Eq. 5 may be sufficient for many needs, especially when the offset ratio associated with other uncertainty

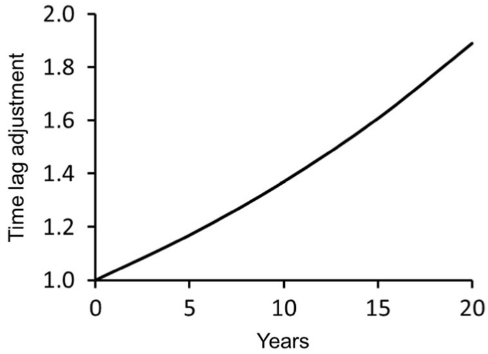

Fig. 3 Multipliers required to achieve equivalency when there is a lag, *t*, between when benefits of the offsetting measures take effect and when residual impacts of the project occur. Discount rate is 3%

elements are much larger than those needed to make up for time lags. Detailed net present value calculations may be required when project impacts and offset benefits vary over time; an example for fish habitat is provided in Clarke and Bradford (2014).

Since prediction uncertainty and adjustments for time lags are independent, the total adjustment for both factors is computed as the product of the two multipliers. For the coho salmon example, using scenario 2 of Table 1 and assuming a 5-year time lag for offset delivery and a 3% discount rate, the total multiplier is $2.2 \cdot 1.16 = 2.55$.

Discussion

I developed a method to account for prediction uncertainty and time lags in biodiversity offset programs for aquatic habitats. When information is available on the biodiversity value of impacted habitats and the biodiversity gain of offsets it is possible to use those data to produce quantitative estimates of the allowances needed to account for the difference in value of impact and offset areas, and the uncertainties associated with those assessments. For two examples for aquatic environments I computed uncertainty and time lag multipliers that are much smaller than other analyses found that sometimes large (3–100 s) and likely infeasible multipliers may be required to ensure a fair exchange (e.g., Robb 2002; Moilanen et al. 2009; Gibbons et al. 2015). This difference is the result of the inclusion of different factors and assumptions about time horizons, efficacy of offsets, and risk tolerances. These factors are discussed below.

Net present value calculations have been proposed as a method to adjust offsetting requirements to account for time lags between project impacts and offsetting benefits (Dunford et al. 2004). Recently it has been suggested that multipliers required to achieve equivalency may be so large to be largely impractical once time lags are fully incorporated (Moilanen et al. 2009; Laitila et al. 2014; Gibbons et al. 2015). I believe this not to be the case for aquatic habitats if appropriate measures are taken in the design and implementation of offsets. The response of many aquatic habitats to manipulations, including restoration measures, may be relatively rapid compared to forest habitats that have been the focus of other analyses (Curran et al. 2014; Gibbons et al. 2015). For riparian communities, 10–15 years are needed for many attributes to be maximized (Lennox et al. 2011; Orzetti et al. 2010) and for stream biota that delay may be less (Decker et al. 2008; Miller et al. 2010). Jones and Schmitz (2009) found the average recovery time for aquatic habitats to recover from human-caused disturbance was about 10 years. If an offset is constructed as soon as the project is started, and it takes 10 years to become fully functional, then the multiplier to account for the delay will be less than 1.3 (assuming a 3% discount rate). If the offset provides partial benefits as it matures, then appropriate adjustments are smaller than this. Thus with policy measures that ensure appropriate design and timely implementation, adjustments for time lags should be far less those suggested for other habitat types.

I demonstrate how prediction uncertainty in equivalency analyses can be quantitatively accounted for through the use of empirically-based risk adjustments. My examples were based on simple models that predict fish abundance or production from habitat area. Uncertainty results from factors not included in those models, including physical habitat features, the chemical environment, biotic interactions and measurement error (Minns and Moore 2003). I found the risk “premium” associated with that uncertainty fell within a relatively narrow range (1.4–2.1) for two examples of freshwater fish habitat. However, I caution that the examples chosen to illustrate the calculation of uncertainty adjustments for offsetting requirements may have generated smaller adjustments that might generally be the case. For coho salmon, robust data were available for specific habitat types that are known to be productive for coho salmon. This likely reduced the variation within and among sites which would have a corresponding effect on the uncertainty surrounding predictions of impact loss or offset gains. In the case of the lake model, the availability of habitat quality variables in a predictive model also reduced uncertainty. Further analysis of different situations, including those that are data sparse or reliant on expert judgement will be needed for the development of more general guidance about uncertainty adjustments.

To illustrate the method I chose a percentile (80th) for M that will result in most individual projects achieving a net gain in the equivalency metric. The choice of percentile is a management decision; and I was largely guided by a similar choice by Cochrane et al. (2015). In theory, repeated application of the 80% percentile should result in a net gain at a regional or program level since most projects are expected to result in a net gain as a result of the uncertainty multipler, although one in five individual projects may fail to achieve no net loss. The failure rate can be reduced by using a more risk-adverse percentile, but the corresponding multipliers increase significantly as the percentile increases. For example, Moilanen et al. (2009) modeled a 95% success rate, which contributed to the large multipliers in their study.

For in-kind exchanges where replacement habitat is constructed near the site of the impact, smaller allowances than those shown in my examples may be appropriate. Many components of prediction uncertainty will be correlated between impact and offset sites if both are located in the same area and if both are affected by the same larger-scale processes that impact fish populations beyond the habitat variables included in the predictive model. It will be difficult to estimate the magnitude of that covariation in most cases; however, the theoretical analysis suggests that a significant reduction the uncertainty adjustment may be justified.

Although I used simple fish-habitat models in the analyses, the risk-based approach can be applied to any quantitative analysis of the relative magnitudes of project impacts and offset benefits. Cochrane et al. (2015) modeled population-level effects of development and out-of-kind compensation measures and used a risk-based analysis to determine the scope of offset measures required to meet policy or regulatory requirements. They also included results of an expert solicitation which is an approach that may be needed when quantitative information is lacking. Interestingly, using a risk tolerance of the 20th percentile for mortality (equivalent to my 80th percentile for M) Cochrane et al. (2015) found an approximate doubling of the offsetting measure was required as a risk adjustment, a result similar to my findings for fish habitat.

An alternative to multipliers for addressing uncertainty is the use of an adaptive approach that allows adjustment of offsetting requirements and measures based on the measured impacts of the project and the results of monitoring the offset. A carefully designed monitoring program and a commitment to respond to monitoring results should reduce uncertainty; in this case multipliers designed to protect against worst case outcomes with regard to both project impacts and offset benefits may not be needed. Some of the resources that would have been used for the implementation of larger amount of offsetting can be directed to monitor and possibly intervene to ensure the success of the offset (Doyle and Shields 2012). Results of monitoring programs can also inform future decision making; failure to learn from past practices has been repeatedly been identified as a shortcoming of habitat compensation and restoration programs (Harper and Quigley 2005; Palmer et al. 2007; Kondolf et al. 2007).

Some published analyses of uncertainty in offsetting incorporate adjustments for efficacy into their calculation of multipliers. Efficacy values 10–50% of natural value for habitat replacement have been assumed (Moilanen et al. 2009; Gibbons et al. 2015) and can be supported by some empirical results (Pickett et al. 2013). In my analysis the determination of offsets requirements in the absence of uncertainty occurs during the equivalency analysis and is separated from adjustments for uncertainty. Equivalency analysis accounts for predicted differences in conservation value of impacts and proposed offsets (as indexed by d_i/d_o in habitat-based analyses). While hypothetical values are useful for illustration, decision making will be best informed by empirical information resulting from the monitoring of compensation and restoration activities. In the case of fish habitat, certain types of fish habitat creation or restoration have been extensively monitored and have been found to be productive (e.g., coho salmon floodplain channels, Roni et al. 2006; Ogsten et al. 2015). In these cases equivalency can be met without a large increase in the amount of offsetting, although my analysis shows an adjustment is still required as a result of prediction uncertainty.

In some circumstances increasing the size of the offset to account for poor function may not be an appropriate means to achieve a balance between losses and gains. Equivalency calculations result in increased offset requirements when predicted biodiversity values for the offset area are lower, but the consequences of creating lower quality habitat on population productivity needs to be considered (Keagy et al. 2005). For example, if the survival of fish eggs and larvae is lower in artificial spawning habitats constructed as an offset, increasing the amount of habitat could cause a decrease in the productivity of the population if artificial habitats are used in preference to natural ones. Similarly, if constructed juvenile habitats do not provide the conditions for fish to achieve a positive rate of energy gain of natural habitats, a loss in adult production may result that cannot be mitigated through the creation of more poorly functioning habitat (i.e., Rosenfeld and Boss 2001). Careful analysis of the effects of the offsetting measure on the ultimate biodiversity goal is required to evaluate the suitability of proposed measures.

Other Sources of Risk

In addition to adjustments for prediction uncertainty and time lags, there are other sources of risk that can result in offsetting programs not meeting their goals (Fig. 1). The feasibility, implementation and sustainability of offsetting measures are critical components for the success of the offsetting program, but I do not include these elements in the calculation of multipliers. Implementation and compliance risk results from proponents not meeting conditions of their permits, including misspecification of the size or magnitude of project impacts, failure to meet their offsetting requirements, and technical issues associated with the design and implementation of offset measures. Race and Fonseca (1996) considered implementation and compliance issues to the major factor in the failure of offset programs, and subsequent reviews find similar results (e.g., Zedler et al. 2001; Quigley and Harper 2006; Tischew et al. 2010; but see Hill et al. 2013).

Some analyses have calculated multipliers that incorporate observed rates of compliance failure and these can be significant when compliance is poor (Robb 2002; Quigley and Harper 2006). Retrospective analyses of compensation or offsetting programs have found about 30% of permit conditions were not met, contributing to a failure to meet program or policy objectives (Quigley and Harper 2006; Brown et al. 2013). Multipliers may be an appropriate strategy to deal with area-based compliance issues; for example, for fish habitat cases in Canada, Quigley and Harper (2006) found many projects that had impact areas larger than planned or offsetting areas smaller than required in their permits. However, increasing the size of the offset may not be an effective measure for other forms of non-compliance. Overton et al. (2013) suggest that legal, regulatory or financial instruments may be more appropriate to manage compliance risk as the use of multipliers penalizes all proponents relative to those that fail to comply. Lack of monitoring and enforcement has been raised as a factor in non-compliance (Quigley and Harper 2006). Brown et al. (2013) cautions that the causes of non-compliant behavior are not well known and the development of a risk management strategy will be aided by understanding the conditions that lead to decreased rates of compliance.

Similar arguments can be made for implementation risk associated with the failure to use the best available scientific and engineering expertise in the design and construction of offsetting measures; these risks can be reduced through the advancement of the science, using monitoring and adaptive management and the institutionalization of cultures of best practise that include guidance for implementation, quantitative targets, and monitoring programs (Hill et al. 2013).

The failure of offsets to deliver benefits is a risk not accounted for in the equivalency analysis. This risk includes catastrophic failures, or the deterioration of the offsetting measure over time. As noted by Pilgrim and Ekstrom (2014) increasing the size of an offset does not reduce the risk of failure, however, bet-hedging by constructing multiple offset works can reduce the probability of complete loss (van Katwijk et al. 2009; Moilanen et al. 2009). Offsets that are prone to complete failure are inconsistent with the principle of long-term self-sustaining benefits identified in Canada's offsetting policy (DFO 2013). Offset measures placed in high energy habitats (coastlines or rivers) are particularly vulnerable to such losses (Frissell and Nawa 1992; van Katwijk et al. 2009). However, other types of measures for offsetting aquatic habitats such as road culvert replacements or side-channel construction (Roni et al. 2006; Ogsten et al. 2015) can provide sustainable benefits. Thus it should be possible to reduce this source of risk by placing constraints on the types of measures that are acceptable, and monitoring and maintenance of offsetting to ensure the benefits are realized over time.

In summary, I demonstrate that scientifically defensible methods for computing adjustments to account for time lags and prediction uncertainty in equivalency are feasible for aquatic habitats. Remaining aspects of the uncertainty and efficacy of the offsetting program may be best managed with other measures, and regulatory oversight. These other factors are significant, suggesting that regulators need to set limits as to what can be offset, provide clear technical guidance for the development of offsetting measures, require adequate monitoring and employ enforcement to ensure permit requirements are met (Brown et al. 2014).

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Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no competing interests.

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