

A Service of

ZBW

Leibniz-Informationszentrum Wirtschaft Leibniz Information Centre for Economics

Gerling, Charlotte; Schöttker, Oliver; Hearne, John

Article — Published Version The 'Climate Adaptation Problem' in Biodiversity Conservation: The Value of Spatial Flexibility in Land Purchase

**Environmental and Resource Economics** 

**Provided in Cooperation with:** Springer Nature

*Suggested Citation:* Gerling, Charlotte; Schöttker, Oliver; Hearne, John (2024) : The 'Climate Adaptation Problem' in Biodiversity Conservation: The Value of Spatial Flexibility in Land Purchase, Environmental and Resource Economics, ISSN 1573-1502, Springer Netherlands, Dordrecht, Vol. 88, Iss. 2, pp. 339-365, https://doi.org/10.1007/s10640-024-00932-4

This Version is available at: https://hdl.handle.net/10419/318625

## Standard-Nutzungsbedingungen:

Die Dokumente auf EconStor dürfen zu eigenen wissenschaftlichen Zwecken und zum Privatgebrauch gespeichert und kopiert werden.

Sie dürfen die Dokumente nicht für öffentliche oder kommerzielle Zwecke vervielfältigen, öffentlich ausstellen, öffentlich zugänglich machen, vertreiben oder anderweitig nutzen.

Sofern die Verfasser die Dokumente unter Open-Content-Lizenzen (insbesondere CC-Lizenzen) zur Verfügung gestellt haben sollten, gelten abweichend von diesen Nutzungsbedingungen die in der dort genannten Lizenz gewährten Nutzungsrechte.



WWW.ECONSTOR.EU

http://creativecommons.org/licenses/by/4.0/

#### Terms of use:

Documents in EconStor may be saved and copied for your personal and scholarly purposes.

You are not to copy documents for public or commercial purposes, to exhibit the documents publicly, to make them publicly available on the internet, or to distribute or otherwise use the documents in public.

If the documents have been made available under an Open Content Licence (especially Creative Commons Licences), you may exercise further usage rights as specified in the indicated licence.





# The 'Climate Adaptation Problem' in Biodiversity Conservation: The Value of Spatial Flexibility in Land Purchase

Charlotte Gerling<sup>1</sup> · Oliver Schöttker<sup>1</sup> · John Hearne<sup>2</sup>

Accepted: 30 September 2024 / Published online: 23 October 2024 © The Author(s) 2024

# Abstract

Existing reserve networks become less suitable as species' ranges shift under climate change and the scarcity and value of habitats change. Reserve sites hence have to be reallocated to reflect these changing values and to remain cost-effective, but restrictions on selling reserve sites limit this adaptation. Under climate change, a novel 'sale' policy that provides resale flexibility by allowing increasingly less cost-effective sites to be sold to free funds for purchasing increasingly cost-effective sites may hence be preferable over a typical 'no sale' policy. Here, we develop a conceptual climate-ecological-economic model to examine under what conditions resale flexibility provides cost-effectiveness advantages. We find that the benefits of resale flexibility provided by the 'sale' policy are particularly high when little funding is available for climate change adaptation. Due to this budget effect, the value of resale flexibility also decreases with increasing interest rates. The value of resale flexibility is particularly high when creating reserves for species that are highly mobile as new reserve sites may then be chosen freely (in comparison to the case that new sites have to be located in the proximity of existing reserve sites). We further examine to what extent the 'sale' policy reduces the permanence of habitat within a specific location ("site habitat permanence"), and identify a new trade-off: 'sale' provides advantages for habitat types that become increasingly rare, while 'no sale' improves site habitat permanence mainly for expanding habitat types.

**Keywords** Climate-ecological-economic model · Climate adaptation · Habitat conservation · Reversible investment · Irreversible investment

Charlotte Gerling charlotte.gerling@b-tu.de

<sup>&</sup>lt;sup>1</sup> Environmental Economics, Brandenburg University of Technology Cottbus-Senftenberg, Erich-Weinert-Str. 1, 03046 Cottbus, Germany

<sup>&</sup>lt;sup>2</sup> School of Science, RMIT University, La Trobe St, Melbourne, VIC 3001, Australia

# 1 Introduction

Climate change is one of the largest threats to global biodiversity (Dasgupta 2021) and poses new challenges as policy instruments for biodiversity conservation may have to be adapted. One such potential adaptation is increasing flexibility in the designation of reserve sites over time: flexibility to add particularly cost-effective sites to the reserve, and flexibility to remove sites of low cost-effectiveness to free funds for purchasing more cost-effective sites. This flexibility becomes relevant under climate change (Heller and Zavaleta 2009; Vincent et al. 2019; Albers 2022) because climate change causes species' ranges to shift (Ponce-Reyes et al. 2017; Campos-Cerqueira et al. 2021; Dasgupta 2021). From an economic perspective, climate change may thus induce changes to the comparative advantage of some sites over others: if climate change leads to a relative increase (decrease) in the "ecological productivity" of new (former) sites, reserve sites have to be reallocated in line with these changes to remain cost-effective.

However, this need for flexibility is often not accounted for in land purchase for reserve creation: reserve expansion is limited by budget constraints, and previously purchased reserve sites are rarely sold (Fuller et al. 2010; Lennox et al. 2017). In addition, selling reserve sites is often not permitted due to the negative ecological consequences of habitat turnover, high transaction costs such as taxes, and costs for the identification of suitable sites for selling (Schöttker and Wätzold 2018). This is "akin to large investments in specific productive capital, which can limit choices for decades" (Meyfroidt et al. 2022 p.7). When selling reserve sites in prohibited and a conservation agency aims to adapt an existing reserve that has not been designed with the impacts of climate change in mind to changing climatic conditions, the decision problem of the agency is to select cost-effective reserve sites, taking into account future changes in comparative advantages.

Given that comparative advantages change over time, a better outcome may be reached if the investment decision was to some degree reversible. This idea translates to a novel version of land purchase that allows for "resale flexibility", in which the decision maker is allowed to sell previously purchased sites. Resale flexibility hence describes the flexibility to replace existing reserve sites provided by the 'sale' policy. This adapted policy may outperform the policy that does not allow for sales (the former represents the lower bound), and recent applied ecological research provides some evidence that the benefits of selling may warrant the option of selling under changing climatic conditions, albeit cautioning against general recommendations to allow the sale of reserve sites (Alagador et al. 2014, 2016). However, the conditions under which resale flexibility provides cost-effectiveness advantages have not been analysed. Given political inertia and potentially high transaction costs of changing existing regulations (Nalle et al. 2004; Burch et al. 2014), understanding the factors that drive the value of resale flexibility is important to inform policy design in the face of climate change.

With this paper we aim to contribute to filling this research gap. The paper is therefore located at the intersection of two strands of the economic literature. The first strand is the literature on the design of policy instruments for biodiversity conservation, which has so far largely relied on the assumption of static climatic conditions (e.g., Armsworth et al. (2012), Wätzold et al. (2016), and Wuepper and Huber (2022). As notable exceptions, Schöttker and Wätzold (2022) assess the cost-effectiveness of different governance modes for biodiversity conservation under climate change, Huber et al. (2017) simulate the outcome of

agri-environment schemes under climate change, and Gerling et al. (2023) examine changes in the cost-effective conservation measures of an agri-environment scheme over time due to climate change.

The second strand is the literature on the value of flexibility in reserve design. Much of this research focuses on the value of flexibility under uncertainty. For example, previous research has examined the value of flexibility in terms of when to select conservation sites (e.g. Westphal et al. 2003; Costello and Polasky 2004; Kassar and Lasserre 2004; Costello and Polasky 2008; Drechsler 2020), often considering the case of irreversible habitat loss (e.g., Arrow and Fisher 1974; Albers 1996; Leroux et al. 2009; Shah and Ando 2016). Lennox et al. (2017) introduce further flexibility by allowing borrowing and budget carry-over. Lewis and Polasky (2018) specifically consider the provision of ecosystem services under climate change uncertainty in case of irreversible land use changes and develop an auction mechanism in which the regulator has the flexibility to allow conversion of habitat to farming in the second period to improve overall outcomes as climatic conditions change. Other research has considered the need for flexibility in biodiversity conservation under climate change for specific case studies, such as conservation on the Iberian peninsula (Alagador et al. 2014, 2016) and conservation of a grassland indicator species in Germany (Gerling et al. 2022). To the best of our knowledge, there is no economic research on the value of resale flexibility to improve the cost-effectiveness of policy instruments under climate change.

In this paper, we aim to gain a conceptual understanding on when adapting land purchase by allowing for sale may be warranted to contribute to the limited economic research on the necessary adaptations of policy instruments for cost-effective biodiversity conservation in the face of climate change. To do so, we draw from insights from the two strands of the literature mentioned above: we compare the cost-effectiveness of the different policy options which differ in terms of flexibility. However, by considering climate changeinduced changes in the "ecological productivity" of potential reserve sites, and the specific policy background of land purchase, we go beyond the previous literature which has largely focused on uncertainty issues and has not made the link to implications for policy instrument design. For example, given the policy context, flexibility in terms of when to buy, but also when to sell reserve sites becomes relevant, while much of the previous literature has focused on the flexibility of when to buy reserve sites.

Specifically, we analyse the implications of the irreversible policy which prohibits sales (henceforth referred to as 'no sale') and an adapted, partially reversible policy option<sup>1</sup> which allows sales (henceforth referred to as 'sale'). Throughout this paper, we evaluate the two policies in a setting in which an initial reserve has been designed without consideration of future climate change impacts, and land purchase is the chosen policy instrument for climate change adaptation of this reserve. We develop a conceptual climate-ecological-economic (CEE) model in which a conservation agency aims to maximise habitat persistence and selects sites to purchase (and, under the 'sale' policy, sell) to analyse the value of resale flexibility in a setting of multiple habitat types. The value of resale flexibility is defined as the percentage improvement in utility received from habitat conservation when moving from the 'no sale' policy to the 'sale' policy. The considered habitat types are influenced

<sup>&</sup>lt;sup>1</sup> Note that this represents a partially reversible investment (Baldwin 1982; Abel and Eberly 1996; Hartman and Hendrickson 2002), as transaction costs do not allow for full recoverability of initial investment costs (Verbruggen 2013; Drechsler and Wätzold 2020). A fully reversible investment would imply full recoverability of initial investment costs (Davis and Cairns 2017).

by climate change in different ways, each representing typical, stylised cases observed in reality (see for example Lamprecht et al. (2018). Specifically, we consider habitat types that expand, shift, and contract due to the impacts of climate change.

In order to compare the two policy options, we follow the economic literature on policy instrument design for biodiversity conservation and evaluate the policies' cost-effectiveness, but within a utility maximisation framework (Davis et al. 2006). In line with previous literature (cp. Doven et al. (2013), Montova et al. (2020), cost-effectiveness hence refers to maximising the habitat area of the rarest habitat type at any point in time for a given budget<sup>2</sup>. To examine whether the size of the budget has an impact on the comparison, we consider different budget constraints, and we vary a range of parameters in sensitivity analyses. This allows us to get an understanding under which circumstances the 'sale' policy provides costeffectiveness advantages, and under which circumstances it does not. In this analysis, we not only consider final results for some chosen point in the future in which climatic conditions have changed, but explicitly examine the optimal adaptation path towards this result. Finally, we consider that climate change dynamics may imply that the cost-effective policy is subject to low site habitat permanence, and that one of the key arguments against the 'sale' policy are potential increases in habitat turnover. We define site habitat permanence of a chosen habitat type as the percentage of the areas covered initially by that habitat type that is maintained throughout the run-time of the model in the same grid cells. We therefore additionally examine the impact of resale flexibility on site habitat permanence.

With this paper, we contribute to the limited economic literature on cost-effective policy design for biodiversity conservation under climate change. Given the important impacts of climate change on biodiversity, the current, very limited economic literature on the topic presents a striking research gap. We specifically address the value of resale flexibility as it emerges in land purchase, an issue that has so far not been examined in the economic literature despite the importance of land purchase in conservation practice. This allows us to gain some insights into when a novel, more flexible version of land purchase provides cost-effectiveness advantages over current policy design.

# 2 The Climate-Ecological-Economic (CEE) Model

#### 2.1 Model Setting: Landscape, Habitat Types and Climate Change

We consider a conceptual landscape consisting of different elevations representing three different climatic regions, each providing ideal conditions for a specific habitat type. Climate change causes changes in the habitat suitability of potential reserve sites in this landscape. While the model landscape is purely conceptual, it represents typical climate change-

<sup>&</sup>lt;sup>2</sup> Cost-effectiveness in biodiversity conservation under climate change may be defined as "the ability of conservation measures to maximize a species' population viability for given costs in a region that experiences climate change" (Gerling and Wätzold 2021, p. 2). In the case of habitat conservation, the conservation impact indicator may be referred to as "habitat persistence". However, when considering several habitat types, the trade-offs between different habitat types also need to be addressed. Here, we adopt a utility maximisation framework to model these trade-offs (Davis et al. 2006) and follow Doyen et al. (2013) and Montoya et al. (2020) in setting the conservation agency's objective function as maximising the habitat area of the rarest habitat type. We chose this utility function due to the impacts of climate change on the habitat types' scarcity. We discuss this further in Chap. 2.2.

induced habitat shifts observed in reality: Species' ranges shift uphill towards previously cooler areas which become increasingly warmer under climate change. This means that mountain top habitats become particularly threatened as a further uphill movement is not possible (see Lamprecht et al. (2018) and Ludovicy et al. (2022) for examples).

The landscape is divided into 200 cells. The elevation of a cell *i* is is given by:

$$e_{i} = \frac{\frac{\left(\sin\left(\left((x+1)-\frac{10}{2}\right)*\frac{\pi}{10}\right)+1\right)}{2} + \frac{\left(\sin\left(\left((y+1)-\frac{20}{2}\right)*\frac{\pi}{20}\right)+1\right)}{2}}{2}}{2}$$
(1)

With  $x \in \{1, 2, .., 10\}$  the x-coordiante of cell *i* and  $y \in \{1, 2, .., 20\}$  the y-coordinate. This generates the landscape shown in Fig. 1.

In this landscape, three different habitat types exist ( $H_k$ :  $H_1$ ,  $H_2$ ,  $H_3$ ):  $H_1$  represents a habitat type predominantly present in low elevations,  $H_2$  can mainly be found in medium elevations, and  $H_3$  is located in high elevations. The distribution of habitat suitability is indicated by the *ecological value*,  $h_{kit}$ , of each habitat type *k* in grid cell *i* at time *t*. This value is particularly high in the respective ideal elevation zone (e.g. in low elevations for  $H_1$ ), but decreases with an increasing distance to the ideal elevation. Moreover, we assume that cells' opportunity costs for conservation are spatially heterogeneous (e.g., Lewis and Polasky 2018) and randomly drawn from a uniform distribution.

Under climate change, the suitability of each grid cell for the different habitat types changes. Hence, the ecological value  $h_{kit}$  is time-dependent. Importantly, climate change



Fig. 1 Landscape grid and grid cells, coloured by elevation level visualising the topography of the landscape

Deringer

impacts each habitat type differently, representing different possible changes in comparative advantages. This is given by:

$$h_{kit} = \frac{\sin\left(\pi * \left(e_i - \frac{t}{2T} + l_k\right)\right) + 1}{2} \quad with \quad l_k = \begin{cases} 0.5 & for \ k = 1\\ 0 & for \ k = 2\\ -0.5 & for \ k = 3 \end{cases}$$
(2)

Where *T* is the time horizon of the model. This equation of motion implies that the cells' ecological value for habitat type  $H_1$ , which is initially high in low elevations, improves over time and leads to  $H_1$  covering a large portion of the landscape. Hence, it becomes increasingly present in areas with higher elevation. The cells' ecological value for  $H_2$ , which is initially high in medium elevations, generally decreases in low and medium elevations, leading to habitat type  $H_2$  to shrink in these regions. Cells of higher altitude however will face an increase in ecological value for habitat type  $H_2$ , meaning that the potential habitat sites tend to move "uphill" towards higher elevations. The cells' ecological value for  $H_3$ , for which the potential habitat sites are initially mainly found in high elevations, decreases over time in all regions. Thus, the potential extent of  $H_3$  shrinks over time and eventually almost disappears as a further upwards movement is impossible. In order to capture the long-term impacts of climate change on conservation, we consider a period of 60 years, divided into 12 periods of 5 years each. Figure 2 illustrates the suitability of each habitat for all elevation levels for time steps  $t = \{1, 4, 8, 12\}$ .

When conserving a grid cell, i.e. when a grid cell is purchased, habitats are instantly created on this grid cell, with each habitat type taking up a share of the grid cell proportional to their respective ecological value  $h_{kit}$ . The proportional coverage of a cell *i* by habitat type *k* is given by:

$$h_{kit}^{norm} = \frac{h_{kit}}{h_{1it} + h_{2it} + h_{3it}}$$
(3)

This means that conserving a specific grid cell will lead to up to 3 habitat types being conserved. Depending on the ecological value  $h_{kit}^{norm}$ , conserving a grid cell *i* may lead to higher outcomes for some habitat types than for others. For example (cp. Figure 2), in t=1, conserving a grid cell of medium elevation will lead to all three habitat types being conserved, but the largest portion of the grid cell will be covered by habitat type  $H_2$  as this area provides only sub-optimal conditions for  $H_1$  and  $H_3$ . In t=8, grid cells with low elevation however will only host 2 habitat types- H1 and H2. H1 will cover the majority of the grid cell as the conditions for this habitat type are better than for  $H_2$ .  $H_3$  will have disappeared from low elevation cells by t=8 as climate change has rendered conditions unsuitable for this habitat type. Any non-conserved grid cell, i.e. any grid cell that has not been purchased yet or has been sold again after being purchased before, does not provide any habitat  $H_k$ . All habitat on a grid cell that is sold is therefore lost immediately. You may consider this a simplified representation of agricultural landscapes, in which taking intensively farmed land out of production may provide habitat to many endangered species at relatively short time scales (Helbing et al. 2023), while this habitat is lost as soon as the reserve site is used for intensive agriculture again.



Fig. 2 Level of ecological value  $h_{kit}$  for different elevation levels at time steps  $t \in \{1, 4, 8, 12\}$  for all three habitat types  $k \in \{1, 2, 3\}$ 

# 2.2 Decision Problem: Optimal Allocation of Reserve Sites

The conservation agency initially (in t=0) owns a network of connected grid cells, the reserve network. The optimisation procedure selecting this initial reserve network is explained in detail in Appendix A. This reserve is optimal under initial climatic conditions (in t=0) but does not consider the impact of future climate change. This implies that we assume that current reserves were not designed and designated to reflect future values. This lack of consideration for climate change in conservation decision-making has frequently been observed in reality, e.g. due to lack of relevant information (Hannah et al. 2002; Prober et al. 2017). Based on this set-up, we determine the optimal time series of reserve networks. Each of the individual reserve networks in this time series is created through the purchase and, under the 'sale' policy, the sale of grid cells by the conservation agency, and hence represents incremental adaptations of the initial reserve network.

#### 2.2.1 Considerations on the Utility of Habitat Under Climate Change

Cost-effectiveness is an important criterion to evaluate policy instruments for biodiversity conservation. Under climate change, cost-effectiveness may be defined as "the ability of conservation measures to maximize a species' population viability for given costs in a region that experiences climate change" (Gerling and Wätzold 2021, p. 2). However, when considering several species or habitat types, this standard definition needs to be further refined. When considering two target habitat types A and B, the decision maker faces tradeoffs between how much to focus on habitat A, and how much on habitat B. One could imagine drawing a Pareto frontier showing this trade-off as there may be several cost-effective (Pareto-optimal) conservation strategies for which the outcome for habitat type A may only be increased if accepting a decrease in the outcome for habitat type B. To choose one of these Pareto-optimal policies (and to formulate the decision problem as an objective function) requires the decision maker to clearly weight the focus on different habitat types. To model this, one may examine cost-effective conservation within a utility maximisation framework (Davis et al. 2006), and explicitly model the utility of conserving different habitat types. Essentially, such frameworks maximise the *value* of biodiversity for given budgets (Davis et al. 2006). A key question is hence what this value of biodiversity looks like. In this sub-chapter, we develop a utility function that captures the utility of conservation of several habitat types under climate change based on the literature. For an alternative utility function, see Appendix B. Based on this, the optimisation problem is then presented in Chap. 2.2.2.

In deciding which sites to purchase (and sell), we assume the decision maker aims to maximise utility, which is a function of the utility of conserving the three habitat types:

$$U = f(U_{k=1}, U_{k=2}, U_{k=3})$$
(4)

A question that has yet to be answered is how this utility function for biodiversity, habitat types or ecosystem services should look like (Paul et al. 2020). It is often assumed (Davis et al. 2003, 2006) or found in empirical research (e.g. Wallmo & Edwards 2008, Lew & Wallmo 2011) that increasing conserved area exhibits diminishing marginal utility. In line with this, we therefore consider the positive concave utility function shown in Fig. 3 (see Appendix C for further details and comparisons to the literature). This implies that increasing amounts of habitat are valued equally for all three habitat types if the habitat types cover an equal amount of the landscape, and differences in their value only arise from differences in scarcity. Hence, a given marginal increase in habitat area for a habitat type with small spatial extent is valued higher than the same increase for a habitat type with larger spatial extent.

Next, one has to consider how the decision maker trades-off different habitat types. Here, we model a conservation agency's decision-making (rather than, for example, maximising social welfare), and hence base the utility function on ecological research (e.g. Montoya et al. 2020)<sup>3</sup>. In line with the literature on cost-effective conservation that focuses on maximising species persistence or population viability for given costs (e.g. Cabeza and Moilanen 2001; Gerling and Wätzold 2021), we assume that the conservation agency aims to maxi-

<sup>&</sup>lt;sup>3</sup> Other research considers additive utility functions. We have chosen the *maximin* problem as we focus on climate change increasing the risk of habitat loss, and the chosen utility function places greater emphasis on scarcity and hence, potential habitat loss.



Fig. 3 Utility function exhibiting diminishing marginal utility

mise habitat persistence, for example to prevent species extinctions (e.g. Calkin et al. 2002). Specifically, the agency aims to maximise the minimum area covered by any one of the habitat types (Montoya et al. 2020) for given costs. Maximising the extent of the rarest habitat type hence represents a specific formulation of cost-effectiveness that maximises the utility received from conservation by considering the trade-offs between habitat types, specifically reflecting the concave nature of the underlying utility function, i.e. the diminishing marginal utility, and also has a strong focus on scarcity:

$$U = min. \{ U_{k=1}, U_{k=2}, U_{k=3} \}$$
(5)

Finally, an important question is how future biodiversity outcomes influence today's utility, i.e. whether and how future biodiversity outcomes should be discounted. Under climate change, however, this issue not only pertains to finding an appropriate discount factor. Rather, given that previously common habitat types may become more threatened due to climate change, and previously rare habitat types may become more common, the value of a habitat type changes over time depending on how climate change influences its rarity. The valuation of future conservation outcomes under climate change is thus a critical question as a decision-maker would consider expected future changes to rarity (and hence, the utility of habitat) in today's decisions.

We are not aware of any previous research addressing the changing value of habitats under climate change. Here, we therefore adhere to the logic of the extinction-focused conservation agency adopted above and assume that the decision-maker is solely focused on maximising habitat persistence within the 60-year planning horizon, without considering whether the benefits of conservation occur now or in the future. The agency therefore aims to maximise the extent of the smallest habitat type *at any time* for given costs to reflect diminishing marginal utility of the three habitat types, even if this means sacrificing possible increases in biodiversity today for future increases. We believe that this valuation may be particularly valid in the case of climate change as climate change may make recolonization of areas where a species has been lost more unlikely, e.g. if the surrounding area is no longer climatically suitable for the species (Gerling and Wätzold 2021), and as the impacts of climate change will increase in the future, implying that biodiversity will become more threatened in the future and the focus of conservation may shift further into the future to mirror the high value of increasingly scarce biodiversity. Moreover, this utility function has been suggested in previous research (e.g. Doyen et al. 2013). When considering the temporal dimension, the utility function given in Eq. 5 therefore becomes:

$$U = min. \{ \mathbf{U}_{k,t} \} \quad \forall t \in \{1, \dots, 12\}, \quad \forall k \in \{1, 2, 3\}$$
(6)

#### 2.2.2 Formulation of the Optimization Problem

The agency receives a budget for climate adaptation of the reserve. At any time t, the conservation agency aims to maximise utility by purchasing (and selling) reserve sites. Based on our discussions in Chap. 2.2.1, the aim of the conservation agency is to maximise the extent of the rarest habitat type at any time (cp. Equation 6). We assume that the agency has perfect information on future developments of the sites' ecological value and is hence able to consider future developments in current decisions. We therefore compare the cost-effectiveness of the dynamically optimal strategy and the option of selling with the dynamically optimal strategy and an additional constraint that prohibits sales. The binary variables  $b_{it}$  and  $s_{it}$ represent the agency's choice of sites to purchase and sell at time t, respectively, and are the agency's choice variables. These choices are subject to a budget constraint (Eqs. 8, 9 and 10), a possible constraint on selling under the 'no sale' policy (Eq. 11), and a proximity constraint (Eq. 12). The proximity constraint is introduced in line with the previous literature (e.g. Jafari and Hearne 2013) as habitat cannot generate itself in isolation, but relies on plant and animal species colonising a newly conserved habitat area, and this colonisation becomes more likely if the site is in close proximity of existing habitat sites (Watts et al. 2020). Table 1 provides an overview of all sets, indices, parameters and variables.

$$max. U = max.[min. \{U_{k,t}\}] \quad \forall t \in \{1, \dots, 12\}, \quad \forall k \in \{1, 2, 3\}$$
(7)

s.t.

$$c(t) \ge \sum_{i \in N} (1+a) \times p_i \times b_{it} \qquad \forall t \in \{1, \dots, 12\}$$
(8)

$$c\left(t=1\right) = K\tag{9}$$

$$c(t) = c(t-1) \times (1+r)^5 + (1+r)^{2.5} \times \sum_{i \in N} (1-a) \times p_i \times s_{it-1} \forall t \in \{2, \dots, 12\}$$
(10)

Deringer

Table 1       Overview of used         sets, indices, parameters,       variables and objectives in the         optimisation       Variables	Symbol	Description			
	Sets and Indices				
	Ν	Set of all sites (cells) in the landscape			
	$M_{i}$	Set of all sites (cells) that make up the Moore neighbourhood of site <i>i</i>			
	Parameters				
	$p_i$	Price of site <i>i</i> (in monetary units (MU))			
	h <sub>kit</sub>	Ecological value of site $i$ for habitat type $k$ at time $t$			
	a	Transaction cost (as a percentage value of $p_i$ )			
	r	Interest rate			
	Κ	Adaptation budget (in monetary units (MU))			
	Variables				
	$\mathcal{Y}_{it}$	Equal to 1 if site $i$ is part of the reserve at time $t$ , 0 otherwise			
	c(t)	Capital available at time t (in monetary units (MU))			
	$b_{it}$	Equals 1 if site <i>i</i> is purchased at time <i>t</i> , 0 otherwise			
	s <sub>it</sub>	Equals 1 if site $i$ is sold at time $t$ , 0 otherwise			

$$y_{t+1} \ge y_t \tag{11}$$

$$b_{it} \leq \sum_{j \in M_i} (y_{jt} - s_{jt}) \quad \forall t \in \{1, \dots, 12\}$$
 (12)

Equation 8 describes the budget constraint in time step t for the conservation agency, where c(t) is the available capital in time t. The cost of purchasing site i is equal to its price,  $p_i$  plus transaction costs a.  $b_{it}$  is a binary variable indicating whether site i is purchased in time t. The available budget in the first period is given by Eq. 9 and is equal to an initial 'adaptation budget' K. The available budget in the subsequent periods is given by Eq. 10: The budget may consist of the budget remainders of the previous period c(t-1) including corresponding interest payments for five years, and, under the 'sale' policy, compounded funds from selling reserve sites. Funds from selling reserve sites consist of the price of site i,  $p_i$  net of transaction costs a. Note that  $s_{it}$  is a binary variable taking the value 1 when site i is sold in time t and 0 otherwise. While sales may occur throughout the 5-year period, for simplicity, we assume that all sites are sold in the middle of each 5-year period, i.e. 2.5 years are used in the respective exponent for compounding.Constraint 11 represents the constraint that prohibits sales in the 'no sale' policy:  $y_{jt}$  is a binary variable indicating that site j is part of the reserve network. By ensuring that  $y_{t+1}$  is never smaller than  $y_t$ , sites that are classified as reserve sites at some point t may not become declassified later on. Constraint 12 ensures that a site is only purchased if at least one neighbouring site is already part of the reserve network: The bracket takes the value 1 for any site *j* that is part of the reserve network and is not sold at time t. Site i may hence only be purchased (i.e.,  $b_{it}$  may only take the value 1) if at least one site within its Moore neighbourhood (indicated by the variable  $M_i$ , cp. Appendix D) is part of the reserve.

#### 2.3 Budget Scenarios

We compare the 'sale' and the 'no sale' policies for different budget constraints in order to see whether the size of the budget has an influence on the value of resale flexibility.

Table 2         Overview of the 10           budget scenarios         \$\$		K=0MU	K=40MU	U K=80MU	K=120MU	K=160MU
	'no sale' 'sale'	no sale– no funding sale– no funding	no sale– low funding sale– low funding	no sale– medium funding sale– medium funding	no sale– high funding sale– high funding	no sale– very high funding sale– very high funding
Table 3 Overview of the sensitivity analyses	Parameter/ assumption		Base	e case	Sensitivity analysis	
	Transaction costs a		a 10%	)	5%, 20%	
	Interest rate r		3%		0%, 5%	
	Price of site $i(p_i)$		Ran disti	dom ibution	2 additional random distribution	
	Sensitivity to climate change		nate H <sub>1</sub> i undo char	s expanding er climate nge	$H_1$ is shifting under climate change	
	Proximity constraint		aint Acc Eq.	ording to 9	No proximity constraint	
	Habitat on non-con- served sites		on- Non sites habi	-conserved provide no tat	Non-conserved sites provide 10% of the habitat they would pro- vide under conservation	

Specifically, we consider that the agency may have no, a low, medium, high or a very high adaptation budget K to adapt the reserve network. In the 'low funding' case, the size of the 'adaptation budget' is 40MU (monetary units), in the medium, high and very high funding cases it is increased to 80MU, 120MU and 160MU, respectively.

We therefore consider 10 budget scenarios (Table 2):

We determine the optimal spatial configuration of reserve networks over time for the 'sale' and 'no sale' policy separately and then assess the value of resale flexibility for different funding levels in the Sec. 3.

### 2.4 Sensitivity Analyses

There are some parameters that are uncertain and that may influence the outcome. We vary these parameters in sensitivity analyses to examine their impact on the value of resale flexibility (Table 3 provides an overview).

First, we vary transaction costs *a* and interest rate *r*. We further examine in how far our assumptions regarding the distribution of prices and ecological values influences overall results by considering two additional random cost distributions, and the case in which  $H_1$  does not expand but shift due to climate change. Regarding the constraints of the optimisation, we examine the impact of relaxing the proximity constraint. The proximity constraint was added as new habitat can only be created if plant and animal species can colonise it and hence represents an ecological constraint. However, the necessary degree of connection between reserve sites depends strongly on the dispersal ability of the species of interest (Watts et al. 2020), and the proximity constraint may not be necessary for habitat for highly



funding level K (in monetary units)

Fig. 4 Utility of conservation

mobile species or in case of species relocation. Finally, we examine in how far our results hold when non-conserved sites also provide some habitat.

## 3 Results

### 3.1 The Value of Resale Flexibility

Figure 4 shows the utility of conservation for the 'sale' and 'no sale' policies for different budget constraints. It can be seen that utility increases as the available budget *K* increases, and that the 'sale' policy is preferable over the 'no sale' policy. However, it can also be seen that with increasing funding, the value of resale flexibility<sup>4</sup> decreases: While resale flexibility increases utility by 77% when no additional funding is available, utilites of the two policies are very similar in the "very high funding" case. When no additional capital is available, allowing for sales is the only option of adapting the reserve network, and resale flexibility decreases as the increasing budget also provides flexibility to adapt the reserve network. These results are robust to other cost distributions (results not shown). We also examine whether the specific assumptions regarding how climate change affects the habitat types influences the value of resale flexibility. We find that results regarding

<sup>&</sup>lt;sup>4</sup> Remember that the value of resale flexibility is defined as the percentage improvement in utility received from habitat conservation when moving from the 'no sale' policy to the 'sale' policy.

the value of resale flexibility for different budget constraints are qualitatively similar (see Appendix E for details).

Flexibility in reserve design may in principle be provided by replacing decreasingly cost-effective reserve sites (here called "resale flexibility"), but also by adding sites with external funding. The value of both of these types of adaptation mechanisms has diminishing marginal utility: The diminishing marginal utility of resale flexibility provided by the 'sale' policy can be observed by the two curves in Fig. 4 moving increasingly close to each other, while the diminishing marginal utility of increasing budgets can be observed by the concavity of the function. We examine this in greater detail in Fig. 5. Figure 5a shows that the marginal utility of increasing the budget by a fixed amount of 40MU as well as the value of resale flexibility decreases rapidly when moving from the "no funding" case to a funding level of 40MU, and starts to level off for higher funding levels. In all cases, increasing the available funding by 40MU leads to slightly higher utility levels than switching to the 'sale' policy. However, Fig. 5b shows that the advantage of increasing the funding level over resale flexibility increases from 6.5% for the case of no funding to 200% for a funding level of 120MU. The decreasing value of resale flexibility for increasing funding levels is thus particularly pronounced, providing further evidence for the value of resale flexibility when conservation budgets are very small.

The utility of habitat is influenced by both the impact of climate change and the impact of resale flexibility. Figure 6 provides an indication of the relative contribution of these factors: The value of resale flexibility is 77% (i.e., allowing for sales improves the utility achieved by 77%). Here, the 'no sale' policy may be interpreted as a case where the conservation agency is unaware of the need to adapt the reserve network to climate change, or is unable to do so due to a lack of adaptation funding and resale flexibility. Allowing resale flexibility then increases the utility of habitat conservation. However, even with resale flexibility, the utility of habitat conservation is 42% lower than in the hypothetical case of no climate change. Climate change therefore poses a strong limitation on the maximum utility of habitat conservation that may be achieved, and the loss in utility caused by climate change can only be partially compensated by providing resale flexibility. Finally, Fig. 6 shows the utility of habitat when conserving the whole landscape under climate change. It can be seen that the utility of this case is only slightly greater than the utility of the reserve under the hypothetical assumption of no climate change. However, it requires a more than



Fig. 5 Diminishing marginal utility of increasing the adaptation budget K by a fixed amount of 40MU and adopting the 'sale' policy (Fig. 5a), and relative advantage of the fixed budget increase over the 'sale' policy (Fig. 5b) for different funding levels



Fig. 6 Utility of habitat conservation in case of 'no sale', 'sale', under no climate change (no funding case) and if the whole landscape was conserved under climate change





12-fold budget increase in comparison to the reserve depicted under "no climate change". A large budget increase would therefore be required to achieve a similar level of utility under climate change as under no climate change.

We conducted a range of sensitivity analyses. Figure 7 shows the impact of changes to the interest rate and changes to the size of transaction costs. Figure 7a shows that an increase in interest rates increases the utility of habitat conservation, and that it reduces the value of resale flexibility from 86% for an interest rate of 0.00 to 2.5% for an interest rate of 0.05.

The reason for this result is that an increasing interest rate increases the available budget (cp. Equation 10), and an increase in budget decreases the value of resale flexibility (cp. Figure 4). Figure 7b shows that increasing transaction costs decrease the utility of habitat conservation slightly. The value of resale flexibility for different transaction costs is similar in all three cases.

Figure 8 shows the impact of relaxing the proximity constraint (Eq. 12) and the case that sites that are not conserved also provide some habitat. Figure 8a shows that relaxing the proximity constraint during the climate change adaptation process improves the utility of habitat conservation achieved. Moreover, it can be seen that the marginal increase in utility is larger for the 'sale' policy than for the 'no sale' policy, implying that the value of resale flexibility increases as the proximity constraint is relaxed. When sales are not allowed, new sites may only be added with the adaptation budget K, leading to fewer transactions than in the 'sale' policy. While relaxing the proximity constraint does provide added flexibility in the choice of reserve sites and hence, additional utility in the 'no sale' policy, the larger number of transactions conducted in the 'sale' policy– each of which may be chosen more freely when relaxing the proximity constraint– leads to a larger added value overall.

Figure 8b shows that when non-conserved sites also provide some ecological benefit, the utility of habitat conservation increases. However, the differences between the case in which non-conserved sites do not provide any habitat and the case in which they do is particularly large for small budgets. This occurs as for smaller budgets, a smaller proportion of the landscape is conserved, implying that large areas are not conserved that then provide additional benefits. As an increasing proportion of the landscape is conserved, the added benefit by this assumption decreases. Moreover, the general insights regarding the value of resale flexibility for different budget constraints also hold under this assumption: The value of resale flexibility is still particularly large for small budget constraints, while it decreases as the budget constraint increases.

#### 3.2 The role of Scarcity in Driving Results

The utility received from the different policies is driven by scarcity. To examine this in more detail, we first consider how the different habitat types develop over time for the 'sale' and 'no sale' policies (Fig. 9). Figure 9a represents the case of no adaptation and may be inter-



Fig. 8 Sensitivity analyses for relaxing the proximity constraint (Fig. 8a) and considering the case that non-conserved sites provide 10% of the habitat value that they would provide if they were conserved (represented by "+") (Fig. 8b)

preted as the direct impact of climate change on the initial reserve. The impact of climate change captured in the equation of motion (Eq. 2) can be clearly seen: while habitat  $H_1$  (blue line) expands, habitat  $H_2$  (yellow line) decreases due to its shifting uphill, and habitat  $H_3$  (red line) contracts strongly as it cannot shift further uphill. Considering other funding levels and the impact of resale flexibility, it can be seen that initially, habitat  $H_1$  is the rarest habitat type but increases over time in all 6 scenarios. Habitat  $H_2$ , which we expect to move 'uphill', increases in its extent in all scenarios except in the case of no additional capital (Fig. 9a and d). Finally, habitat  $H_3$  becomes increasingly rare in all scenarios under climate change. However, when the reserve network may be expanded by providing sufficient additional capital (Fig. 9b and c) and/or allowing for sales (Fig. 9d-f), the loss in  $H_3$  is less pronounced than without these adaptation options (Fig. 9a).

Overall, Fig. 9b-f thus show the conservation impact of alternative policies relative to the "do nothing" policy depicted in Fig. 9a. For all habitat types, all considered alternative policies are strictly preferable at the final time step over the "do nothing" policy. However, the conservation impact in terms of additional habitat conserved/ avoided habitat loss depends strongly on the habitat type and policy under consideration. For example, the relative conservation impact of the 'sale' policy (in comparison to the 'no sale' policy) is largest for the increasingly rare habitat type  $H_3$  in case of no additional funding (cp. Figure 9a and d).

We now analyse the conservation impacts of the 'sale' policy relative to the 'no sale' policy in more detail. The 'sale' policy is strictly dominant to the 'no sale' policy in the no funding case, as the conservation impact of the 'sale' policy is at least as large as that of the 'no sale' policy for all three habitat types. Under medium funding, no policy is strictly preferable to the other, as the conservation impact of the 'sale' policy is higher for one habitat type, but the impact of the 'no sale' policy is higher for others. In this case, choosing the preferred policy hence depends on the underlying utility function. In the very high funding case, the 'no sale' policy is dominant to the 'sale' policy, as the conservation impact of the 'no sale' policy is at least as high as the impact of the 'sale' policy for all three habitat types (considering that the outcome for  $H_3$  under both policies is very similar).

Focusing on the increasingly rare habitat type  $H_3$ , the advantage of the 'sale' policy decreases as the amount of additional funding increases. In case of no funding, the 'sale' strategy improves the final conservation impact for the rarest habitat type by 143% (from 0.7 to 1.7 (Fig. 9a and d)). In the medium funding case, the 'sale' policy only improves the conservation impact by around 14% (from 2.8 to 3.2 (Fig. 9b and e)), while the impact in the very high funding case is comparable for both policies (4.1 vs. 4.2 in Fig. 9c and f). This result, combined with the observation that only in case of no funding all three habitat types receive a positive conservation impact from adopting the 'sale' policy, is the key driver explaining the value of resale flexibility observed in Fig. 4.

Similarly, for habitat type  $H_2$  we find that the conservation impact of the 'sale' policy only exceeds the conservation impact of the 'no sale' policy in the case of no funding. For habitat type  $H_1$ , the 'sale' policy has no clear advantage over the 'no sale' policy for any funding level. All in all, examining the habitat type-specific conservation impacts of the two policies shows that the advantages of the 'sale' policy is particularly pronounced for increasingly rare habitat types.

To examine the role of scarcity in more depth, we consider Fig. 10, which shows the production possibility frontiers (PPFs) for the three habitat types over time. It can be seen that the "ecological productivity" in the case study overall increases for H<sub>1</sub>; decreases, espe-



**Fig. 9** Area covered by each habitat type over time for the 'no sale' (Fig. 9a-c) and 'sale' policies and levels of additional funding ranging from none (Fig. 9a, d), medium (K=80MU) (Fig. 9b, e) to very high (K=160MU) (Fig. 9c, f)

cially in the second half of the model timeframe for  $H_2$ ; and decreases strongly for  $H_3$  (consider the shifts in the respective PPFs over time). This is a direct result of the habitat types' equations of motion (Eq. 2), and the shifts in the PPFs represent the habitat types' scarcity.

The PPFs may then be used to visualise how the changes in relative scarcity drive the overall outcome. In t=1, actual outcomes are relatively close to the PPFs in all three cases, as no habitat is particularly threatened. The outcomes achieved for the different habitat types in t=1 (blue data points) are hence a similar distance to the habitat type's PPF in t=1 (blue curves)– no habitat type is particularly preferred by the decision maker over another. Over time, the PPFs of H<sub>3</sub> (and H<sub>2</sub>) move closer to the origin, while the PPF of H<sub>1</sub> moves outwards. At the same time, reserve sites are adapted in a way that by t=12, actual outcomes for H<sub>3</sub> are very close to the PPF, while outcomes for H<sub>1</sub> are increasingly far from its potential PPF. This visualises how the scarcity of habitat types drive the optimisation. Regarding H<sub>3</sub> in particular, it can be seen that the outcome achieved is very close to the maximum that can still be achieved given climate change, and that outcomes achieved in earlier time steps are now beyond the PPF of H<sub>3</sub>. This effect occurs as reserve sites are selected to maximise the extent of H<sub>3</sub> in the final time step as the PPF of H<sub>3</sub> experiences the strongest contraction, visualising the high value of conserving this habitat type due to its high scarcity.

Finally, we would like to highlight that none of the chosen allocations lie on the PPFs, which might suggest an inefficient allocation of resources at first glance. However, there are four reasons for the discrepancies between the actual allocations and the PPFs: first, the budget constraint may not allow reaching the PPF (the PPF was drawn for a budget constraint of 1001 monetary units, representing the value of the reserve in t=12 in the "no sale, very high funding" case). This budget is not available in earlier time steps or for scenarios of smaller budget constraints. Hence, it can be seen that for higher funding levels, the outcomes achieved move closer towards the bottom right, representing a shift outwards and towards habitat conservation over "resources for other purposes". Second, the PPFs show only one habitat type each. The actual allocations not only create a single habitat type, but all three. Third, the optimisation procedure selects sites by considering future conditions, while the PPF may be interpreted as optimal at a given point in time. This means that the optimisation may select sites that may not be optimal initially, but improve the outcome in later time steps. This may be observed by considering that the outcome of H<sub>3</sub> moves increas-



**Fig. 10** Production possibility frontiers (PPFs) showing the trade-offs between resources available for other purposes in monetary units (MU) and conservation of habitat type 1 (Fig. 10a), habitat type 2 (Fig. 10b) and habitat type 3 (Fig. 10c) over time. The common budget constraint is 1001 monetary units (i.e., the value of the reserve in t=12 in the "no sale, very high funding" case). A visualisation of the development of the PPFs over time has been uploaded separately as Video for Fig. 10



Fig. 11 Site habitat permanence measured as the % of the original habitat areas maintained in all time steps depending on the level of funding

ingly close to the maximum in the late time steps. Finally, the PPF assumes that sites may be reallocated at no cost (i.e., transaction costs equal zero), while the actual allocations are subject to path dependencies and transaction costs. The impact of these factors explains the remaining differences between the PPF and the actual outcomes achieved.

# 3.3 Site Habitat Permanence

A potential loss in site habitat permanence<sup>5</sup> is typically one of the key arguments brought forward against the 'sale' policy. In this section, we therefore briefly examine to what extent the 'sale' policy decreases site habitat permanence for each habitat type (Fig. 11). For a more detailed discussion on these aspects, see Appendix F1.

Considering the 'no sale' policy, Fig. 11 shows that even if sales are not permitted, 100% of initial habitat areas can be maintained over time only for  $H_1$  (the expanding habitat type). This occurs as given climatic changes, maintaining a stable reserve network does not necessarily lead to site habitat permanence, as climatic changes themselves lead to changes in habitat distributions within reserve networks, even when the sites are maintained continuously over time (Lewis and Polasky 2018). For  $H_2$  (the shifting habitat type), even under 'no sale' only 64% of initial habitat areas can be maintained under climate change, while for  $H_3$  (the contracting habitat type), only 9% of initial habitat areas can be maintained due to changing climatic conditions. The curves representing the 'no sale' policies hence show the extent to which climate change causes habitat turnover even in a static reserve network.

<sup>&</sup>lt;sup>5</sup> Remember that site habitat permanence of a chosen habitat type is measured as the percentage of the areas covered initially by that habitat type that is maintained throughout the run-time of the model in the same grid cells.

Comparing the 'sale' and 'no sale' policies, it can be seen that site habitat permanence tends to be lower under the 'sale' policy. However, two effects can be observed: (1) site habitat permanence increases under the 'sale' policy with increasing levels of funding. The reason for this is that as the budget increases, fewer sites are sold (see Appendix F2 for details). (2) The difference between the 'sale' and 'no sale' policies is smallest for the habitat type that becomes increasingly rare,  $H_3$ . Sites that contain  $H_3$  are therefore unlikely to be sold even if they could be, as the optimisation maximises the extent of the scarcest habitat type. This means that allowing for sale tends to decrease site habitat permanence when compared to the 'no sale' case, but mainly for the expanding and thus not threatened habitat type.

## 4 Discussion and Conclusion

Under climate change, the comparative advantage of potential reserve sites to "produce" habitat may change. To maintain cost-effectiveness, reserves need to be adapted to these dynamic changes and conservation sites need to be reallocated according to their "ecological productivity". In this paper, we analysed two land purchase policies in order to assess their suitability for the climate adaptation problem in biodiversity conservation: a typical 'no sale' policy and a novel 'sale' policy that provides resale flexibility for climate change adaptation. We developed a climate-ecological-economic model to analyse under what conditions the novel 'sale' policy provides cost-effectiveness advantages.

We find that the value of resale flexibility is particularly high when the conservation agency only has a small budget to adapt the reserve network to climate change. This "budget effect" implies that the value of resale flexibility is also higher in case of low interest rates as high interest rates increase the available budget over time. A similar budget effect has been identified before in a different conservation setting (Lewis et al. 2011), and is caused by increasing budgets leading to a large proportion of the landscape being conserved, which implies that the likelihood of including sites that become cost-effective in the future increases.

However, in the setting of changing comparative advantages the budget effect occurs for an additional reason: An increasing budget acts as an additional source of flexibility as it allows the agency to add increasingly cost-effective sites to the reserve network, hence acting as a means of climate adaptation of the reserve. For low budgets, increasing the budget or adopting the 'sale' policy may to some extent be seen as alternative climate adaptation measures, both of which improve the utility received from habitat conservation. However, for low budgets the value of resale flexibility in comparison to the utility received from increasing the budget is particularly high. At higher funding levels, the value of resale flexibility ceases as there is no longer a need to replace existing reserve sites following the argumentation by Lewis et al. (2011)– but buying new reserve sites still provides ecological benefits. The *relative* value of increasing budgets in comparison to resale flexibility hence increases with increasing budgets. However, the marginal value of increasing budgets also exhibits diminishing marginal returns as increasingly less cost-effective sites are added.

When considering ecosystems of highly mobile species, one may consider relaxing the proximity constraint which allows the conservation agency to select new reserve sites more freely. In this case, the value of resale flexibility is particularly high as the option of sales increases the number of transactions relative to the 'no sale' policy, and the value of each of

those transactions is increased when relaxing the proximity constraint: Without the proximity constraint, the agency may choose sites to maximise utility, while the proximity constraint restricts the choice of sites. In research on agglomeration payments to incentivise contiguous habitat creation with landowners, a similar effect has been termed the "patch restriction effect" (Drechsler et al. 2010). Our results are robust to the considered alternative cost distributions, climate change sensitivities of the habitat types, and the case in which non-conserved sites provide some ecological value.

Previous research has suggested that allowing the sale of reserve sites may improve cost-effectiveness by facilitating adaptation of reserves to the dynamic conditions of climate change (Alagador et al. 2014, 2016), but possible increases in habitat turnover are the key reason against the 'sale' policy (Appendix F1). To reduce the negative impacts of habitat turnover, previous research has suggested to tie the 'sale' option to restrictions on future land use (Hardy et al. 2018), and that temporary protection may be particularly valuable for early-successional habitats (Ranius et al. 2022). Our research suggests that two additional important factors influencing this trade-off are (1) whether, and to what extent, other sources of funding are available for adapting the reserve network, and (2) how climate change influences the relative occurrence of the habitat type in question. More specifically, our model allowed us to identify a new trade-off between the 'sale' and 'no sale' policies: 'sale' provides advantages for habitat types that become increasingly rare, especially for low levels of additional funding, as 'sale' then provides an important option to adapt the reserve network to changing climatic conditions. 'No sale' on the other hand improves site habitat permanence mainly for the expanding habitat type, while only little permanence is lost under 'sale' for the increasingly rare habitat type.

Our paper builds on the scarce previous literature on cost-effective policy instrument design for biodiversity under climate change (Huber et al. 2017; Schöttker and Wätzold 2022; Gerling et al. 2023). We consider land purchase as a typical policy instrument for biodiversity conservation and examine the value of resale flexibility provided by the 'sale' policy. Our paper therefore also has strong links to previous research on the value of flexibility in reserve design (e.g., Costello and Polasky 2008; Shah and Ando 2016; Lewis and Polasky 2018). However, this research has largely focused on the issue of uncertainty and irreversible land use. By contrast, we assume that decision makers have perfect knowledge of future conservation costs and impacts of reserve sites, and the value of flexibility arises from changes in comparative advantages. If decision makers did not have perfect information on future costs and impacts, the value of resale flexibility is therefore likely to increase (e.g. Westphal et al. 2003; Costello and Polasky 2004; Kassar and Lasserre 2004; Costello and Polasky 2008; Drechsler 2020). Changes in comparative advantages of sites to produce habitat as well as uncertainty regarding future costs and ecological productivity of the sites hence both contribute to the value of flexibility in conservation decision making. Similarly, much of the previous research has focused on permanent habitat loss (e.g., Arrow and Fisher 1974; Albers 1996; Leroux et al. 2009; Shah and Ando 2016), while we consider that sites that are not conserved may generate habitat if they are set under conservation. In our model context, shifting reserve sites towards newly cost-effective sites in the key pattern observed in spatial habitat designation over time. This shift would likely not be as prominent if assuming irreversible habitat loss, as the optimisation of our model would likely focus in the early time steps much more on sites that become cost-effective in later time steps. Irreversible habitat loss hence represents an ecological constraint on spatial flexibility. Finally, we assume that there is no uncertainty about the future availability of reserve sites (unlike, e.g., Costello and Polasky (2004). Considering restricted availability would limit future site choices, implying that cost-effectiveness decreases. Moreover, considering that sites may no longer be available for conservation in the future may lead to other site selection strategies: sites that will become highly cost-effective may be selected sooner to ensure that they are under conservation in later time steps.

As in any model, we had to make some assumptions to simplify reality. First, we assume that the habitat is generated instantly once a grid cell is conserved. Considering the time lag between conservation action and outcomes essentially creates a "conservation credit" (as opposed to an "extinction debt" when species do not become extinct immediately after a habitat is degraded) (Watts et al. 2020). Including a time lag would reduce the benefits in conservation strategies relying on frequent habitat turnover. Given the conceptual nature of the model, we decided against specifically modelling this aspect: The size of the time lag may vary between years and centuries or even millennia (Watts et al. 2020), and depend largely on the habitat type in question (Drechsler and Hartig 2011; Wilson et al. 2011; Possingham et al. 2015), the initial conditions of the site to be restored (Wilson et al. 2011), the connectivity to the existing reserve network (Watts et al. 2020), and the dispersal ability of the species inhabiting the chosen habitat type (Watts et al. 2020). We refer the interested reader to previous research on the topic (Drechsler and Hartig 2011; Possingham et al. 2015). Similarly, we chose habitat area as the ecological indicator of interest. Habitat area is a popular ecological indicator due to its relative simplicity (zu Ermgassen et al. 2019), and it is often expected that habitat conservation will lead to certain species inhabiting this site (Simpson et al. 2022). However, the creation of a habitat site may not necessarily translate to all expected species actually inhabiting that site. For example, the distance between two habitat sites plays a large role in determining colonisation success of a new habitat site (Westphal et al. 2003). If the aim is to conserve a specific species, species-based approaches rather than a habitat-based approach may therefore be more suitable (Simpson et al. 2022).

Second, the economic model contains some simplifications. We assume that the prices of the grid cells remain constant. Under climate change, opportunity costs of land parcels will likely be affected in a spatially heterogeneous manner (Schöttker and Wätzold 2022; Gerling et al. 2023). Regarding conservation in agricultural landscapes, for example, some areas may become more suitable for agricultural activity (for example areas in higher altitudes), while others may become less suitable (for example areas suffering from increasing droughts), which may lead to opposing developments of opportunity costs (Ray et al. 2019; Lachaud et al. 2021). If climate change leads to a relative decrease (increase) in conservation costs of newly (formerly) cost-effective sites, the cost-effectiveness of the static reserve network is further reduced (increased) in comparison to the case of static opportunity costs. However, we decided to ignore this aspect as the changes to opportunity costs depend largely on which area is considered. Future research based on specific case study areas or on conceptual models investigating the interactions between the relevant variables may include these aspects in a similar model.

Research on the value of resale flexibility within the context of policy instrument design under climate change has so far only received little attention by economists– despite being an important issue in climate adaptation for biodiversity conservation. In particular, understanding under what conditions the value of flexibility is particularly high is valuable given political inertia and potentially high transaction costs of changing existing regulations (Nalle et al. 2004; Burch et al. 2014). The CEE model presented in this article is able to assess the cost-effectiveness of different land purchase policies for climate adaptation of biodiversity conservation. A similar model may be used in future research to address other questions of adapting conservation to climate change that are beyond the scope of this paper. In particular, we would like to suggest the following topics for future research: First, in our model we consider two extreme cases of the 'sale' and 'no sale' policies for land purchase. Future research could examine other policy instruments with medium levels of flexibility– e.g. in (temporary) conservation easements (Rissman et al. 2015; Owley et al. 2018), conservation contracts (Schöttker and Wätzold 2022), offsets (Bull et al. 2013; Gerling and Wätzold 2021), or 'purchase, protect, resale' programmes (in which land may be sold with restrictions on future land use) (Hardy et al. 2018). Given the policies' level of flexibility, they provide promising alternatives to land purchase in the face of climate change and analysing their cost-effectiveness may provide interesting insights for conservation practice.

Second, we consider the case of adapting existing reserves that have not been designed with the impacts of climate change in mind. Parts of the value of resale flexibility will hence arise to remedy past "misallocations". However, given that climate change impacts the comparative advantages of different conservation sites in "producing" habitat, resale flexibility may be expected to continue to provide value even if climate change was considered from the beginning: Resale flexibility allows the conservation agency to adapt in line with changes in comparative advantages, rather than having to choose the allocation that is optimal on average over time. Examining these effects in detail remains a question for future research. In particular, it may be interesting to consider the impact of uncertainty in conservation decision-making in this context– for example considering questions pertaining to when the agency receives information, and in how far this information is subject to uncertainty.

Finally, we consider a one-off adaptation budget and assume that funding may be invested to be used in future time periods. Previous research has shown that this is an additional source of (valuable) flexibility (Lennox et al. 2017). Future research could examine the value of resale flexibility in different funding models, e.g. considering regular adaptation budgets and restrictions to the roll-over of funding. Again, examining in how far the value of resale flexibility in different funding models (e.g. one-off payments vs. regular payments) depends on assumptions regarding perfect information or uncertainties seems particularly interesting.

Given the rapid loss of global biodiversity (Dasgupta 2021) and the need to adapt existing reserve networks to climate change (Pyke and Fischer 2005; Fung et al. 2017; Graham et al. 2019; Lawler et al. 2020), we believe that further (economic) research on how to design policy instruments of biodiversity conservation for the climate adaptation problem will provide valuable information to conserve biodiversity cost-effectively in the future.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10640-024-00932-4.

Acknowledgements We would like to thank two anonymous reviewers for their helpful and interesting comments and suggestions.

Funding Open Access funding enabled and organized by Projekt DEAL.

#### Declarations

Conflict of interest The authors have no relevant financial or non-financial interests to disclose.

**Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by/4.0/.

# References

Abel AB, Eberly JC (1996) Optimal investment with costly reversibility. Rev Econ Stud 63(4):581-593

- Alagador D, Cerdeira JO, Araújo MB (2014) Shifting protected areas: scheduling spatial priorities under climate change. J Appl Ecol 51(3):703–713
- Alagador D, Cerdeira JO, Araújo MB (2016) Climate change, species range shifts and dispersal corridors: an evaluation of spatial conservation models. Methods Ecol Evol 7(7):853–866
- Albers HJ (1996) Modeling ecological constraints on tropical forest management: spatial interdependence, irreversibility, and uncertainty. J Environ Econ Manag 30(1):73–94
- Albers HJ (2022) Protected Area Network Expansion and Management: Economics to improve conservation outcomes. Environ Resource Econ 83:955–972
- Armsworth PR, Acs S, Dallimer M, Gaston KJ, Hanley N, Wilson P (2012) The cost of policy simplification in conservation incentive programs. Ecol Lett 15(5):406–414
- Arrow KJ, Fisher AC (1974) Environmental preservation, uncertainty, and irreversibility. Q J Econ 88(2):312
- Baldwin CY (1982) Optimal sequential investment when capital is not readily reversible. J Finance 37(3):763-782
- Bull JW, Suttle KB, Singh NJ, Milner-Gulland EJ (2013) Conservation when nothing stands still: moving targets and biodiversity offsets. Front Ecol Environ 11(4):203–210
- Burch S, Berry P, Sanders M (2014) Embedding climate change adaptation in biodiversity conservation: a case study of England. Environ Sci Policy 37:79–90
- Cabeza M, Moilanen A (2001) Design of reserve networks and the persistence of biodiversity. Trends Ecol Evol 16(5):242–248
- Calkin DE, Montgomery CA, Schumaker NH, Polasky S, Arthur JL, Nalle DJ (2002) Developing a production possibility set of wildlife species persistence and timber harvest value. Can J for Res 32(8):1329–1342
- Campos-Cerqueira M, Terando AJ, Murray BA, Collazo JA, Aide TM (2021) Climate change is creating a mismatch between protected areas and suitable habitats for frogs and birds in Puerto Rico. Biodivers Conserv 30(12):3509–3528
- Costello C, Polasky S (2004) Dynamic reserve site selection. Resour Energy Econ 26(2):157-174
- Costello C, Polasky S (2008) Optimal harvesting of stochastic spatial resources. J Environ Econ Manag 56:1–18
- Dasgupta P (2021) The Economics of Biodiversity: the Dasgupta Review. HM Treasury, London
- Davis GA, Cairns RD (2017) The odd notion of reversible investment. J Banking Finance 81:172-180
- Davis FW, Stoms DM, Costello C, Machado EA, Metz J, Gerrard R, Andelman S, Regan H, Church R (2003) A framework for setting land conservation priorities using multi-criteria scoring and an optimal fund allocation strategy. Report. https://www.researchgate.net/profile/Frank-Davis-9/ publication/238712362\_A\_framework\_for\_setting\_land\_conservation\_priorities\_using\_multi-criteria\_ scoring\_and\_an\_optimal\_fund\_allocation\_strategy/links/00b4952787886d40c5000000/A-frameworkfor-setting-land-conservation-priorities-using-multi-criteria-scoring-and-an-optimal-fund-allocation-strategy.pdf
- Davis FW, Costello C, Stoms D (2006) Efficient conservation in a utility-maximization Framework. Ecol Soc, 11(1)
- Doyen L, Cissé A, Gourguet S, Mouysset L, Hardy P-Y, Béné C, Blanchard F, Jiguet F, Pereau J-C, Thébaud O (2013) Ecological-economic modelling for the sustainable management of biodiversity. CMS 10:353–364

- Drechsler M (2020) Conservation management in the face of climatic uncertainty- the roles of flexibility and robustness. Ecol Complex 43:100849
- Drechsler M, Hartig F (2011) Conserving biodiversity with tradable permits under changing conservation costs and habitat restoration time lags. Ecol Econ 70(3):533–541
- Drechsler M, Wätzold F (2020) Biodiversity conservation in a dynamic world may lead to inefficiencies due to lock-in effects and path dependence. Ecol Econ 173:106652
- Drechsler M, Wätzold F, Johst K, Shogren JF (2010) An agglomeration payment for cost-effective biodiversity conservation in spatially structured landscapes. Resour Energy Econ 32(2):261–275
- Fuller RA, McDonald-Madden E, Wilson KA, Carwardine J, Grantham HS, Watson JEM, Klein CJ, Green DC, Possingham HP (2010) Replacing underperforming protected areas achieves better conservation outcomes. Nature 466(7304):365–367
- Fung E, Imbach P, Corrales L, Vilchez S, Zamora N, Argotty F, Hannah L, Ramos Z (2017) Mapping conservation priorities and connectivity pathways under climate change for tropical ecosystems. Clim Change 141(1):77–92
- Gerling C, Wätzold F (2021) An economic evaluation framework for land-use-based conservation policy instruments in a changing climate. Conserv Biol 35(3):824–833
- Gerling C, Drechsler M, Leins J, Keuler K, Radtke K, Schulz B, Sturm A, Wätzold F (2022) Combining ecological-economic modelling and climate science for the cost-effective spatio-temporal allocation of conservation measures in the face of climate change. QOpen2(1):qoac004
- Gerling C, Drechsler M, Keuler K, Sturm A, Wätzold F (2023) Time to consider the timing of conservation measures: designing cost-effective agri-environment schemes under climate change. Agric Resour Econ Rev 52(2):231–249
- Graham V, Baumgartner JB, Beaumont LJ, Esperón-Rodríguez M, Grech A (2019) Prioritizing the protection of climate refugia: designing a climate-ready protected area network. J Environ Planning Manage 62(14):2588–2606
- Hannah L, Midgley GF, Millar D (2002) Climate change-integrated conservation strategies. Glob Ecol Biogeogr 11:485–495
- Hardy MJ, Fitzsimons JA, Bekessy SA, Gordon A (2018) Purchase, protect, resell, repeat: an effective process for conserving biodiversity on private land? Front Ecol Environ 16(6):336–344
- Hartman R, Hendrickson M (2002) Optimal partially reversible investment. J Economic Dynamics Control 26(3):483–508
- Helbing F, Fartmann T, Morkel C, Poniatowski D (2023) Rapid response of vascular plants and insects to restoration of montane grasslands. Front Ecol Evol 11:1148266
- Heller NE, Zavaleta ES (2009) Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biol Conserv 142:14–32
- Huber R, Snell R, Monin F, Brunner SH, Schmatz D, Finger R (2017) Interaction effects of targeted agrienvironmental payments on non-marketed goods and services under climate change in a mountain region. Land Use Policy 66:49–60
- Jafari N, Hearne J (2013) A new method to solve the fully connected reserve network design problem. Eur J Oper Res 231:202–209
- Kassar I, Lasserre P (2004) Species preservation and biodiversity value: a real options approach. J Environ Econ Manag 48(2):857–879
- Lachaud MA, Bravo-Ureta BE, Ludena CE (2021) Economic effects of climate change on agricultural production and productivity in Latin America and the Caribbean (LAC). Agric Econ 53(2):321–332
- Lamprecht A, Semenchuk PR, Steinbauer K, Winkler M, Pauli H (2018) Climate change leads to accelerated transformation of high-elevation vegetation in the central alps. New Phytol 220(2):447–459
- Lawler JJ, Rinnan DS, Michalak JL, Withey JC, Randels CR, Possingham HP (2020) Planning for climate change through additions to a national protected area network: implications for cost and configuration. Philosophical Trans Royal Soc B: Biol Sci 375(1794):20190117
- Lennox GD, Fargione J, Spector S, Williams G, Armsworth PR (2017) The value of flexibility in conservation financing. Conserv Biol 31(3):666–674
- Leroux AD, Martin VL, Goeschl T (2009) Optimal conservation, extinction debt, and the augmented quasioption value. J Environ Econ Manag 58(1):43–57
- Lewis DJ, Polasky S (2018) An auction mechanism for the optimal provision of ecosystem services under climate change. J Environ Econ Manag 92:20–34
- Lewis DJ, Plantinga AJ, Nelson E, Polasky S (2011) The efficiency of voluntary incentive policies for preventing biodiversity loss. Resour Energy Econ 33(1):192–211
- Ludovicy S, Noroozi J, Semenchuk P, Moser D, Wessely J, Talebi A, Dullinger S (2022) Protected area network insufficiently represents climatic niches of endemic plants in a global Biodiversity Hotspot. Biol Conserv 275:109768

- Meyfroidt P, de Bremond A, Ryan CM, Archer E, Aspinal R, Chhabra A et al (2022) Ten facts about land systems for sustainability. PNAS 119(7):e2109217118. https://doi.org/10.1073/pnas.2109217118
- Montoya D, Gaba S, de Mazancourt C, Bretagnolle V, Loreau M (2020) Reconciling biodiversity conservation, food production and farmers' demand in agricultural landscapes. Ecol Model, 416
- Nalle DJ, Montgomery CA, Arthur JL, Polasky S, Schumaker NH (2004) Modeling joint production of wildlife and timber. J Environ Econ Manag 48(3):997–1017
- Owley J, Cheever F, Rissman AR, Shaw RM (2018) Climate Change challenges for Land Conservation: Rethinking Conservation Easements, strategies, and Tools. Denver Law Rev 95(3):727–779
- Paul C, Hanley N, Meyer S, Fürst C, Weisser WW, Knoke T (2020) On the functional relationship between biodiversity and economic value. Sci Adv, 6(5)
- Ponce-Reyes R, Plumptre AJ, Segan D, Ayebare S, Fuller RA, Possingham HP, Watson JE (2017) Forecasting ecosystem responses to climate change across Africa's Albertine Rift. Biol Conserv 209:464–472
- Possingham HP, Bode M, Klein CJ (2015) Optimal conservation outcomes require both restoration and protection. PLoS Biol 13(1):e1002052
- Prober SM, Colloff MJ, Abel N, Crimp S, Doherty MD, Dunlop M, Eldridge DJ, Gorddard R, Lavorel S, Metcalfe DJ, Murphy HT, Ryan P, Williams KJ (2017) Informing climate adaptation pathways in multi-use woodland landscapes using the values-rules-knowledge framework. Agric Ecosyst Environ 241:39–53
- Pyke CR, Fischer DT (2005) Selection of bioclimatically representative biological reserve systems under climate change. Biol Conserv 121(3):429–441
- Ranius T, Widenfalk LA, Seedre M, Lindman L, Felton A, Hämäläinen A, Filyushkina A, Öckinger E (2022) Protected area designation and management in a world of climate change: a review of recommendations. Ambio 52:68–80
- Ray DK, West PC, Clark M, Gerber JS, Prishchepov AV, Chatterjee S (2019) Climate change has likely already affected global food production. PLoS ONE 14(5):e0217148
- Rissman AR, Owley J, Shaw MR, Thompson B, (B) (2015) Adapting conservation easements to Climate Change. Conserv Lett 8(1):68–76
- Schöttker O, Wätzold F (2018) Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area. Biodivers Conserv 27(6):1327–1345
- Schöttker O, Wätzold F (2022) Climate change and the cost-effective governance mode for biodiversity conservation. Environ Resource Econ 82:409–436
- Shah P, Ando A (2016) Permanent and Temporary Policy incentives for conservation under stochastic returns from competing Land uses. Am J Agric Econ 98(4):1074–1094
- Simpson K, de Vries F, Dallimer M, Armsworth P, Hanley N (2022) Ecological and economic implications of alternative metrics in biodiversity offset markets. Conserv Biol 36(5):e13906
- Verbruggen A (2013) Revocability and reversibility in societal decision-making. Ecol Econ 85:20-27
- Vincent C, Fernandes RF, Cardoso AR, Broennimann O, Di Cola V, D'Amen M, Ursenbacher S, Schmidt BR, Pradervand J-N, Pellissie L, Guisan A (2019) Climate and land-use changes reshuffle politicallyweighted priority areas of mountain biodiversity. Global Ecol Conserv 17:e00589
- Watts K, Whytock RC, Park KJ, Fuentes-Montemayor E, Macgregor NA, Duffield S, McGowan PJK (2020) Ecological time lags and the journey towards conservation success. Nat Ecol Evol 4(3):304–311
- Westphal MI, Pickett M, Getz WM, Possingham HP (2003) The use of stochastic dynamic programming in optimal landscape reconstruction for metapopulations. Ecol Appl 13(2):543–555
- Wilson KA, Lulow M, Burger J, Fang Y-C, Andersen C, Olson D, O'Connell M, McBride MF (2011) Optimal restoration: accounting for space, time and uncertainty. J Appl Ecol 48(3):715–725
- Wuepper D, Huber R (2022) Comparing effectiveness and return on investment of action- and results-based agri-environmental payments in Switzerland. Am J Agric Econ 104(5):1585–1604

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.