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Poorly designed biodiversity loss-gain models facilitate biodiversity loss in New Zealand

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Abstract: Biodiversity offsetting and compensation proposals are routinely employed through the resource consenting process to address development-induced indigenous biodiversity losses in Aotearoa/New Zealand. Determining the quantum of demonstrable biodiversity gain required to adequately account for development induced losses is a fundamental component of designing a biodiversity offset. However, trading biodiversity is complex and must account for substantial uncertainties. Therefore, biodiversity offset models that account for losses and gains are a necessary tool for determining the adequacy of an offset proposal. Yet there is currently no accepted standard approach to loss-gain calculations. Models of insufficient ecological and mathematical robustness can perpetuate systematic biodiversity losses and distract decision-makers from discussions regarding real-world ecological consequences of development. We discuss these issues and present a case study to demonstrate how poorly designed biodiversity models that are currently in use in Aotearoa/New Zealand facilitate biodiversity loss. Model development and implementation has been hampered by a tension between competing requirements: (1) simple models that are user-friendly and not resource intensive to parameterise, and (2) models that are sufficiently complex to represent ecological values at an appropriate resolution. It is imperative that newly developed models adhere to standards employed in other ecological modelling domains to curb current and future biodiversity loss. Ecological practitioners and decision-makers are often unable to assess the quality of models and a lack of guidance and oversight of biodiversity offset modelling by the wider ecological and academic community is evident. We conclude that biodiversity offset modelling is a critical research area and that advancements within this space are urgently needed to halt ongoing biodiversity declines.

Keywords: biodiversity offsets, biodiversity protection, compensation, no-net-loss, RMA

Introduction

Tools that reliably inform resource management decision-making are increasingly critical as human pressures on species and ecosystems are steadily contributing to the global biodiversity crisis (Mendoza & Martins 2006; Rouse & Norton 2010; Namany et al. 2019). Our ability to measure, monitor, and forecast trends has not kept pace with the rate and extent of biodiversity decline (Gonzalez et al. 2023). Land-use and land management change, relative to other pressures, has the largest negative impact on biodiversity for terrestrial and freshwater ecosystems mainly through habitat loss and degradation (Collen et al. 2014), although other pressures such as introduced fauna and invasive plant species also contribute to biodiversity decline (Doherty et al. 2016; Shabani et al. 2020). To-date, three-quarters of the world's land surface has been significantly altered by humans (UN Report 2019). To understand this biodiversity loss, identify its causes, and predict the outcomes of management interventions, ecologists utilise a wide range of tools including predictive models (Wood et al. 2018). Models can be powerful tools that may aid resource management decision-making, including those involving the use of biodiversity offsetting to address development-induced impacts. However, the use of models can result in more confidence being placed in predictions than is warranted. Therefore, it is essential that any advances in model development are robust, transparent, and capable of being relied upon by decision makers.

Like the rest of the world, there is evidence that biodiversity loss driven by development (e.g. infrastructure, resource extraction, urban expansion, intensification of farming) is occurring in Aotearoa/New Zealand (Walker et al. 2006; Myers et al. 2013; Parliamentary Commissioner for the Environment 2015; Monks et al. 2019; MfE & Stats NZ 2021). Of the nearly 11 000 terrestrial species assessed under the Aotearoa/New Zealand Threat Classification System 811 species (7%) are

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ranked as Threatened and 2416 species (22%) At Risk. Since humans first arrived in Aotearoa/New Zealand, many species including 57 birds that rely on land and/or freshwater habitats, have been lost to extinction (Department of Conservation 2020). These losses were largely because of the introduction of pest species, habitat loss, and environmental degradation (Innes et al. 2010). Around half of the total land area in Aotearoa/New Zealand has been transformed from indigenous vegetation cover to agriculture, production forestry, and urban uses (MfE & Stats NZ 2021). Pressure on biodiversity that is driven by development is likely to increase in line with Aotearoa/New Zealand's population growth (5.13 million in 2022 to between 5.28 and 5.85 million in 2033; Stats NZ 2022). In this context, the development of appropriate policies, methods, and tools to prevent further biodiversity loss is critical. The focus of this paper is on biodiversity accounting models for use in the design of biodiversity offsets, their inherent risks and limitations, and the potential significant conservation outcomes with their use.

The role of offsetting and compensation in curbing biodiversity losses driven by development

Biodiversity offsetting is a tool to manage risks associated with biodiversity losses driven by development activities (Table 1). Biodiversity offsetting emerged from an international recognition that existing approaches to conservation have failed to halt biodiversity decline resulting from development and that an urgent shift to a more sustainable development model is required globally to reduce adverse biodiversity impacts (Maron et al. 2016). The purported aim of biodiversity offsetting is to achieve No Net Loss (NNL) and preferably a Net Gain (NG) of biodiversity. These gains are achieved through targeted conservation actions (e.g. creation of new ecosystems, restoration of existing degraded ecosystems, and removal of threats such as invasive species). Biodiversity offsetting is based on a set of widely accepted principles, including the recognition of limits to offsetting, evaluation of ecological equivalence (across type, amount, time, and space), and the requirement of biodiversity gains to be additional to what would have occurred in the absence of an offset action (Maseyk et al. 2018). Critically, the practice of offsetting necessitates a strict adherence to the effects management hierarchy (also internationally known as the mitigation hierarchy) whereby offsets only apply to residual impacts on biodiversity after options to avoid, minimise, and remedy effects have been exhausted. Within Aotearoa/New Zealand, the effects management hierarchy and biodiversity offsetting principles are also reflected in the National Policy Statement for Freshwater Management 2020 Amendment No 1 (MfE 2023a) and the National Policy Statement for Indigenous Biodiversity 2023 (MfE 2023b). Biodiversity offsetting is different from other conservation interventions because it is a response to project-specific residual losses and therefore the gains proposed by an offset need to be adequate to address these specific losses. This requires that both biodiversity losses due to the development and anticipated biodiversity gains generated by the proposed offset measures be quantified within a numerical or statistical framework that accounts for time-lags (between losses and gains) and uncertainty. Where it is not possible to accurately quantify biodiversity losses or anticipated biodiversity gains, decision-makers may consider proposals to provide biodiversity compensation (definition in Table 1) as the subsequent and last step in the effects management hierarchy.

Biodiversity offsetting is a high-risk tool for managing biodiversity because it involves trading guaranteed biodiversity

Table 1. Glossary of terms used in biodiversity loss-gain modelling.

Term	Description of term
Biodiversity compensation	Actions (excluding biodiversity offsets) to compensate for residual adverse biodiversity effects arising from activities after all appropriate avoidance, minimisation, remediation, and biodiversity offset measures have been subsequently applied. Gains generated by compensation actions must be additional to those that would have occurred anyway in the absence of those actions.
Biodiversity offset	A measurable outcome resulting from actions designed to compensate for residual adverse biodiversity effects arising from activities after appropriate avoidance, minimisation, and remediation measures have been subsequently applied and that achieves No Net Loss or preferably a Net Gain.
Like for like	The concept of exchanging the same type of biodiversity within an offset or compensation calculation.
Net Gain (NG)	The objective for a biodiversity offset to generate (at a specified point in time) gains in target biodiversity values that counterbalance and exceed the biodiversity losses resulting from a project. Net Gain offsets are demonstrated by a like-for-like quantitative loss and gain calculation of the type, amount, and condition of the biodiversity value. A Net Gain is achieved when the ecological values at the offset site exceed (accounting for time lag and uncertainty) those being lost at the impact site.
No Net Loss (NNL)	The objective for a biodiversity offset to generate (at a specified point in time) sufficient gains in target biodiversity values that balance the biodiversity losses associated with a project. No Net Loss is demonstrated through a like-for-like quantitative loss and gain calculation that estimates ecological equivalence across type, amount, and condition. No Net Loss is achieved when the ecological values at the offset site are equal to (accounting for time lag and uncertainty) those being lost at the impact site.

losses in the present for estimated future gains resulting from targeted management interventions. Very little evidence concerning the efficacy of offsets or compensation has been published within the Aotearoa/New Zealand context because many of the consented projects that require offsets or compensation have not yet begun or been completed preventing the evaluation of whether biodiversity gains achieved from offset or compensation actions were in fact sufficient to balance losses. Further where projects have commenced, compliance monitoring and enforcement of consent conditions pertaining to effects management have been inadequate (Brown et al. 2013). Recent international reviews on the effectiveness of biodiversity offsetting have ranged from scathing to neutral. Josefsson et al. (2021) show that real-world data regarding the efficacy of biodiversity offsets is limited, making efficacy assessments of offsets against their NNL target difficult. In a global review, zu Ermgassen et al. (2019) found weak to no support for the effectiveness of NNL policies.

Within the Aotearoa/New Zealand context, biodiversity compensation is differentiated from biodiversity offsetting (Maseyk et al. 2018; Department of Conservation 2014; MfE 2023a; MfE 2023b). Biodiversity compensation is designed to compensate for losses but is not held to the same high threshold as biodiversity offsetting in that compensation is not required to fully account for and balance specific losses and gains and thus, demonstrate a NNL or NG outcome for target biodiversity elements (Maseyk et al. 2018). Thus, methods to determine the quantum of compensation offered are typically qualitatively informed or subjective. Compensation is often akin to “doing good things” which, albeit preferable to doing nothing, increases the likelihood that there will be a loss in the impacted biodiversity for which NNL or NG objectives are not met. Therefore, biodiversity compensation poses an even greater risk than biodiversity offsetting for the persistence of the targeted biota, and therefore should be the last, least preferable, step in the application of the effects management hierarchy when addressing adverse effects. This is now reflected in the national policy statements (MfE 2023a; MfE 2023b).

A key challenge for biodiversity offsetting is determining the quantum of gain in the biodiversity target required to appropriately and adequately balance the development-induced losses accounting for time and uncertainty. The development of models that can evaluate losses and gains, accurately and transparently communicate trade-offs when offsetting (or compensating), and that facilitate robust discussion and robust application has been a necessary development. However, both model development and subsequent implementation is complicated by the tension between a desire by proponents of development for simpler, less resource intensive models and the difficulty of adequately capturing the complexity of biodiversity, or at least the elements perceived to be important without increased investment. Presently, parameterising a model adequately on a project-by-project basis can be construed as cost prohibitive for an applicant. However, applicants regularly invest heavily in a project (e.g. research, equipment engineering, design), prior to receiving resource consent and a typical Aotearoa/New Zealand infrastructure project spends, on average, 5.5% of their total project budget seeking a resource consent (Moore et al. 2021). Conversely, ecological issues are often seen as a secondary concern. Robust offset or compensation models can help to appropriately internalise the true ecological costs of development projects and should give a true representation of the difficulty of offsetting and/or compensation requirements. The ability to estimate the cost of offsetting or compensation upfront will also allow applicants to determine the most cost-effective pathway for their project and will likely incentivise stronger use of avoidance and impact minimisation measures (Phalan et al. 2018; Kujala et al. 2022).

Current and emerging use of loss-gain models and metrics in designing biodiversity offsets or compensation

An overview of the governance of offsetting around the world found that there is diversity of offsetting governance in different local, institutional, and sector contexts (Droste et al. 2022). Unlike Aotearoa/New Zealand where national policy guiding the implementation and evaluation of offsetting has only just emerged, offsetting models and metrics used in other jurisdictions are directly linked to legislation and policy and thus can be highly prescriptive. However, even for these existing and emerging tools, the need for standards and guidance in their application is recognised. For example, the Australian Federal Government’s Environmental Offsets Policy (under the Environment Protection and Biodiversity Conservation Act 1999) has included a model (the Offsets Assessment Guide) in the policy since 2011 (Miller et al. 2015). The Offsets Assessment Guide is applied on a project-by-project basis, and despite its relative prescriptiveness, some components are sensitive to user-derived inputs leading to inconsistencies in determining required offset actions (Maseyk et al. 2020). Additional guidance has been produced to improve the application of the Offsets Assessment Guide (Maseyk et al., 2017). At the state level, the Western Australian Department of Water and Environmental Regulation released its environmental offsets metric (comprising a calculator and guideline on its use) in 2021 (DWER 2021; DWER 2022a). Additional guidance on using the metric was released for public consultation in 2022 (DWER 2022a; DWER 2022b). In England a mandatory biodiversity NG policy came into effect in 2023. Ahead of this policy the UK government ran a technical consultation on the latest version of the Biodiversity Metric: the biodiversity accounting tool that applicants will use to calculate biodiversity NG for terrestrial and intertidal development (DEFRA 2022). Within condition metrics (used to generate a currency to calculate losses and gains), modelling is seldom utilised (Borges-Matos et al. 2023).

In Aotearoa/New Zealand, biodiversity models have been used since 2006 to help quantify and manage the effects of development. The first edition of the Stream Ecological Valuation (SEV) was released in 2006 (Rowe et al. 2006), and the second edition (Rowe et al. 2008) provided updated formulae for calculating Environmental Compensation Ratios (ECR) for streams. In 2010 the SEV method was reviewed, resolving some performance issues and improving the practical guidance for undertaking SEV assessments. Both the SEV and ECR have had widespread use as a method for quantifying the functional values of streams and determining the quantum of necessary stream restoration required to provide NNL of some attributes of ecological function (but not all biodiversity values) (Storey et al. 2011). A biodiversity offset accounting system focused on terrestrial biodiversity was developed in 2016 with the aim of improving estimation of ecological equivalency and transparency in communicating loss-gain calculations (Maseyk et al. 2016). This model uses a disaggregated area by condition currency, thus explicitly and transparently accounting for individual biodiversity elements. The model has since been used in a number of projects around Aotearoa/New Zealand to evaluate biodiversity offset proposals for terrestrial biodiversity (e.g. Te Ara o Te Ata Mt Messenger Bypass, Te Ahu a Turanga – Manawatū Tararua Highway, the proposed Auckland Regional landfill at Wayby Valley). However, the disaggregated model has been criticised by some users (Baber et. al. 2021) on the basis that the level of data required for its parameterisation, commensurate with the complexity of affected biodiversity, can be large. Despite rapid advances in both available technology and tools to support model development in the intervening years, no further advances in offsetting modelling have been made in Aotearoa/New Zealand.

In 2021 a Biodiversity Compensation Model (BCM) (Baber et al. 2021) was developed as an alternative and simpler version of existing offsetting accounting models. The BCM relies on an aggregated qualitative habitat score to make inferences about the adequacy of proposed management actions to generate sufficient biodiversity gains. The model structure and data requirements mean that the BCM cannot demonstrate NG or NNL of biodiversity. These limitations occur for several reasons. For example, rather than the use of disaggregated quantitative measurements on a range of habitat and vegetation characteristics that describe a particular habitat type, a BCM would include an aggregated qualitative biodiversity value score. This could, for example, be a single value ranging from 1 to 5 to indicate a subjective assessment of habitat quality for a particular species or assemblage of species. However, in some cases (e.g. the Te Kuha coal mine proposal on the West Coast, and the proposed Auckland Regional landfill at Wayby Valley) the BCM has been used to support a "likely" NNL or NG outcome without the rigour of a quantitative offset model. A case study is presented below to clarify how these types of structural issues in models can lead to biodiversity loss if model outputs are provided to decision-makers without a comprehensive explanation of their limitations (Walker et al. 2009).

Necessary characteristics for robust predictive models

Evaluating ecological equivalence when exchanging biodiversity components requires models that accommodate measures of quantity and quality of target biota at impact and offset sites, both pre-impact and post offset. Thus, any loss-gain model will include a component of estimating a future state. Predicting the future is difficult and uncertain and thus models used for biodiversity offsetting or compensation must be designed and implemented in a manner that accounts for these inherent risks.

Predictive models should be accurate, robust, and transparent regarding their assumptions and objectives (Bodner et al. 2020). Depending on the availability of data and knowledge and the precise question, various modelling approaches can be used. However, good ecological models that are useful for conservation management all share a similar set of attributes (Schuwirth et al. 2019). Firstly, a mechanistic understanding regarding causality is needed. We should also expect that models have clear and stringent data requirements as the use of biased data can lead to biased results and it should be the responsibility of the model builder to give users explicit guidance on this. Secondly, there needs to be an alignment of model inputs and outputs with the response of target biodiversity to management interventions. Model predictions need to be translated to meaningful outputs (e.g. quantity of planting required to create specified extent of desired habitat). These then can be used to meet defined outcomes necessary to offset losses (e.g. the structure, composition, and condition of created habitat) or to provide forecasts for decision-makers (Pollock et al. 2020). Models also need to incorporate appropriate spatial and temporal resolutions; a balance is required between the scale of the management problem and the level of ecological detail needed. Importantly, any model that creates predictions in complex systems such as ecosystems must have a robust and transparent way of evaluating uncertainty. Models also need to have sufficient predictive performance, i.e. be sensitive enough for the relevant management action and be transferable to a variety of situations. Lastly, if models are to effectively guide biodiversity offset or compensation management interventions, outputs must be easily translatable to non-statisticians, including those who monitor and adaptively manage outcomes against model predictions.

Appraising the performance of complex ecological models can be challenging for users and decision-makers. To answer the question "is this a good or useful model?" Saltelli et al. (2020) distil five simple principles. The first principle is to mind the assumptions, which is crucial to assess both uncertainty and sensitivity within the model (especially true when models use qualitative data). The second principle is to mind the hubris; additional complexity is not always useful, because as more parameters are added, uncertainty builds, and error may increase to the point at which predictions become useless. Thirdly, the framing of the model needs careful consideration; end users need to be aware that results from models will at least partly reflect the interests, disciplinary orientations, and biases of the model developers. When a new model is developed, it must be validated and verified. The fourth principle is to pay attention to consequences; when presented with a number, audiences may ignore other possible explanations or estimates. Thus, a full explanation and justification are needed to accurately interpret results and should always be expected. The last principle is to mind the unknowns; transparently communicating what is not known is crucial as models can mask ignorance.

Model developers can help end users and promote the standardisation and transparency of model evaluation by using methods such as the OPE (objectives, patterns, evaluation) protocol (Planque et al. 2022) to document model evaluation. We suggest that models must be accompanied by user manuals that provide guidance on how to present model findings in the context of model limitations and include rules to address potential concealed losses (biodiversity elements that are not explicitly accounted for and may be lost in the exchange). It is critical that ecological concepts (e.g. linkages or dependencies within ecosystems) are appropriately incorporated into the model structure, that qualitative data inputs are transparent and rigorously developed (e.g. via structured elicitation), that uncertainty estimates are included, and that a transparent reporting structure is created that includes a discussion of outputs in the context of the model's limitations. Poor models avoid substantive discussion on parameterisation choices (a critical aspect of robust modelling), provide insufficient clarity on output interpretation (i.e. currency limitation), and/or compare different types of biodiversity in the absence of accepted methods for such comparisons. In addition, we suggest that application of the effects management hierarchy is undermined if models are used to support compensation proposals where offsetting is feasible. The utility of mathematical models is predicated on the ability to answer two key questions: (1) whether the mathematical structure of a model is sufficiently realistic, and (2) how that model should be parameterised to best represent reality (Stouffer 2019).

Case Study

It is possible, and even likely that, poorly designed biodiversity models will facilitate biodiversity loss when relied upon by decision-makers (Walker et al. 2009). Here we present an example from Aotearoa/New Zealand of the risks of relying on a model that is not robust. In 2017 the Buller District Council and West Coast Regional Council granted a resource consent to allow the development of a new coal mine at Te Kuha, on the West Coast of the South Island, Aotearoa/New Zealand. This decision was appealed and subsequently overturned by the Environment Court in 2023 (Royal Forest and Bird Protection Society of New Zealand Inc. v West Coast Regional Council and Buller District Council [2023] NZEnvC 68). Although the application for consent was ultimately declined, the case provides a useful illustration of the application of the BCM (Baber et al. 2021) to support an effects management package (a combination of all measures to address adverse effects on biodiversity including avoidance, mitigation, offset and compensation measures) proposed as part of the resource consent application.

Expected ecological impacts from the development included the loss of ecologically significant undisturbed coal measures vegetation, the largest known population of the endemic At Risk (Relict) forest-ringlet butterfly (*Dodonidia helmsii*) and the potential loss of the only known population of a leaf veined slug (*Pseudoneitea* sp.) which is yet to be formally described and assigned a threat classification. The applicant proposed compensating for these difficult-to-offset losses through an out-of-kind exchange involving enhancing populations of several Threatened or At-Risk avifauna species by undertaking predator control. The BCM was used to guide the compensation proposal, but the quantum of compensation required was not estimated due to an absence of data. In the Te Kuha case, the design of the BCM obscured the out-of-kind trade as the BCM output is a unitless percentage that has only relative meaning and is difficult to interpret. The use of this unitless output misrepresented the fungibility of the different types of biodiversity and did not guide the court to questions that were in fact critical to evaluating an out-of-kind exchange. Namely: how many tūī (*Prosthemadera novaeseelandiae*) equal a forest ringlet butterfly? In addition to actively contributing to decreased transparency in the Te Kuha case, this output type violates several of the necessary characteristic of robust models detailed above.

The BCM for Te Kuha predicted large increases in avifauna populations within the offset sites as a result of the proposed predator control (e.g. 112% for mātātā South Island fernbird (*Bowdleria punctata punctata*; At Risk – Declining), 188% for tūī (Not Threatened) and 431% for kakaruai South Island robin (*Petroica australis*; At Risk – Declining). Subsequent sensitivity analyses showed that the calculated NG outcomes for all bird species were sensitive to minor fluctuations in inputs (e.g. ± 10% of the input data) and the outcomes predicted were therefore speculative. Large gains predicted by the BCM lost their weight when accompanied by error estimates. For example, gains of mātātā fernbird 112 ± 139%; tūī 188 ± 209%, and kakaruai robin 431 ± 326% indicated that the project could not ensure that increases in habitat quality would result in increases in avifauna populations. The sensitivity testing in this case study illustrates how the absence of uncertainty and sensitivity estimates in the BCM (i.e. not "minding the assumptions") impedes the ability of decision makers to assess the real-world consequences of modelled trades. This model limitation was further illustrated during the Court proceedings, as back calculations were used to convert unitless percentages to estimated 5-Minute Bird Count (5MBC) measures. Estimates of "real-world" numbers were less impressive than model outputs: fernbirds were predicted to increase from 0.75 to 0.85 individuals per hectare; tūī calls would increase from 0.40 to 0.42 per 5MBC; and robin calls would increase from 0.10 to 0.11 calls per 5MBC over the compensation area. These alternative presentations of the same model predictions are likely to influence decision-makers towards different biodiversity outcomes, highlighting the consequences of using models that do not adhere to principles of good model building and reporting.

The inputs required for the model additionally violate several requirements for robust models. Rather than using detailed measurements to describe a habitat or vegetation characteristics, the BCM required an expert to provide a subjective habitat value score (e.g. ranging from 0 to 5) for losses on the development site and gains at the restoration site. The shortcomings of the data required by the BCM resulted in several issues in the case. Firstly, the BCM did not provide clear and stringent data requirements, which resulted in insufficiently substantiated debate regarding input values. Secondly, the BCM did not require mechanistic causality to drive the model. The output was described as a quantified Net Gain for each species, despite calculations solely considering changes in the qualitative rank score of the species' habitat, not a quantifiable gain in the biodiversity of interest (e.g. abundances of these avifauna species) and the relationship between these was not validated. Thirdly, the BCM did not address the uncertainty associated with biased data; likely to be an issue when relying on subjective, poorly corroborated data. These limitations were built into the structure of the BCM. The model user guide does not provide sufficient guidance to facilitate the discussion of outputs within the context of model limitations.

The use of the BCM in the Te Kuha appeal was contentious throughout the proceedings and was ultimately set aside by the Environment Court on the basis that the Court was not the forum to resolve technical debates (Royal Forest and Bird Protection Society of New Zealand Inc. v West Coast Regional Council and Buller District Council [2023] NZEnvC 68). Ultimately, the Court found the ecological effects of the proposal outweighed the anticipated positive effects and the consent was declined.

The Te Kuha case study illustrates some of the risks that poor models pose to biodiversity, and the need for ecologists to use better discipline with models. Depending on how they are designed and used, models can considerably undervalue existing biodiversity and overvalue certain management interventions. Bias in models can lead to consistently negative ecological outcomes across many development projects. Omissions, miscalculations, and directional biases in the assumptions of a model can aggregate to large errors in predictions. Given the cost of parameterising complex ecological models, there is an argument for the use of simpler models that work as an aid or sense check to indicate the likely magnitude of effort required to address residual adverse effects. However, even preliminary models must provide calculations that are more likely to be correct than incorrect. Although not the case in this instance, it is conceivable that decision-makers may be inclined to place undue weight on predictions in situations where uncertainties and assumptions are obscured in the presentation of model outputs. As indicated by the Te Kuha case study, not all models currently used by practitioners in Aotearoa/New Zealand produce reliable calculations. Thus, a general upskilling of practitioners in model use and evaluation would be beneficial. If biodiversity offsetting and compensation remains an important component of effects management in Aotearoa/New Zealand, accurately quantifying the type and amount of offset or compensation required is a research priority.

The development and adoption of models into the future

Based on technological trends, an improvement in accuracy and reliability of biodiversity offsetting models should be expected. It is now easier than ever to develop and access models. Consequently, the use of complex statistical analyses in ecology has proliferated and the number of model users will continue to increase (Seidl 2017). This is partly due to the increasing availability of specialised statistical libraries and statistical software platforms such as R (R Development Core Team 2016). Thus, the use of models of insufficient quality is a regressive step both within the context of ecological modelling and of progressing best practise offsets and compensation to decelerate biodiversity loss in Aotearoa/New Zealand.

To date, mathematical models have made enormous contributions to conservation biology. Models have been used to anticipate the risk of extinction (Akçakaya & Sjögren-Gulve 2000), map species distributions (Rodríguez et al. 2007), predict the course of biological invasions or diseases (He et al. 2019) along with answering many other questions. In fact, modelling biological systems to compare different management strategies can be just as useful as conducting additional field trials or gathering more empirical data collection (e.g. Bertolino et al. 2020). The recent developments in ecological modelling illustrate that loss-gain models can be developed to enable the “back-end” of the model to be sufficiently complex and ecological rigorous, while providing a user-friendly, transparent “front-end” (Marxan, Watts et al. 2009). Funding bodies should recognise biodiversity accounting modelling as a critical research gap with significant real world biodiversity implications. Any future developments of new biodiversity offsetting models will require an appropriate level of peer review. Ideally this is completed prior to submitting model outputs in support of development applications, to ensure biodiversity outcomes sought are not compromised by poor model structure and misapplication of mathematical principles.

Conclusion

The widespread acceptance and use of poorly designed ecological models will result in real biodiversity loss in Aotearoa/New Zealand and will contribute to the global biodiversity crisis. We suggest that any efforts to simplify or discount the costs of effects management be treated with caution and subject to scrutiny. While the development of better suited models is urgently needed, we recommend the following in the interim:

(1) Models which facilitate compensation over attempting offsetting in the first instance are not accepted.

(2) The use of overly aggregated currencies, metrics, and models is discouraged, particularly for complex, vulnerable, or culturally important biota and ecosystems.

(3) Models designed to test compensation outcomes are not used to suggest NNL or NG outcomes.

(4) Model outcomes are provided together with estimates and discussions of uncertainty.

Models are necessary to assist decision-making of complex or uncertain biodiversity trades and the appropriate use and interpretation of peer-reviewed, ecologically robust, and mathematically reliable models is usually preferable to not using models. However, there are specific characteristics of models that are essential to reduce the risk of falsely predicting, or misrepresenting, the outcomes of a biodiversity offset.

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