Choosing prevention or cure when mitigating biodiversity loss: trade-offs under 'no net loss' policies

Joseph W. Bull* [a, b], E.J. Milner-Gulland [c]

[a] *Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Canterbury, CT2 7NZ, U.K.*

j.w.bull@kent.ac.uk; +44 (0) 7837 172 886 (corresponding author)*

[b] *Department of Food and Resource Economics & Center for Macroecology, Evolution and Climate, University of Copenhagen, Rolighedsvej 23, 1958 Copenhagen, Denmark.*

[c] *Department of Zoology, University of Oxford, Oxford, OX1 3SZ, U.K.*

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Abstract

1. Biodiversity cannot always be conserved. Economic development activities can result in biodiversity losses, but also increase human wellbeing, so trade-offs must sometimes be made between conservation and development. An alternative strategy to avoidance of impacts through the strict protection of biodiversity (‘prevention’) is to permit certain biodiversity losses and fully compensate for them through offsets elsewhere (‘cure’).
2. Here, we build a stochastic simulation model to explore trade-offs between biodiversity loss prevention and cure, in the context of development under ‘no net loss’ (NNL) biodiversity policies. Our model implements a Management Strategy Evaluation framework, monitoring outcomes using four different performance metrics: total biodiversity, net biodiversity, total economic activity, and development activity.
3. We find that a "cure" strategy can potentially perform just as well as a prevention strategy in terms of biodiversity objectives, whilst outperforming the latter from an economic perspective. However, this does not undermine the need for a mitigation hierarchy, and the best-performing strategy depends strongly upon both the degree of compliance with the NNL policy and upon underlying ecological parameters.
4. Perhaps counterintuitively, when evaluated as advised by the technical literature (i.e. against an appropriate counterfactual scenario), we find that net biodiversity outcomes are highest when natural ecosystem recovery rates are slow (so long as development rates are also slow).
5. Finally, using the illustrative example of US wetlands, we suggest that real-world NNL policies could already be driving landscape-scale avoidance of development impacts under a "prevention" approach.
6. *Policy implications*. No net loss (NNL) biodiversity policy is currently being developed or implemented by over 100 countries worldwide and incorporated into environmental safeguards by multinational lenders. The socio-ecological model presented here can be used to advise decision makers about the best structure for nascent NNL policy on the basis of region-specific ecosystem recovery rates, development activity, legal compliance and monitoring uncertainty. Further, the model presents a means for

estimating the degree to which biodiversity impacts are avoided by developers under NNL – an important monitoring consideration given that ensuring high levels of avoidance is crucial to robust NNL policy, but which has to date evaded assessment through purely empirical means.

Keywords: Biodiversity, biodiversity offsets, development, impact avoidance, management strategy evaluation, net gain, no net loss

Introduction

It would be unrealistic to suggest that all biodiversity should be conserved at any cost. Economic development activities result in biodiversity losses (Venter et al., 2016), but also often increase human wellbeing – and, given the resultant ethical dilemma, there are trade-offs to be made between achieving ecological and social objectives. An alternative strategy to the outright protection of biodiversity (‘prevention’ of losses) is to permit certain biodiversity losses, but compensate for them with ecological gains elsewhere (‘cure’ for losses). Our question, then, is when is it best to protect extant biodiversity and when is it best to compensate for its loss? This question goes to the heart of emerging environmental policies designed to fully compensate for biodiversity impacts associated with development: ‘no net loss’ policies (Arlidge et al., 2018).

Conservation interventions characterised by an overarching policy objective that seeks no net loss (NNL) of biodiversity or better are being implemented far and wide (Maron et al., 2016). NNL interventions are implemented in response to ecologically damaging economic activities, requiring anticipated biodiversity impacts to be quantified and then either prevented or fully compensated for. NNL thus involves some calculation of development-associated biodiversity losses and gains, and some demonstration that gains balance losses (usually with a multiplier built in such that expected gains overcompensate expected losses, as a buffer against uncertainty; Moilanen et al., 2009). NNL is generally delivered through the application of a mitigation hierarchy, which involves sequentially seeking to avoid, minimise, remediate and finally offset any predicted impacts (Gardner et al., 2013).

The preventative stages in the mitigation hierarchy (avoidance and minimisation) are often considered preferable from a conservation stakeholder standpoint (Lindenmayer et al., 2017; Phalan et al., 2017). But it cannot be assumed that avoidance of impacts always leads to the best outcomes for nature conservation under NNL (Bull et al., 2014) or more generally (Possingham et al., 2015). Moreover, avoidance, if leading to constraints on development activities, may not lead to optimal outcomes for social wellbeing once the full range of costs and benefits are taken into account (e.g. Bidaud et al., 2017). A crucial line of enquiry for NNL is therefore to explore the degree to which a balance can be struck between preventative (avoidance, minimisation) and compensatory (remediation, offsets) actions. This makes NNL policy a good testbed for exploring the balance between biodiversity prevention and cure more generally.

Empirical data on the balance between prevention and cure in the implementation of NNL policies are focused primarily on the final stage of the mitigation hierarchy, biodiversity offsetting. Offset data are typically poor quality and incomplete, and this is even more true when it comes to data on prevention measures (Bull et al., 2018; Bull & Strange, 2018; Sonter et al., 2018). Partly this is because NNL is implemented in complex socio-economic systems, where it is difficult to attribute causation e.g. for why a particular area of land was or was not developed, and so say whether this was as a result of a prevention strategy or coincidental. Anecdotally, some authors consider it likely that avoidance measures contribute substantially towards achieving NNL at the policy scale, but this is usually unrecorded (e.g. Levrel et al., 2017). Overall, it is difficult to use existing data to make empirical judgements concerning the extent to which NNL policy drives avoidance of development impacts.

Given this difficulty, there is a need to turn to alternative approaches to establish the degree to which impacts have been prevented, such as simulation modelling approaches. One potentially appropriate modelling framework in this regard is Management Strategy Evaluation. The Management Strategy Evaluation (MSE) conceptual framework underlies simulation-based decision support tools that have been applied extensively in the context of marine fisheries (Plagányi et al., 2013; Fulton et al., 2014). MSE has been employed in terrestrial situations (Chee & Wintle, 2010; Bunnefeld et al., 2011; Milner-Gulland, 2011), but never in relation to NNL policy. A strength of MSE is that it can be used to model the response of a natural resource stock to management whilst also including the actions of the managers themselves – both in monitoring the resource, and setting rules relating to its exploitation. Thus, the performance of different management strategies can be compared in the context of both the inherent system uncertainties, and against various stakeholder objectives (Milner-Gulland, 2011). Indeed, MSE provides a useful model framework when “policies are sought that are feasible, robust to uncertainty, and which provide adequate management performance with respect to multiple criteria” (Bunnefeld et al., 2011).

The MSE framework is composed of a feedback loop between four components: an operating model (the dynamics of a system, for example a natural resource and its users), an observation model (the process by which the system is monitored), an assessment model (describing how managers use the information generated by observation to set management rules) and an implementation model (in which the rules are applied to the system). When translated by Milner-Gulland (2011) to the terrestrial realm, the MSE operating and observation models were extended beyond the resource alone (traditional in fisheries applications) to include social components of the system. In turn, versions of the framework that incorporate subsistence alongside commercial harvesters have been developed for fisheries (Plagányi et al., 2013). More recently, MSE has been proposed as a possible means for building consensus in developing international conservation targets (Maxwell et al., 2015).

Here, we treat terrestrial biodiversity as a natural resource, and use MSE to compare prevention (avoidance) versus cure (offset) measures as strategies for conserving biodiversity under NNL. In so doing, we search for robust approaches to the implementation of the mitigation hierarchy that balance competing objectives, whilst acknowledging the substantial uncertainties inherent in operationalising NNL. Subsequently, we illustrate how the model might be applied to evaluate the behaviour of a specific real-world compensatory biodiversity policy, using the example of US NNL wetlands policy.

Materials and Methods

Simulation model

We use analogues for traditional components of the MSE conceptual framework to capture the approach taken in implementing NNL policies (Fig. 1). The key components of our model are thus the operating model (biodiversity dynamics), the assessment model (the policymaker, who sets and maintains rules on the basis of observations to deliver the relevant strategy), the implementation model (developers, who we assume both carry out development activities and implement associated mitigation), and the observation model (monitoring by the policymaker, of both biodiversity and developer activities). Following Milner-Gulland (2011), we differentiate between monitoring *within* the model (by the policymaker) – which is subject to uncertainty both in terms of the data available to the policymaker and inherent stochasticity – and the performance metrics P1–P4, which represent the 'true' state of variables within the model (Fig. 1).

Our focus is the degree to which the policymaker requires prevention of impacts versus cure for impacts. The set of strategies is therefore on a spectrum between total avoidance and total permission of impacts (with full compensation required). Given a known starting point for all model parameters, our model applies all MSE components once consecutively during each time step, and then repeats for 100 further time steps (unless otherwise specified). Change in variables is monitored throughout each simulation, and at the end of that specific simulation the final value for the four performance metrics is recorded. The simulation is repeated 50 times, and the mean final value of each performance metric is reported. Model components are expressed in a set of dynamic equations coded in R (R Development Core Team, 2017) (Bull & Milner-Gulland, 2019), as follows.

<u>Biodiversity</u>

'Biodiversity' B represents some hypothetical component of an ecosystem of interest. In reality this might be e.g. species, habitats, some composite measure. It is common for policymakers to use a single indicator to track losses and gains under NNL policy (Quétier & Lavorel, 2011), making the use of a single variable B appropriate for biodiversity here, however, the model could readily be expanded to track multiple biodiversity indicators. B takes a normalised value between 0 and 100. At time t = 0, biodiversity (B_0) = 99, i.e. biodiversity is very high at the beginning of the simulation, to model the introduction of NNL policy to a yet to be heavily exploited landscape. Note that in many real world cases NNL policy is applied to historically heavily modified and low biodiversity landscapes (Bull & Strange, 2018) – a situation which we capture in one of our scenarios and in the US wetlands case study (see below). The inherent trend in biodiversity is then for it to recover, and to do so following a logistic equation – a reasonable assumption for either species or habitat recovery (Mace et al., 2008; Gordon et al., 2011).

$$B = \frac{K.B_0.e^{r.t}}{K + B_0(e^{r.t} - 1)}$$

Equation (1)

$$\frac{dB}{dt} = r.\left(1 - \frac{B}{K}\right).B$$

Equation (2)

Where: B = biodiversity; t = time; r = intrinsic growth rate; K = carrying capacity = 100.

The independent model parameter governing biodiversity is the biodiversity recovery rate *r* (equivalent to the intrinsic rate of population increase in population dynamics). Note also that the magnitude of the incremental change in biodiversity for each step is subject to some stochasticity, which we achieve by randomly selecting a value at each time step governed by a normal distribution (standard deviation = *0.2 x dB/dt*) of the deterministic incremental change in biodiversity that would be realised under equation (2) for that time step.

Monitoring

The state of biodiversity is monitored by the policymaker with some uncertainty. Observed biodiversity is randomly selected at each time step from a normal distribution around 'true' biodiversity (standard deviation = 10). The policymaker also monitors the perceived outcomes of mitigation achieved by developers – but this is subject to uncertainty due to the level of compliance demonstrated by developers. Compliance with the policy on the part of the developers is unknown but important, and in many real world systems is likely rather less than 100% (Bull et al., 2014). Therefore, we conservatively choose a value for minimum proportional implementation (baseline scenario = 0.3) and randomly select a value between this and a value for maximum proportional implementation (baseline scenario = 0.5) for each time step, meaning that somewhere between 30-50% of required mitigation measures are actually implemented in practice by developers. The policymaker considers the requisite amount of mitigation to have been implemented, but the 'true' amount implemented is rather less. There are multiple additional sources of uncertainty that might arise in a real system

(Kujala et al., 2012), but we assume those already mentioned to be the key sources with respects to our simulation model (Bull et al., 2014).

<u>Policymaker</u>

The policymaker sets development constraints and mitigation requirements for the subsequent time step, based upon their observations of the state of biodiversity, guided by their overall strategy (see below). Requirements applied to developers are either to: (a) develop without mitigation; (b) develop but offset any associated impacts; or, (c) avoid development impacts entirely. The requirement is determined by biodiversity thresholds chosen for triggering (a – c), which are independent model parameters representing a different weighting applied to avoidance vs. offset strategies:

$$\Delta impacts_{t+1} = \begin{cases} \Delta impacts_{t+1}\ if\ (B_t > T_M)\ \Rightarrow offsets_{t+1} = 0 \\ \Delta impacts_{t+1} = -offsets_{t+1}\ if\ (T_M > B_t > T_A) \\ 0\ if\ (B_t < T_A)\ \Rightarrow offsets_{t+1} = 0 \end{cases}$$

Equation (3)

Where: T_M = threshold for requiring mitigation measures; T_A = threshold for requiring avoidance measures; offsets = amount of compensation implemented.

The main independent model parameters governing policymaking are therefore T_M and T_A, with the latter being the focus of attention in this study. Indeed, we set T_M = 99 for all simulated scenarios reported here (meaning that all biodiversity impacts are subject to mitigation measures of some kind) as our interest is in trade-offs *within* the mitigation hierarchy. However, T_M is retained for future implementations of the model. Note also that we do not incorporate multipliers into this version of the model (i.e. factors >1 applied to compensation requirements, which are common in many policies; Moilanen et al., 2009) as we prefer that our findings are conservative in terms of conservation outcomes, although to do so in future versions of the model would be trivial.

<u>Developers</u>

We assume that development tends to impact negatively upon biodiversity, and that over the course of the simulation (100 time steps) development would, if permitted to carry on as usual, eventually lead to a total clearance of biodiversity at some time $t = 1 \rightarrow 100$. It is also assumed that the magnitude of development impacts decreases as the biological component in question becomes scarce across the entire system (an exponential decay function; equation 4). Such an assumption is consistent with development impacts that do not specifically *target* biodiversity, but instead create residual impacts upon it – meaning conceptually that development is simply constrained over time, as human development requirements reach equilibrium. In turn, simulated impacts do not currently extend to exploitation of specifically targeted resources (e.g. fish stocks), consistent with current NNL policies (Arlidge et al., 2018).

$$B_t = B_0 e^{-\lambda . t}$$

Equation (4)

Where: λ = exponential decay constant; B_0 *= initial biodiversity = 99.*

The equation governing development impacts upon biodiversity for each time step is the differential of equation (4):

$$\Delta impacts_{t+1} = \frac{dB_t}{dt} = -\lambda . B_t$$

Equation (5)

The main independent model parameter governing development impacts is the exponential decay constant (*λ*).

The decision about whether to develop in any time step, apart from in cases of non-compliance, depends upon the policymaker. If no mitigation is required ($B > T_M$), then development is implemented with negative biodiversity impacts. If offsets are required ($T_M > B > T_A$), then development goes ahead and the equivalent amount of compensation is required as an offset – the assumption being made that capacity exists within the simulated ecosystem to physically increase biodiversity somehow without reversing the development (see 'model applications'). A record is kept of the size of the necessary offset, and it is delivered (to the extent governed by compliance) over the subsequent time steps. The amount of offsets delivered during each time step is some proportion of the total offsets due (i.e. the ecological restoration debt) at that point in the simulation, a parameter that we vary within the model (for baseline scenario = 0.2, meaning the offset is delivered over a 5-year period). However, if no development is permitted and avoidance is required ($B<T_A$), then the associated development impacts are prevented for that time step. The 'true' negative impact upon biodiversity again depends upon compliance (and is therefore typically non-zero).

Note: the assumption in the general case is that avoided development impacts are genuinely avoided i.e. not simply shifted elsewhere. In reality, avoidance might mean that development is shifted to another region ('leakage'; Moilanen & Laitila, 2015) – but the focus of our model is on the impacts affecting some subcomponent of biodiversity more broadly, so we ignore leakage here (again, see 'model applications'). Further, we have assumed for clarity that development impacts upon biodiversity are primarily negative, and those of conservation interventions are primarily positive – both assumptions are not strictly true in general for real world impact mitigation, but are reasonable approximations to make in the interests of not initially overcomplicating the model.

Performance metrics

We capture four performance metrics, representing the 'true' state of the system after 100 time steps (Fig. 1).

P1: <u>Absolute value of biodiversity</u>

The main objective for conservationists is likely the absolute value of biodiversity remaining in the system (i.e. B).

P2: <u>Net value of biodiversity</u>

From the point of view of NNL policy, the key outcome is net value of biodiversity compared to some counterfactual (Gordon et al., 2011; Bull et al., 2014). Here, our counterfactual is the state of biodiversity had there been no anthropogenic intervention whatsoever i.e. the 'no development' counterfactual. Given our experimental set up, such a counterfactual scenario would mean that if biodiversity value was high (initially, or otherwise) it would remain high, and if it was in any way depleted it would recover following Equation 2. In turn, the net value of biodiversity =

[final value of biodiversity – final value of biodiversity given the counterfactual trajectory in the absence of development and mitigation].

P3: Total development plus offset activity

From the perspective of actors in the system who are neither conservationists nor implementing the policy, a key metric is likely the total economic activity. So, our third metric = [sum total of development carried out + the sum total of offset activity carried out] (treating offsetting as an economic activity which provides utility to some actors, supported by fact that offset provision has become a high value market in countries with established offset policies).

P4: Total development only

For developers, it may be that the key metric is not total economic activity (including offsets) but the amount of traditional development only. For completeness, our fourth metric therefore captures the sum total of development.

Scenarios

The scenarios modelled are:

a) The baseline scenario, parameters specified throughout the Methods;
b) As (a), except biodiversity does not 'bounce back' if development stops (r = 0);
c) As (a), but with high compliance (minimum = 0.8, maximum = 1.0);
d) As (a), but with a lower starting value for biodiversity (B_0 = 10). We choose this otherwise arbitrary value for B_0 to represent a historically impoverished ecosystem.

The purpose of the initial scenario-based approach is to investigate and illustrate the difference in results obtained through various alternative implementations of the model. In particular, we are interested in the difference in outcomes across performance metrics, for a prevention (avoidance) versus cure (offset) approach, and for a range of values in r and λ (both of which we varied across several orders of magnitude).

We explore and contrast the outcomes across performance metrics P1 – P4 when varying key variables, including:

- Whether increasing compliance results in better or worse performance outcomes than shifting the avoidance and compensation thresholds;
- To what extent the policymaker can achieve better or worse performance outcomes by focusing on ensuring compliance; and,
- The range of possible performance outcomes given uncertainty about the ecological variables r and λ for recovering biodiversity.

Finally, we implement the model using parameters that approximate the real world US NNL wetlands case study, one of the first modern conservation policies to include a ‘no net loss’ requirement (for wetland area and function). US wetlands policy provides a useful illustrative application of the model, being one of the more mature NNL policies (with correspondingly high implementation spanning a number of decades) and having some of the most readily available data (Bull & Strange, 2018). US developments that would negatively impact certain physical characteristics of extant wetlands (in practice, mainly the area) must either seek to avoid those impacts through redesign, or compensate for them through wetland habitat restoration/creation elsewhere in the state. In the contiguous US, 53% of the wetlands present when European settlers first arrived have been lost in 200 years (Dahl, 2010), so we set B_0 ~ 47 and λ ~ 0.003. Recovery times for wetlands (in terms of ‘brackish’ systems returning to a pre-disturbance state on the basis of multiple indicators) are ~ 30 years (Jones & Schmitz, 2009), so we set r ~ 0.5 to give the appropriate restoration curve (equations 1, 2). We acknowledge that many US wetlands are not brackish systems, but again this analysis is intended to be illustrative – a more in-depth analysis would consider different wetland types with corresponding restoration curves. We run the simulation model for 50 years (approximate time the US wetland programme has been in operation; Maron et al., 2016). Assuming the approximate current area of wetland offsets as a percentage of overall area of wetlands in the contiguous US = 0.55 (calculated from Bull & Strange, 2018), we implement the model to estimate the value of T_A in the US wetlands case. The application of the model to US wetlands is illustrative only, being based upon secondary datasets subject to considerable uncertainties (Bull & Strange, 2018).

Results

Prevention vs. cure in the baseline case

In the baseline case (a), absolute biodiversity (P1) is not significantly influenced by the strategy chosen, and ends up greater than zero (though less than B_0) under a spectrum of strategies from prevention-focused to cure-focused. However, an avoidance strategy leads to slightly worse (more negative) net biodiversity outcomes P2. P2 is always negative, indicating a slight net loss rather than achievement of NNL. As the preference for avoidance exceeds a threshold of T_A ~ 50%, the total economic activity P3 begins to drop sharply (Fig. 2). The unimodal dip in net biodiversity outcomes at higher values of T_A, which consistently coincides with the value of T_A at which total economic activity begins to drop off, is probably related to the imposed downwards trend in economic activity and lag on restoration in the model.

Performance under other scenarios

Under scenario (b), characterised by a deteriorating background trend in biodiversity, the performance metrics demonstrated similar functional forms as in the baseline scenario (a). A comparable amount of economic activity was undertaken (P3), but there was a far worse final outcome for absolute biodiversity P1. Crucially, scenario (b) resulted in an extremely negative net outcome for biodiversity (i.e. a large *net loss*), P2. Similar outcomes were achieved under scenario (d), characterised by a very low value of initial biodiversity B_0. However, it was under scenario (d) alone that absolute biodiversity P1 was both higher at the end of the simulation than at the beginning, and also significantly higher using an avoidance strategy than an offset strategy (Table 1). This is a property of the logistic curve: absolute biodiversity gain is highest at 50% of the pristine natural biodiversity level (or carrying capacity, K), whilst the rate of biodiversity gain is highest when the amount of biodiversity B is very small, as the growth rate approximates the intrinsic recovery potential of the area (r). Therefore, it should not be surprising that offsetting outperforms avoidance as biodiversity approaches K.

Scenario (c) was the case in which developers demonstrated high levels of compliance with rules set by the policymaker. In turn, this scenario resulted in the best outcomes for absolute biodiversity P1. But in addition, it was only under scenario (c) that net biodiversity P2 was substantially positive (i.e. a net gain for biodiversity was achieved). Furthermore, this scenario resulted in the highest values for total economic value P3 (Table 1).

Under most scenarios, final absolute (P1) and net (P2) biodiversity outcomes were only marginally different under an avoidance versus an offset strategy (Table 1), but with a clearly identifiable minimum value (see Fig. 2). Outcomes were perhaps unsurprisingly significantly lower for an avoidance vs. an offset strategy for metrics P3 and P4 – that is, whilst absolute and net biodiversity outcomes (P1, P2) were comparable, economic outcomes were substantially worse.

Outcomes under varying compliance

Whilst always significant, the quantitative difference in performance between an avoidance and offset strategy for economic outcomes P3 was substantially less as compliance with the policy decreased. Avoidance strategies performed better against this measure (i.e. higher values for P3) as compliance decreased, whereas offset strategies performed worse (Fig. 3). Of note in relation to monitoring of outcomes by the policymaker: if the policymaker's expectation is that developers will largely comply with the policy, as modelled, then the gap between high and low compliance outcomes (Fig. 3) is also the gap between the policymaker's predicted and observed outcomes, respectively. When monitored using P3 as a metric, an avoidance strategy might in fact lead to better outcomes than the policymaker expects, as a result of poorer-than-expected compliance. The opposite is true of P1 and P2, the mean values for which are significantly different at the end of the simulation under a high (0.9) vs. a low (0.3) compliance rate (high compliance: P1 = 83.5±1.0, P2 = 10.2±1.0, n = 50. Low compliance: P1 = 61.5±0.5, P2 = -11.8±0.5, n = 50. Independent t-test; $p \sim 0$, $\alpha = 0.05$).

Outcomes for biodiversity when varying r and λ

Outcomes for biodiversity (P1, P2) are dependent upon the fixed parameters chosen for the model: the trend in development impacts (biodiversity decay rate λ), and the recovery potential of biodiversity (r). As might be expected, when λ is low, absolute biodiversity outcomes (P1) are high for all r, whereas high λ results in low absolute biodiversity outcomes P1 for all r (Fig. 4a).

Perhaps more counterintuitive are net biodiversity outcomes (P2; Fig. 4b). Net biodiversity is high and positive (P2 ~ +50) as both r and λ approach their lower limit (= 0.0001), but becomes marginally negative (P2 ~ –10) as r increases to the upper limit (= 0.63) since the counterfactual scenario in which there is no development is so high in relative terms. Equally, at large λ and small r, P2 is negative (P2 ~ –50), because there are major development impacts; and finally, as both λ and r increase to the upper limit, P2 reaches its minimum value (P2 ~ –90).

US NNL of wetlands

Simulation outcomes for the estimated values of T_A in US NNL wetland policy are shown in Figure 5a. To achieve the observed extent of wetland offset activity noted in the US, under these parameters, a high level of avoidance (T_A = 84-94% of wetland habitat by area) would be necessary (the model returns a value between T_A = 90-91, but post-hoc analysis suggests T_A = 90 is only significantly different from T_A = 83 and from T_A = 95). Notably, in all simulation runs the net outcome for biodiversity P2 in the US wetland case is marginally, but not significantly, negative, i.e. NNL is approximately achieved. In circumstantial support of the proposition that substantial avoidance has been taking place, US wetland extent has declined less rapidly in recent years, despite economic growth and population increase (Fig. 5b).

Model applications

The analytical simulation model we implement here is generalised, and does not reflect a real world case study (other than the illustrative application to US wetlands). It should be noted though that, in seeking to apply the model to case studies, certain constraints or modifications would need to be considered. We illustrate this here using three examples: a closer look at the aforementioned US wetlands case; the alternative example of bycatch in an industrial fishery; and, of infrastructure development in a region of pristine forest.

Example 1 – US wetlands: 'biodiversity' can be conceptualised in terms of the total area of wetland. All developments and wetland offsets take place within a continuous landscape, but the model considers only parcels of wetland nested within that broader mosaic. This means that 'avoided' development cannot be pushed into another wetland within the landscape, although the development can be pushed into a different habitat type (which we ignore in the development metrics, P3 and P4, as we are interested in development that has value in relation to wetlands only). Offsets can however take place elsewhere within the landscape, and be captured within metrics P1–P3, as wetland can be created (and since we are interested in amount of wetland, we count that and ignore loss of any other habitat types). *Constraint*: wetland offset opportunities are limited by the area of land available for wetland restoration outside of the model.

Example 2 – fishery bycatch: application of NNL to fisheries bycatch has recently been discussed (Milner-Gulland et al., 2018). 'Biodiversity' in this case could be conceptualised as the population of a specific bycatch species (e.g. albatross), with the application of the mitigation hierarchy to impacts upon that species. Avoided impacts on biodiversity in the bycatch case are genuinely avoided (i.e. no leakage to other albatross populations). Offsets would be implemented somewhere elsewhere entirely in geographical terms (e.g. restoring breeding colonies for that specific albatross population) but still increase the value of biodiversity within the model. In such a case, there is no immediate limit to the amount of offsetting effort that can be implemented to increase the albatross population, until that population reaches K. *Constraint*: extent to which opportunities exist for increasing bycatch species' population size.

Example 3 – pristine forest: consider the case of pristine forests, with biodiversity measured in terms of 'habitat condition x area' (condition-area) of forest (Quétier & Lavorel, 2011). If development impacts upon a pristine forest are avoided, they either do not happen or they leak outside the region of interest (impacting upon unrelated biodiversity in another region), and we do not consider them further. If development does take place in the forest, we reduce the condition-area of the forest. However, we now have no acceptable offset options, because the only possibility for offsetting within the otherwise pristine area of forest is to undo the recently implemented development. So the model would have to be modified, either: (a) reducing the threshold for requiring any impact mitigation whatsoever, such that a certain amount of development is possible before any mitigation is required, and then monitor which of these development impacts are reversible (to allow offsets); or (b) include an area beyond the forest within scope, say incorporating potential offset receptor sites (e.g. rangeland). In either case, there needs to be some constraint on the maximum possible development and offset opportunity, as biodiversity would be tied to a finite and limited resource (i.e. area of forest). *Constraint*: the degree to which undeveloped land is available for forest restoration within the model.

Discussion

Avoidance vs offsetting strategies

The choice of conservation strategy (represented by T_A) can be important in determining biodiversity outcomes under NNL. Our findings suggest there are multiple values of T_A that can give desirable biodiversity outcomes, but the values of T_A giving less desirable outcomes for biodiversity depend utterly upon the scenario and associated parameter values – i.e. it cannot be said that avoidance or offsetting will lead to better outcomes for biodiversity *in general*. This is important, given that conservation stakeholders will often tend to assume that prevention is the preferred strategy over cure (see also Possingham et al., 2015). Note that this conclusion applies to biodiversity that is replaceable or capable of recovering, not that which is irreplaceable (impacts upon which would ideally always be avoided).

Indeed, in the baseline scenario, we find not only that absolute biodiversity outcome is not significantly influenced by the choice of strategy, but also that under a high avoidance strategy there is potential for a lose-lose: with both net outcomes for biodiversity and total economic activity at a minimum. That is, for the baseline scenario, across all four performance metrics, the avoidance strategy performs *worse* than the offset strategy. It cannot be assumed that economic activity correlates with social utility nor that it always negatively impacts biodiversity, but for some real world regions both are likely. In these specific cases (but *not* in general) it would be rational to encourage offsetting over avoidance measures.

To heavily caveat the previous point, achievement of a positive outcome depends upon compliance with the NNL policy. The NNL policy in general only delivers a true ‘no net loss or better’ for biodiversity when compliance is high. This underlines the importance of compliance in implementing NNL policies, and demonstrates that a lack of ability to ensure policy compliance is much more likely to result in policy failure than uncertainty about biodiversity recovery, or even the type of conservation strategy chosen.

Absolute biodiversity outcomes are generally higher when the intrinsic biodiversity recovery rate is high and development rate is low, and outcomes are worse when the opposite is true. This is not unexpected. However, this does not translate into net biodiversity outcomes against the counterfactual scenario in which there is no development, which is technically the preferred means for evaluating outcomes (Maron et al., 2018). In the latter case, absolute development rate is not so influential upon outcomes as the interaction between development rate and inherent ability of biodiversity to recover – such that the best outcomes result when both biodiversity recovery and development rate are low, and the worst outcomes when both are high. Again, it cannot be assumed that systems with high development rates or low ecosystem recovery rates *necessarily* result in the worst outcomes for biodiversity.

Real world avoidance measures

Since relatively little attention has been paid to avoidance measures as a component of the mitigation hierarchy (Phalan et al., 2017), it is unknown to what extent NNL discourages impacts. Although some authors find the implementation of the Environmental Impact Assessment process to more often result in the reduction of impacts rather than their avoidance (Bigard et al., 2017), it is likely that avoidance plays a crucial, if under-reported role in the real-world realisation of NNL.

Whilst not empirical evidence, the application of our model to US wetland policy provides implicit support that avoidance measures could be making an overwhelmingly large contribution to US NNL outcomes. This would be consistent with qualitative suggestions that avoidance of impacts on wetlands, rather than wetland creation, is a key factor in US wetland impact mitigation policy (Dahl, 2010; Levrel et al., 2017). Note that real world applications of the model should be considered in the context of our many assumptions – the US result is entirely contingent, for instance, on our development function being valid.

Our simulation model outcomes are qualitatively robust to changes in scenario, yet they do demonstrate quantitative sensitivity to the value of underlying key parameters (e.g. degree of compliance, initial biodiversity) and assumed counterfactual (deteriorating vs. improving biodiversity trend; Table 1). Despite incorporating stochasticity throughout the model, our outcomes exhibit small confidence intervals. This is a shortcoming of the model: the apparently overwhelming strength of built-in deterministic trends in relation to stochastic components. Given these findings in combination, we would not use our model as a predictive tool for NNL strategies in different socio-ecological systems. However, the model provides a useful basis upon which key parameters and uncertainties can be considered. Therefore, our model has further applications both in terms of the scientific exploration of NNL, and in developing robust real world NNL policy.

Authors' contributions

J.W.B. and E.J.M.G. conceived of and designed the simulation model. J.W.B. built the simulation model, generated results and carried out statistical analyses. J.W.B. and E.J.M.G. wrote the manuscript. Both authors approve the manuscript for publication.

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Data availability statement

Simulation model code has been made publically available online (Bull & Milner-Gulland, 2019; DOI: 10.5281/zenodo.3490963).

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Table 1*: mean value for performance metrics P1 – P4 after 100 years, for scenarios (a) – (d), under an 'avoidance' strategy (T_A = 95) vs. an 'offset' strategy (T_A = 5). Light shading = significant difference between the mean value for prevention vs. cure strategy, for the same performance metric and scenario (independent t-test; t = 2.00, p < 0.01). Uncertainty bounds represent 95% confidence intervals (α = 0.05, N = 50).*

Scenario	**Key parameter values**	**Focus of strategy**	**Performance metric**			
			P1: absolute biodiversity	**P2: net biodiversity**	**P3: total economic activity**	**P4: total development**
(a) baseline	*B_0 = 99; r = 0.01; Comp.= 0.3-0.5; λ = 0.01*	*Avoidance*	65.6 ± 1.2	-7.7 ± 1.7	52.1 ± 0.3	51.3 ± 0.2
		Offsets	65.6 ± 0.1	-7.7 ± 0.1	99.0 ± 0.4	77.8 ± 0.1
(b) no recovery	*B_0 = 99; r = 0; Comp.= 0.3-0.5; λ = 0.01*	*Avoidance*	52.8 ± 1.3	2.6 ± 0.1	48.3 ± 0.2	47.6 ± 0.2
		Offsets	52.9 ± 2.0	2.7 ± 0.2	92.4 ± 0.4	72.3 ± 0.1
(c) high compliance	*B_0 = 99; r = 0.01; Comp.= 0.8-1.0; λ = 0.01*	*Avoidance*	83.6 ± 0.3	10.3 ± 0.3	31.3 ± 0.9	29.3 ± 0.6
		Offsets	85.0 ± 0.3	11.7 ± 0.3	133.4 ± 0.9	88.1 ± 0.2
(d) low initial biodiversity	*B_0 = 10; r = 0.01; Comp.= 0.3-0.5; λ = 0.01*	*Avoidance*	13.2 ± 0.1	-60.1 ± 0.1	7.0 ± 0.0	7.0 ± 0.0
		Offsets	12.7 ± 0.1	-60.6 ± 0.1	12.4 ± 0.1	10.1 ± 0.1

***Figure 1**: conceptual framework for the MSE simulation model, in the case of NNL policy. P1 – P4 = the four performance indicators; sc. a – sc. d = the four main scenarios modelled.*

***Figure 2**: outcomes of the baseline scenario, as T_A varies from 1 – 100 (x-axis; low T_A = cure strategy, high T_A = prevention strategy). Metrics are absolute biodiversity [P1], net biodiversity [P2], total economic activity [P3], total development [P4]. Error bars are 95% confidence intervals in outcomes ($\alpha = 0.05$, $N = 50$). Inset: small scale version of the plot for P2, showing functional form.*

***Figure 3**: plot of total economic activity [P3] against mean compliance, under a 'prevention' ($T_A = 95$) and 'cure' strategy ($T_A = 5$). Vertical error bars represent 95% confidence intervals ($\alpha = 0.05$, $N = 50$), horizontal error bars represent the simulated range in compliance rate (range = 0.2). All values of P3 were significantly different unless denoted on the figure (one-way ANOVA with post-hoc Tukey test, $\alpha = 0.05$).*

***Figure 4**: 3-dimensional plots of outcomes under variation in the independent variables r (biodiversity recovery) and λ (development impacts). (a) Absolute biodiversity outcome [P1], (b) net biodiversity outcome [P2]. Here, $T_A = 50$, $B_0 = 99$, average compliance = 0.4, $K = 100$.*

***Figure 5**: The US NNL for wetlands case study. (a) Proportional area of offsets created at the end of the simulation against variation in the value of T_A ($r = 0.5$, $\lambda = 0.003$). Dashed line = estimate of proportional wetland area occupied by wetland offsets; blue shade = values of T_A which are not significantly different from the value of T_A predicted by the model (one-way ANOVA with post-hoc Tukey test, $\alpha = 0.05$). (b) Recent change in wetland area across the contiguous USA (secondary y axis), human population and gross domestic product in 2009 USD, adjusted for inflation (both primary y axis). Inset: rho and p-values (Spearman's Rank Correlation). Data sources: US Fish and Wildlife Service, US Census Bureau, Council of Economic Advisors respectively.*