Systematic Nature Positive Markets

Abstract

Environmental markets are a rapidly emerging tool to mobilize private funding to support landholders to undertake more sustainable land management. One aim of such markets is to incentivize ecosystem restoration world-wide. How we measure and subsequently trade units of biodiversity within these markets creates key challenges both ecologically and economically, since it determines whether environmental markets will be ecologically successful in delivering net gains in biodiversity, and economically efficient in lowering the costs of conservation. Our innovation in this paper is to develop and then test a new metric for such markets based on the well-established principle of irreplaceability from Systematic Conservation Planning. Irreplaceability as a metric allows us to capture the multidimensional nature of ecosystems (e.g., habitats, species, ecosystem functioning) yet simultaneously achieve cost-effective, land manager-led investments in conservation. Using an integrated ecological modelling approach, we tested whether using irreplaceability as a metric is more ecologically and economically beneficial than the simpler biodiversity offset metrics typically used in net gain and no-net-loss policies. Taken together, our results demonstrate that irreplaceability can deliver no net loss of biodiversity, avoids the limitations of like-for-like trading, reduces costs of offsetting to developers and society, ensures land managers are fairly rewarded for the opportunity costs of conservation, and safeguards sites critical to achieving conservation goals. More generally, our study highlights the benefits of integrating economic data and approaches within Systematic Conservation planning as a means of incentivizing the most ecologically and economically efficient investments in nature recovery.

Keywords: irreplaceability, prioritization, offset market, biodiversity net gain

Article Impact Statement: Trading credits based on irreplaceability efficiently guides Nature positive investment across the complexity of ecosystems.

Introduction

More than 75% of the Earth's land is degraded, and this has led to widespread biodiversity loss, undermining the well-being of billions of people, as well as our efforts to combat climate change (IPBES 2019). Current evidence suggests multiple planetary boundaries have been exceeded (Steffen et al. 2015) and business-as-usual is highly likely to result in catastrophic collapse across many ecosystems (Armstrong McKay et al. 2022). Yet numerous global commitments to reduce, stop or even reverse current rates of biodiversity loss have not been met (Tittensor et al. 2014; Díaz et al. 2019). Instead, reversing global terrestrial biodiversity trends will only be achievable if we adopt strategic, coordinated, and above all ambitious, action (Leclère et al. 2020). To "bend the curve" toward a more nature-positive future, private sector funding of biodiversity conservation needs to be increased to complement longer-established publicly funded programs. The ENACT initiative (Enhancing Nature-based Solutions for an Accelerated Climate Transformation) launched at COP27 calls for the mobilization of private finance to support action on nature and climate-related targets the world over, accompanied by robust environmental and social safeguards (IUCN 2022).

Environmental markets are one such tool to mobilize private finance which can incentivize landholders to undertake more sustainable land management actions (Schmalensee & Stavins 2017). Such markets create income streams in the form of tradeable credits for landholders in return for undertaking actions to protect and/or enhance specified environmental goods and services, for example biodiversity, carbon sequestration or water quality. Demand (and thus buyer willingness to pay) for these credits can be voluntary, arising from a demand from individuals or companies who wish to offset their negative environmental impacts; or else are created through government regulation, for example through requiring developers to purchase credits to offset new house building (Needham et al. 2019; Radu et al. 2020).

In this paper, we focus on regulated markets for biodiversity offsets where developers must purchase credits to mitigate impacts on biodiversity as a result of development activities such as mine construction, housing, road building or hydroelectric dams. Credits are supplied by landowners who switch their current land management (such as arable farming) to a more conservation-orientated alternative (such as wetland creation). In regulated offset markets, state-sanctioned intermediary bodies such as offset banks validate credits and enforce offset requirements placed on developers. By establishing an appropriate rate of exchange between sellers (landowners) and buyers (developers), biodiversity offset markets can, in principle, achieve no net loss of biodiversity or a net gain within some defined area at the lowest overall economic cost to society, and are thus potentially economically efficient (Needham et al, 2019).

Within both regulated and voluntary nature markets, the choice of biodiversity metric plays a pivotal role in determining their ecological and economic performance (Simpson et al. 2021). This metric establishes the units in which biodiversity is traded, determining how a regulator or offset bank measures the gains in biodiversity resulting from restoration actions undertaken by landowners, and balances those against the expected biodiversity lost due to development impacts. Simple metrics based on a combination of the area and condition of habitat are often preferred by regulators (Bull et al. 2014; zu Ermgassen et al. 2019), easing the task of identifying matching biodiversity units, and assuming that habitat classes indirectly capture benefits on other aspects of the ecosystem (Marshall et al. 2020). However, numerous studies have demonstrated that these approaches rarely benefit biodiversity in the manner intended, or else fail to deliver gains in biodiversity in an economically efficient manner (Maron et al. 2012; Bull et al. 2014; zu Ermgassen et al. 2021).

In this study, we develop and then apply a new metric for application in biodiversity offset markets, and in environmental markets more broadly, that derives from the Systematic Conservation Planning (SCP) literature (Margules & Pressey 2000; McIntosh et al. 2017). SCP tools are designed to minimize the cost of achieving conservation targets. The importance of any specific site to achieving conservation targets is measured by its *irreplaceability*. A site that is essential to achieving targets is completely irreplaceable (and its loss could not be offset), whereas irreplaceability is low for sites which can be easily substituted for many others to contribute to conservation targets. Crucially, irreplaceability can aggregate the importance of a specific site over multiple biodiversity features, integrating the likelihood that actions are successful across space with ensuring that overarching targets for the whole landscape are achieved. This integration represents a step change away from like-for-like compensation regimes in existing biodiversity offset markets (for example BBOP 2009; Natural England 2022; NSW DPE 2022). Furthermore, if conservation targets are chosen to exceed their existing availability in a landscape, this embeds net-gain as an implicit outcome where this is needed to meet specific targets.

Our contribution is to demonstrate that an offset market steered by a metric derived from irreplaceability ensures the opportunity to achieve conservation targets is always protected, and results in the network of conserved sites selected being more economically efficient than that obtained using simpler offset metrics. Irreplaceability as a metric thus offers a step-change in the design of biodiversity offset and environmental markets, which is important given the current fast rate of expansion in such nature markets globally.

Materials and Methods

Irreplaceability recast for biodiversity offset markets

Systematic Conservation Planning is a rigorous, repeatable, and structured approach to designing protected areas that efficiently meet conservation objectives (Margules & Pressey 2000). At an analytical level, the task is a classic resource allocation problem that either maximizes conservation outcomes within a given resource budget, or else minimizes the cost of achieving specified conservation targets (Moilanen et al. 2009). This structure has led to the use of SCP in supporting conservation decisions across the globe (McIntosh et al. 2017). A key strength of SCP is that it can incorporate a wide variety of data types, including attributes of ecosystems at all levels of structural, taxonomic, and functional organization, as well as accounting for social, financial and political constraints and opportunities (Knight et al. 2011; Ban et al. 2013). Suitable targets are often based on the principle of adequacy, which aims to maintain the viability and persistence of those features (Kukkala & Moilanen 2013). Species-level targets may be informed by population viability analyses, or habitat-level targets by species-area relationships, and functional targets may be informed by our need for particular services across landscapes (Bryan et al. 2010). The value of any specific site is based on its *marginal* contribution to achieving the conservation targets by complementing what features are already secured. A key feature therefore of SCP is that, unlike ranking procedures, properties of reserve systems emerge from the combination of areas either through the complementarity of their composition, or by their connectivity in space. This suggests a strong potential advantage for using a metric derived from SCP within biodiversity offset markets, where a need exists to be able to compare ecological gains and losses across space between development sites (where biodiversity declines) and offset supply sites (where biodiversity is increased due to the action of the landowner). Moreover, a biodiversity offset metric needs to make sense in the context of an overall policy target of no net loss or net gain in a specific aggregate indicator of biodiversity. This combination of an aggregate target with the need to compare gains and losses across space suggests that a metric derived from SCP could have important advantages over the kinds of metrics investigated so far in the offset markets literature (Simpson et al. 2022).

Provided with data on feature values for all planning units, planning unit costs, and the desired targets for protection, systematic conservation planning tools identify which sets of sites deliver conservation targets most efficiently (Moilanen et al. 2009). For convenience, we refer to “features” and “planning units” as *species* and *sites* hereafter. Often targets can be achieved by many different combinations of sites because alternatives exist with similar, or at least complementary, values. The importance of any specific site to achieving conservation targets is measured by its *irreplaceability*. A site that is essential to achieving targets is irreplaceable (and its loss could not be offset), whereas irreplaceability is low for sites which can be substituted by many others. An exact calculation of irreplaceability rapidly becomes intractable as the number of combinations to test grows exponentially with the number of planning units (Pressey et al. 1993), and alternatives to estimate irreplaceability have been proposed (Ferrier et al. 2000). Most recently, Baisero et al. (2021) proposed a new metric for describing irreplaceability (α) that defines the extent to which a site *k* is essential for achieving the conservation of species *s* as:

$$\alpha_{k,s} = \begin{cases} 0 & \text{if } t_s = 0 \\ 0 & \text{if } t_s \geq R'_s \text{ } and \text{ } R_{k,s} = 0 \\ 1 & \text{if } t_s \geq R'_s \text{ } and \text{ } R_{k,s} > 0 \\ \min\left(\frac{R_{k,s}}{R'_s - t_s}\right) & \text{otherwise} \end{cases} \quad (1)$$

where the difference between the total availability of a species in the landscape R'_s and its target t_s indicates how much of that availability a site can contain ($R_{k,s}$) before it becomes irreplaceable. Baisero et al. (2021) defined β as the combined irreplaceability of a site by taking the complement of the product of replacement probabilities $\beta = 1 - \Pi(1-\alpha_{k,s})$. However, this constrains site irreplaceability to between 0 and 1, and consequently no longer indicates whether a site was irreplaceable for one or many species. To retain this distinction and make comparisons among sites within an offset market equivalent, we use summed α-irreplaceability. We note Ferrier et al. (2000) also summed irreplaceability in their study for a similar reason (albeit with a different formulation for each species), and therefore from now on our paper specifically refers to the sum of α-irreplaceability ($\sum \alpha_{k,s}$), which we abbreviate here to ∑α.

The biodiversity offset market

The structure of the biodiversity offset market was based on the model developed by Simpson et al. (2021). A single agent controls each land parcel or site within a landscape. Each agent decides to either develop their land for housing, generate biodiversity offset credits by adopting a conservation land management practice, or remains in the current land use of agriculture. For an agent to develop their land, each hectare acquired for new housing development requires a number of offset credits to be purchased from an offset provider equal to the measured biodiversity value of the site. The developer’s maximum willingness to pay (WTP) for an offset credit is determined by the expected value of land for housing development and the need to purchase offset credits. Ranking this WTP from highest to lowest yields a downward-sloping demand curve for offset credits. This WTP varies over space due to variations in house prices and the value of each site for biodiversity. We assume the offset credits are supplied by agents on agricultural land ("farmers"). Farmers change their current agricultural land management practices in a way which increases the biodiversity by a measured amount at the site. Every hectare given up to benefit biodiversity means one less hectare for agricultural production. Furthermore, the farmer may incur restoration costs in creating an offset credit. Therefore, the conversion cost to the farmer consists of the opportunity costs of the foregone agricultural output plus any associated restoration costs. This sum is the farmer's minimum price they will sell an offset credit for, known as their minimum Willingness to Accept (WTA). Since agricultural productivity and profits vary across space (due, for example, to variations in soil quality or site altitude), the minimum WTA of farmers to create biodiversity credits will also vary over space. Ranking farmers from lowest WTA to highest WTA generate a supply curve for offsets. Farmers and developers interact in this market to generate an equilibrium, market-clearing price for offsets where marginal WTP and marginal WTA are equal, that is, where supply for credits equals the demand for credits.

Simulation

Inputs: To demonstrate the operation of a biodiversity offset market using the ∑α-irreplaceability metric we simulated the probability of species occurrence within a 64 x 64 patch (or site) landscape. We used the R packages *NLMR* and *landscapetools* to control the degree of spatial autocorrelation in the baseline environmental gradient (Sciaini et al. 2018). Note however that α-irreplaceability is determined by the global availability of that species, not their distribution, and that the simulation of maps was solely intended to communicate the parallels with field-data and empirical models. We subsequently simulated three communities, each with 200 species whose distributions were either equally distributed across the environmental gradient, or moderately and highly skewed towards one extreme to produce an overall gradient in richness (Leroy et al. 2016). We ran offset market simulations based on subsets of species from each community, rising from 5 to 50 species and repeated 10 times each. More complex arrangements in response to multiple gradients are easily generated, but not considered further here. Likewise, we did not account for time lags or uncertainties in the ability of conservation actions to generate offset credits.

Four further pieces of information were generated for each site. The values of each patch of land for agriculture and for housing development were generated by defining their correlation to the environmental gradient (ranging from 0-1), although without a clear rationale for how these costs are expected to co-vary, both correlation coefficients were set to zero in our simulations. To reduce the likelihood that market trading stalls when WTA<WTP (and where therefore potential gains from trade still exist), the mean development value was set to double that of agricultural value. Next, each site is assigned to one of three initial land use classes: agriculture, conservation, and development in a 70:20:10 split. Lastly, a "habitat" layer is generated to indicate where habitat, and hence species, currently occur on agricultural land and in conserved sites to define the baseline from which "gains" should be compared. Agricultural land patches without habitat, but with suitable environmental conditions for species to occur, are treated as areas with restoration potential. The final inputs are the conservation targets for each species. To illustrate a scenario of net-gain, rather than no-net-loss, we set targets in all scenarios to be the equivalent of each species existing availability plus 20% of their restoration potential at agricultural sites.

Market: After each offset trade the ∑α-irreplaceability is recalculated for all sites. An agricultural site that is not irreplaceable for any species (all individual α<1) and has the greatest WTP per unit loss in the metric (£/∑α), is selected for development. If the development site ∑α is 0, either because the site has no species potential at all, or because all species with potential have already achieved their targets, then no offset is required. Otherwise, an offset site with the lowest WTA per unit gain in the metric (£/∑α) is selected and either all or a fraction of species values at that site are assigned to conservation status. The species values at the developed sites are removed from the global total R'_s and the values added by the offset are deducted from the remaining targets t_s. These steps are then repeated until all species conservation targets have been achieved, or there are no mutually beneficial opportunities to trade in biodiversity credits remaining (that is, a market equilibrium where for all remaining sites WTP<WTA).

Performance: To rate the performance of an offset market based on $\sum\alpha$-irreplaceability we compared the efficiency with which targets were achieved using alternative metrics for the same landscape. Firstly, the R package *prioritizr* was used to identify the exact optimal combination of sites that achieved all conservation objectives for minimal cost (Hanson et al. 2022). Secondly, the offset market was re-run using three alternative site-based metrics that increasingly reduced the need for the information involved in strategic planning. The first offset metric (OM1) weighted site scores by the inverse of each species range, thereby favoring the rarest taxa in the landscape (Crisp et al. 2001). OM1 scores were also continually updated to reflect changes in global availability due to the market. OM1 assumes the same degree of knowledge as required for the $\sum\alpha$-irreplaceability, but without setting targets. Updates to planning unit scores reflect species' global availability, but not complementarity to areas already protected. Offset metric 2 (OM2) is equivalent to OM1, but values for each planning unit are not updated over time meaning weights for each species were fixed at their starting value. This metric required the same initial understanding of species distributions but does not require an updating register of species affected by previous offset transactions. Finally, offset metric 3 (OM3) was based solely on how many species were present in each site, but not which species, meaning only a map of species richness would be required to guide a market.

The code and a full description of the results reported in the paper are provided in the supplementary material.

Results

Irreplaceability achieves conservation targets in an economically efficient manner

Our simulations demonstrated that using $\sum\alpha$ within an offset market resulted in continuous incremental progression was made toward conservation targets (Fig. 1a). The potential economic gains from trade were realized as long as developers WTP exceeded farmers WTA and this trading allowed all species to achieve their conservation targets. Economic gains from trade are initially high when trading first takes place (WTP>>WTA; Fig. 1b, note the log scale) but rapidly decline as more expensive and less irreplaceable offsets are required to meet demand. At each stage the market favored the greatest gains towards targets at minimal cost, making more likely an economically efficient solution to achieving the targets. Conversely, $\sum\alpha$-irreplaceability strongly dis-incentivized developments from taking place on land with high $\sum\alpha$ scores, because the number of offset sites typically required to replace their loss is typically prohibitive (Fig. 1c).

$\sum\alpha$-irreplaceability does not specifically prioritize sites that contain species rarely found in the landscape; it values sites based on the difficulty of achieving conservation targets without them. Nonetheless, as there are typically fewer opportunities to conserve rare species (i.e. low replaceability), sites that contain those species tend to score highly. In our model, once a species target was reached (green line Fig. 1d), their contribution to the $\sum\alpha$ of remaining agricultural was zero, meaning there was no benefit to its presence within new offsets, or cost associated with its occurrence at new development sites. Nonetheless, some species could eventually exceed their targets because they were present at offset sites added later to achieve targets of other species (Fig. 1a and d). As the $\sum\alpha$ contribution of species that have met their targets is zero, this reduces the burden for developers and increases their WTP for offsets at sites that contain species whose targets have been achieved (red line Fig. 1d).

Accounting for more species in the market does not necessarily increase costs, or require more offsets, or a greater conserved area to meet targets

The distribution of biodiversity, in particular the degree to which multiple targets overlap with others, determines the degree to which additional sites are required to protect additional species. As illustrated by our simulations (Fig. 2), the network is specific to the assemblage, and how the ecological community correlates spatially with economic land values. Our results showed that accounting for conservation targets of more species did not in itself increase the cost of conservation solutions, or require more trades, or more space to meet targets (Fig. 2b and 2c). However, in all cases, the wide variation in outcomes for small subsets of taxa illustrated the risks associated with conservation policies reliant on small numbers of indicator species whose suitability to represent the conservation needs of biodiversity and ecosystem processes is unknown (Yong et al. 2018).

Irreplaceability-led offsetting is comparable to optimal prioritization

Our modelling framework allowed us to compare site prioritization generated by SCP optimization with site selection through the ∑α-led offset market (Fig. 3). Site prioritizations generated by SCP were mathematically optimal, minimizing the cost of land needed to achieve all conservation targets. But rather than being reliant on landowners WTA, the SCP solutions assumed that regulators or planners have full control over site selection and management. This is rarely the case where much land is privately owned. Consequently, our results showed that if we assume developer's WTP is sufficient to support continued trading, the ∑α offset market could achieve all targets using very different networks of sites than the SCP solution (Fig.3b), and could even require fewer sites in total (Fig.3a), but the total cost of conserved sites are always equal to, or more likely, greater than SCP solutions. Our simulation results showed the total cost of sites selected for conservation in an ∑α offset market was only 2-11% greater than that using SCP, but that this gap narrowed as the numbers of species increased (Fig.3c) because the flexibility by which all targets could be achieved was reduced. Conservation solutions selected by the market were more expensive than networks selected by SCP because these minimize the total agricultural value of properties included (WTA), but do not consider whether the sites also provide high returns to developers (WTP). While high ∑α values and associated offset costs would incentivize developers to consider alternatives for development to some sites in the SCP network, if WTP is still sufficiently high to be profitable, then the offset market must settle for a more expensive complement of sites to replace them. The basis of SCP is that priorities are not simply cheapest or the most ecologically diverse, but those sites that best complement and add to what is already conserved. This principle ensures that given the changing constraints present at the time of trading, all conservation targets are still met as efficiently as possible, minimizing the overall cost to society.

Irreplaceability is ecologically and economically superior to simpler offset metrics

Finally, we compared the ecological and economic performance of an ∑α led offset market with three alternative offset metrics (OM) for the same simulated landscape; OM1 weighted site scores by the inverse of each species range, thereby favoring the rarest taxa in the landscape (Crisp et al. 2001); OM2 was equivalent to OM1, but values for each planning unit were not updated over time meaning weights for each species were fixed at their starting value; and OM3 was based solely on how many species were present. Our results showed that markets where trade was governed by these three alternative metrics typically failed to achieve all their targets (2%, 22% and 1% for OM1-OM3 respectively), even when property values were increased to support continued trading (Fig. 4). OM2, in which sites were weighted by species rarity, was only more successful because targets in all our scenarios were directly proportional to their availability, and hence this was the only situation where fixed weighting could sometimes be appropriate. Yet the few occasions when alternative metrics did achieve all targets relied upon the subset of species selected to have narrow distributions which restricted the flexibility of selection. Where successful, solutions were achieved with a higher number of sites and at greater cost (115-130%), and none were successful for a larger number of species.

Discussion

Land use and land management are central to addressing challenges of global biodiversity conservation, as well as food security, poverty alleviation and climate change mitigation (Meyfroidt et al. 2022). The failure to coordinate appropriate and effective actions across sectors not only undermines commitments to drive a recovery of Nature, but it also further risks the sustained wellbeing of people. In this study, we have demonstrated that if relevant parties engage in trading of biodiversity credits based on a metric derived from ∑α-irreplaceability, an offset market can support the most efficient trajectory toward all conservation targets. That is, designing an offset market with $\sum\alpha$-irreplaceability as its metric delivers a low-cost way of meeting biodiversity targets.

Our approach challenges the current school of thought that to ensure no net loss (or achieve a net gain in biodiversity), “like-for-like” trading should be mandatory within a policy design (Bull et al. 2015; zu Ermgassen et al. 2020). As a metric, $\sum\alpha$-irreplaceability relaxes the need for equivalent species in each transaction, and instead motivates restoration of species and ecosystems in greatest need (relative to targets), and where that action is most efficient economically. This element of prioritization ensures offsetting conserves the most important sites and at-risk species first, irrespective of whether they face direct development pressure. Indeed, the rationale for such prioritization is entirely transparent, and although many targets are combined to effectively rank each site, this can easily be traced back to its value for different conservation targets. Previous research has hypothesized that increasing the complexity of offset trading metrics, in a similar vein to $\sum\alpha$-irreplaceability, is likely to reduce the number of trades and hence the economic efficiency of the policy instrument (Needham et al. 2019). In contrast, we demonstrate that simpler metrics are unlikely to achieve their primary goal or guide effective progress toward conservation targets. We also show that the economic cost of solutions based on $\sum\alpha$-irreplaceability were not dependent on the number of conservation targets considered. In line with previous research, we demonstrate that the location of offset sites and overall cost of conservation actions is dictated by the overlap among ecological targets, and with ecological and economic heterogeneity across the landscape (Doyle & Yates 2010; Kangas & Ollikainen 2019; Drechsler 2021; Simpson et al. 2022). Finally, if we select conservation targets that exceed species’ initial availability because we anticipate restoration potential, then net gain, rather than no net loss, is achieved at the market-scale.

The adoption of systematic planning tools allows conservation objectives to be achieved efficiently, but rather than relying on new national parks and reserves to stall biodiversity loss, our intention is to recognize the value of effective off-reserve management (Wilson et al. 2007), and engaging private finance in conservation. Systematic conservation planning algorithms may define "optimal" solutions to meet all conservation targets, but in practice these networks are hard to implement when land is privately owned and landowner decisions are based on the relative payoffs from alternative uses (Knight et al. 2011; McIntosh et al. 2017). By introducing regulations requiring developers to offset the predicted impacts of development on biodiversity, a biodiversity offset market generates a positive financial return for farmers investing in conservation that does not exist prior to this market being created. Our study demonstrates that ∑α-irreplaceability is an effective market metric to allow farmers and developers to independently engage in trades, while ensuring an underlying strategic approach is taken to secure the targets deemed critical to biodiversity conservation.

An ongoing problem in the successful implementation of biodiversity offset markets, and environmental markets more broadly, is the lack of regulatory capacity to implement the program with an emphasis on the follow up monitoring of newly created sites (BenDor et al. 2009; Brownlie et al. 2017; zu Ermgassen et al. 2021). Similarly, a market based on the ∑α metric could potentially result in higher transactions costs. The metric is dynamic as the values of sites would ideally be recomputed after each successful trade. The uncertainty these updates create may lead to lower gains from trade being realised, eroding the ability of the offset market to deliver conservation actions cost-effectively. We have not addressed these potential costs in our study.

How can we avoid previous mistakes? Effective asset management requires monitoring.

The quality of our knowledge of biodiversity is critical to estimating the appropriate allocation of land for conservation and to quantify trade-offs. Rather than rating performance according to the resources or finance committed, ∑α credits provide the greatest reward to landowners able to deliver high marginal gains in ecological outcomes at low financial cost (Pressey et al. 2021). However, to identify the importance of a site to achieving conservation targets, ∑α-irreplaceability credits combine knowledge of how ecological assets are distributed throughout the market's jurisdiction, not just within sites associated with offset trading. Such information is not static and should also be updated routinely by the market metric to reflect their changing stocks. Note the same information would still be required to weight the alternative metrics in Figure 4, but they typically failed to achieve conservation goals because they cannot recognize when losses would be regarded as irreplaceable. Given that inadequate monitoring has been cited as a key constraint to global action for many years (Pressey et al. 2021), as well as in the context of prior attempts to organize biodiversity markets (Maron et al. 2012; zu Ermgassen et al. 2021; Kujala et al. 2022) a change in approach is required if biodiversity is to be valued correctly.

Firstly, a key principle underpinning ∑α-irreplaceability market offsets is that losses to development are not sanctioned if they cannot be replaced. It is key the market should represent as many asset types as possible, even if their distribution is uncertain, to avoid unintentional losses of biodiversity being permitted because those features were absent from ∑α calculation (Popov et al. 2022). In this context the value of ecological monitoring data gains new meaning. If our understanding of an ecological feature like species distribution, is poor, we should err on the side of caution and protect a higher number of sites to be confident we have reached a target (IUCN 2007). Without this prudent approach, land and ecological assets upon which society depends may be lost before we have the knowledge to react. If caution due to data shortages leads to an over-estimation of the area required to achieve targets, this increases the difficulty of achieving targets and consequently the financial costs of offsetting for developers. It would therefore be in the interests of both market regulators and developers to improve monitoring to minimize the uncertainty of site's ∑α-irreplaceability, balancing the cost of further monitoring against expected efficiency gains for the market (Bolam et al. 2019; Eyvindson et al. 2019). In addition, while the cost of monitoring has traditionally been prohibitive, modern tools such as acoustics, molecular methods, automated imaging and remote surveys from drone and satellites have dramatically increased our ability to monitor many ecological systems at scale (Keitt & Abelson Eric 2021; Besson et al. 2022). It is beyond the scope of this paper to provide an overview of these methods, but the capacity to efficiently verify restoration outcomes is growing, particularly if sampling design can be strategically adapted to minimize uncertainties in ∑α (Brown et al. 2013).

The biodiversity market is created by a demand for credits. In our simulated market, trading is enforced by a regulator, rather than emerging from a voluntary demand for credits. However, the guarantees that conservation targets will be safeguarded and eventually achieved cannot be made if developers participation in offset trading is voluntary. The market regulator receives updates from monitoring sources to maintain oversight of each asset's progress toward targets at the market-scale, thereby determining local site ∑α scores and the credits required for trades (Kujala et al. 2022). The regulator is also able to intervene in the economic efficiency of the market, for example by subsidizing restoration costs on farms to increase the market supply of ∑α-irreplaceability credits. While we recognize defining site ∑α-irreplaceability based on the potential recovery of a site is challenging (Sutherland 2022), including forecasting of the timeframe and risks (Laitila et al. 2014; Ladouceur et al. 2023), those uncertainties are motivations for targeted research, rather than barriers to adoption (Bolam et al. 2019; Eyvindson et al. 2019). Public support and trust will be strengthened by the transparency with which individuals can understand how local, and potentially highly visible, losses are accompanied by secure landscape gains designed to benefit society and the economy (Cvitanovic et al. 2021). We also note that landowners with spatial, strategic advantages due to the location of their land may be able to leverage payments from developers which are well in excess of their opportunity costs, where their property is key to achieving a conservation target (Lennox et al. 2012).

Beyond biodiversity offset markets

Even with introduction of planning regulation, to avert substantial biodiversity loss and degradation of ecosystem services, we must raise our ambitions to begin restoring ecosystems (Leclère et al. 2020). The resources available for conservation action are woefully inadequate compared to the resources invested in activities that further degrade or destroy nature (Dasgupta 2021), and yet the expected benefits of conservation investment often far outweigh the costs

(Bradbury et al. 2021; DEFRA 2022). The evidence of an ecological crisis is so serious that any action or investment is seen as positive, but this lack of discrimination also weakens the motivation of individuals and companies to support more transformative change. ∑α-irreplaceability credits can be used to recognize and reward private investment in conservation because they provide a comparable metric of performance within a market, even if two sites or actions impact different ecological assets.

Within an ∑α-irreplaceability market, an investor could anticipate the relative costs of their actions and define the performance of their investments in restoration and conservation for biodiversity in "net" terms. ∑α-irreplaceability could therefore be key to allowing fair recognition of investors' contributions, while building public trust that companies statements of environmental responsibility match their claims.

The debates associated with pathways to sustainability and a nature positive recovery are highly value laden, "wicked" problems (DeFries & Nagendra 2017; Meyfroidt et al. 2022), but we cannot expect ecosystem recovery to emerge from a piecemeal approach. Land is finite, and reconciling demands and interactions of complex multisector systems requires strategic oversight to avoid scenarios of ecological, economic and societal collapse (Steffen et al. 2015; Shin et al. 2022). Ecologists can identify what targets are required as a *minimum* to sustain species, ecosystem or process, but targets must ultimately be defined collaboratively with economists, social scientists, health economists and politicians. Incentivizing outcomes using insights from systematic planning will become increasingly important as the collective benefits of multiple land uses diverge (Moilanen et al. 2005; Jung et al. 2021). Adopting ∑α-irreplaceability would enable authorities to identify and minimize the conflict between conservation targets and other land uses, thereby incentivizing greater private sector investment in actions that accelerate the speed with which we can achieve Nature's recovery.

References

Armstrong McKay DI, Staal A, Abrams JF, Winkelmann R, Sakschewski B, Loriani S, Fetzer I, Cornell SE, Rockström J, Lenton TM. 2022. Exceeding 1.5°C global warming could trigger multiple climate tipping points. Science **377**:eabn7950.

Baisero D, Schuster R, Plumptre AJ. 2021. Redefining and mapping global irreplaceability. Conservation Biology **36**:e13806.

Ban NC, et al. 2013. A social–ecological approach to conservation planning: embedding social considerations. Frontiers in Ecology and the Environment **11**:194-202.

BBOP. 2009. Business and Biodiversity Offsets Programme: Biodiversity Offset Design Handbook. Washington D.C., Available from: https://www.forest-trends.org/publications/biodiversity-offset-design-handbook/.

BenDor T, Sholtes J, Doyle MW. 2009. Landscape characteristics of a stream and wetland mitigation banking program. Ecological Applications **19**:2078-2092.

Besson M, Alison J, Bjerge K, Gorochowski TE, Høye TT, Jucker T, Mann HMR, Clements CF. 2022. Towards the fully automated monitoring of ecological communities. Ecology Letters **25**:2753-2775.

Bolam FC, Grainger MJ, Mengersen KL, Stewart GB, Sutherland WJ, Runge MC, McGowan PJK. 2019. Using the Value of Information to improve conservation decision making. Biological Reviews **94**:629-647.

Bradbury RB, et al. 2021. The economic consequences of conserving or restoring sites for nature. Nature Sustainability **4**:602-608.

Brown JA, Salehi M M, Moradi M, Panahbehagh B, Smith DR. 2013. Adaptive survey designs for sampling rare and clustered populations. Mathematics and Computers in Simulation **93**:108-116.

Brownlie S, von Hase A, Botha M, Manuel J, Balmforth Z, Jenner N. 2017. Biodiversity offsets in South Africa – challenges and potential solutions. Impact Assessment and Project Appraisal **35**:248-256.

Bryan BA, Raymond CM, Crossman ND, Macdonald DH. 2010. Targeting the management of ecosystem services based on social values: Where, what, and how? Landscape and Urban Planning **97**:111-122.

Bull JW, Hardy MJ, Moilanen A, Gordon A. 2015. Categories of flexibility in biodiversity offsetting, and their implications for conservation. Biological Conservation **192**:522-532.

Bull JW, Milner-Gulland EJ, Suttle KB, Singh NJ. 2014. Comparing biodiversity offset calculation methods with a case study in Uzbekistan. Biological Conservation **178**:2-10.

Crisp MD, Laffan S, Linder HP, Monro A. 2001. Endemism in the Australian flora. Journal of Biogeography **28**:183-198.

Cvitanovic C, Shellock RJ, Mackay M, van Putten EI, Karcher DB, Dickey-Collas M, Ballesteros M. 2021. Strategies for building and managing 'trust' to enable knowledge exchange at the interface of environmental science and policy. Environmental Science & Policy **123**:179-189.

Dasgupta P. 2021. The Economics of Biodiversity: The Dasgupta Review. London.

DEFRA. 2022. Environment Act targets: Summary of evidence and approach. . Online: https://consult.defra.gov.uk/natural-environment-policy/consultation-on-environmental-targets.

DeFries R, Nagendra H. 2017. Ecosystem management as a wicked problem. Science **356**:265-270.

Díaz S, et al. 2019. Pervasive human-driven decline of life on Earth points to the need for transformative change. Science **366**:eaax3100.

Doyle MW, Yates AJ. 2010. Stream ecosystem service markets under no-net-loss regulation. Ecological Economics **69**:820-827.

Drechsler M. 2021. Bundling of Ecosystem Services in Conservation Offsets: Risks and How They Can Be Avoided. Land DOI: 10.3390/land10060628.

Eyvindson K, Hakanen J, Mönkkönen M, Juutinen A, Karvanen J. 2019. Value of information in multiple criteria decision making: an application to forest conservation. Stochastic Environmental Research and Risk Assessment **33**:2007-2018.

Ferrier S, Pressey RL, Barrett TW. 2000. A new predictor of the irreplaceability of areas for achieving a conservation goal, its application to real-world planning, and a research agenda for further refinement. Biological Conservation **93**:303-325.

Hanson JO, Schuster R, Morrell N, Strimas-Mackey M, Edwards BPM, Watts ME, Arcese P, Bennett J, Possingham HP. 2022. prioritizr: Systematic Conservation Prioritization in R. R package version 7.2.2, Available at https://CRAN.R-project.org/package=prioritizr.

IPBES. 2019. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science- Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany.

IUCN. 2007. Guidelines for applying the precautionary principle to biodiversity conservation and natural resource management. As approved by the 67th meeting of the IUCN council 14–16 May 2007.

IUCN. 2022. Egyptian COP27 Presidency, Germany and IUCN announce ENACT Initiative for Nature-based Solutions. Online. Available at: https://cop27.eg/#/presidency/initiative/enact.

Jung M, et al. 2021. Areas of global importance for conserving terrestrial biodiversity, carbon and water. Nature Ecology & Evolution **5**:1499-1509.

Kangas J, Ollikainen M. 2019. Economic Insights in Ecological Compensations: Market Analysis With an Empirical Application to the Finnish Economy. Ecological Economics **159**:54-67.

Keitt T, H., Abelson Eric S. 2021. Ecology in the age of automation. Science **373**:858-859.

Knight AT, Grantham HS, Smith RJ, McGregor GK, Possingham HP, Cowling RM. 2011. Land managers' willingness-to-sell defines conservation opportunity for protected area expansion. Biological Conservation **144**:2623-2630.

Kujala H, et al. 2022. Credible biodiversity offsetting needs public national registers to confirm no net loss. One Earth **5**:650-662.

Kukkala AS, Moilanen A. 2013. Core concepts of spatial prioritisation in systematic conservation planning. Biological Reviews **88**:443-464.

Ladouceur E, Isbell F, Clark AT, Harpole WS, Reich PB, Tilman GD, Chase JM. 2023. The recovery of plant community composition following passive restoration across spatial scales. Journal of Ecology **111**:814-829.

Laitila J, Moilanen A, Pouzols FM. 2014. A method for calculating minimum biodiversity offset multipliers accounting for time discounting, additionality and permanence. Methods in Ecology and Evolution **5**:1247-1254.

Leclère D, et al. 2020. Bending the curve of terrestrial biodiversity needs an integrated strategy. Nature **585**:551-556.

Lennox GD, Dallimer M, Armsworth PR. 2012. Landowners' ability to leverage in negotiations over habitat conservation. Theoretical Ecology **5**:115-128.

Leroy B, Meynard CN, Bellard C, Courchamp F. 2016. virtualspecies, an R package to generate virtual species distributions. Ecography **39**:599-607.

Margules CR, Pressey RL. 2000. Systematic conservation planning. Nature **405**:243-253.

Maron M, Hobbs RJ, Moilanen A, Matthews JW, Christie K, Gardner TA, Keith DA, Lindenmayer DB, McAlpine CA. 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biological Conservation **155**:141-148.

Marshall E, Wintle BA, Southwell D, Kujala H. 2020. What are we measuring? A review of metrics used to describe biodiversity in offsets exchanges. Biological Conservation **241**:108250.

McIntosh EJ, Pressey RL, Lloyd S, Smith RJ, Grenyer R. 2017. The Impact of Systematic Conservation Planning. Annual Review of Environment and Resources **42**:677-697.

Meyfroidt P, et al. 2022. Ten facts about land systems for sustainability. Proceedings of the National Academy of Sciences **119**:e2109217118.

Moilanen A, Franco AMA, Early RI, Fox R, Wintle B, Thomas CD. 2005. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. Proceedings of the Royal Society B: Biological Sciences **272**:1885-1891.

Moilanen A, Wilson KA, Possingham H 2009. Spatial Conservation Prioritization: Quantitative Methods and Computational Tools. Oxford University Press, Oxford.

Natural England. 2022. Biodiversity Metric 3.1: Auditing and accounting for biodiversity – User Guide. London.

Needham K, de Vries FP, Armsworth PR, Hanley N. 2019. Designing markets for biodiversity offsets: Lessons from tradable pollution permits. Journal of Applied Ecology **56**:1429-1435.

NSW DPE. 2022. Biodiversity credit pricing guide: Guidance for pricing biodiversity credits under the Biodiversity Offsets Scheme. Parramatta, NSW.

Popov V, Shah P, Runting RK, Rhodes JR. 2022. Managing risk and uncertainty in systematic conservation planning with insufficient information. Methods in Ecology and Evolution **13**:230-242.

Pressey RL, Humphries CJ, Margules CR, Vane-Wright RI, Williams PH. 1993. Beyond opportunism: Key principles for systematic reserve selection. Trends in Ecology & Evolution **8**:124-128.

Pressey RL, Visconti P, McKinnon MC, Gurney GG, Barnes MD, Glew L, Maron M. 2021. The mismeasure of conservation. Trends in Ecology & Evolution **36**:808-821.

Radu C, Caron M-A, Arroyo P. 2020. Integration of carbon and environmental strategies within corporate disclosures. Journal of Cleaner Production **244**:118681.

Schmalensee R, Stavins RN. 2017. The design of environmental markets: What have we learned from experience with cap and trade? Oxford Review of Economic Policy **33**:572-588.

Sciaini M, Fritsch M, Scherer C, Simpkins CE. 2018. NLMR and landscapetools: An integrated environment for simulating and modifying neutral landscape models in R. Methods in Ecology and Evolution **9**:2240-2248.

Shin Y-J, et al. 2022. Actions to halt biodiversity loss generally benefit the climate. Global Change Biology **28**:2846-2874.

Simpson KH, de Vries F, Dallimer M, Armsworth PR, Hanley N. 2021. Understanding the Performance of Biodiversity Offset Markets: Evidence from An Integrated Ecological - Economic Model. Land Economics:030420-030032R.

Simpson KH, de Vries FP, Dallimer M, Armsworth PR, Hanley N. 2022. Ecological and economic implications of alternative metrics in biodiversity offset markets. Conservation Biology **36**:e13906.

Steffen W, et al. 2015. Planetary boundaries: Guiding human development on a changing planet. Science **347**:1259855.

Sutherland WJ. 2022. Transforming Conservation: A Practical Guide to Evidence and Decision Making. Open Book Publishers https://doi.org/10.11647/OBP.0321, Cambridge, UK.

Tittensor DP, et al. 2014. A mid-term analysis of progress toward international biodiversity targets. Science **346**:241-244.

Wilson KA, et al. 2007. Conserving Biodiversity Efficiently: What to Do, Where, and When. PLOS Biology **5**:e223.

Yong DL, Barton PS, Ikin K, Evans MJ, Crane M, Okada S, Cunningham SA, Lindenmayer DB. 2018. Cross-taxonomic surrogates for biodiversity conservation in human-modified landscapes – A multi-taxa approach. Biological Conservation **224**:336-346.

zu Ermgassen SOSE, Baker J, Griffiths RA, Strange N, Struebig MJ, Bull JW. 2019. The ecological outcomes of biodiversity offsets under “no net loss” policies: A global review. Conservation Letters **12**:e12664.

zu Ermgassen SOSE, Maron M, Corlet Walker CM, Gordon A, Simmonds JS, Strange N, Robertson M, Bull JW. 2020. The hidden biodiversity risks of increasing flexibility in biodiversity offset trades. Biological Conservation **252**:108861.

zu Ermgassen SOSE, Marsh S, Ryland K, Church E, Marsh R, Bull JW. 2021. Exploring the ecological outcomes of mandatory biodiversity net gain using evidence from early-adopter jurisdictions in England. Conservation Letters **14**:e12820.

Figures

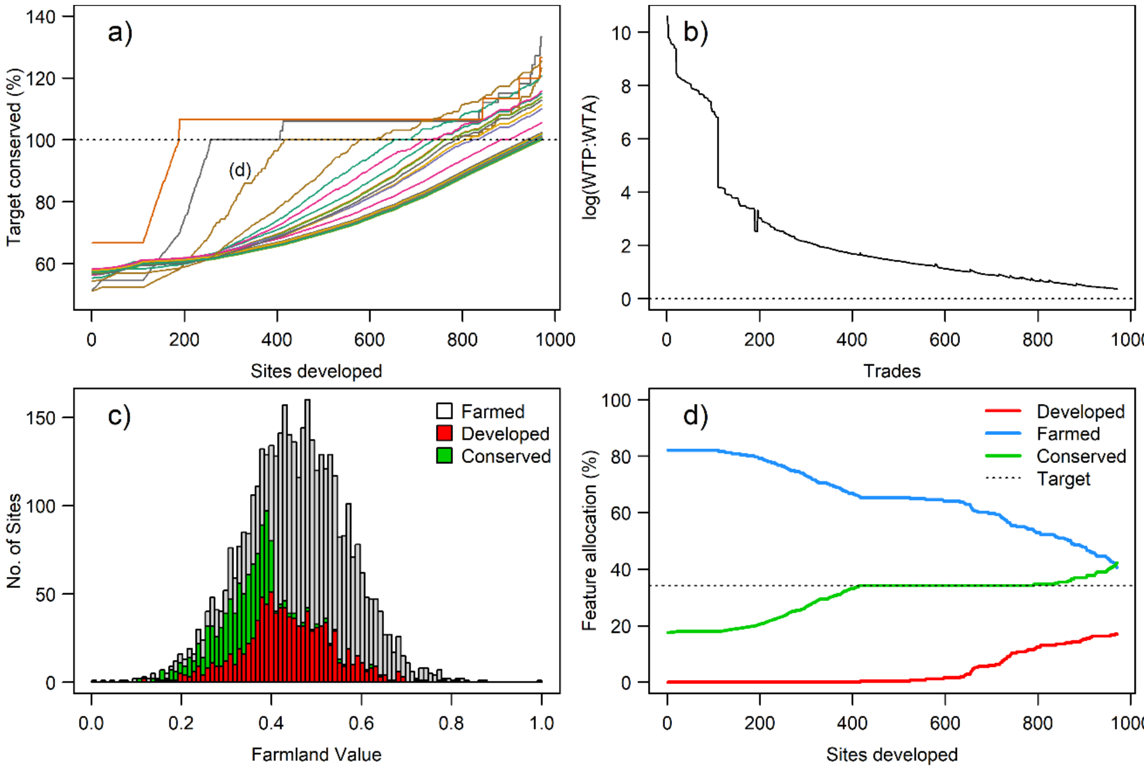

Figure 1. Example of ∑α-irreplaceability offset market for 25 simulated species. Panel a) indicates the progress of each species toward its conservation targets (dotted line) as new developments requiring offsets take place. Panel b) illustrates the decline in the log ratio between willingness to pay (WTP) and willingness to accept (WTA) as representative of gains from trade, and c) displays the distribution of values for purchasing agricultural land in this simulated landscape, and the final proportion of those that were selected for development and conservation offsets. Panel d) displays the changes in the allocation of a single species (also identified in panel a) among land types as trading progresses.

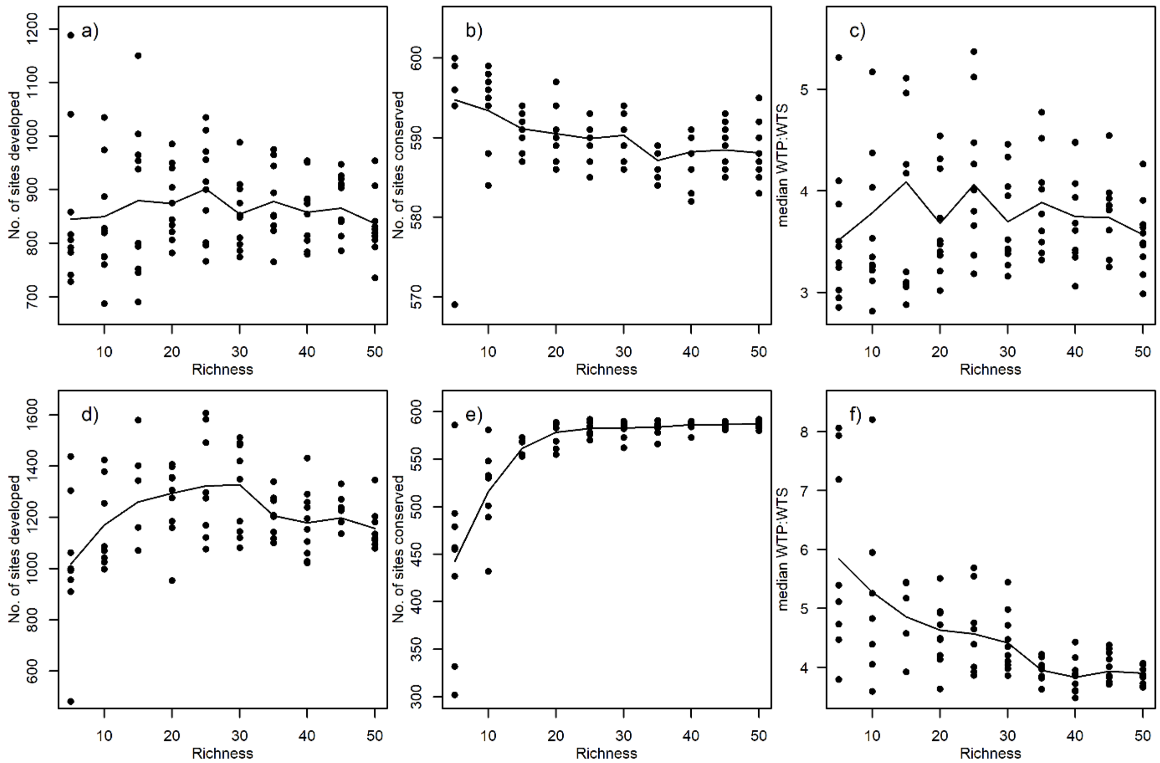

Figure 2. Variation ∑α-irreplaceability market trading outcomes when the richness of communities is increased. Assemblages were drawn from communities of 200, with either a strong richness gradient (a,b,c) or no richness gradient (d,e,f). The columns show the number of sites selected for development (a & d), the number of sites required for conservation (b & e) and the median ratio between willingness to pay (WTP) and willingness to accept (WTA) (c & e). All conservation targets were achieved in each market simulation and lines of best fit were added based on local polynomial regression.

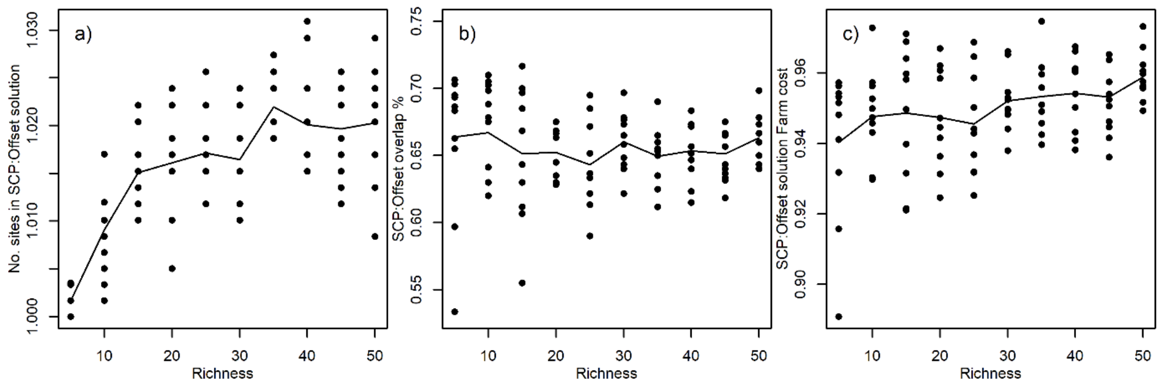

Figure 3. Illustration of a comparison between conservation networks selected by ∑α-irreplaceability market trading and “optimal” planning outcomes for simulated community with a strong richness gradient. Panel a) plots the ratio of network size when the richness of communities is increased; panel b) the percentage of planning units that are shared with the optimal network, and panel c) the ratio of network cost.

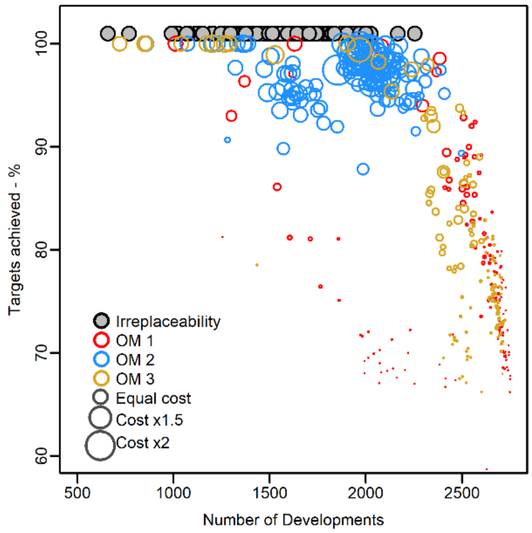

Figure 4. Comparison of conservation solution efficiency when guided by systematic conservation planning (SCP), or an offset market based on ∑α-irreplaceability, and three alternative offset metrics described in the main text (O1-O3). Panel a) displays the total cost of agricultural land with the increasing richness of simulation scenarios, and panel b) displays the number of planning units that were Developed or entered into Conservation offsets. To make outcomes comparable only solutions that achieved >99% of targets are displayed. Note the solutions proposed by SCP are not associated with Development but are added to the plot to indicate the number of planning units conserved.