Dear Panel and Secretariat,

Please find attached another submission by MiningWatch Canada for the technical hearings.

I'm also attaching 3 documents for the official record, others that we request be added within our submission can be found online.

I would appreciate acknowledgement of the receipt of our submission and the other three attachments.
Supplemental Submission for Fisheries Technical Hearings
Prosperity Mining Project – CEAA Panel Review

MiningWatch Canada

April 16, 2010

In preparation for the upcoming topic specific hearings on fish and fish habitat, MiningWatch Canada submits the attached documents and following summary comments relevant to our concerns about the destruction of the aquatic ecosystems in the Fish Lake watershed and Taseko’s proposed habitat compensation plan.

These comments focus on the record of success of fish habitat compensation projects in Canada and the ability of DFO to demonstrate that such projects are meeting the required objectives of No Net Loss. Such contextual information is of vital importance to the Panel in order to weigh the degree of risk and the acceptability of the risks associated with attempting to compensate for the loss of the natural features and functions of the watershed. Judging these risks is, in turn, critical to determining whether or not the proposed project is likely to contribute to sustainable development and provide a net ecological and social benefit – an assessment required by the Guidelines for the EIS.

The proponent is asking the Canadian government to approve its mine project and the destruction of the Fish Lake watershed on the premise that the company can develop and implement a successful habitat compensation plan into the indefinite future. We submit that there is substantial evidence that the proposal has a high risk of failure on several different accounts including:

1. Fundamental inadequacies in the plan itself;
2. Poor track record of success for conventional, single-component compensation projects;
3. Greatly increased risks and uncertainties of success in implementing a multi-component compensation plan of this scale and complexity;
4. Inability of DFO to effectively monitor and enforce compliance with commitments at present and the decreasing resources available for future habitat management and protection activities.

The inadequacy of the plan is clearly a crucial element in evaluating the sustainability of the project. The failure of the plan to conceptually achieve the No Net Loss objective has been well documented in our previous submissions, as well as the DFO’s March 12 submission. Accordingly, we will only address points 2 to 5 in the present submission.

Our submission includes the following documents for the record, most are available online and urls are included with full references at the end of the document. Those not available online will be submitted as attachments.

2. Assessment of Techniques for Rainbow Trout Transplanting and Habitat Management in British Columbia. Canadian Manuscript Report of Fisheries and Aquatic Sciences No. 2562 (Hartman and Miles 2001; online)

3. Habitat Compensation Case Study Analysis. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2576. (Lange et al. 2001; online)

4. Compliance with Canada’s Fisheries Act: A Field Audit of Habitat Compensation Projects. (Quigley and Harper 2006a; attached)

5. Effectiveness of Fish Habitat Compensation in Canada in Achieving No Net Loss (Quigley and Harper 2006b; attached)

6. Why Bartering Biodiversity Fails (Walker et al 2009; attached)

Below we summarize the key points of these documents and offer additional illustrative points from other sources.

**The Poor Track Record for Conventional Compensation Projects**

In evaluating the proposal to mitigate for the destruction of the Fish Lake watershed it is important to assess the risk associated with the plan’s implementation in terms of both actual work completed and if implemented how likely the works perform as predicted. Insight into these questions can be gained by reviewing the success of existing compensation plans in achieving the basic objective of No Net Loss.

DFO, as explained below, does not have a reliable system for monitoring and documenting success. Nonetheless, a few studies are available, and we have submitted some of the most recent audits in our submission. All the documents we located that review the performance of habitat compensation projects in Canada generally report discouraging results. It should be noted that most, if not all, these projects are small in area compared to the project at hand, and are typically single-component projects being implemented within an existing functional ecosystem.

The following are some of the conclusions from these reviews.

“The results of evaluations of the success in the application of the Fisheries Act to prevent habitat loss in the face of development reveal a relatively low level of achievement”. (Bitwell et al. 2005)

“In almost all cases, actual compensation ratios were smaller than required compensation ratios.”
“Noncompliance with HADD (harmful alteration, disruption or destruction of fish habitat) and compensation areas contributed to substantial losses of habitat. The prevalence and magnitude of larger HADD areas and smaller compensatory works far exceeded the gains in fish habitat due to authorisations with smaller HADD areas or larger compensation. Habitat loss as a result of improperly installed or designed compensatory structures (e.g., perched culverts, impassable weirs, dry channels) was also considerable. In many cases, these habitat losses exceeded the original HADD that necessitated the compensation habitat.” (Quigley and Harper 2006a)

“We found that although success improved with artificial [compensation] ratios of 2:1, a substantial proportion of compensation projects still did not achieve NNL” (Quigley and Harper 2006b)

“Compensation science and institutional approaches need to improve in Canada if the conservation policy of NNL of habitat productivity is to be met, as evidenced by the compensation projects assessed in this study, of which only 37% achieved this goal.” (Quigley and Harper 2006b)

“….compensation ratios are dramatically reduced in projects with HADD areas greater than 30,000 m$^2$ [3 ha], suggesting that the ability to apply ‘No Net Loss’ is very difficult in large projects.” (Lange et al 2001)

In the Pacific Region, almost a quarter of examined compensation projects were considered to be poor or failures. (Lange et al. 2001)

The above quotes speak to compensation projects in general, however Hartman and Miles (2001) provide some specific comments on a variety of types of habitat improvements or creation for rainbow trout. One might expect that, given the extensive body of research on rainbow trout, compensation projects would typically be highly successful. Unfortunately, this was not the case for several of the different classes of projects that were reviewed.

The authors reviewed the success of six lakes created for rainbow trout habitat. While habitat managers reported that one of the projects had limited success, three had moderate success and two were outstanding successes, the authors provided several important qualifications to the reported “successes”. The definition of success used by the managers varied greatly and included “put and take” fisheries that were not self sustaining, lakes that needed ongoing interventions at spawning areas, and a lake (Trojan Pond at Highland Valley) with fish too contaminated with copper to eat. The findings were further qualified by the authors finding that there was insufficient information about long-term viability or productivity of the trout populations to project the qualified successes into the future.

Taseko’s proposed fish habitat compensation plan is dependent on the performance of a number of components functioning in concert, including the success of an artificial spawning channel.
Hartman and Miles (2001) had this to say about the success rate of creating spawning channels for rainbow trout.

“Our analyses indicate that less than half of the spawning or spawning/rearing channel projects were successful. In addition, many successful sites required either pumps to provide a water supply or regular maintenance to ensure gravel quality. For this reason, many of the constructed channels are not self-sustaining and on-going maintenance funding is required.”

and

“The documented success rates for spawning/rearing channels and ecologically effective diversion channels are a concern as most projects have been recently constructed. Longer term success rates are therefore expected to be even lower than the documented 38% to 43% and additional research will be required to determine and quantify the factors which contributed to failure or success.”

Increased Complexity Leads to Increased Risks and Uncertainties

What Taseko is proposing is not a simple single-component compensation plan that is typical of the approved fish habitat compensation projects included in the above reviews. What is being proposed here is the reconstruction of an entire aquatic ecosystem, complete with water delivery systems, retention ponds for control, spawning habitat, a new lake for over-wintering and supporting fish populations, tributary fish-bearing streams, a variety of fish-supporting riparian habitats, wetlands, and so forth. As such, the proposed compensation plan is larger in scale and more complex than the majority of compensation plans that have been approved in Canada. In fact, after reviewing a key documents on large scale fish habitat compensation (Bitwell et al. 2005, Hartman and Miles 2001, Lange et al. 2001, Packman et al. 2006) we have found no Canadian example of a successfully implemented plan to compensate for such a large area of highly valued, productive and complex freshwater ecosystems as the current proposal by Taseko. Other major compensation projects have not attempted the whole-scale re-establishment of self-contained aquatic ecosystems, particularly those with a flourishing sustainable (and exploitable) fishery.

“The behaviour of natural systems is characterized by complexity and uncertainty” (Hartman and Miles 2001) and as shown in Figure 1 the degree and uncertainty and likelihood of failure increase as the scale and complexity of a habitat compensation plan increases (Birtwell et al. 2005, Hartman and Miles 2001, Lange 2001).
Figure 1. Increasing degrees of risk and uncertainty at increasing scales of habitat compensation (after Bitwell et al. 2005)

Unlike most habitat compensation activities that address smaller areas of specific fish habitat, this project contemplates the destruction of 117.6 ha of lake habitats and 6.4 ha of stream habitats plus adjacent riparian areas. The watershed currently function as an integrated unit that provides multiple services and among many other resource values, the maintenance of a productive rainbow trout population. Replacing the multiple functions of the watershed (even just from a fish perspective) requires the layering of many different ecological features and functions including supply of water with appropriate chemical and thermal characteristics, spawning beds, a food supply and appropriate shelter for various life stages. If any one of these functions fails, the entire compensation project fails.

While the proponent believes there is a high degree of certainty about the effectiveness of its proposal, the evidence suggest otherwise. Trying to recreate the interconnected functionality of the lakes, and streams of the Fish Lake watershed takes this proposal well into a high degree of high speculation and overwhelming odds against success.

The attached report Why Bartering Biodiversity Fails casts a worrisome doubt over the prospects of success for any compensation project, regardless of how robust the compensation plan is. This is not only because of ecological failure from design deficiencies, but because of biased institutional behaviour and political forces that ultimately favour development over protection goals when biodiversity is on the bartering block. The abstract says it all:

“Viable trading requires simple, measurable, and interchangeable commodities, but the currencies, restrictions, and oversight needed to protect complex, difficult-to-measure, and noninterchangeable resources like biodiversity are costly and intractable. These safeguards compromise trading viability and benefit neither traders nor regulatory officials. Political theory predicts that (1) biodiversity protection interests will fail to counter motivations for officials to resist and relax safeguards to facilitate exchanges and resource development at cost to biodiversity, and (2) trading is more vulnerable than pure administrative mechanisms to institutional dynamics that undermine environmental protection. Delivery of no net loss or net gain through biodiversity trading is thus administratively improbable and technically unrealistic. Their proliferation without
credible solutions suggests biodiversity offset programs are successful “symbolic policies,” potentially obscuring biodiversity loss and dissipating impetus for action.” (Walker et al. 2009)

The Problem of Monitoring and Enforcement

The next issue relating to ultimate success of a compensation project that needs examination is the issue of follow-up by the regulator. If this project were to be approved and a fish habitat compensation plan were to be implemented, how effective is the monitoring and enforcement of the conditions of the plan likely to be? Will there be a viable mechanism in place to track and document the outcome, and to take action if compliance is not being achieved? Sadly, the answer is probably not.

The authors of the reviews cited above noted the limited and insufficient nature of monitoring and enforcement as a key contributor to the lack of success in achieving No Net Loss and in their inability to adequately determine the success of compensation projects.

The reasons for these difficulties are made clear in the Commissioner of the Environment and Sustainable Development’s (CESD) 2009 audit of DFO habitat management. The results are profoundly disturbing since they imply that it is unlikely that the federal government will be able to determine whether or not this project is implemented as approved and whether the approved measures were adequate. Conclusions of the audit included:

- DFO has little of the documentation required by departmental policy that is necessary to track and evaluate impacts on fish habitat and compensation projects.

- While proponents are normally required to carry out project monitoring activities, and the Department may monitor projects directly or rely on monitoring by the proponent, the CESD audit found that the Department does not have a risk-based approach to monitoring proponents’ compliance with the terms and conditions of ministerial authorizations and letters of advice. It found that proponents had carried out the required monitoring in only 6 of 16 (38 percent) sample items involving ministerial authorizations, and 1 of 30 sample items involving letters of advice. Further, the Department directly monitored the proponent’s compliance in only one of the cases reviewed, but there was no documentation found to show that the Department had followed up or evaluated the effectiveness of its decisions—that is, whether implementing the conditions of the ministerial authorizations or letters of advice had resulted in no net loss of habitat.

- The Department does not have a systematic approach to monitoring proponents’ compliance with the conditions of its project approvals. Nor does it evaluate whether its decisions on mitigating measures and compensation are effective in meeting the no net loss principle. As a result, projects may be causing damage to habitat beyond the amount authorized, and mitigating measures and compensation may not be effective.
• The audit could not determine whether the Department is following the Compliance and Enforcement Policy. There was no evidence of what, if any, actions the Department had taken to inspect or investigate alleged violations or what enforcement actions it had taken.

• The Department’s ongoing challenges in collecting data and selecting habitat indicators means that it still does not know whether it is progressing toward the Habitat Policy’s long-term objective of a net gain in fish habitat.

One of the explanations for the poor performance of DFO in monitoring and enforcement is an insufficient and decreasing amount of dedicated resources. The CESD audit found that:

• Between 2003 and 2007 Conservation and Protection fishery officers time spent on habitat-related enforcement matters decreased from 6.4 percent to 3.3 percent of total time.

• The total number of full-time equivalent staff in the Habitat Management Program had decreased from 460 to 430.

With the deficiencies identified by the CESD it is worrying that the current spending estimates for Habitat Management Program identify a further 22% reduction of planned spending from 2009-2010 to 2012-2013. (DFO 2010). This risks exacerbating existing problems.

Conclusion

Given the poor track record of relatively simple compensation projects, the scale and complexity of the proposed compensation project and the risks and uncertainty that these create, and the poor performance of DFO in monitoring and enforcing approvals, we are left with little confidence that the proposed plan can be successful – even if it were conceptually sound – which it is not. Without a viable mitigation for the extensive negative effects that would occur to the aquatic ecosystems of the Fish Lake Watershed, the proposal for the Prosperity gold-copper mine can not be considered sustainable or to provide a net ecological benefit. For this reason and those we have raised in earlier submissions, we once again urge the panel to recommend that the project not be approved.
References


Why bartering biodiversity fails
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Abstract
Regulatory biodiversity trading (or biodiversity “offsets”) is increasingly promoted as a way to enable both conservation and development while achieving “no net loss” or even “net gain” in biodiversity, but to date has facilitated development while perpetuating biodiversity loss. Ecologists seeking improved biodiversity outcomes are developing better assessment tools and recommending more rigorous restrictions and enforcement. We explain why such recommendations overlook and cannot correct key causes of failure to protect biodiversity. Viable trading requires simple, measurable, and interchangeable commodities, but the currencies, restrictions, and oversight needed to protect complex, difficult-to-measure, and noninterchangeable resources like biodiversity are costly and intractable. These safeguards compromise trading viability and benefit neither traders nor regulatory officials. Political theory predicts that (1) biodiversity protection interests will fail to counter motivations for officials to resist and relax safeguards to facilitate exchanges and resource development at cost to biodiversity, and (2) trading is more vulnerable than pure administrative mechanisms to institutional dynamics that undermine environmental protection. Delivery of no net loss or net gain through biodiversity trading is thus administratively improbable and technically unrealistic. Their proliferation without credible solutions suggests biodiversity offset programs are successful “symbolic policies,” potentially obscuring biodiversity loss and dissipating impetus for action.

Introduction
Biodiversity trading programs (which include biodiversity compensation, offsets, banking, and biobanking) have proliferated internationally, and are promoted by policy makers and developers as facilitating both conservation and development. Like programs developed for simpler environmental commodities such as air pollutants (Pedersen 1994), most biodiversity trading has a regulatory or statutory basis that prohibits an activity (e.g., indigenous vegetation clearance, species habitat destruction, filling of wetlands) and later permits it conditionally (Salzman & Ruhl 2000).

As a regulatory incentive mechanism (Figure 1), environmental trading relies on developers’ self-interest and resources in addition to administrative enforcement (Gustafsson 1998:268). Compared with pure administrative mechanisms (e.g., rules, standards), such market mechanisms are often purported to (1) allocate natural resources more efficiently, (2) satisfy developers better (increase access to resources, reduce compliance costs, and/or enhance green credentials; ten Kate et al. 2004), and (3) provide improved environmental protection (see Gustafsson 1998; Kroeger & Casey 2007). In trading biodiversity, some programs aim to reduce rates of biodiversity loss (e.g., Lueck & Michael 2003; Chomitz 2004). Others, perhaps increasingly, propose to achieve no net loss or a net gain in biodiversity (e.g., WHOEP 1993; VDNRE 2002; WA EPA 2006).

So far, evaluations suggest that biodiversity trading has not produced its promised biodiversity outcomes. Typically, development proceeds while offsets fall short...
Inadequate biodiversity currencies

The test of a currency’s adequacy is “...can [it] capture the significant values exchanged or do some important features remain external to the trades?” (Salzman & Ruhl 2000:614, Table 1). Simple environmental goods are easiest to commodify in currency: for example, a kilogram of sulfur dioxide provides a simple, relatively measurable, and adequate exchange currency for a unit of air pollution. But for biodiversity, there is no simple currency that adequately “…capture[s] what we care about” (Salzman & Ruhl 2000:623) (see also Robertson 2000). Biodiversity—the variety of living organisms—is hierarchical, with levels of organization from genes to ecosystems, an extraordinary number of elements at each level that vary in time and space, and diverse interactions within and between levels (e.g., Gaston 2000). Such complexity makes it exceptionally difficult to measure biodiversity, and to estimate an element’s contribution to the whole.

Furthermore, if “what we care about” is persistence of the full variety of life, contributions of different biodiversity elements are noninterchangeable. This noninterchangeability can be conceived of in three different dimensions (Salzman & Ruhl 2000): type (e.g., endangered frog habitat is neither equivalent to nor exchangeable for endangered tree habitat; captive-bred subpopulations do not replicate a diverse population gene pool); space (e.g., isolated and contiguous habitat patches are not equivalent); and time (e.g., genetic bottlenecks alter population characteristics irreversibly; early and late seral stages of an ecosystem type support different species suites).
Incomplete measurement, imprecise valuation, and noninterchangeability mean biodiversity exchange is strictly not commodity trading, but barter: “individuals haggling over goods and services with unique attributes” (Salzman & Ruhl 2000:614). But unlike barter in private goods, exchanges in environmental goods affect interests beyond direct participants; trading can erode the public’s interest in public resources (Gustafsson 1998; Salzman & Ruhl 2000; Kroeger & Casey 2007). Unavoidably, simple biodiversity currencies are inadequate; they facilitate nominal biodiversity accounting, but omit, obscure, or conceal biodiversity features and noninterchangeabilities (Robertson 2000; Salzman & Ruhl 2000; e.g., see Stein et al. 2000; McCann et al. 2004; Fox & Nino-Murcia 2005). And in any exchange, a characteristic not counted is protected only by chance, which facilitates its loss. Simple currencies simultaneously enable poor accountability for biodiversity outcomes and provide opportunity for damage to biodiversity, bringing a need for restrictions on exchanges if public interests are to be protected (Salzman & Ruhl 2000).

Exchange restrictions to compensate for currency inadequacy

The literature describes many restrictions on biodiversity exchange intended to compensate for currency inadequacies in the three noninterchangeability dimensions. In each case, a test of adequacy asks: “is this restriction adequate to ensure against biodiversity loss?” (Table 1).

(1) Type. Exchanges of dissimilar biodiversity risk loss of biodiversity components and functions (Salzman & Ruhl 2000; ten Kate et al. 2004). To counter this problem, some trading programs propose no-go areas to prohibit trading of critical assets (e.g., WA EPA 2006) but permit exchanges of noncritical biodiversity. Others limit exchanges to the same species, communities, or ecosystem type (e.g., VDNRE 2002; Brownlie et al. 2007), relying on simplified biodiversity classification tools. Some suggest out-of-kind exchanges (“like for like or better” or “trading up”; ten Kate et al. 2004:61; WA EPA 2006:10; Brownlie et al. 2007:6) might offer greater value if affected biodiversity is secure and more imperilled biodiversity is protected, although credible guidelines based on measures of complementarity (Justus & Sarkar 2002) have been slow to emerge.

(2) Space. The location of individuals, populations, and communities profoundly influences ecological interactions and biodiversity persistence (Hanski 1998); and ecosystems in different locations serve dissimilar functions (e.g., Mitsch 1998). To maintain biodiversity, exchanges must replace ecological interactions and functions lost in development, and restoration projects must not displace other natural ecosystems. Yet quantifying spatial dependency is data demanding, even for single species (e.g., Ovaskainen & Hanski 2004), and adverse effects of spatial displacement are poorly recognized and
rarely remedied in biodiversity trading. Some programs use a rule-of-thumb preference for nearby replacements over distant ones (ten Kate et al. 2004). Others restrict trades to within geographic zones (e.g., wetland service areas; Salzman & Ruhl 2000), or concentrate replacements in aggregated sites, intending to overcome fragmentation (e.g., Fox & Nino-Murcia 2005). Still others apparently ignore the problem (see Burgin 2008).

(3) Time. Development is usually permanent, life cycles of companies are finite, and ecosystem reconstruction seldom, if ever, succeeds in structure, composition, or function (e.g., Zedler & Callaway 1999; Hilderbrand et al. 2005; Quigley & Harper 2005a, b; Morris et al. 2006; Gibbons & Lindenmayer 2007; Matthews & Endress 2008). Even temporary losses may permanently damage populations and engender or aggravate cumulative effects. To provide certainty that development will not cause biodiversity loss, new, equivalent habitat must be created before existing habitat is destroyed or modified (Veltman 1995; Crooks & Ledoux 2002). This would restrict exchanges to a few, simple, predictable, quickly maturing ecosystem types (Morris et al. 2006). In biodiversity trading practice, time noninterchangeability is dealt with in three ways. First, permanent drawdown trading overlooks it, and exchanges destruction of existing ecosystems or species habitats for improved protection of other, existing ecosystems or habitats (as in USA’s conservation banking) (Fox & Nino-Murcia 2005; Carroll et al. 2008) and Brazil’s forest set-aside trading (Chomitz 2004)). Second, interim drawdown programs permit ecosystem or species habitat destruction before reconstruction (e.g., Australian states; VDNRE 2002; Gibbons & Lindenmayer 2007). Such programs generate immediate ecosystem or habitat loss, interrupt ecological processes (see Fig. 4 of Gibbons & Lindenmayer 2007:30), and risk permanent loss through restoration failure (Moilanen et al. 2008). Third, true banking programs nominally address time noninterchangeability by requiring biodiversity replacement before development occurs. This eliminates interim biodiversity loss and risk of restoration failure (though such requirements appear to be seldom enforced; see Salzman & Ruhl 2000; Mack & Micacchion 2006).

Further ecological problems

The above scan reveals persistent deficiencies in information and practice that facilitate net biodiversity loss through nonequivalent exchanges. Further problems span all three noninterchangeability dimensions. For example, the biodiversity data needed to inform exchange restrictions usually exceed those that governments, developers, or habitat bankers have been willing to fund. Less comprehensive data bring greater uncertainty about biodiversity characteristics and hence increase potential for biodiversity loss. Also, researchers developing exchange restrictions at project scales often overlook cumulative (often nonlinear, synergistic, and indirect) negative effects of multiple nonequivalent exchanges in type, space, or time (Bedford & Preston 1988; Quigley & Harper 2005a; Mack & Micacchion 2006; but see Brownlie et al. 2007; Ves & al. 2008). Another problem concerns ratios (or multipliers) applied to compensate for noninterchangeability in type, space, or time. Some have a statistical or ecological basis. For example, high offset ratios are needed to avoid risk of unfavorable biodiversity outcomes when restoration effectiveness is uncertain, failure is correlated among sites, or restoration is delayed (Moilanen et al. 2008). Brownlie et al. (2007) recommend multipliers to protect specified minimum areas, addressing the question “what ratio will achieve the biodiversity outcome sought?”. Elsewhere, the basis for multipliers seems unsound: providing several times something different cannot replace a lost species or unique ecosystem; restoring something to higher abundance later may not compensate for consequences of a loss now. Similarly, financial insurance can neither restore the unrestorable nor remedy permanent loss.

Oversight of biodiversity barter

The currency and exchange inadequacies that beset biodiversity barter place a heavy burden on precautionary oversight (a review mechanism) to control exchanges sufficiently to protect biodiversity. Salzman & Ruhl (2000) suggest adequate oversight should ensure meaningful valuation of the public goods exchanged and fair apportioning of costs and risks, and counteract the agencies’ and trading parties’ incentives to transact bad deals (Table 1). Time and again, researchers report procedural and enforcement failures in biodiversity trading programs, and urge improvement, through more or better frameworks, resourcing, or insurance (e.g., Gibbons & Lindenmayer 2007; Matthews & Endress 2008; Norton 2008). But these suggestions do not address the political and administrative causes of inadequate review.

Administrative problems

Salzman & Ruhl (2000) observed an administrative playing field of biodiversity barter tilted toward development. We propose that classic theories of politics predict this tilt, and that biodiversity’s poor measurability and noninterchangeability exacerbate it. Together, political, and
ecological factors create two fundamental problems for public administration of biodiversity barter:

(1) Thin markets. For a viable trading program to operate in practice, currencies must be simple, review cannot be onerous, and restrictions must be straightforward and few (Pedersen 1994; Salzman & Ruhl 2000). But to protect biodiversity, high-quality data must inform precautionary exchange restrictions. Such restrictions create transaction costs and allow few exchanges, constraining an otherwise well-supplied trading market (Salzman & Ruhl 2000; see e.g., Chomitz 2004).

(2) Inequality, divergence, and coincidence of interests. Precautionary exchange is also unlikely because of the unequal power and different goals of participants. This is foreseen by the public choice theory of politics, which predicts that rational actors act in their own self-interest, and that some actors are more powerful than others (e.g., see McCubbins et al. 1987; Eskridge 1988). Specifically, the motivated few will be more powerful than the disorganized many (Olson 1965); so public choice theory predicts private interests—such as developers—will often defeat public interests—such as biodiversity protection—and reap most policy benefits. As Eskridge (1988:294) observed, “[t]he legislative market is one that works badly. The public goods that government ought to be providing . . . are seldom passed by the legislature, because demand for them is usually not strong and legislators gain too little from sponsoring them . . . Conversely, rent-seeking statutes—primarily, concentrated benefit, distributed cost measures—seem inevitable.”

Three interests compete in biodiversity barter:

(a) Traders (developers and restoration/offset providers) have a financial, or vested, interest in obtaining permits to conduct business. Such traders in environmental goods need not be conscious of the quality of environmental outcomes if a permit is forthcoming (Gustafson 1998; Floumoy 2000; Salzman & Ruhl 2000; Kroeger & Casey 2007). This encourages developers seeking permits to underestimate (perhaps unintentionally) environmental impacts, and restoration providers to exaggerate (maybe unwittingly) the value of biodiversity goods offered in exchange. Neither trader profits from investment in data to support independent assessment, robust exchange restrictions, and meaningful review. Instead, they benefit from simple currencies that are inexpensive to measure, plentiful trading options with few exchange restrictions, and limited review to minimize risk that a permit will be overturned.

(b) Biodiversity protection interests usually have no vested interest in biodiversity barter. They benefit from exchanges that are fully measured, exchange restrictions that are robust and upheld, and review mechanisms that are meaningful and effective in protecting biodiversity.

(c) Regulatory officials are those appointed to enforce trading conditions, and are both referee and representative of the public’s interest in biodiversity. Because traders have little incentive to control quality, officials shoulder the full burden of enforcement. But officials are not disinterested “billiard balls,” faithfully implementing democratically determined rules (Wilson 1989:88). Without inferring corruption or malfeasance, public choice theory predicts that officials often have motivations that are different from their statutory mandates, and that, given freedom to choose, officials will often pursue their own self interest (e.g., Niskanen 1971; McCubbins et al. 1987; O’Toole 1988). In environmental regulation, incentives on officials often coincide more strongly with development than environmental interests: Winter (1985) even suggests that governments rarely fund full enforcement, and sometimes directly discourage officials from frustrating powerful vested interests. Therefore, officials can and sometimes do reduce their financial or political costs by offering development interests more palatable and less environmentally demanding options (Winter 1985; Salzman & Ruhl 2000:648–665; Brower 2008:20–22; 84–108). Simple inexpensive biodiversity currencies, weak or ambiguous exchange restrictions, and limited review benefit both officials and traders because they are cheap and offer flexibility, or utility (see Pedersen 1994; Parkes et al. 2004). Coincidentally, they also facilitate development at the expense of biodiversity.

The playing field on which these interests compete is far from level; the “default setting” (Brower 2008:14) predicted by Olson (1965) is that development will defeat biodiversity. To address biodiversity decline, policy instruments must level this playing field. But theory predicts biodiversity barter will reinforce, rather than correct, this default setting.

First, mandates to barter biodiversity weaken existing statutory constraints on biodiversity harm by allowing officials discretion to circumvent them; for example, the Habitat Conservation Plan provision of the USA’s Endangered Species Act erodes its absolute prohibition on
species take (Ruhl 1999). Even in situations of routine noncompliance, legitimizing barter may produce worse environmental outcomes than policy regimes in which officials barter with developers “outside the shadow of the law” (Ellickson 1991:52), but the existence of a clear statute constrains their bartering leeway (see Winter 1985:240). More generally, in giving officials discretion to work toward unspecified outcomes, barter increases opportunity for officials already motivated to “skip rather lightly past avoidance and minimization and proceed instead directly to compensation” (Bean & Dwyer 2000:10537), while reducing public power to specify rules and goals through democratic processes (see Salzman & Ruhl 2000:683).

Second, the case-by-case decision making inherent in biodiversity barter reinforces dominance of vested development interests by constraining the effectiveness of biodiversity protection interests. Case-by-case decision making keeps biodiversity loss off the national radar and limits its importance, hence weakening the environmental voice (see Schattschneider 1960; Pralle 2006). It is more costly and less feasible for environmental interests to marshal the resources to challenge proposals case-by-case than through high-level orchestrated campaigns (Brower 2008:57).

Third, problematic measurement and case-by-case barter each render biodiversity trading especially vulnerable to information asymmetry—the situation in which insiders (traders and officials) know more than outsiders (biodiversity protection interests and the public), who are unable to measure the quality of biodiversity deals. Information asymmetry creates slack, or “a zone of freedom of action for regulators...in which they can operate with lessened fear of punishment by the polity for decisions that deviate from those the polity would adopt on its own” (Levine 1998:269). When officials’ and developers’ interests coincide in negotiating permits, a pattern of informal and less-than-transparent deals can result (Winter 1985; Freeman 2000; Brower 2008) with norms of behavior and standards of fairness that benefit insiders, but deviate from statutes and ideas of fairness held to protect outsiders—the public (Ellickson 1991). Thus, information asymmetry will systematically favor development over protection.

No net loss as symbolic policy

Absence of opportunity for public input in case-by-case decisions often renders ecological scientists the most vocal critics of biodiversity trading. But scientists appear reluctant to abandon hope that biodiversity offsets might yet deliver no net loss (see Gibbons & Lindenmayer 2007; Burgin 2008; Norton 2008). We see compelling reasons for skepticism.

Some ecologists insist biodiversity barter could achieve no net loss—if only there were better currencies, informed exchange restrictions, and attention to review (e.g., ten Kate et al. 2004; Gibbons & Lindenmayer 2007). They assume that if improved information and measures were available, and rules were clear and transparently defensible on ecological grounds, officials would use and implement them. Empirical evidence shows that officials have repeatedly failed to do so (e.g., Salzman & Ruhl 2000; Fox & Nino-Murcia 2005; Burgin 2008); and public choice theory predicts this failure. Others might see opportunities to leverage funds for improved biodiversity data and measurement; developers, agencies, and governments are likely to resist this. Those recognizing the primacy of administrative problems posit carefully designed review might counter motivations of traders and officials (Salzman & Ruhl 2000:693). But this would constrain exchanges to the detriment of developers and officials, and no such review institution has emerged. In the absence of credible solutions to these problems, biodiversity trading is likely to continue to facilitate development at the expense of biodiversity.

In addition, biodiversity exchange has potential to postpone social and legislative changes needed to address the basic problem of biodiversity loss (see Pedersen 1994; Gustafsson 1998:271). We see two reasons. First, bartering focuses parties’ attention on immediate steps, rather than stimulating them to proceed “according to some larger progressive principle” (Winter 1985:246). This resembles displacement behavior in which “organizational means become transformed into ends-in-themselves and displace the principal goals of the organization” (Merton 1957). Conservation programs with a preference for near-term, achievable, procedural goals can deflect attention from long-term, more difficult goals for ecological outcomes (Brower et al. 2001).

Second, no net loss and net gain slogans themselves may be effective political diversions. We have argued that achieving no net biodiversity loss through barter is an illusion that crumbles under scrutiny from ecological and political science. But Edelman (1960, 1964) suggests that some policies are never intended by politicians to be more than hollow promises. Such symbolic policies promise much but guarantee little, and allow the motivated few to reap most of a policy’s benefits while leaving the disorganized many unaware, or lulled into “political quiescence” (Edelman 1964). No environmentalist will disagree with the goal of no net biodiversity loss. In attaching the slogan “no net loss” to biodiversity barter, politicians can appear to take action while continuing to serve development interests, and ignoring or perhaps
Exacerbating biodiversity loss. In engaging ecologists’ collaboration in a symbolic but illusory goal, biodiversity barter may succeed by “keeping friends close and enemies closer” (Brower 2008:58) thus defusing potential opposition (Robertson 2000). Developers, politicians, and officials embrace biodiversity barter under “no net loss” or “net gain” flags (Robertson 2000; Salzman & Ruhl 2000; Burgin 2008) because it benefits them to do so. Support from ecological scientists, whether tacit or active, sustains and authenticates the illusion.

**Conclusions**

Viable biodiversity barter and meaningful biodiversity protection seem mutually exclusive. We can achieve one or the other, but not both. Although compensation and no net loss are laudable ideals, ecological and political problems appear intractable, and mean that bartering is likely to accomplish more harm than good for biodiversity.

Ecological and political factors combine in bartering biodiversity to produce currencies, exchange restrictions, and oversight that are inadequate to protect biodiversity. Because biodiversity is complex and its elements non-interchangeable, there is no simple currency to measure fairness of exchange, and restrictive exchange rules and robust review institutions are needed to protect it. But a functioning exchange program requires simple currencies, few restrictions, and undemanding review. This gulf between market and ecological viability seems to render biodiversity trading doomed to fail—more specifically, to fail biodiversity. Indeed, the simplistic currencies, lax exchange restrictions and inadequate review that benefit both traders and officials are predicted by political theory and observed in practice. All come at a cost to biodiversity.

We further conclude that inequalities, divergence, and coincidence among interests in biodiversity barter mean that improved biodiversity measures and exchange restrictions recommended by ecologists will rarely be adopted. Few academics and practitioners have understood and tried to address these non-ecological causes of failure (Salzman & Ruhl 2000:693).

The administrative playing field described in this article shapes the outcomes of not only biodiversity trading, but also all environmental policy. However, political theory predicts that biodiversity exchange policies—because of biodiversity’s complexity, poor measurability, and non-interchangeability—will be more vulnerable to the institutional failings that undermine environmental protection than simple (albeit imperfectly enforced) prohibitions. Public choice theory suggests officials and traders have more incentive to facilitate barter than to ensure biodiversity protection. Thus, given the option of saying to developers “yes, with conditions” rather than “no,” officials will prefer “yes, with conditions”—particularly when compliance with conditions cannot be credibly measured and officials can avoid accountability for outcomes. Legitimized bartering can thus create a policy situation “obscure enough to please all parties . . . and so ill-defined that failures . . . will be difficult to detect and impossible to litigate” (Walker et al. 2008:226; see also Winter 1985).

Furthermore, recent proliferation of offset programs, with the promise of no net loss or net gain, is consistent with effective use of symbolic policy to “give the rhetoric to one side and the decision to the other” (Edelman 1960:703). Symbolic policy may cost conservation by obscuring biodiversity loss and dissipating impetus for social activism and forthright conservation planning.

In sum, while compensation and no net loss are worthy goals, and bartering biodiversity might appear more promising than simple and weakly enforced prohibitions, this article suggests policies that enable biodiversity trading may perversely yield worse biodiversity outcomes. All theoretical predictions point to further biodiversity loss paving the way for development in any biodiversity trading program, while a no net loss tag-line defuses potential opposition and impetus for change.

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**Supporting Information**

Additional Supporting Information may be found in the online version of this article:

**Appendix S1:** Major regulatory biodiversity trading programs and references given in the manuscript.

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RESEARCH
Compliance with Canada’s Fisheries Act: A Field Audit of Habitat Compensation Projects

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ABSTRACT / Loss of fish habitat in North America has occurred at an unprecedented rate through the last century. In response, the Canadian Parliament enacted the habitat provisions of the Fisheries Act. Under these provisions, a “harmful alteration, disruption, or destruction to fish habitat” (HADD) cannot occur unless authorised by Fisheries and Oceans Canada (DFO), with legally binding compensatory habitat to offset the HADD. The guiding principle to DFO’s conservation goal is “no net loss of the productive capacity of fish habitats” (NNL). However, performance in achieving NNL has never been evaluated on a national scale. We investigated 52 habitat compensation projects across Canada to determine compliance with physical, biological, and chemical requirements of Section 35(2) Fisheries Act authorisations. Biological requirements had the lowest compliance (58%) and chemical requirements the highest (100%). Compliance with biological requirements differed among habitat categories and was poorest (19% compliance) in riparian habitats. Approximately 86% of authorisations had larger HADD and/or smaller compensation areas than authorised. The largest noncompliance in terms of habitat area occurred in riverine habitat in which HADDs were, on average, 343% larger than initially authorised. In total, 67% of compensation projects resulted in net losses of habitat area, 2% resulted in no net loss, and 31% achieved a net gain in habitat area. Interestingly, probable violations of the Fisheries Act were prevalent at half of the projects. Analyses indicated that the frequency of probable Fisheries Act violations differed among provinces. Habitat compensation to achieve NNL, as currently implemented in Canada, is at best only slowing the rate of habitat loss. In all likelihood, increasing the amount of authorised compensatory habitat in the absence of institutional changes will not reverse this trend. Improvements in monitoring and enforcement are necessary to move towards achieving Canada’s conservation goals.

Loss of fish habitat, a leading factor in the decline of Canada’s fisheries resources (Beamish and others 1986; Pearse 1988), has occurred at an unprecedented rate through the last century. For example, in the world’s premiere salmon-producing watershed, the Fraser River (Levy 1992), approximately 90% of the fish habitat in the lower watershed has been lost during the 20th century (Levings and Nishimura 1996). Human population growth, and the concomitant increase in landscape development, is likely a key factor in this downward trajectory (Lackey 2001). Indeed, there is a striking negative relationship between wild salmon populations and human population density (Hartman and others 2000).

In response to this habitat loss, the Canadian Parliament enacted the habitat provisions of the Fisheries Act in 1976, which effectively made the Fisheries Act one of the strongest pieces of environmental legislation in Canada. A “harmful alteration, disruption, or destruction to fish habitat” (HADD) cannot occur unless authorised via Section 35(2) of the Fisheries Act by Fisheries and Oceans Canada (DFO) (see Goodchild 2004 for detailed overview). Implementation of Section 35(2) of the Fisheries Act is guided by the Policy for the Management of Fish Habitat (DFO 1986) (hereafter the Habitat Policy), the cornerstone of DFO’s habitat management program, which states that the guiding principle to DFO’s conservation goal is “no net loss of the productive capacity of fish habitats” (NNL). If a proposed development project (e.g., mine development, highway construction, etc.) is deemed to result in a HADD after project relocation and redesign have

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been considered, DFO will only issue a *Fisheries Act* authorisation if NNL can be achieved through the construction of compensatory habitat to offset the residual impacts from the development project. The amount and type of compensation habitat required is guided by national policy that recommends a minimum compensation ratio of 1:1 and a hierarchy of compensation options, with like-for-like habitat at the site as the most preferred option (DFO 1986, 2002a). An authorisation legally binds the proponent to design, develop, monitor, and maintain the compensation habitat.

Systematic, quantitative, and independent evaluations of DFO’s performance in achieving NNL through compensation habitat have rarely been undertaken. In fact, since the inception of the Habitat Policy, more than 2500 authorisations have been issued yet only 103 compensation projects have been evaluated to determine their performance in achieving NNL (Harper and Quigley 2005a). In a national study, Drodge and others (1999) documented that DFO’s habitat management staff allocate 1.7% of their workload to compliance monitoring. Consequently, the long-term success rates and efficacy of fish habitat compensation projects are not well known (DFO 1997; Lister and Bengseyfield 1998; Lange and others 2001).

The few evaluations of compensation habitat that have occurred have indicated that proponent compliance with the authorisation’s requirements has been poor (Harper and Quigley 2005a). Postconstruction monitoring is a legally binding requirement of proponents in virtually all fish habitat compensation projects, yet it is completed less than 43% of the time (Harper and Quigley 2005b). Even when completed, monitoring requirements are generally superficial, resulting in qualitative reports that simply document that compensation was completed (e.g., photographic record) (Goodchild 2004), rather than provide measurable indices of whether the compensation project achieved NNL. Compliance with authorisation requirements cannot be gleaned from monitoring programs in many instances (Harper and Quigley 2005b). Moreover, bias can be introduced into monitoring conducted to evaluate the success of compensation projects because self-assessments are often completed by proponents with an invested interest in the outcome, and in many cases by the same individuals involved in the design. Such a systemic lack of effective monitoring has likely constrained DFO’s ability to adaptively manage its habitat management program.

The importance of an ongoing evaluation of DFO’s performance in achieving the objectives of the Habitat Policy has been recognised by the Auditor General of Canada (Government of Canada 1997), the Standing Committee on Public Accounts (Government of Canada 1998), and DFO itself (Lange and others 2001). It is critical for regulatory agencies to conduct evaluations of the performance of such core policies in order to build on their successes and learn from past mistakes to be able to improve habitat conservation for the future. In this article, we describe a field audit of fish habitat compensation projects across Canada that was conducted to determine compliance with physical, biological, and chemical requirements of Section 35(2) *Fisheries Act* authorisations. Specifically, our objective was to quantify the achievement of NNL in the field and investigate factors related to noncompliance in order to suggest improvements to habitat conservation practices.

**Methods**

All authorisations issued between 1994 and 1997 were collected from five provinces: British Columbia, Manitoba, Ontario, New Brunswick, and Nova Scotia. Geographic stratification into five provinces ensured a mixture of coastal and interior regions of Canada (Figure 1). A subset of the authorisations were then selected randomly from each province. Field audits were completed from May to October of 2000 and 2001; therefore, the projects had a postconstruction age range of 4 to 8 years. Authorisations were partitioned based on the type of fish habitat that had been impacted by the HADD. The habitat categories included riverine (both in-channel and off-channel fluvial habitat), standing water (marine, estuarine, and lacustrine habitat), and riparian. A hierarchy of compensation options, from most to least preferred, that compare the habitat type and ecological unit of compensation habitat relative to the impacted habitat is provided in the Habitat Policy (DFO 1986, 1998). We described compensation projects based on a modified hierarchy of preferences. This included three basic classifications: (1) like for like habitat: create similar habitat at or near the site in the same ecological unit (e.g., replace off-channel habitat with off-channel habitat); (2) like for unlike habitat: create or increase the productivity of unlike habitat in the same ecological unit (e.g., replace in-channel habitat with off-channel habitat); and (3) increasing like habitat productivity; increase the productivity of like habitat at or near the site (e.g., enhance existing in-channel habitat to compensate for in-channel habitat loss). Ecological unit was defined as “populations of organisms considered together with their physical environment and the interacting processes amongst them” (DFO 2002a).
Field Methods

The legally binding requirements in each authorisation were partitioned into the following categories. Compliance with each requirement was audited in the field.

HADD Area Requirements

Requirements in this category included the physical area (square meters) of each component of the authorised HADD (most projects specified HADD areas in multiple habitat categories). Many authorisations specified the HADD area both in writing and in a scale drawing appended for further detail. Scale drawings were digitised using AutoCad 2000i (Autodesk) to compare to written HADD areas. Preliminary review of the files indicated that, in many cases, HADD areas described in writing and in scale drawings were inconsistent, thereby providing the proponent with greater latitude in interpreting the habitat area legally allowed to be destroyed. Discrepancies in HADD areas between scale drawings and the authorisation text were recorded as authorisation contradictions. During field audits of projects with authorisation contradictions, a noncompliant score was assigned only if the project’s actual HADD area exceeded the larger of these two areas. In some cases, the HADD area was not clearly distinguishable during field audits. In these situations, the proponent was assumed to have been compliant and scored accordingly. A surveyor’s chain was used for compliance measurements.

Compensation Area Requirements

The physical area (square meters) of each component of the authorised compensatory habitat was included in this category. The compensation areas were often specified both in writing and in a scale drawing.

Figure 1. Location of compliance audits across Canada (n = 52).
AutoCad 2000i (Autodesk) was used to digitise scale drawings for comparison to written compensation areas. Preliminary review of the files indicated that compensation areas described in writing and in scale drawings were often not equivalent, thereby providing the proponent with a broader range of habitat area legally required for compensation. Discrepancies in compensation areas between scale drawings and the authorisation text were recorded as authorisation contradictions. During field audits of projects with authorisation contradictions, a noncompliant score was assigned only if the project’s actual compensation area was smaller than the lesser of these two areas. A surveyor’s chain was used for compliance measurements. Authorisations often specified a compensation ratio of habitat area to be created relative to destroyed. Field measurements allowed comparisons of authorised versus actual compensation ratios.

Construction Specifications

Construction specifications represent requirements in the authorisation that pertained to the construction and development of the HADD or compensatory works. These construction specifications indirectly influenced or mitigated impacts to fish and fish habitat. Requirements categorised as construction specifications included elevations of structures, channel gradients, culvert lengths, culvert widths, culvert characteristics (e.g., open vs. closed bottom), riparian buffer areas, and bank stabilisation specifications (e.g., size and characteristics of rip-rap). A surveyor’s level and rod, surveyor’s chain, and clinometer (Suunto PM-5) were used in accordance with their manufacturers’ instructions for compliance measurements.

Habitat Features

Requirements in this category included specifications for habitat features that functioned to directly benefit fish and fish habitat. Examples of requirements in the habitat features category include in-channel habitat complexing with boulders or large woody debris, spawning gravel addition, pool and riffle creation, weirs, baffles, and fishways. Habitat features also included channel creation such as low flow channels, diversion channels, and off-channels. Presence, absence, and characteristics (number, dimensions) of habitat features were measured according to standard methodologies (Schuett-Hames and others 1994; RIC 1997).

Mitigation

The mitigation category included requirements designed to minimise impacts to fish and fish habitat during the construction period. Mitigation requirements typically included specifications to maintain water quality and prevent introduction of deleterious substances to fish habitat (e.g., water treatment ponds, silt fences, straw bales, rock-lined ditches, etc.). Often, mitigation requirements were not possible to evaluate because we audited projects many years after the construction period. However, in some cases evidence of mitigation existed at the time of the audit (e.g., presence of sediment and erosion control structures).

Biological

Requirements in the authorisations that were biological in nature were included in this category. These requirements included fish utilisation (different age-classes and life history stages), fish biomass and densities, benthic invertebrate re-colonisation, riparian re-vegetation (percent survivorship, stem density, diversity), and fish access. Water velocity was measured with a flow meter (Halltech FP 101, Guelph, Ontario) to evaluate potential for fish passage. Fish sampling was completed by electroshocking (Smith Root 12C) with the two-pass removal method (Seber and LeCren 1967). Surber samplers were used to measure macro-invertebrates (RIC 1997). Riparian vegetation was sampled in either 1 m² and 50 m² quadrats (Koning 1999) or total stem counts within the entire compensation area.

Chemical

The chemical category included requirements that specified particular water quality parameters such as water temperature, dissolved oxygen, and suspended sediment concentration. A portable digital meter (Hanna HI9143) was used for field measurements of temperature and dissolved oxygen (DO). Water samples (1L) were taken and suspended sediment concentration was determined in the laboratory by filtration.

Compliance Scoring

Authorisation requirements were often composed of many characteristics. Each characteristic of a requirement was evaluated separately in order to determine degrees of compliance. For example, a construction specification requirement for an open bottom culvert with a width of 3 m was evaluated as three requirements (culvert presence/absence, open vs. close bottom, width). If a given requirement was met or exceeded from a habitat conservation perspective (such as if the culvert width was greater than three metres), a score of 1 (compliant) was assigned. A
noncompliant finding resulted in a score of zero. If a noncompliant finding rendered subsequent requirements unachievable (such as absence of the above-mentioned culvert), these additional requirements (open-bottom, width) were not scored. An authorisation could not be scored noncompliant twice for the same requirement. We did not attempt to determine whether a noncompliant finding was due to failure over time or because the requirement was not completed initially. Either way, if the requirement was not met at the time of the audit, a noncompliance score was assigned. The mechanism of failure did not alter compliance scoring.

Compensation ratios (compensation area:HADD area) were investigated to determine whether some habitat categories were more difficult to compensate than others. The difficulty in compensating within each habitat category was characterised as a percentage failure rate \(f\) by subtracting from 1.0 the actual compensation ratio observed in the field \((a)\) divided by the required compensation ratio in the authorisation \(r\) \((sensu\ Robb\ 2002)\):

\[
f = (1.0 - (a / r)) \times 100
\]

The ratio \(q\) necessary to overcome these failure rates and achieve the required compensation ratios was calculated by dividing the required ratio \(r\) by the failure rate \(f\) subtracted from 1.0.

\[
q = r / (1 - f)
\]

A separate category, probable violations of the *Fisheries Act*, was used to categorise new, unforeseen HADDs (Section 35(1)), or introductions of deleterious substances to fish-bearing waters (Section 36(3)) that occurred outside the scope of the authorisation. Most findings of noncompliance with the requirements of an authorisation could in essence be considered a violation of the *Fisheries Act*. Therefore, in order to distinguish between noncompliance with a requirement and additional ecological impacts beyond the scope of the authorisation, these noncompliance findings were not included in this separate category. For example, if a given authorisation required 1000 m² of off-channel compensation habitat to be created and we measured the compensatory works to be 600 m², this would be recorded once as a noncompliant finding. However, if we discovered that construction crews had deposited multiple bags of concrete in the off-channel compensatory habitat, this incident, clearly outside the scope of the requirements in the authorisation, would be recorded as a probable violation.

Data Analyses

Descriptive statistics (mean ± 1 standard error (SE)) were used to summarise and describe the compliance results. Compliance with the HADD area and compensation area requirements was most often either 100% or zero. Therefore, logistic regression analyses were used to assess relations between compliance with the HADD area and compensation area requirements and the following independent variables: compensation project age (years), HADD area \(m^2\), compensation area \(m^2\), financial security per square meter of compensation habitat \(\$/m^2\), total financial security \$(\), monitoring reports required \((n)\), authorisation contradictions \((n)\), monitoring compliance \((Yes\ or\ No)\), DFO field inspection \((Yes\ or\ No)\), development activity \(\text{roads/urban/other}\), habitat category \(\text{riverine, standing water, riparian}\), hierarchy of compensation preferences \((1, 2, 3)\), and geographic region \(\text{West = BC; Central = MN, ON; East = NB, NS}\). We used analysis of covariance (ANCOVA) to determine whether compliance with habitat features, biological requirements, construction specifications, and number of probable violations was associated with the same independent variables \(\text{(the covariates were the continuous variables and the factors were the class variables)}\) as in the logistic regression analyses. After deleting nonsignificant independent variables, either regression analyses \(\text{(for continuous variables)}\) or analysis of variance (ANOVA) \(\text{(for class variables)}\) were used in the final model. Multiple comparisons of class variables were completed using Tukey’s HSD test. For all analyses, a Shapiro–Wilks test statistic was used to test for normality, and data were visually inspected for homogeneous variances. Compliance data in the habitat features requirement category were square root transformed to minimise the effects of heterogeneous variances. Outliers \((> 3\ SD\ from\ the\ mean)\) were removed from calculation of the means \((Sokal\ and\ Rohlf\ 1981)\). All tests were considered to be significant to a \(P < 0.05\). Statistical analyses were completed using SAS statistical software, release 8.02 \((\text{SAS Institute 2001})\).

Results

A total of 52 authorisations and associated habitat compensation projects were audited across Canada in British Columbia \((n = 36)\), Manitoba \((n = 5)\), Ontario \((n = 4)\), New Brunswick \((n = 3)\), and Nova Scotia \((n = 4)\) \((\text{Figure 1})\). This sample represents approximately 42% of the total number of authorisations \((N = 124)\) issued in these provinces during 1994 to
Table 1. Mean number of requirements and the mean compliance in each requirement category per authorisation (n = 52)a

<table>
<thead>
<tr>
<th>Requirement category</th>
<th>Mean number of requirements (n ± 1 SE)</th>
<th>Mean compliance (% ± 1 SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>HADD</td>
<td>1.9 ± 0.2</td>
<td>72 ± 13</td>
</tr>
<tr>
<td>Compensation area</td>
<td>2.1 ± 0.2</td>
<td>62 ± 13</td>
</tr>
<tr>
<td>Construction specifications</td>
<td>14.2 ± 3.3</td>
<td>74 ± 7.5</td>
</tr>
<tr>
<td>Habitat features</td>
<td>19.7 ± 4.4</td>
<td>71 ± 7.9</td>
</tr>
<tr>
<td>Mitigation</td>
<td>2.0 ± 0.5</td>
<td>77 ± 22</td>
</tr>
<tr>
<td>Biological</td>
<td>12.6 ± 2.9</td>
<td>58 ± 9.4</td>
</tr>
<tr>
<td>Chemical</td>
<td>0.2 ± 0.1</td>
<td>100 ± 0</td>
</tr>
<tr>
<td>Total</td>
<td>52.8 ± 7.5</td>
<td>n/a</td>
</tr>
</tbody>
</table>

aHADD, harmful alteration, disruption, or destruction to fish habitat.

1997 inclusive (excluding Ontario). The mean age of compensation projects was 4.4 years (SE = 0.3). Authorisations were a result of the following development activities: urban development (n = 19), roads and highways (n = 18), forestry (n = 6), industrial (n = 2), agriculture (n = 2), private land (n = 2), mining (n = 2), and oil and gas (n = 1). Many authorisations included HADDs in multiple habitat categories. The frequency of HADDs in each habitat category included: riverine (n = 37), standing water (n = 10), and riparian (n = 43). The mean number of requirements per authorisation was 52.8 (Table 1).

HADD Area Requirements

Mean compliance per authorisation with the HADD area requirements was 72% (Table 1). The total and mean HADD areas were 439,971 m² and 8626.9 m², respectively. The largest mean HADD area occurred in the riparian habitat category and the smallest occurred in the riverine habitat category (Table 2). Approximately 37% (19) of authorisations had larger HADD areas than authorised, and only 7.8% of authorisations had smaller HADD areas. The mean size of the larger HADD areas was not minor. On average, larger HADD areas were 234% greater than the authorised value. This difference was particularly noticeable for larger HADD areas that occurred in the riverine habitat category which, on average, were 343% larger than authorised (Table 3). In contrast, the mean difference for authorisations with smaller HADD areas was not nearly as great as projects with larger HADD areas. The mean size of the smaller HADD areas was 35% less than the authorised HADD areas (Table 3).

Compensation Area Requirements

Mean compliance per authorisation with the compensation area requirements was 62% (Table 1). The total and mean compensation areas were 1,037,086 m² and 8741.7 m², respectively; however, one authorisation had an exceptionally large compensation area that accounted for 600,000 m². Approximately 21% of authorisations had larger compensatory areas than authorised. In contrast, 71% of authorisations had compensation areas that were smaller than authorised. Larger and smaller compensatory works were 46% and 48% different, respectively, from authorised values (Table 3).

Net Balance

In total, 86% of the 52 authorisations had either larger HADD areas and/or smaller compensatory areas than authorised. Only 24% had smaller HADD areas and/or larger compensation than authorised. Authorisations that affected riparian habitat had the greatest noncompliance in terms of area. Approximately 91% (39) of authorisations in the riparian habitat category had larger HADD areas and/or smaller compensation than authorised.

Table 2. Mean HADD area and mean net balance (compensation area minus HADD area) per authorisation in each habitat categorya

<table>
<thead>
<tr>
<th>Habitat category</th>
<th>Mean HADD (m²)</th>
<th>SE (m²)</th>
<th>Max (m²)</th>
<th>Min (m²)</th>
<th>Mean net balance (m²)</th>
<th>SE (m²)</th>
<th>n</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riverine</td>
<td>3315.1</td>
<td>1119.5</td>
<td>31,300.0</td>
<td>10</td>
<td>−839.3</td>
<td>755.3</td>
<td>34 (32)</td>
</tr>
<tr>
<td>Standing water</td>
<td>5534.0</td>
<td>2400.0</td>
<td>24,500.0</td>
<td>490</td>
<td>−145.1</td>
<td>4148.1</td>
<td>3499.7</td>
</tr>
<tr>
<td>Riparian</td>
<td>6632.2</td>
<td>1722.9</td>
<td>60,930.9</td>
<td>10</td>
<td>−3154.2</td>
<td>954.3</td>
<td>43</td>
</tr>
<tr>
<td>Total</td>
<td>8626.9</td>
<td>2990.3</td>
<td>92,250.9</td>
<td>40</td>
<td>−2103.1</td>
<td>1237.3</td>
<td>52 (50)</td>
</tr>
</tbody>
</table>

aHADD, harmful alteration, disruption, or destruction to fish habitat.

Note that two authorisations in the riverine category were removed as outliers in calculation of the mean net gain (sample size for net balance calculations indicated in parentheses).
Overall, the mean net balance of habitat area (compensatory habitat area minus HADD area) per authorisation was $-2103.1\ m^2$. Two authorisations in the riverine category, with net gains of $599,500\ m^2$ and $118,700\ m^2$, were removed as outliers from calculation of this mean. Because the mean net balance of habitat area was a much smaller number than expected, we explored the potential outcome if DFO had not been involved. If DFO had not required any habitat compensation in the authorisations we audited, the mean net balance per authorisation would have conservatively been $-8627\ m^2$, not including the subtraction of habitat gains from relocation, redesign, and mitigation resulting from the authorisation process. The largest mean net balance occurred in the standing water habitat category ($4148.1\ m^2$). As a consequence of the considerable noncompliance in both HADD and compensation areas in the riparian category, the mean net balance was smaller ($-3154.2\ m^2$) in this category than in authorisations that occurred in the riverine habitat category ($-899.3\ m^2$) (Table 2).

In total, 67% (35) of authorisations resulted in net losses of habitat area. Approximately 2% (1) resulted in no net loss and 31% (16) achieved a net gain in habitat area. In terms of habitat category, 72% of authorisations with HADDs in riparian habitat, 30% in standing water, and 49% in riverine habitat resulted in net losses of habitat area (Figure 2).

### Construction Specifications

Mean compliance per authorisation in the construction specifications category was 74% (Table 1). The most common findings of noncompliance were related to culvert characteristics (dimensions, gradient, embedment) ($n = 21$), channel characteristics (gradient, stability, elevation) ($n = 17$), rip-rap characteristics (dimensions, encroachment, diameter) ($n = 16$), riparian buffer zone attributes (width, exclusion fencing) ($n = 9$), and removal of old road fill ($n = 4$).

### Habitat Features

The mean compliance per authorisation in the habitat features category was 71% (Table 1). Common findings of noncompliance were with respect to rock weirs (absence, height, notched) ($n = 65$), organic weirs/digger logs (absence, dimensions, spacing) ($n = 35$), large woody debris (absence, dimensions) ($n = 25$), boulders (absence, diameter) ($n = 25$), and pools/riffles (absence, dimensions) ($n = 23$). Compliance with the habitat features requirement category was negatively associated with the amount of financial security retained weighted by compensation area (Table 4).

### Mitigation

Mean compliance with mitigation requirements was 77% (Table 1). The absence of sediment and erosion

---

**Table 3. Mean size (m$^2$) of larger and smaller HADD and compensation areas$^a$**

<table>
<thead>
<tr>
<th>Habitat category</th>
<th>Mean size (m$^2$)</th>
<th>Mean difference in area relative to authorisation (%)</th>
<th>N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riverine</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Larger HADD</td>
<td>34,586.8</td>
<td>343</td>
<td>37</td>
</tr>
<tr>
<td>Smaller HADD</td>
<td>670.9</td>
<td>33</td>
<td>8</td>
</tr>
<tr>
<td>Larger compensation</td>
<td>350.8</td>
<td>101</td>
<td>9</td>
</tr>
<tr>
<td>Smaller compensation</td>
<td>1818.7</td>
<td>50</td>
<td>15</td>
</tr>
<tr>
<td>Standing water</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Larger HADD</td>
<td>3080.8</td>
<td>2934</td>
<td>1</td>
</tr>
<tr>
<td>Smaller HADD</td>
<td>904</td>
<td>16</td>
<td>1</td>
</tr>
<tr>
<td>Larger compensation</td>
<td>972.7</td>
<td>63</td>
<td>5</td>
</tr>
<tr>
<td>Smaller compensation</td>
<td>5634.2</td>
<td>22</td>
<td>5</td>
</tr>
<tr>
<td>Riparian</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Larger HADD</td>
<td>1865.4</td>
<td>198</td>
<td>10</td>
</tr>
<tr>
<td>Smaller HADD</td>
<td>7730.2</td>
<td>47</td>
<td>3</td>
</tr>
<tr>
<td>Larger compensation</td>
<td>139.3</td>
<td>9.4</td>
<td>4</td>
</tr>
<tr>
<td>Smaller compensation</td>
<td>1718.6</td>
<td>49</td>
<td>25</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Larger HADD</td>
<td>3815.6</td>
<td>234</td>
<td>19</td>
</tr>
<tr>
<td>Smaller HADD</td>
<td>5420.2</td>
<td>35</td>
<td>4</td>
</tr>
<tr>
<td>Larger compensation</td>
<td>957.1</td>
<td>46</td>
<td>10</td>
</tr>
<tr>
<td>Smaller compensation</td>
<td>2727.8</td>
<td>48</td>
<td>34</td>
</tr>
</tbody>
</table>

$^a$HADD, harmful alteration, disruption, or destruction to fish habitat.

The mean difference is expressed as a percentage relative to the authorised area. The total represents all 52 authorisations audited with HADD and compensatory areas combined regardless of habitat category.
control measures was the most common form of non-compliance (n = 3).

Biological
The mean compliance per authorisation with biological requirements was 58% (Table 1). Common findings of noncompliance were riparian revegetation (absence, area, survivorship, species, height) (n = 151), fish passage (n = 9), and fish utilisation (n = 7). Only 19% of authorisations were compliant with riparian vegetation requirements. Authorisations deemed to be noncompliant with the requirement for fish utilisation were often due to inhospitable water quality (low DO) or an absence of the habitat itself that would allow fish utilisation. We found that compliance with biological requirements differed statistically among habitat categories (Table 5). Compliance with biological requirements in the riparian habitat category was less than in standing water (Figure 3).

Chemical. Mean compliance per authorisation with chemical requirements was 100% (Table 1); however, only 10% (5) of authorisations had chemical requirements.

Compensation Ratios
In almost all cases, actual compensation ratios were smaller than required compensation ratios (Table 6). Compensation ratios, weighted by area, ranged from 0.1:1 to 20.4:1. The DFO requested ratios of 1.6:1 and 0.7:1 for replacement of riverine and riparian habitat...
categories (like for like) and achieved ratios of 1:1 and 0.5:1, respectively. In terms of total hectares, authorisations in the riverine habitat category (like for unlike) had the largest gain (71.55 ha), whereas those in the riparian habitat category (like for like) had the largest loss (-11.86 ha). Compensation ratios of less than 1:1 were documented in the following habitat categories: riverine (like for like and increase like productivity), riparian (like for like and like for unlike), and standing water (like for like). Five of eight habitat category and compensation option combinations resulted in negative net balances of habitat area (Table 6).

The DFO would have needed to require a compensation ratio of 2.5:1 in order to overcome the 37% failure rate documented for riverine (like for like) compensation projects (Table 7). The compensation ratio data were not partitioned into those projects that failed temporally versus those that were never implemented; however, the majority of authorisations (90%) implemented some degree of compensation, yet failed over time. Compensatory works were not implemented for less than 10% of the authorisations.

Probable Fisheries Act Violations

Of the 52 authorisations audited, the total number of probable Fisheries Act violations was 26, exclusive of findings of noncompliance with the requirements of the authorisations. Sixteen authorisations (31%) had one or more probable violations. The mean number of probable Fisheries Act violations per authorisation was 0.51 (SE = 0.1). The frequency of probable Fisheries Act violations differed statistically among provinces (Table 5). Probable Fisheries Act violations were more frequent in central Canada than western Canada (Figure 4). Examples of probable violations include permanent riparian habitat loss and isolation of seasonal off-channel habitat due to channel hardening (n = 5), deposit of deleterious substances (e.g., concrete, sediment, etc.) (n = 3), smothering of compensatory lacustrine spawning habitat with filter fabric and construction materials (n = 1), obstructions to fish passage such as instream silt fences perpendicular to flow, illegal dam construction, impassable culverts (perched, impassable baffles), and impassable compensatory riffle and weir construction (n = 6), loss of riparian vegetation to acquire large organic debris for compensatory habitat features (n = 2), creation of compensatory habitats with anoxic water quality due to fecal coliform contamination or stagnated ponds (n = 3), de-watering of compensatory habitat for dust control during road construction (204,000 L/day) (n = 1), stranding of fish and sedimentation of compensatory habitat due to dam removal (n = 1), blocked access to side-channel habitat in attempts to enhance performance of in-channel habitat features (n = 3),

Table 5. ANOVA model statistics, their degrees of freedom, and probability levels of significance for compliance with biological requirements and frequency of probable Fisheries Act violations

<table>
<thead>
<tr>
<th>Variable</th>
<th>df</th>
<th>F statistic</th>
<th>P value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Compliance with biological requirements*</td>
<td>2.45</td>
<td>4.155</td>
<td>0.0221</td>
</tr>
<tr>
<td>Frequency of probable Fisheries Act violations*</td>
<td>2.48</td>
<td>6.241</td>
<td>0.0059</td>
</tr>
</tbody>
</table>

Asterisk indicate variables that differed (P < 0.05).
<table>
<thead>
<tr>
<th>Habitat Category</th>
<th>Required HADD area (ha)</th>
<th>Required COMP area (ha)</th>
<th>Required COMP ratio</th>
<th>Actual HADD area (ha)</th>
<th>Actual COMP area (ha)</th>
<th>Actual COMP ratio</th>
<th>Difference between required and actual HADD (ha)</th>
<th>Difference between required and actual COMP (ha)</th>
<th>Difference between actual HADD and actual COMP (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riverine: like for like</td>
<td>25</td>
<td>4.49</td>
<td>1.55:1</td>
<td>4.79</td>
<td>4.67</td>
<td>0.97:1</td>
<td>-0.30</td>
<td>-2.31</td>
<td>-0.12</td>
</tr>
<tr>
<td>Riverine: like for unlike</td>
<td>4</td>
<td>0.78</td>
<td>15.47</td>
<td>19.78:1</td>
<td>3.69</td>
<td>75.22</td>
<td>20.37:1</td>
<td>2.91</td>
<td>59.75</td>
</tr>
<tr>
<td>Riverine: increase like productivity</td>
<td>5</td>
<td>2.36</td>
<td>0.40</td>
<td>0.17:1</td>
<td>2.79</td>
<td>0.27</td>
<td>0.10:1</td>
<td>-0.12</td>
<td>-2.52</td>
</tr>
<tr>
<td>Standing water: like for like</td>
<td>6</td>
<td>3.10</td>
<td>4.22</td>
<td>1.36:1</td>
<td>3.40</td>
<td>1.00:1</td>
<td>0.31</td>
<td>-0.82</td>
<td>-0.01</td>
</tr>
<tr>
<td>Standing water: like for unlike</td>
<td>1</td>
<td>0.30</td>
<td>1.00</td>
<td>3.33:1</td>
<td>0.30</td>
<td>0.70</td>
<td>2.34:1</td>
<td>0.00</td>
<td>-0.30</td>
</tr>
<tr>
<td>Standing water increase like productivity</td>
<td>3</td>
<td>1.91</td>
<td>5.54</td>
<td>2.90:1</td>
<td>2.60</td>
<td>9.64</td>
<td>3.71:1</td>
<td>0.69</td>
<td>4.10</td>
</tr>
<tr>
<td>Riparian: like for like</td>
<td>38</td>
<td>26.78</td>
<td>19.20</td>
<td>0.72:1</td>
<td>25.64</td>
<td>13.78</td>
<td>-1.14</td>
<td>-5.42</td>
<td>-11.86</td>
</tr>
<tr>
<td>Riparian: like for unlike</td>
<td>3</td>
<td>1.53</td>
<td>0.05</td>
<td>0.18:1</td>
<td>1.55</td>
<td>0.03</td>
<td>0.16:1</td>
<td>0.02</td>
<td>-1.53</td>
</tr>
</tbody>
</table>

*HADD, harmful alteration, disruption, or destruction to fish habitat.

The difference in area between required and actual HADD areas, required and actual compensation areas, and actual HADD and actual compensation areas is calculated.
poor compliance we observed. Disproportionate loss of riparian habitats relative to other habitat types has been problematic in the United States as well, according to reviews of compensatory mitigation through Section 404 of the Clean Water Act (Kunz and others 1988; Sifneos and others 1992a; Cole and Shafer 2002).

Noncompliance with HADD and compensation areas contributed to substantial losses of habitat. The prevalence and magnitude of larger HADD areas and smaller compensatory works far exceeded the gains in fish habitat due to authorisations with smaller HADD areas or larger compensation. Habitat loss as a result of improperly installed or designed compensatory structures (e.g., perched culverts, impassable weirs, dry channels) was also considerable. In many cases, these habitat losses exceeded the original HADD that necessitated the compensation habitat. Poorly designed compensatory works also caused habitat fragmentation by obstructing or impeding juvenile migration, resulting in isolation of individuals from the rest of the population. Clearly, the limited amount and qualitative nature of monitoring (Harper and Quigley 2005a, 2005b) and enforcement (Drodge and others 1999) have constrained the achievement of NNL.

In general, there appears to be a dearth of expertise in compensatory science in Canada, likely due to the lack of monitoring (Drodge and others 1999; Harper and Quigley 2005a, 2005b) and subsequent adaptive management. In some instances, poorly designed compensatory works resulted in negative impacts to fish habitat. These additional HADDs were due to a lack of expertise on behalf of proponents, consultants, or community groups (community groups receiving cash to implement compensatory projects). Many cases could likely have been prevented with a hydrological and engineering review of compensation proposals by DFO.

Ambiguous requirements and authorisation contradictions also contributed to large losses of habitat at compensation sites. Many authorisations contained open-ended requirements such as “the proponent shall monitor water temperature and measure vegetative cover,” rather than provide measurable thresholds based on parametric and dimensioned units in reference sites. Poorly defined requirements gave rise to

### Table 7. Comparing the actual compensation ratios to the required ratios generates failure rates for each habitat category

<table>
<thead>
<tr>
<th>Failure rate (%)</th>
<th>Ratio to overcome failure rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riverine: like for like</td>
<td>37</td>
</tr>
<tr>
<td>Riverine: like for un-like</td>
<td>N/A</td>
</tr>
<tr>
<td>Riverine: increase like productivity</td>
<td>41</td>
</tr>
<tr>
<td>Standing water: like for like</td>
<td>26</td>
</tr>
<tr>
<td>Standing water: like for un-like</td>
<td>30</td>
</tr>
<tr>
<td>Standing water: increase like productivity</td>
<td>N/A</td>
</tr>
<tr>
<td>Riparian: like for like</td>
<td>25</td>
</tr>
<tr>
<td>Riparian: like for un-like</td>
<td>11</td>
</tr>
</tbody>
</table>

The compensation ratio necessary to achieve the required ratios and overcome the failure rates is calculated.

### Figure 4. Mean frequency of probable Fishes Act violations in each geographic region. Means not connected by the same letter are significantly different (P < 0.05). (West = British Columbia, n = 36); (Central = Manitoba and Ontario, n = 9); (East = New Brunswick and Nova Scotia, n = 7). Error bars represent 1 SE.
situations where proponents were entirely compliant (e.g., the channel was physically stable) yet functional success of the compensation habitat was doubtful (e.g., the channel was dry and disconnected from the watershed).

Some compensation projects were exceptionally successful in achieving large net gains in habitat area. These projects were characterised by compensation ratios that exceeded 5:1. Compensation ratios are intended to offset the inherent risk of project failure, the value of the impacted habitat, and the temporal loss of a compensatory habitat is fully functioning. Although compensation ratios are intended to increase proceeding through the hierarchy of preferences (like for like, like for unlike, increase like productivity) (Minns 1995; DFO 1998, 2002a; Lange and others 2001), this trend was not present in the projects audited.

The failure rates we calculated were a snapshot in time, 4 to 8 years postconstruction. Because habitats are rarely static, it is likely that over time the compensatory works and failure rates will continue to change. Additionally, the failure rates we calculated for each habitat category were both a function of noncompliance and failure (biological and physical) over time. It was not possible to discern the true cause of project failures due to the post-hoc design employed in this study. Ultimately, this distinction does not affect the determination of performance in achieving NNL, but the mechanism of failure does provide insight into recommendations for improvement. Compensation projects that were not completed and/or noncompliant suggest that lack of monitoring and enforcement were more the cause rather than deficiencies in current compensatory science (e.g., regarding appropriate ratios). However, 90% of the projects made attempts to compensate for habitat losses, suggesting that failure over time was likely the primary factor in many cases. Enhanced monitoring and enforcement and improvements in compensatory science are both necessary to address project failure and poor compliance. These institutional failures and lack of scientific understanding have also been suggested as causes to account for the failure of wetland mitigation banking in the United States (Shabman and others 1996; Brown and Lant 1999).

It is important to note that both the required ratios and the ratios we calculated to overcome the failure rates were based strictly on habitat quantity and assumes that the habitats are equivalent in terms of productivity. Accordingly, the ratios calculated to overcome the measured failure rates were contingent upon the assumption that the required ratios were appropriate in the first place from an ecosystem functionality perspective. However, Minns and Moore (2003) argued that fish-habitat linkages have a high degree of uncertainty, providing the impetus to adopt a precautionary approach and implement larger compensation ratios. For many authorisations, habitat losses in some habitat categories (e.g., riparian) did not require compensation, resulting in required ratios that were much less than 1:1. Clearly it would be difficult to achieve NNL of habitat productivity if a given project is only compensating for a fraction of the habitat lost on an areal basis. The smaller that fraction becomes, the greater the productivity of the compensatory habitat needs to be for NNL of habitat productivity to be met. We echo Race and Fonseca’s (1996) comment that “concerns about function are eclipsed by concerns about generating habitat in the first place.” Based on the magnitude of noncompliance and the failure rates documented in this article, it is clear that both compliance and compensation ratios need to increase if Canada’s policy of NNL of habitat productivity is to be achieved.

The lifespan of compensatory works should be commensurate to the longevity of the HADD. The impacts to the landscape from development (e.g., highways, urban development) generally last into perpetuity. However, whether compensation efforts persist over the long term (>50 years) remains uncertain. The long-term prognosis for freshwater compensatory habitats can be tenuous, considering the dynamic nature of watersheds. We observed many compensation projects positioned in landscape locations that will not ensure sustainability (i.e., prone to isolation or destruction during channel-forming flood events).

Financial security is retained by DFO to repair failed compensation habitat as a contingency for complex or risky compensation projects. In theory, riskier compensation techniques should retain greater financial security. In practice, we found compliance with the habitat features requirements was negatively associated with the amount of financial security per square metre of compensation habitat. It is unlikely that this is a causal relationship. Rather, projects that had larger financial security were likely riskier and therefore more prone to failure (i.e., low compliance). Although using financial security as a contingency factor could be an excellent mechanism to work towards conserving fish habitat over the long term, it is rarely implemented. In fact, less than one third of projects in Canada retained financial security and none exercised this option to repair failed compensatory works during the 1994 to 1997 time frame (Harper and Quigley 2005b).

The prevalence of probable Fisheries Act violations at compensation projects was surprisingly high, and even
more alarming is that this was exclusive of findings of noncompliance with the HADD and compensation area requirements. In many cases, these probable violations compromised compensation efforts and likely reduced their efficacy. In the last 5 years, more than 2529 authorisations have been issued across Canada (DFO 2002b), yet DFO has only ever charged a proponents for noncompliance with the requirements of an authorisation on three occasions (Regina vs. Wright, Regina vs. GBA Logging Ltd., Regina vs. BHP Diamonds Inc.). Based on the findings of this study, in the last 5 years there may be more than 1300 authorisations that are in potential contravention of the Fisheries Act. This estimate is based upon the frequency of probable violations, which did not include occurrences of larger HADD or smaller compensation areas, and is therefore likely conservative because nearly all of the authorisations audited had either larger HADD and/or smaller compensation areas than authorised. The rarity of DFO field inspections and monitoring (Harper and Quigley 2005b) are likely contributing factors to the prevalence of probable violations.

The geographic disparity in probable violations is interesting and may be an artifact of institutional differences. The DFO’s habitat management program in British Columbia has long been considerably resourced, enabling a balanced habitat program including education and stewardship initiatives, guideline development, and multi-stakeholder participatory planning initiatives that contribute to successes in conservation of fish habitat. Habitat management has only recently become resourced comparatively in other parts of Canada (e.g., Manitoba, Ontario) (Goodchild 2004), which may partly explain this finding.

The prevalence of authorisation contradictions was an unexpected finding. These authorisation contradictions generally arose due to a lack of confirmation, on behalf of DFO, that areas contained in the scale drawings provided by the developer conformed to the negotiated areas documented in the authorisation text. The outcome was that these authorisations provided two different areas that were legally permitted to be impacted and conversely compensated. This provided the developer with a much broader range of habitat area legally allowed to be impacted and compensated. These discrepancies were considerable, because the HADD areas on scale drawings were, on average, nearly double the value negotiated in the body of the authorisation.

Noncompliance with habitat conservation requirements is not unique to Canada. In a comprehensive examination including nine studies of permitted compensatory mitigation requirements pursuant to Section 404 of the Clean Water Act in the United States, actual compensation ratios were never met (Zedler and others 2001). The average compliance rate with required ratios was 69% in these studies (Zedler and others 2001), which is similar to compliance rates we documented in Canada. Several recommendations have arisen in the United States to address the low compliance rates with Section 404 permits and compensatory wetlands. These include retention of financial security, legally binding monitoring requirements, and legally binding performance measures (Zedler and others 2001). Interestingly, all of these measures have been in place in Canada, yet poor compliance is still pervasive. These measures may also not be adequate to achieve compliance in the United States if additional strategies are not employed.

Rarity of monitoring and enforcement activities has been cited frequently as the primary contributing factors to poor compliance in both Canada (Millar and others 1997; Drodge and others 1999) and the United States (Kusler and Kentula 1990; Sifneos and others 1992a, 1992b; Holland and Kentula 1992; Race and Fonseca 1996; Zedler and others 2001). For instance, Millar and others’ (1997) study of nonlegally binding letters of advice from regulatory agencies in British Columbia, Canada found extremely poor compliance (range 15–40%) with requirements to protect fish and fish habitat, and they recommended increased monitoring and enforcement as a solution. Findings of noncompliance in Canada are not surprising considering that in a national study, Drodge and others (1999) documented that DFO habitat management staff allocated only 1.7% and 1.3% of their workload on compliance monitoring and enforcement, respectively. Although recommendations to improve monitoring and enforcement have occurred during major federal funding initiatives in Canada to address these shortfalls (e.g., Green Plan (Environment Canada 1992); Blueprint (Drodge and others 1999)), the deficiency persists. For effective monitoring and enforcement to become reality, a substantial change in the nature and structure of regulatory involvement in compensation habitat, supported by simultaneous changes in human and financial resources, should be considered to address these recommendations.

Since the inception of Canada’s Habitat Policy (DFO 1986), there has been a proliferation of authorisations issued, and a strong and growing reliance upon this process as a mechanism to conserve fish habitat in Canada. In the late 1980s, the annual number of authorisations issued numbered in the dozens, whereas in 2002, 426 were issued nationally (DFO
Habitat compensation, as currently implemented in Canada, is slowing but not stopping the rate of habitat loss (DFO 1997; Metukosh 1997). Increasing the amount of authorised compensatory habitat in the absence of institutional changes in implementation will not reverse this trend. Improvements in monitoring, enforcement, and compensation ratios are necessary for the authorisation process to move towards achieving Canada’s conservation goal of NNL. Increasing our experience and understanding of habitat compensation will hopefully provide an important means to reversing current trends in fish habitat loss.

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Effectiveness of Fish Habitat Compensation in Canada in Achieving No Net Loss

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ABSTRACT / Fish habitat loss has been prevalent over the last century in Canada. To prevent further erosion of the resource base and ensure sustainable development, Fisheries and Oceans Canada enacted the habitat provisions of the Fisheries Act in 1976. In 1986, this was articulated by a policy that a “harmful alteration, disruption, or destruction to fish habitat” (HADD) cannot occur unless authorised with legally binding compensatory habitat to offset the HADD.

Canada contains approximately one quarter of the world’s wetlands (Rubec 1994), which support a rich biodiversity of more than 198 fish species (Scott and Crossman 1998). Losses of wetlands have occurred at an alarming rate in the last century. In fact, approximately one seventh (20 million ha) of Canada’s wetlands have been lost (Rubec 1994). Habitat loss has been identified as a key factor in the decline of Canada’s freshwater fisheries resources (Pearse 1988; Beamish and others 1986). In North American freshwaters, 73% of fish extinctions can be attributed to habitat alterations (Miller and others 1989).

To prevent further erosion of the resource base, in 1976 Fisheries and Oceans Canada (DFO) enacted the habitat provisions of the Fisheries Act, one of the strongest pieces of environmental legislation in Canada. A “harmful alteration, disruption, or destruction to fish habitat” (HADD) cannot occur unless authorised via Section 35(2) of the Fisheries Act. In 1986, the Policy for the Management of Fish Habitats (hereafter the Habitat Policy; DFO 1986) was implemented to ensure sustainable development by requiring authorised HADDs to be offset by legally binding habitat compensation. The guiding principle behind the requirement for habitat compensation is the achievement of no net loss (NNL) of the productive capacity of fish habitats, the primary conservation goal of the Habitat Policy.

Thus, the putative solution for conserving Canada’s rich biodiversity of fish and the habitats they depend upon, while allowing development to continue, is through compensation habitat. Canada has received accolades for its progressive conservation policies (Brouha 1993), yet in practice, the effectiveness of compensation habitat in achieving NNL has never been tested on a national scale. Excessive workload in DFO results in reactive, crisis habitat management such that follow-up monitoring and adaptive management do not occur (Harper and Quigley 2005a). In fact, nationally only 2.1% of DFO’s habitat management workload is spent conducting follow-up monitoring to...
determine efficacy of compensatory works (Drodge and others 1999). Independent evaluations of the effectiveness of compensation habitat in Canada are even rarer (Harper and Quigley 2005b). Furthermore, the vast majority of evaluations and monitoring that have occurred has been short term (1–3 yrs), judgement based, and qualitative rather than quantitative (Harper and Quigley 2005a). Most studies have based their NNL determinations simply upon area gained or lost rather than scientifically defensible assessments of the true productive capacity of fish habitats. A possible reason for this is the difficulty in assessing productive capacity as it is defined in the Habitat Policy.

Productive capacity is characterised as “the maximum biomass of organisms that can be sustained on a long term basis by a given habitat, analogous to carrying capacity” and “the measure of a habitat to produce fish and/or food organisms in natural or restored conditions” (DFO 1998). Indeed, the difficulty in operationally defining and assessing productive capacity has been recognized (Jones and others 1996; Levings and others 1997) since the inception of the Habitat Policy and has likely impeded applied, practical research into the performance of DFO’s conservation policy. Efforts in providing operational definitions of productive capacity have received considerable resources and effort (Minns and Moore 2003; Minns 1995, 1997; Levings and others 1997), as have inventories of the productive capacity of various habitats (Gordon and others 1997; Amiro 1997; Welch 1997; Williams and others 1997). However, with respect to compensation habitat, DFO’s core mechanism to conserve habitat, practical evaluations of the attainment of NNL of the productive capacity of fish habitats have rarely been undertaken, and if so have been local in scope (Scruton 1996; Minns and others 1995; Levings and Nishimura 1996).

Productive capacity was intended to measure the capacity of the habitat (carrying capacity potential) and not solely current fish production (Scruton 1996). We agree with Minns (1997) and Levings and others (1997) that the Habitat Policy was drafted to conserve habitat quality and ecosystem productivity akin to the United States’ legislation (Section 404, Clean Water Act) designed to conserve wetland functionality. Productive capacity can be considered an intrinsic potential property of habitat, which is not only difficult to quantify but is “logically inoperable” (Minns 1997). However, the ability of DFO to achieve NNL of future productivity may be a moot issue if current habitat productivity is not being effectively conserved. Indeed, the productivity of compensation habitats relative to impacted habitats is the key unanswered question of habitat managers in Canada (Metikosh 1997).

As such, we investigated the effectiveness of habitat compensation in achieving NNL of current habitat productivity by measuring both the area and the productivity of compensatory habitats. Consistent with the latest trends in conservation biology (Underwood 1995; Walters and Holling 1990), we treated DFO’s management actions with respect to habitat conservation (i.e., compensation projects) across Canada as experiments. In this article, we describe a field evaluation of fish habitat compensation projects completed across Canada to determine effectiveness of compensation projects in achieving NNL of habitat productivity.

Methods

Habitat compensation projects were selected randomly across Canada with geographic stratification in five provinces: British Columbia, Manitoba, Ontario, New Brunswick, and Nova Scotia. Field evaluations were completed from May to October of 2000 and 2001. We selected projects that had been completed between 1994 and 1997, which ensured a postconstruction age range of 4 to 8 years.

A hierarchy of compensation options, from most to least preferred, that compare the habitat type and ecological unit of compensation habitat relative to the lost habitat is provided in the Habitat Policy (DFO 1986, 1998). We described compensation projects based on a modified hierarchy of preferences. This included three basic classifications: (1) like for like habitat: create similar habitat at or near the site in the same ecological unit (e.g., replace off-channel habitat with off-channel habitat); (2) like for unlike habitat: create or increase the productivity of unlike habitat in the same ecological unit (e.g., replace in-channel habitat with off-channel habitat); (3) increasing like habitat productivity: increase the productivity of like habitat at or near the site (e.g., enhance existing in-channel habitat to compensate for in-channel habitat loss). Ecological unit was defined as “populations of organisms considered together with their physical environment and the interacting processes amongst them” (DFO 2002a).

Each compensation project was partitioned into treatment sites (n = 2–4). Unimpacted reference sites (n = 2–4) were selected to represent the HADD site prior to the impact. Pre-impact assessment reports, photographs, and on-site visits with the DFO biologist responsible for the authorisation assisted in reference site selection. In some compensation projects, the HADD site and the compensation habitat were spatially distinct. In these cases, treatment sites were selected in both the HADD site (n = 2–4) and the compensatory...
site (n = 2–4) and data were pooled to develop mean response values. In this way, we were able to evaluate the habitat productivity of the compensatory, modified (HADD site), and lost habitats (reference site).

For each project, the evaluations consisted of two general components: determining the overall areal extent of habitat change and the magnitude of change per unit area (Minns 1995). Total surface area of gains and losses in habitat were measured and compensation ratios (habitat area gained:habitat area lost) were calculated (sensu Quigley and Harper 2005).

Taking an ecosystem approach, we selected four variables as a proxy to productive capacity to quantify magnitude of change in habitat productivity. This multimetric approach included biomass of periphyton, macroinvertebrate density, fish biomass, and areal cover of riparian vegetation. These variables were measured at both treatment and reference sites. For some projects, all four variables were not measured because of logistical constraints or because a given indicator was not applicable for a particular project.

Treatment and reference sites were netted off and the areas were measured so that response variables could be quantified per unit area. Periphyton was sampled from each site by selecting five rocks, using a random stratified approach along a transect in the centre of the channel. Sediment was first removed from each rock with a washbottle. Then a cordless drill with nylon brush was used to emulsify periphyton from a known area on each rock defined by 3.8-cm sections of polyvinyl chloride pipe of varying diameters (5.08 cm, 7.62 cm, or 10.16 cm, depending upon substrate size). Emulsified periphyton was rinsed into sample bottles and quantified in the laboratory by filtration (g/m²). Five invertebrate samples were randomly taken per site using a Surber sampler (RIC 1997). Densities (number/m²) and diversity of invertebrates were recorded. Fish were sampled by electroshocker (Smith Root 12C), and densities were calculated using a two-pass removal method (Seber and LeGren 1967). Fish biomass (g/m²) and species diversity were recorded for each site. Riparian vegetation was sampled at each site using a random stratified approach along a transect parallel to the channel. Total percent coverage of 1-m² quadrats as well as diversity of woody and nonwoody riparian species were quantified at five locations per site.

Treatment response variables were weighted by the difference in area between the compensation and HADD areas (i.e., compensation ratio). For example, if the total compensation area exceeded the HADD area by a factor of 1.2, then all of the mean treatment response variables would be multiplied by 1.2 to estimate the total production for that variable. Response variables were contrasted between treatment and reference sites. Most projects were composed of an in-channel and a riparian component, which were evaluated separately. A project was deemed to have resulted in a net gain if one or more of the response variables were statistically greater in treatment sites than reference sites and the remaining variables were not different. A project was deemed to have resulted in a net loss if one or more variables were statistically greater in reference sites than treatment sites. Projects achieved NNL if all of their response variables did not differ between reference and treatment sites.

Two additional sets of analyses were also completed whereby artificial ratios of 1:1 and 2:1 were used with the mean treatment response variables rather than the actual compensation ratios measured. These analyses were completed because many projects had compensation ratios less than 1:1 and we wanted to ascertain the effect that larger compensation ratios might have on the achievement of NNL.

It is possible to have no change in production (biomass) in a particular indicator but have a shift in species composition. Diversity of fish species, invertebrate orders, and riparian nonwoody and woody species was measured to capture changes in community structure.

Data Analyses

Data were visually inspected for normality and homogeneous variances. We used log transformations to minimise heterogeneous variances. For each compensation project, we used analysis of variance to compare response variables between reference and treatment sites. Least-square means were used to calculate means for graphical presentations. Values are reported as means ± 1 standard error (SE). Statistical analyses were completed using graphical presentations. Values are reported as means ± 1 standard error (SE). Statistical analyses were completed using SAS statistical software, release 8.02 (SAS Institute 2001). All tests were considered to be significant to a P ≤ 0.05.

Results

A total of 16 habitat compensation projects were evaluated across Canada in British Columbia (n = 7), Manitoba (n = 3), Ontario (n = 2), New Brunswick (n = 2), and Nova Scotia (n = 2) (Figure 1). This sample represents approximately 13% of the total number of authorisations (N = 124) issued in these provinces during 1994 to 1997 inclusive. The mean age of projects was 4.3 years (SE = 0.5) (Table 1). Habitat compensation projects evaluated were a result of the
following development activities: roads and highways ($n = 7$), urban development ($n = 4$), forestry ($n = 3$), agriculture ($n = 1$), and oil and gas ($n = 1$) (Table 1). The HADDs and compensatory habitats occurred in two habitat categories: in-channel and riparian. Many projects included HADDs and compensation in both habitat categories. Common compensation techniques included riparian revegetation, channel creation, and habitat complexing through addition of boulders, large woody debris, or pools (Table 1).

In the in-channel habitat category, approximately 58% of projects had HADD areas that were larger than authorised. Smaller-than-authorised HADD areas were less common, occurring 8% of the time (Figure 2A). The mean size of the authorised and actual HADDs was 2493 m$^2$ and 5393 m$^2$, respectively. In contrast, compensation habitat tended to be smaller than required. Approximately 50% of projects had compensation habitat smaller than required, whereas 17% were larger than required (Figure 2B). The mean size of the

Figure 1. Location of compensation projects evaluated across Canada ($n = 16$).

![Figure 1. Location of compensation projects evaluated across Canada ($n = 16$).](image-url)
Table 1. Descriptive information for compensation projects studied across Canada

<table>
<thead>
<tr>
<th>Project</th>
<th>Province</th>
<th>Age (yrs)</th>
<th>HADD description</th>
<th>Compensation description</th>
<th>Hierarchy option</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Manitoba</td>
<td>5</td>
<td>Highway realignment resulted in a loss of in-channel riverine habitat and of riparian habitat.</td>
<td>River diversion created in-channel riverine habitat and riparian habitat. Constructed riffles and deep pools incorporated as compensation features.</td>
<td>Like for like</td>
</tr>
<tr>
<td>2</td>
<td>British Columbia</td>
<td>3</td>
<td>Forestry road realignment and culvert installation resulted in a loss of in-channel riverine habitat and riparian habitat.</td>
<td>Creation of in-channel riverine habitat and riparian habitat. Habitat complexing with large woody debris was a compensation feature.</td>
<td>Like for like</td>
</tr>
<tr>
<td>3</td>
<td>Nova Scotia</td>
<td>7</td>
<td>Highway twinning resulted in stream diversion and culvert installation destroying in-channel riverine habitat and riparian habitat.</td>
<td>Habitat complexing was completed to enhance productivity by installing digger logs.</td>
<td>Increase like productivity</td>
</tr>
<tr>
<td>4</td>
<td>Ontario</td>
<td>7</td>
<td>Municipal road construction and bridge installation resulted in stream channelisation and diversion. In-channel riverine habitat and riparian habitat was lost.</td>
<td>Creation of in-channel riverine habitat and riparian habitat.</td>
<td>Like for like</td>
</tr>
<tr>
<td>5</td>
<td>Manitoba</td>
<td>3</td>
<td>Construction of a dam and spillway to create an agricultural water reservoir resulted in a loss of riverine in-channel habitat and riparian habitat.</td>
<td>Creation of lacustrine/reservoir habitat, riparian habitat and fishway.</td>
<td>Like for unlike</td>
</tr>
<tr>
<td>6</td>
<td>British Columbia</td>
<td>3</td>
<td>Installation of an outfall structure for discharge from a water treatment plant resulted in a loss of riparian habitat.</td>
<td>Riparian revegetation.</td>
<td>Like for like</td>
</tr>
<tr>
<td>7</td>
<td>New Brunswick</td>
<td>4</td>
<td>Highway construction and installation of twin bridges resulted in a river diversion and channelisation and a loss of riparian and in-channel habitat.</td>
<td>Creation of in-channel habitat complexed with digger logs, boulders, and large woody debris.</td>
<td>Like for like</td>
</tr>
<tr>
<td>8</td>
<td>Ontario</td>
<td>3</td>
<td>Road construction, culvert installation and stormwater retention pond to service new subdivision resulted in a loss of in-channel riverine habitat and riparian habitat.</td>
<td>Riparian revegetation and creation of in-channel habitat complexed with large woody debris.</td>
<td>Like for like</td>
</tr>
<tr>
<td>9</td>
<td>New Brunswick</td>
<td>2</td>
<td>Highway construction and installation of four culverts, channelisation, and two stream diversions resulted in a loss of in-channel riverine habitat and riparian habitat.</td>
<td>Riparian revegetation and creation of in-channel habitat complexed with digger logs, boulders, and large woody debris.</td>
<td>Like for like</td>
</tr>
<tr>
<td>10</td>
<td>British Columbia</td>
<td>3</td>
<td>Major river channelisation and creation of a spur-dyke to protect downstream forestry mill resulted in a loss of in-channel riverine habitat and riparian habitat.</td>
<td>Riparian revegetation and preservation of adjacent side-channel access. Irregular edge habitat and groynes incorporated into rip-rap design as compensation features.</td>
<td>Like for like</td>
</tr>
<tr>
<td>11</td>
<td>British Columbia</td>
<td>2</td>
<td>Condominium development resulted in a loss of riparian habitat.</td>
<td>Riparian revegetation.</td>
<td>Like for like</td>
</tr>
</tbody>
</table>

Continued

Effectiveness of Habitat Compensation in Canada

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| Table 1. Continued |
|-------------------|---------------------|---------------------------------------------------------------------------------|-----------------|
| Project | Province | Age (yrs) | HADD description | Compensation description | Hierarchy option |
| 12 | British Columbia | 7 | River diversion to protect forestry mill resulted in a loss of in-channel riverine habitat and riparian habitat. | Riparian revegetation and creation of in-channel habitat complexed with boulders and large woody debris. | Like for like |
| 13 | Nova Scotia | 9 | Highway construction, culvert installation, and stream diversion and realignment resulted in a loss of in-channel riverine habitat and riparian habitat. | Enhanced productivity of in-channel habitat through installation of digger logs. | Increase like productivity |
| 14 | Manitoba | 5 | Road construction, stream realignment, and bridge installation resulted in a loss of in-channel riverine habitat and riparian habitat. | Creation of in-channel habitat complexed with boulders and deep pools. | Like for like |
| 15 | British Columbia | 3 | River channel hardening and straightening with rip-rap to protect gas pipeline resulted in a loss of in-channel riverine habitat and riparian habitat. | Creation of off-channel habitat complexed with large woody debris and pools and riparian revegetation. | Like for unlike |
| 16 | British Columbia | 3 | Road realignment and channelisation resulted in a loss of riparian habitat. | Riparian revegetation and incorporation of groynes in the rip-rap to create edge habitat. | Like for like |

The differences in riparian productivity were large and unequal in these eight projects. The differences in riparian productivity were large and unequal in these eight projects.
There were no differences in diversity of fish or invertebrates between treatment and reference sites in any of the projects (Table 3). Three compensation projects had differences in diversity of riparian vegetation between treatment and reference sites. Project 10 had a greater diversity of nonwoody riparian species in reference sites (0.67/m²) in comparison to treatment sites (0.33/m²). Project 6 had a greater diversity of woody riparian species in reference sites (3/m²) compared to treatment sites (0.5/m²). Project 11 had a greater diversity of nonwoody species in treatment sites (2.1/m²) relative to reference sites (0/m²), yet had a greater diversity of woody species in reference sites (2.0/m²) compared to treatment sites (0.45/m²) (Table 3). There were no differences between diversity of nonwoody or woody riparian species in any of the other projects.

Figure 2. Required and actual HADD areas (A, D), compensation areas (B, E), and compensation ratios (area gained:area lost) (C, F) in the in-channel and riparian habitat categories. Values that exceeded the scale are indicated above the bar except for project 5, which had a required HADD of 60,000 m² and an actual HADD of 69,931 m² in the riparian category (D). Project 5 had a required and actual in-channel compensation area (B) of 150,000 m² and riparian compensation area (E) of 60,000 m², respectively. Bars that are absent indicate a zero value except for project 14, in which actual HADD and compensation areas were not measured in the in-channel category.
Discussion

Inherent ecosystem variability meant that differences had to be large in order to detect responses. In this respect, our results can be considered conservative because we defaulted to a NNL outcome on many projects that may not have achieved this goal. Indeed, Mapstone (1995) reports that many environmental impact assessments conclude that a development had no effect because an 80–100% change in the measured variable would have been required to detect change. Although more replicates would have assisted in

![Figure 3. Percentage of projects achieving a net gain (NG), no net loss (NNL), and a net loss (NL) of habitat productivity based on the mean actual compensation ratios (A) indicated under each bar, and artificial ratios of 1:1 (B) and 2:1 (C).](image-url)
determining differences in habitat productivity, the gross disparity in physical area of compensated versus impacted habitats was an overriding factor for many projects. Unquestionably it is exceedingly difficult to achieve equivalent habitat productivity when replacing only a fraction of the habitat impacted (Quigley and Harper 2005).

However, even if compliance was 100% it is unlikely that the compensation projects would have achieved NNL. Ambrose (2000) also demonstrated that compliance success does not ensure ecological success and highlighted the importance of quantitative rather than subjective evaluations. National guidelines recommend that DFO should “aim for minimum compensation ratios of 1:1” (DFO 2002a), yet in our study close to half of the projects would not have achieved NNL with this ratio. In order to achieve NNL, Minns and Moore (2003) advocate compensation ratios larger than 2:1.
Figure 4. Periphyton biomass (A), invertebrate density (B), fish biomass (C), and riparian coverage (D) in control and treatment sites of compensation projects across Canada (based on actual compensation ratios). Asterisk indicates means that differed. Means are based on the number of reference and treatment sites in each project (n = 2–4). Error bars represent 1 SE.
We found that although success improved with artificial ratios of 2:1, a substantial proportion of compensation projects still did not achieve NNL, a finding supported by others (Kistritz 1996). Thus, even if projects were entirely compliant and created twice as much compensation habitat compared to the HADD, the

Table 3. Analysis of variance model statistics, their degrees of freedom, and probability levels of significance for the diversity of fish, invertebrates, woody and nonwoody riparian species

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<th>Project</th>
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<th>df</th>
<th>F statistic</th>
<th>P value</th>
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<td>Fish</td>
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<tr>
<td></td>
<td>Nonwoody riparian</td>
<td>1, 2</td>
<td>7.47</td>
<td>0.1118</td>
</tr>
</tbody>
</table>

*Asterisk indicates variables that differed between treatment and reference sites (P < 0.05). Note that Project 2 contained a single fish species in all sites rendering diversity comparisons between treatment and reference sites not applicable (n/a).
Habitat Policy goal of NNL would still not always be achieved. This is alarming considering that the average compensation ratio for all projects completed in Canada between 1994 and 1997 was 1:1:1 (Harper and Quigley 2005a), indicating that many projects did not achieve NNL. In the present study, projects that successfully achieved a net gain in habitat productivity were characterised by actual ratios of approximately 5:1, although required ratios were up to an order of magnitude larger for these projects. The need for larger compensation ratios has been echoed in the United States (Allen and Feddema 1996; Brown and Lant 1999).

Based on the simple metric of habitat area, it would appear that Canada should be achieving a net gain of habitat productivity (Harper and Quigley 2005a). However, upon closer analyses, the actual areas of compensation habitats are much less than required and actual HADD areas are much larger than that stated in authorisations (Quigley and Harper 2005). Poor compliance rates, and the inability of file reviews to determine actual gains in habitat areas, are common findings in the United States as well (Ambrose 2000; Zedler and others 2001). In Canada, not only is NNL not being met spatially, but it is also not being achieved temporally and functionally. Temporal losses of habitat productivity are inevitable when compensation habitats are developed after the HADD occurs. Furthermore, temporal losses are exacerbated due to the time lag until compensatory habitats function ecologically in a manner comparable to preimpact conditions. In many cases, the time lag may be considerable because some projects will likely never achieve equivalent functionality. Time between HADD occurrence, compensation development, and compensation functionality was not a leading (and in many cases present) consideration in the authorisations we studied. Similar shortcomings have been identified in the United States (Brown and Veneman 2001; Kunz and others 1988; Zedler 1996).

It seems clear that compensatory works were not successful in completely offsetting the losses of habitat, although they were successful in slowing the rate of habitat loss. This is not altogether surprising, considering the general consensus of habitat managers in Canada is that DFO is not achieving NNL (DFO 1997; Metikosh 1997). Most fish habitat managers and scientists agree that we are losing, and will continue to lose, habitats and species if the magnitude, frequency, and type of anthropogenic disturbances continue (Applegate and others 1996).

However, limited success in achieving NNL to date does not erode or invalidate the value of this goal of the Habitat Policy; rather, it provides an impetus for change. It is important to note that in our study, more than one third of projects evaluated achieved either a net gain or NNL in habitat productivity, indicating potential to build on these successes. Challenges in achieving functional equivalency at compensatory habitats have also been reported in the United States (Sudol and Ambrose 2002), and recommendations for improvement have been compiled (Zedler and others 2001). It is conceivable that NNL may be attained if some fundamental changes to compensation science and institutional approaches are incorporated into DFO’s habitat management program. Modifying management approaches based on the results of monitoring and evaluation programs is a critical component of adaptive management, yet often neglected (La Peyre and others 2001). Although broad environmental policy reviews utilising ecological indicators at the national and international level are increasing, they have yet to be integrated into daily environmental decision-making (La Peyre and others 2001). It is critical for Canada’s fisheries resources for DFO to engage and respond to feedback loops that foster the refinement, and in particular the implementation, of environmental policies.

Productivity can be considered the current yield of a habitat (Gordon and others 1997), whereas productive capacity incorporates the future potential. Our evaluations were only a snapshot in time, and it could be argued that some of the compensatory habitats will achieve NNL in the future or at a different season of the year. However, the HADDs exist year-round and will doubtless last into perpetuity in many cases. We would argue that compensatory habitats should offset the HADD today, tomorrow, and into perpetuity, rather than in any particular season or future period. Simply stated, compensatory habitat should be achieving NNL on any given day. Otherwise, Canada’s habitat base will slowly erode due to accumulating temporal losses of fish habitat.

Lack of pre-impact assessment baseline data and limited monitoring data have challenged researchers’ abilities to draw conclusions in NNL studies (Cole and Shafer 2002; Kentula and others 1992; Harper and Quigley 2005a). Only one compensation project we evaluated had quantitative pre-impact data (fish biomass), and none had previously determined reference sites. Ability to detect changes in productivity and power of statistical analyses would be greatly improved if reference sites (Brinson and Rheinhardt 1996) and quantitative pre-impact data were routinely required for compensation projects and rigorous experimental designs were employed in monitoring programs (Underwood 1991, 1993; Stewart-Oaten and Bence 2001; Pearson and others 2005).
The fact that we did not detect considerable differences in diversity of species may be due to the tendency for most of the projects to have implemented in-kind compensation (rather than like for unlike). This practice has been lauded due to its propensity to maintain biodiversity (Race and Fonseca 1996; Allen and Feddema 1996). Lack of an in-kind replacement policy in the United States has resulted in an increase in homogeneous wetland types and a decline in vegetation diversity (Allen and Feddema 1996). However, insistence on like for like in highly disturbed landscapes (such as urbanised areas) is not always advisable because the original landscape has essentially disappeared (Race and Fonseca 1996) and other ecological or biophysical bottlenecks may frustrate compensation attempts. Attention to limiting factors and compensation options lower on the hierarchy of preferences (DFO 1998) would likely be more successful in these instances.

In general, we found that compensation sites were selected opportunistically rather than based on ecological bottlenecks and potential for success, which influenced the success of compensation habitats in achieving equivalent productivity. Natural sites selected for compensation often had environmental and biological limitations that were largely ignored. For example, compensation sites selected for riparian planting tended to have very low success in the present study and others (Cole and Shafer 2002; Robb 2002; Race 1985). The difficulty in establishing vegetation at barren sites is not altogether surprising, because there are generally good reasons why riparian vegetation is not currently flourishing at these locations. An absence of vegetation maintenance programs such as irrigation, fertilisation, and weeding is likely a contributing factor. Vegetation survival and therefore replacement of functional values can be successful in compensation projects that employ maintenance programs (e.g., a large-scale drip irrigation system) (Allen and Feddema 1996; Sudol and Ambrose 2002). However, requiring sites that do not currently support riparian vegetation to be artificially irrigated may not be a wise strategy. If natural hydrologic processes do not support a riparian community, requirements to irrigate may only achieve a partial community and result in sites that are unlikely to be self-sustaining (Sudol and Ambrose 2002). Furthermore, considering poor compliance rates (Quigley and Harper 2005; Zedler and others 2001), irrigation may never occur or certainly be short lived and therefore the site will eventually revert to the natural community it supported prior to compensation efforts.

Our paper quantitatively examined four components of fish habitat, at three distinct trophic levels, to determine efficacy of compensatory habitat in replicating habitat quality. In our study, it appeared that indicators lower on the trophic level such as periphyton and invertebrates were more responsive and/or less variable and thus better at representing gross differences in habitat productivity than fish biomass. However, invertebrates and periphyton are rarely measured in assessments of compensatory projects (Breaux and Sereffiddin 1999); rather, fish biomass (Scruton 1996; Scruton and others 1997) and vegetative cover (Allen and Feddema 1996; Breaux and Sereffiddin 1999) have primarily been used to infer habitat productivity. Our multimetric approach provided a more complete picture of habitat productivity, rather than simply using fish biomass as an indicator. Invariably, habitat alterations do not exclusively affect a particular species in isolation of other biota (Minns and others 1996). Furthermore, fish can be rather poor indicator species because of their mobility, cyclical populations, exposure to confounding influences (ocean productivity, fisheries, etc.), and divergent life histories. Indeed, for anadromous species it is possible to have low escapements and pristine freshwater habitat, as is the reciprocal.

An array of ecological indicators is preferable to detect responses to habitat alterations (Minns and others 1996). In many cases, selecting one surrogate of habitat productivity, rather than an array of ecological indicators at different trophic levels, would have led to erroneous conclusions. For example, in Project 2, greater biomass of fish was measured at the impacted habitat (culvert) in comparison to the compensatory habitat. However, invertebrate density and riparian vegetation were all negligible at the HADD site in contrast to robust populations in the compensatory habitat. Had we only evaluated fish as an indicator, we would have missed important attributes of the ecological picture of this site.

Compensation science and institutional approaches need to improve in Canada if the conservation policy of NNL of habitat productivity is to be met, as evidenced by the compensation projects assessed in this study, of which only 37% achieved this goal. In the United States as well, replacement of functional values of wetlands has been limited (Sudol and Ambrose 2002; Ambrose 2000; Race and Fonseca 1996; Zedler and others 2001). Canada’s poor performance in achieving NNL is especially sobering considering that our study only focused on site-specific impacts and ignored hydrological affects and disruption to landscape processes. Indeed, Hartman and Miles (1997) demonstrated that one of the earliest compensatory spawning channels (created in 1956) failed half a
century later due to cumulative watershed impacts from other development activities. The NNL policy would be best practiced in a watershed or ecosystem-based management context to ensure that landscape processes that build and maintain habitats are considered. Although cumulative impacts due to poor performance of Section 35(2) Fisheries Act authorisations and associated compensation habitats are likely occurring, our study and the monitoring requirements for habitat compensation in Canada are poorly scaled to capture long-term (>50 yrs) and cumulative ecosystem effects.

Institutional shortcomings such as lack of monitoring and maintenance have been identified as the causes for poor compliance with required compensation areas (Shabman and others 1996; Brown and Lant 1999). Race and Fonseca (1996) argue that “concerns about function are eclipsed by concerns about generating habitat in the first place.” The focus on habitat quantity only may be flawed because we demonstrated that artificially increasing compensation areas to ratios of 2:1, by itself, was not sufficient to achieve a net gain in habitat productivity for all projects. Likewise, Sodul and Ambrose (2002) demonstrated compliance with regulatory requirements was not sufficient to replace wetland functions in the United States. Clearly, the ability to replicate ecosystem function is limited, and both improvements in compensation science and institutional approaches are necessary. Recommendations to improve success include larger compensation ratios, creation and documentation of the functionality of compensation habitats prior to and concurrent with HADDS, maintenance programs, increased monitoring, and enforcement, and attention to limiting factors on a watershed basis (Zedler and others 2001). Improvements in these areas will advance the success of habitat compensation toward NNL. However, it is important to acknowledge that it is simply not possible to compensate for some habitats. Therefore, the option to compensate for HADDS may not be viable for some development proposals demanding careful exploration of alternative options including redesign, relocation, or rejection. Failure to acknowledge the limitations of compensatory science will hinder Canada’s efforts to conserve fish habitat and achieve the goal of NNL.

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Literature Cited


