Land for biodiversity conservation – to buy or borrow?

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Abstract

The conservation of endangered species and habitats frequently requires a certain type of land use which, however, leads to opportunity costs compared to profit-maximising land-use. In such a setting biodiversity conservation organisations have two main options: (1) The 'buy alternative' where they buy the area of interest and either carry out the necessary land-use measures themselves or hire firms to do so, or (2) the 'borrow alternative' where they 'borrow' the land for conservation from private landowners who agree to carry out biodiversity-enhancing land-use measures over a certain period while the conservation organisation compensates them for their opportunity costs. Comparing both alternatives raises the question of budget efficiency, i.e. which alternative will lead to a higher level of biodiversity conservation for available financial resources? In this paper we present a conceptual ecological-economic model, and then apply the model to analyse how changes in ecological and economic parameters influence the relative efficiency performance of the two alternatives.

Keywords

Agri-environment scheme, conservation investment, cost-effectiveness, ecological-economic modelling, make-or-buy decision, payments for environmental services

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1 Introduction

The conservation of endangered species and habitats frequently requires a certain type of land use which, however, leads to opportunity costs compared to the most profit-maximising land-use. A typical example is the conservation of biodiversity in European grasslands where many species and habitats are under threat due to the intensification of agriculture and the abandonment of marginal farming areas (Young et al. 2005; Henle et al. 2008; Metera et al. 2010). Although extensive farming measures and the maintenance of farming in marginal areas are better for conservation, they are costly to farmers. If in such a situation property rights are allocated in a way that land users cannot be forced to carry out land-use measures that are beneficial to biodiversity, administrations, foundations and NGOs working in the field of biodiversity conservation are left with two options.

The first alternative is to buy the area of interest and carry out the land-use measures themselves or hire firms to carry out the land-use measures required to conserve biodiversity (henceforth referred to as the 'buy alternative'). For example, many foundations and NGOs such as the Stiftung Naturschutz Schleswig-Holstein in Germany (Stiftung Naturschutz Schleswig-Holstein 2012) and the Royal Society for the Protection of Birds in the UK (Moss et al. 2011) have acquired grasslands and hire firms or farmers to mow or graze the grasslands in a way that supports their conservation aims.

The second alternative is that the conservation organisations 'borrow' the land for conservation for a certain period of time and offer land users payments to compensate them for the opportunity costs that arise as a result of carrying out biodiversity-enhancing land-use measures. Land users are free to decide whether they will participate in the payment scheme offered to them. If they decide to participate, a contract is signed which obliges the land users to carry out certain measures for a specified period of time after which the land users decide again whether to participate in the payment scheme for a further period (henceforth referred to as 'borrow alternative'). For example, several

German federal states offer farmers a five-year contract which guarantees them annual payments if they agree to adopt a mowing regime which improves biodiversity in endangered grasslands but reduces the profit of farmers (Drechsler et al. 2007).

Conservation organisations are faced with the question of budget efficiency (Wätzold and Schwerdtner 2005; Engel et al. 2008; Wätzold and Drechsler 2014), i.e. which of the two alternatives will lead to a higher level of biodiversity conservation for the budget available to an organisation? In the case of the 'buy alternative' a high initial payment needs to be made to purchase the land which then, however, can be used for conservation purposes for as long as the organisation desires. In the case of the 'borrow alternative' there is no need for a high initial payment and the budget can be invested. The return from this investment can then be used to finance payments to landowners participating in later periods. Some landowners, however, may decide not to renew their contract whereas other landowners who initially did not accept the compensation payments may decide to accept payments in later periods. In this way, the payment alternative leads to habitat patch destruction and creation and thus generates a certain habitat turnover. Some species cannot cope with this habitat turnover and so it has negative implications for biodiversity conservation (Van Teeffelen et al. 2012).

The purpose of this paper is to describe a conceptual ecological-economic model, and apply the model to analyse how changes in ecological and economic parameters influence the relative budget efficiency performance of the two alternatives. To our knowledge this is the first paper to explore whether conservation agencies should buy land and manage it themselves or compensate land users for managing the land in a biodiversity-enhancing manner. Our model builds on insights from research on the impact of habitat turnover on species conservation (Drechsler and Johst, 2010; Johst et al., 2011, 2012) and on ecological-economic modelling to optimise the policy instrument of compensation payments for biodiversity-enhancing land-use measures (Drechsler et al. 2007; Bamière et al. 2011; Cong et al. 2014; Mouysset et al. 2014).

2 The Model

2.1 Landscape structure, conservation costs and landscape dynamics

We consider a landscape which consists of *N* = 100 patches. Each patch can be managed either intensively or extensively. An intensively used patch generates maximum profit for the landowner π_i but there are no benefits for biodiversity. An extensively used patch generates less profit but is beneficial for biodiversity. For simplicity, we assume that the patches are of equal size and the spatial location of the patches does not matter for conservation (Hart et al. 2014).

The profits π_i in the landscape are not equal for all patches *i* but vary in a range from $(\bar{\pi} - \sigma)$ to $(\bar{\pi} + \sigma)$. Heterogeneous profits are not uncommon in real agricultural landscapes due to e.g. differences in soil productivity. The profit associated with each patch is randomly drawn from a uniform distribution.

Two types of landowners are considered: the conservation agency and private landowners. We assume that if the land is owned by the conservation agency and managed extensively, profits are lower than if the land is owned by private landowners and managed in the same way. The reason for this assumption is that private landowners specialise in profit-maximising land management and are experts in this field whereas agencies specialise in conservation management coordination and have less expertise in, and equipment for, such types of agricultural management. Alternatively, the agency may contract private landowners to manage a patch extensively. However, it is likely to be less profitable for the contracted landowners to manage this patch extensively compared to their own patches. If landowners offer one of their patches for extensive management they will select the patch with the lowest costs including transportation costs. In contrast, if the agency offers them a patch to manage they cannot make this choice but will rather demand more money if managing the patch is more costly, for example, because it is further away from their farms.

We calculate the profit for an extensively managed patch by multiplying the potential profit off this patch if it were intensively managed with a landscape-wide factor $(1-f)<1$. To take into account the profit differences for extensively managed patches between private landowners and the conservation agency in a simple manner we assume that $f=1$ if an extensively managed patch is owned by the conservation agency and $0<f<1$ if it is owned by a private landowner. For a discussion of these assumptions see section 5.

In the model we consider 50 periods of equal length, starting in period 0. A period covers the length of a contract between the conservation agency and the landowner to manage the land extensively. The decision on which type of management to implement begins in period 0. For the buy alternative, it is decided in this initial period which patches are managed extensively and afterwards no changes take place until T = 49. For the borrow alternative, it is decided anew in each period which patch is managed extensively (for the reasons see the next section), hence a certain turnover of extensively managed patches occurs.

2.2 The decision problem of private landowners

Landowners are assumed to be profit maximisers, which implies that they manage their land intensively. In principle, landowners might also be willing to manage their land extensively if they receive a compensation payment which at least covers their profit losses. However, we assume that each landowner is willing to manage his land extensively only with a certain probability, $P_{part} < 1$, even if their profit losses are covered or over-compensated (Falconer 2000).

One motivation for this assumption is that some landowners may want to sell their farm and if the buyer is bound in any way by an existing contract, this may negatively affect the selling price (Van Herzele et al. 2011). This implies that a landowner may decide to manage his land extensively in one period but may reverse this decision in the next period if he intends to sell his farm in that period. Another motivation for the assumption that landowners change their management is that landowners often only have expected values about the costs of managing their land extensively and they only receive the full information about costs if they or their close neighbours actually manage their land in an extensive manner. This information may then cause the landowners to reverse their management decision (Frondel et al. 2012).

We therefore assume that in each period landowners will make a new decision about whether to manage their land extensively. They do so with probability P_{part} if the compensation payment they are offered covers at least their opportunity costs of participation. With probability $1 - P_{part}$ they choose not to manage their land extensively, even if their opportunity costs are equal to or lower than the payment.

2.3 The decision problem faced by the nature conservation agency

To induce extensive land use the conservation agency can choose between the options of buying the land and carrying out extensive management itself or offering compensation payments to induce landowners to manage their patches extensively. Consequently, depending on the agency's decision, it faces two different alternatives with different implications for landscape dynamics and habitat patch number. In order to render the outcomes of both alternatives comparable we designed the model in a way that the budget for both alternatives is equal in the initial period 0 and also in the final period *T*.

2.3.1 Buy alternative

In the buy alternative, the agency purchases patches and manages them extensively. We assume that the agency's aim is to maximise biodiversity conservation and therefore it uses the budget in the initial period to buy as many patches as possible. As the location of individual patches has no conservation impact in our model only the buying price and the budget are relevant for the decision of the agency. To maximise the number of patches bought, the agency buys the cheapest patch first, followed by the second cheapest etc. until the budget is exhausted. The price of each patch is calculated on the basis of the discounted future expected profits from intensive land use with the help of the capitalisation formula frequently used in spatial economics (Burt 1986). The price of patch *i* therefore is equal to:

$$price_i = \sum_{t=0}^{\infty} \frac{\pi_i}{(1+r)^t} \quad (1)$$

in which $\pi_i \in [(\bar{\pi} - \sigma), (\bar{\pi} + \sigma)]$ is the randomly assigned profit value of patch *i* and *r* the market discount rate.

Formally, the resulting inter-temporal budget constraint of this alternative reads as follows (with B_0 being the initial budget of the agency, B_T the agencies budget in the final period *T*, $price_{buy,i}$ the buying price of an individual patch, $price_{sell,i}$ the selling price of an individual patch, and $x_{buy,i}$ a dummy variable which equals 1 if patch *i* is bought in period 0, and 0 if it is not bought):

$$B_{T,buy} = B_0 - \sum_{i=1}^{N} price_{buy,i}\, x_{buy,i} + \sum_{i=1}^{N} price_{sell,i}\, x_{buy,i} \quad (2)$$

In period 0 the agency's budget is entirely used up for patch purchases. The purchased patches are then extensively managed throughout all periods, and sold after period *T*. For simplicity, we assume that land prices do not change over time. This assumption, together with the assumption that extensive management made or organised by the agency leads to zero profit, implies that the initial budget and the budget at the end of the final period are identical.

Because patches are bought only once (in period 0) and are sold after the final period *T*, their allocation remains unchanged throughout the simulation time frame. The resulting landscape therefore is static with respect to the location of patches.

2.3.2 Borrow alternative

Following Drechsler et al. (2010) and Wätzold and Drechsler (2014) we assume that the agency knows the average profit for the landscape $\bar{\pi}$, the range of profits σ, and the proportion P_{part} but has no information about the profit for an individual patch and the individual profit loss if the patch is managed extensively. The agency is therefore not able to differentiate payments according to the profit losses of the individual landowners and offers a homogeneous compensation payment to landowners for extensive management. Hence, the payment equals the costs of extensive land management for the marginal landowner.

The payments require periodical expenses which are generated by the agency in the following way. Initially (period 0), the agency is equipped with a certain budget B_0 which is equal to the budget for the buy alternative. This budget is invested in long-term government bonds so that it generates a secure and stable periodical income depending on the interest rate of the bonds *r*. For simplicity, we assume that interest rates are fixed in the time frame of our analysis. This implies that in each period *t* a return of $B_t = B_0 * r$ is generated which is spent on compensation payments in period *t*. Potential leftovers are compounded and transferred to the following period. Based on this constant return, the agency offers in each period – from t = 0 to T = 49 – compensation payments to landowners who in turn decide whether to participate in the borrowing scheme for one period based on their opportunity costs and their participation willingness P_{part}.

Because for each patch the willingness to participate in the scheme is randomly re-drawn in each period the landscape continuously changes with the proportion of patches that remain extensively managed from one period to the other determined by P_{part}.

Assuming that the goal of the conservation agency is to maximise biodiversity conservation the number of extensively managed patches is maximised in each period:

$$\max \sum_{i=1}^{N} x_{comp,i,t} \tag{3}$$

subject to the budget constraint:

$$\sum_{i=1}^{N} x_{comp,i,t} * compensation\ payment \ \leq \ B_t \tag{4}$$

Here, $x_{comp,i,t} = 1$ with probability P_{part} if the compensation payment offered is higher than the costs of extensive management for patch *i* in period *t*. Otherwise it equals 0. To cover the compensation expenses, the periodical budget B_t is used. The compensation payment is recalculated in each period in the aforementioned way.

The resulting discounted final budget for this management scheme is as follows:

$$B_{T,comp} \ = B_0 + \sum_{t=0}^{T} \frac{B_0 * r - \overline{cp}_{comp}}{(1+r)^t} \tag{5}$$

where $\overline{cp}_{comp} = \sum_{i=1}^{N} compensation\ payment * x_{comp,i,t}$ with $x_{comp,i,t} = 1$ if the corresponding patch *i* is compensated in period *t,* and 0 if it does not qualify for compensation. As we assume that all periodical returns on the initial budget are spent entirely on compensation payments, the final budget $B_{T,comp}$ equals B_0.

2.4 Ecological benefit function

An ecological benefit function is required to assess the conservation performance of the two alternatives and to identify the budget-efficient solution (i.e. which of the two alternatives performs better with the same budget).

To obtain the ecological benefit function we have to consider that the borrow alternative and the buy alternative generate two different landscapes with respect to habitat dynamics. The buy alternative generates a static landscape in which certain patches are bought in the initial period and kept as habitat for the entire time frame *T*, resulting in a constant habitat patch location and number $N_{buy} = a * N$ with the fraction *a* depending on the budget. Habitat patch turnover is zero.

In contrast, the borrow alternative generates a dynamic landscape with a habitat turnover described by a patch destruction rate μ and a patch creation rate λ. As turnover in each period is driven by the randomly drawn outcome of the participation willingness of landowners, we calculate average rates $\mu = \sum\mu_t/50$ and $\lambda = \sum\lambda_t/50$, over the 50 periods simulated.

To compare the ecological benefits of the two alternatives we take an analytical formula developed by Drechsler and Johst (2010) which is designed to calculate the mean metapopulation lifetime T_{meta} of a certain species in static (without habitat turnover) or dynamic (with habitat turnover) landscapes in a straightforward way. The formula for T_{meta} reads as follows:

$$T_{meta} \approx \frac{1}{\tilde{e}} \sum_{i=1}^{N_{dyn}} \sum_{k=1}^{N_{dyn}} \frac{1}{k} \frac{(N_{dyn}-i)!}{(N_{dyn}-k)!} \frac{1}{(N_{dyn}-1)^{k-i}} q^{k-i} . \quad (6)$$

Quantity $\tilde{e}$ in eq. (6) is the geometric mean over the local extinction rates of N_{dyn} patches in the landscape:

$$\tilde{e} = \prod_{i=1}^{N_{dyn}} (e_i + \mu)^{1/N_{dyn}} = \varepsilon + \mu. \tag{7}$$

Eq. (7) comprises population extinction due to habitat destruction with rate μ and without habitat destruction by rate e_i. Rate e_i is related to patch area A_i by $e_i = \varepsilon A_i^{-\eta} = \varepsilon$ with the species specific parameter ε describing the extinction risk of a species assuming $A_i = 1$.

The quantity q in eq. (6) is an aggregated colonisation-extinction ratio defined as

$$q \approx \frac{\bar{c}}{\tilde{e}} H = \frac{m}{\varepsilon + \mu} \tag{8}$$

The patch connectivity measure *H* in eq. (8) is set to *H* = 1 implying (since we ignore spatial effects in our model) that a species can reach all patches in the landscapes equally well. The colonisation rate $\bar{c}$ in eq. (8) is the power mean of the local colonisation rates c_i over N_{dyn} patches, with b/η a scaling exponent:

$$\bar{c} = \left(\frac{1}{N_{dyn}} \sum_{i=1}^{N_{dyn}} c_i^{\eta/b}\right)^{b/\eta} = m \tag{9}$$

The colonisation rates c_i are related to patch area by $c_i = mA_i^b = m$ with the species-specific parameter *m* describing the colonisation rate of a species assuming patch size $A_i = 1$. Due to the simplifications, the parameters b and η play no further role in our model.

Depending on the alternative being evaluated, we have to take into account the extent of habitat dynamics. For dynamic landscapes, i.e. the borrow alternative, the number of habitable patches is calculated as

$$N_{dyn} = N_{borrow} = round\left(\frac{\lambda N}{\mu + \lambda}\right) \tag{10}$$

while in a static landscape, i.e. the buy alternative, the N_{dyn} is equal to the total number of bought patches

$$N_{dyn} = N_{buy} = aN \tag{11}$$

with a defining a fraction of overall available patches N being bought, dependent on the parameters B_0, r, and σ. For an exact evaluation of N_{buy} see section 2.3.1.

3 Analysis

To analyse the outcome of the model we define a base case scenario with a specific combination of the parameters. We then modify each parameter individually.

The parameters B_0, r, $\bar{\pi}$, and σ influence the economic conditions which affect both the buy and borrow alternatives. The parameters f and P_{part} influence the level of landscape dynamics in the borrow alternative but have no impact on the buy alternative. Finally, the species-specific parameters ε and m affect the ecological benefit resulting from the management alternatives. Variations in these parameters represent species with different ecological characteristics described by species-specific colonisation and extinction rates of a patch. Table 1 shows the base case parameterisation and the possible parameter variations.

[Table 1 somewhere here]

In our analysis we first investigate the base case scenario. For the given base case parameters of ε, γ, $\bar{\pi}$, σ, f and P_{part} we simulate a matrix of combinations with an initial budget B_0 ranging from values of 50 to 150 (in steps of 5 units) and interest rates r ranging from 0.01 to 0.06 (in steps of 0.0025). For each combination we simulate the economic model 100 times to capture the stochastic variability of the economic outcome and calculate average values for the patch creation rate and patch destruction rate (see section 2.4). Afterwards we evaluate the outcome for each of the two management alternatives with the ecological mean metapopulation lifetime. To measure the relative efficiency performance difference we use the difference of the logarithmic mean metapopulation lifetime from the borrow alternative and the buy alternative ($\log(T_{meta}^{borrow}) - \log(T_{meta}^{buy})$) as an indicator to analyse changes in the budget efficiency of the two alternatives as a function of parameter changes.

Afterwards, we individually vary the parameters f, P_{part}, σ, ε and γ on this result matrix to analyse the possible effects of changes in theses parameters on their relative efficiency performance difference. This is done by setting each of the parameters separately on a low and high level (see Table 1).

The exact size of the initial budget (B_0), the average profit of each patch ($\bar{\pi}$), the profit variability in the landscape (σ) and the interest rate (*r*) are related. They together determine the proportion of the landscape which is included in buy or borrow activities and thus the number of participating patches. For example, for small budget and interest rate values (e.g. $B_0 = 50, r = 0.02$), this proportion is rather small, i.e. 2% of the total available patches participating in the borrow alternative. For large values ($B_0 = 150, r = 0.06$), this proportion is high, i.e. 14% of available patches participating in the borrow alternative (see Fig. 1). Corresponding effects can be observed for the buy alternative, though the effect of increasing B_0 and r is smaller and the amount of bought patches is always smaller than the amount of compensated patches for the same B_0 and *r* values.

[Figure 1 somewhere here]

In the base case, the parameter values for *r* range from 0.01 to 0.06. The level of $r = 0.03$ can be considered an average value which is roughly equal to the long-term interest rate of a government bond (note that we ignore inflation and *r* represents the real interest rate).

The base case values and ranges for ε and $m = \gamma * \varepsilon$ are chosen as follows (see also Johst et al. (2011)). The base case value of $\varepsilon = 0.1$ represents a species with local population extinction risk of 10% per period, corresponding to a mean population lifetime of 10 periods which can be seen as an average value. Lower (higher) values of ε suggest species with lower (higher) population extinction risks. The base case value of $\gamma = 8$ represents a species with a colonisation rate which is eightfold higher than its extinction rate. This means that the species has a good dispersal propensity and can easily colonise new patches. Lower (higher) values of γ indicate species which can less (more) easily colonise new patches.

The values for f and P_{part} are percentage values. f represents the forgone proportion of profit which is lost due to implementation of conservation measures and is set to an intermediate level of 0.7 in the base case. It is used in our model to constitute the cost difference between the buy and borrow alternatives. As we assume buying to result in zero profits for the agency, *f* is set to 1 in this case, while it is set to a value smaller than 1 in case of borrowing, thus allowing positive profits for the landowners. P_{part} represents the participation willingness of landowners in the borrow alternative and we consider a value of 0.95 as intermediate.

The landscape-specific parameters, i.e. in case of a dynamic landscape the number of habitable patches N_{dyn}, or in case of a static landscape the total number of available patches N_{buy}, as well as the patch destruction and creation rates μ and λ are determined by the economic model. They are subject to management decisions and result from the different management option choices of the agency and the corresponding subsequent simulation. The species parameters m and ε are varied within certain ranges reflecting different colonisation-extinction ratios $\gamma = m / \varepsilon \in [4, 12]$.

4 Results

4.1 Effects of variations in B_0 and r

For all scenarios we find that the budget efficiency of the two management options depends on the initial budget B_0 and the interest rate *r*. This dependency is shown in Fig. 2 for the base case scenario. Consider the initial budget first.

[Figure 2 somewhere here]

In Fig. 2 we can observe that with an increasing initial budget the efficiency of the buy alternative increases in comparison to the borrow alternative. In order to understand the reason for this consider for both alternatives what happens in the case of an increasing budget. In the buy alternative more costly patches can be purchased additionally, but the price of the low-cost patches which could have been already bought with a lower budget remains the same. In the borrow alternative an increasing budget enables the agency to increase the compensation payment in order to increase the amount of extensively managed patches. While on the one side more farmers participate, on the other side the already extensively managed low-cost patches also receive the higher compensation payment as payments are homogeneous. Hence, the management of these patches generates an additional producer surplus which increases with increasing compensation payments, and therefore also with an increasing initial budget.

Similarly, though somewhat counter-intuitive, with increasing interest rates the efficiency of the buy alternative increases compared to the borrow alternative. In the borrow alternative a higher interest rate leads to a higher periodical income for the agency which itself leads to more patches being extensively managed. In the buy alternative, an increase in the interest rate implies that prices for patch purchase decrease according to eq. (1) due to the increasing discounting effect on future profits from extensive land management. Therefore, more patches can be bought. But while the effect on the increasing income in the borrow alternative is linear as it only affects the income for the respective next period (though of course for all periods subsequently), the decreasing effect on land prices in the buy alternative is exponential (cf. eq. (1)) as it becomes increasingly relevant for periods farther in the future. Consequently, in the buy alternative more patches can be extensively managed with increasing interest rates due to the exponential influence.

4.2 Effects of variations in γ and ε

For the interpretation of the results recall that the borrow alternative generates a dynamic landscape, i.e. a landscape with habitat turnover (including habitat destruction and creation). The patch destruction rate μ increases $\tilde{e}$ in eq. (7) and decreases q in eq. (8), both decreasing metapopulation lifetime (see eq. (6)). Therefore, the borrow alternative can only perform better than the buy alternative if habitat turnover is compensated by a sufficiently larger habitat patch number $N_{borrow} > N_{buy}$. We found that N_{borrow} is always larger than N_{buy} in the investigated scenarios of B_0 and *r* (Fig. 1). At low B_0 and *r* the difference is sufficient to overcompensate the habitat turnover of the borrow alternative. Therefore, metapopulation lifetime is larger in the borrow alternative (green area in the left lower corner of Fig. 2). At high B_0 and *r* the difference is still positive but no longer sufficient to compensate the habitat turnover. Therefore, metapopulation lifetime is larger in the buy alternative (darker area in the right upper corner of Fig. 2).

A change in the species parameter γ (Fig. 3) describing the colonisation potential of a species does not reverse the efficiency performance pattern of the base case scenario of Fig. 2. Nevertheless, we can observe for species with low levels of γ (weaker dispersers; $\gamma = 4$ in Fig. 2) an alleviated effect of an increase in B_0 and *r* on the efficiency increase of the buy alternative in comparison to the borrow alternative.

[Figure 3 somewhere here]

A change in the species parameter ε (Fig. 4), however, has a much larger effect. Increasing ε considerably decreases the performance of the buy alternative and may even reverse the performance difference. An increase in ε increases the total local extinction risk $\tilde{e}$, thereby decreasing the relative contribution of the habitat destruction rate μ in both eq. (7) and eq. (8). This in turn decreases the relative impact of habitat destruction and thus patch turnover on the metapopulation lifetime (see eq. (6)), and strengthens the advantage of higher habitat patch numbers. As a consequence, the higher N_{borrow} (i.e. $N_{borrow} > N_{buy}$) plays the major role for the metapopulation lifetime resulting in a generally better performance of the borrow alternative.

[Figure 4 somewhere here]

4.3 Effects of f, P_{part} and σ

We find that with an increasing proportion of profit *f* which farmers lose by using the land extensively the efficiency of the buy alternative increases compared to the borrow alternative (Fig. 5). This is not surprising as with high values of *f* the opportunity costs for managing a patch extensively increase and compensations to landowners need to be higher (whereas in the buy alternative a change in *f* has no effect).

[Figure 5 somewhere here]

Conversely, increasing values of P_{part}, and thus decreasing probabilities of farmers not to participate in compensation schemes, imply that the efficiency of the borrow alternative increases in comparison to the buy alternative (Fig. 6). There are two reasons for this result. The first is that decreasing values of P_{part} imply an increased patch turnover in the borrow alternative which is not beneficial to biodiversity. The second is increasing compensation costs. Higher compensation costs arise because a low willingness to participate means that a relative high number of low-cost patches are not managed extensively, requiring higher compensation payments by the agency to achieve a certain number of patches. For a given budget this means that a lower amount of patches can be managed extensively resulting in a lower ecological outcome.

[Figure 6 somewhere here]

An increasing variability σ of the profit levels π_i leads to an increasing efficiency performance of the buy alternative in comparison to the borrow alternative (Fig. 7). This is because for low values of σ the opportunity costs of patches qualifying for participation in the borrow alternative are relatively close implying a relatively small amount of producer surplus in the borrow alternative. This changes with increasing values of σ, which leads to higher amounts of producer surplus. A growing amount of producer surplus means that for a given budget less money is available for compensating opportunity costs, i.e. fewer patches participate in the borrow alternative. A second consequence of increasing σ is that the purchasing price according to eq. (1) and the compensation payment (see section 2.3) of the least expensive patches decreases. The result is that for a given budget with increasing σ more patches can be bought which in turn increases the efficiency performance of the buy alternative. At the same time, more patches can be compensated which increases the efficiency performance of the borrow alternative. However, in the borrow alternative, as the compensation payment is determined by the most costly compensated patch whose costs only change slightly due to changes in σ, the expenditure changes are smaller than in the buy alternative, where the buying prices of all patches change and contribute to potential savings.

[Figure 7 somewhere here]

5 Summary and discussion

The conservation of endangered species and habitats frequently requires a certain type of human land use which is, however, costly for landowners. Given that landowners cannot or should not be forced to carry out conservation measures, conservation organisations have, in principle, two options. The first alternative is to buy land and carry out the land-use measures themselves or pay firms to carry them out (buy alternative). The acquired land can be used for conservation purposes as long as the organisation desires, however, a high initial payment is needed to purchase the land. The second alternative is to offer landowners payments to compensate them for the opportunity costs of implementing biodiversity-enhancing land-use measures (borrow alternative). For this alternative no high initial payment is needed. The budget can instead be invested, and the financial return is used to finance the compensation payments to landowners. One disadvantage from an ecological point of view is that landowners may not participate continuously in the compensation scheme. This leads to habitat turnover, i.e. patch destruction and creation, which is disadvantageous for some species. To our knowledge the decision problem of whether to buy or borrow land for biodiversity conservation has not been analysed so far. To contribute to filling this research gap we have developed an ecological-economic model to assess how changes in ecological and economic conditions influence the relative performance of the two alternatives in terms of budget efficiency.

We find that an increase in the initial budget B_0 as well as in the interest rate *r* – both of which determine the proportion of the landscape which is managed with biodiversity-enhancing land-use measures – favour the performance of the buy alternative. Regarding the interest rate this result is somewhat surprising as with an increasing interest rate more financial returns are available from the initially invested budget in the borrow alternative. This effect, however, is overcompensated by the dampening effect of high interest rates on land prices.

Regarding the economic parameters, we find that the efficiency performance of the buy alternative in comparison to the borrow alternative rises with increasing cost variations in the landscape, more fluctuation of landowners in the compensation scheme, and higher profit losses if biodiversity-enhancing land-use measures are applied. For the ecological parameters, our results show that the efficiency performance of the buy alternative compared to the borrow alternative increases with increasing colonisation-extinction ratio and decreases with increasing local extinction ratios.

We made some restrictive assumptions which require a brief discussion. First, we assumed that the conservation agency does not make any profit from managing the land if it buys the land and uses the area for conservation. We made this assumption as it generates identical initial and final budgets (in period 0 and *T* respectively) for both alternatives. This allows us to make the efficiency analysis in a simple manner, i.e. to compare the ecological outcomes of the alternatives for identical budgets. The analysis would have been much more complicated if we had assumed that the conservation agency – similar to the private landowner – also makes a positive profit (albeit less than the private landowner) with a patch used for conservation. Obviously, this admittedly more realistic assumption would increase the performance of the buy alternative compared to the borrow alternative but we are confident that it would not change our results qualitatively.

Second, we assumed that landowners decide about participation in each time period with a certain probability. Thus, they can switch between participation and non-participation multiple times. We made this assumption as the implementation of the mean metapopulation lifetime calculation as suggested by Drechsler and Johst (2010) requires the habitat dynamics to happen randomly across the landscape without any spatial or temporal correlations. By allowing patches to change their willingness to participate only once or a limited amount of times (which would be more plausible), this random aspect would vanish and the ecological benefit could not be evaluated with the applied formula. Again, we are confident that this simplifying assumption does not change our results qualitatively.

Third, we assume perfect knowledge about the future development of key economic parameters such as profits made from land management and interest rates, and therewith also land prices (eq. (1)). In reality, however, knowledge about the future development of these parameters is imperfect and fluctuations of these parameters may substantially influence the performance comparison of the two options. Consider as an example an increase in profit from land management and hence an increase in opportunity costs after the agency has opted for the borrow alternative. Then it gets fewer patches than foreseen to be conserved with the interest. At this stage it is however too late to buy land because land prices have also increased (cf. eq. (1)). Further research will be needed to analyse how the consideration of such imperfect knowledge might influence the efficiency performance of the two alternatives under consideration.

Obviously, our model is of a conceptual nature and its main benefit lies in an improved understanding about the ways in which changes in economic and ecological parameters influence the efficiency performance of the two alternatives. However, the model also provides a framework for case studies in which the budget efficiency of an existing (or planned) borrow or buy option is compared with the respective alternative. For such case studies economic data on interest rates r, land prices and profit reductions f, behavioural data on the participation probability of landowners in payment schemes (P_{part}), and biological data on colonisation and extinction rates (m and ε) of the species of conservation concern would need to be collected and fed into the model. Such case studies might provide valuable recommendations in terms of policy improvements, similar to other case studies where ecological and economic data have been fed into ecological-economic models (cf. Drechsler et al. 2010; Bamière et al. 2011; Armsworth et al. 2012).

Our work was motivated by typical conservation problems encountered in human-dominated landscapes which require a certain active type of land use and where a habitat can be restored quite easily, for example, in the case of the conservation of many endangered species in grasslands. The model was designed to capture the main features of conservation measures in such situations. However, the decision problem of whether to buy land for conservation or borrow it also exists in other circumstances which may be different from those captured by our model. For example, international NGOs often have to decide whether to buy forests in developing countries to conserve endangered biodiversity or pay landowners not to clear the forest for timber production. Whereas our model captures some features of this decision problem there are also differences. For example, we assume that patches can be restored in a short time for conservation through extensive management whereas the restoration of a virgin forest is not feasible within a short time frame. Nevertheless, we think that our model provides a useful starting point for analyses and believe that further research in this field could be fruitful.

6 References

Armsworth PR, Acs S, Dallimer M, et al. (2012) The cost of policy simplification in conservation incentive programs. Ecol Lett 15:406–414

Bamière L, Havlík P, Jacquet F, et al. (2011) Farming system modelling for agri-environmental policy design: The case of a spatially non-aggregated allocation of conservation measures. Ecol Econ 70:891–899

Burt OR (1986) Econometric Modeling of the Capitalization Formula for Farmland Prices. Am J Agric Econ 68:10

Cong R-G, Smith HG, Olsson O, Brady M (2014) Managing ecosystem services for agriculture: Will landscape-scale management pay? Ecol Econ 99:53–62

Drechsler M, Johst K (2010) Rapid viability analysis for metapopulations in dynamic habitat networks. Proc Biol Sci 277:1889–97

Drechsler M, Johst K, Ohl C, Wätzold F (2007) Designing cost-effective payments for conservation measures to generate spatiotemporal habitat heterogeneity. Conserv Biol 21:1475–86

Drechsler M, Wätzold F, Johst K, Shogren JF (2010) An agglomeration payment for cost-effective biodiversity conservation in spatially structured landscapes. Resour Energy Econ 32:261–275

Engel S, Pagiola S, Wunder S (2008) Designing payments for environmental services in theory and practice: An overview of the issues. Ecol Econ 65:663–674

Falconer K (2000) Farm-level constraints on agri-environmental scheme participation: a transactional perspective. J Rural Stud 16:379–394

Frondel M, Lehmann P, Wätzold F (2012) The impact of information on landowners' participation in voluntary conservation programs – Theoretical considerations and empirical evidence from an agri-environment program in Saxony, Germany. Land use policy 29:388–394

Hart R, Brady M, Olsson O (2014) Joint Production of Food and Wildlife: Uniform Measures or Nature Oases? Environ Resour Econ 59:187–205

Henle K, Alard D, Clitherow J, et al. (2008) Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe–A review. Agric Ecosyst Environ 124:60–71

Van Herzele A, Dendoncker N, Acosta-Michlik L (2011) Mobilisation capacity for agri-environmental management. J Environ Manage 92:1023–32

Johst K, Drechsler M, van Teeffelen AJA, et al. (2011) Biodiversity conservation in dynamic landscapes: trade-offs between number, connectivity and turnover of habitat patches. J Appl Ecol 48:1227–1235

Johst K, Hartig F, Drechsler M (2012) Offsetting policies for biodiversity conservation: the need for compensating habitat relocation. In: Jordán F, Jørgensen SE (eds) Model. Ecol. hierarchy - from Mol. to Ecosph. Elsevier, pp 413–430

Metera E, Sakowski T, Słoniewski K, Romanowicz B (2010) Grazing as a tool to maintain biodiversity of grassland – a review. Anim Sci Pap Reports 28:315–334

Moss R, Hill N, Glen F, et al. (2011) RSPB Reserves 2011.

Mouysset L, Doyen L, Pereau J-C, Jiguet F (2014) Benefits and costs of biodiversity in agricultural public policies. Eur Rev Agric Econ 1–26

Stiftung Naturschutz Schleswig-Holstein (2012) Geschäftsbericht 2010/2011.

Van Teeffelen AJ a., Vos CC, Opdam P (2012) Species in a dynamic world: Consequences of habitat network dynamics on conservation planning. Biol Conserv 153:239–253

Wätzold F, Drechsler M (2014) Agglomeration payment, agglomeration bonus or homogeneous payment? Resour. Energy Econ (in press)

Wätzold F, Schwerdtner K (2005) Why be wasteful when preserving a valuable resource? A review article on the cost-effectiveness of European biodiversity conservation policy. Biol Conserv 123:327–338

Young J, Watt A, Nowicki P, et al. (2005) Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. Biodivers Conserv 14:1641–1661

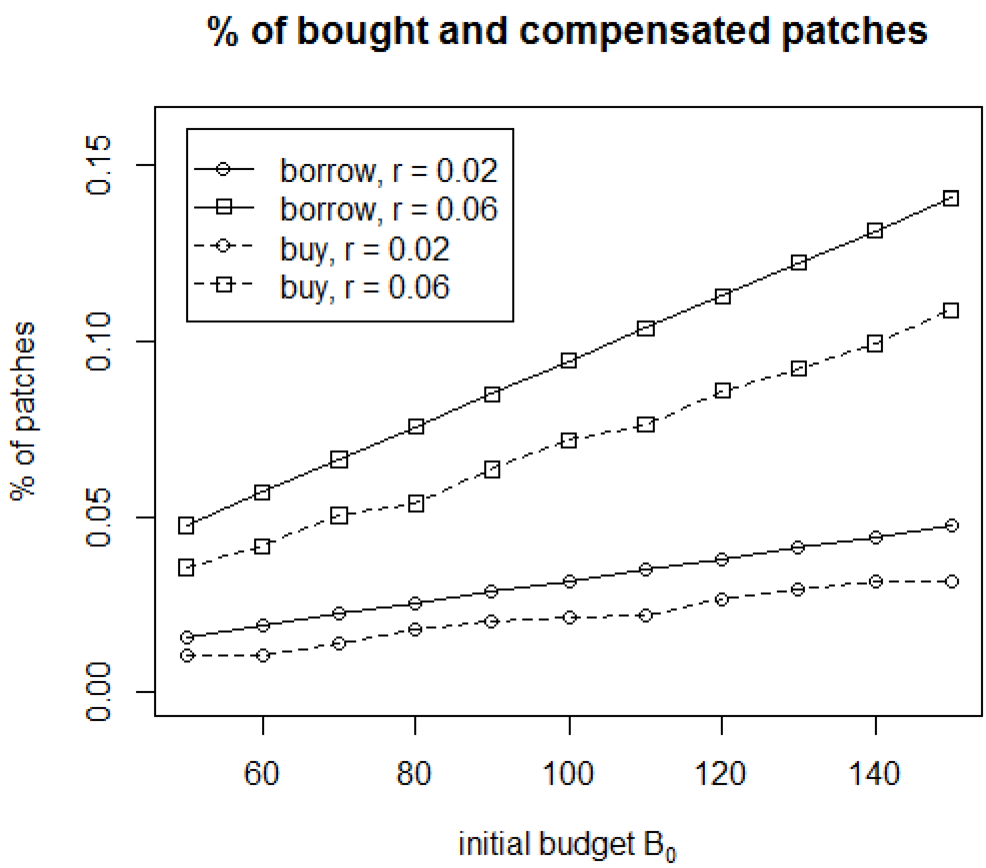

Figure 1: The percentage share of bought and compensated patches, depending on the initial budget B_0, shown for the two levels of interest rate $r = \{0.02, 0.06\}$, increases.

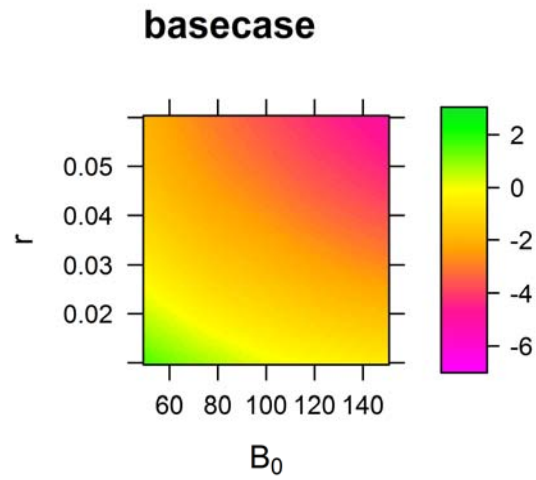

Figure 2: Relative efficiency performance of the ecological benefit (mean metapopulation lifetime) of the borrow and buy alternatives, expressed in logarithms ($\boldsymbol{log}(\boldsymbol{T}_{meta}^{borrow}) - \boldsymbol{log}(\boldsymbol{T}_{meta}^{buy})$) and plotted as a function of interest rate *r* and available initial budget $\boldsymbol{B_0}$. Increasing (decreasing) numbers and greener (more reddish) areas indicate a better (worse) relative efficiency performance of the borrow alternative compared to the buy alternative. All other parameters are set to their base case values (see Table 1).

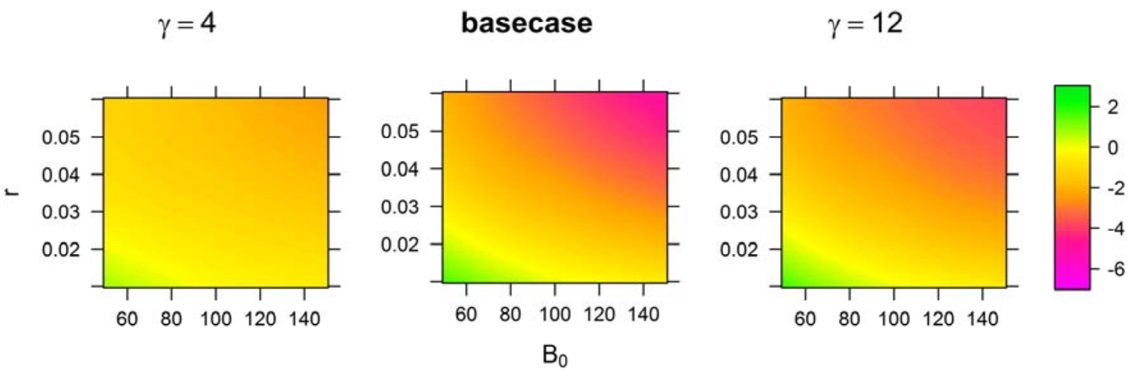

Figure 3: Impact of changes in species specific colonisation-extinction ratio (γ) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.

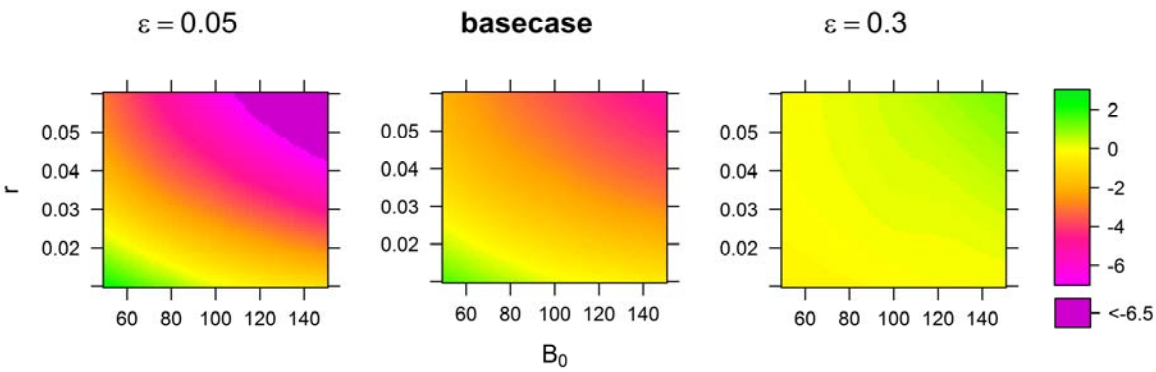

Figure 4: Impact of changes in the species-specific local extinction ratio ($\boldsymbol{\varepsilon}$) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2. The dark magenta area indicates the difference of the logarithmic ecological benefits to be smaller than -6.5, i.e. an even better relative efficiency performance of the buy alternative compared to the borrow alternative.

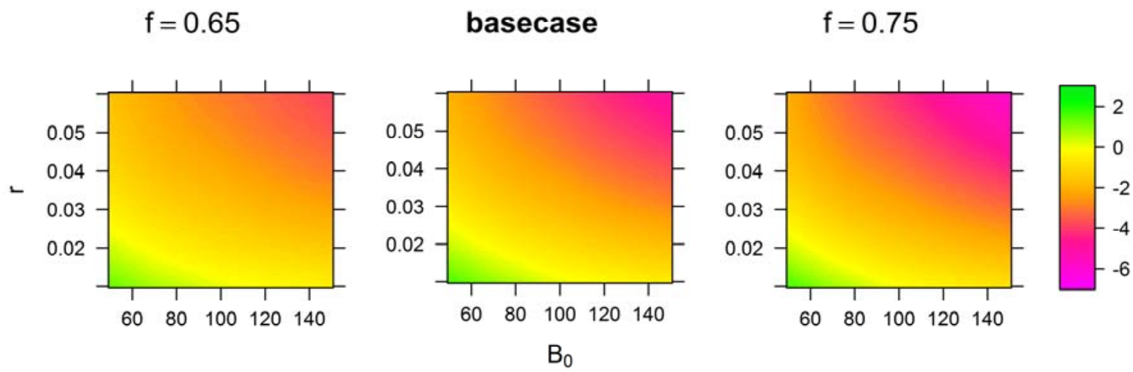

Figure 5: Impact of changes in the foregone proportion of profit (*f*) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.

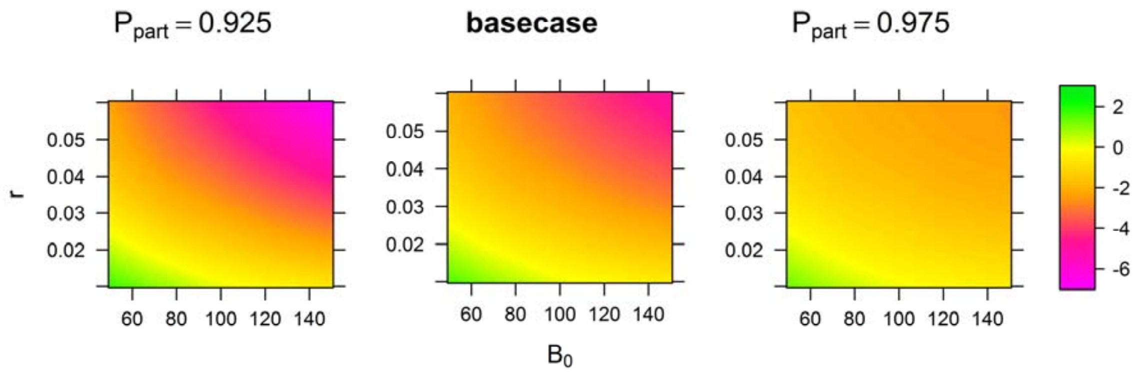

Figure 6: Impact of changes in the participation probability ($\boldsymbol{P_{part}}$) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.

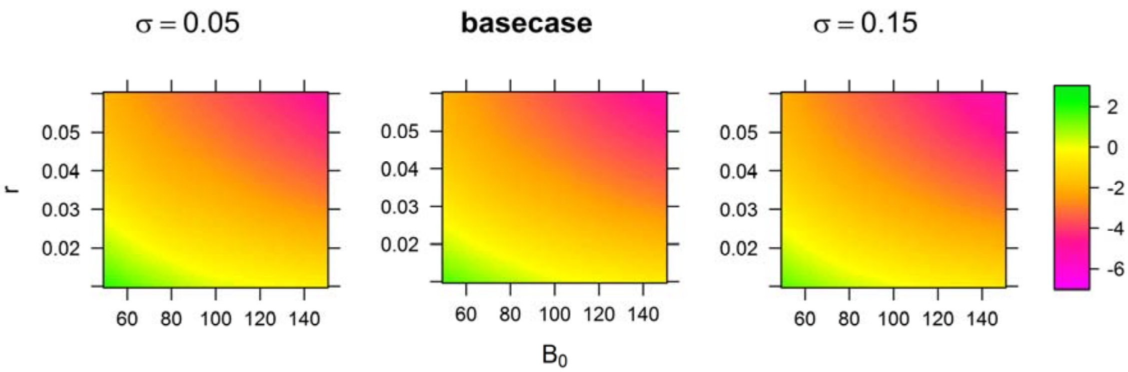

Figure 7: Impact of changes in the profit variation in the landscape (***σ***) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.

variable	motivation	low value	base case value	high value
parameters for base case result matrix				
B_0	initial budget	50	-	150
r	interest rate	0.01	-	0.06
$\bar{\pi}$	average profit	-	1.0	-
economic parameters				
σ	profit variation in the landscape	0.05	0.10	0.15
f	foregone proportion of profit	0.65	0.70	0.75
P_{part}	participation probability	0.925	0.95	0.975
ecological parameters				
ε	species extinction rate	0.05	0.10	0.30
m	species colonisation rate	0.4	0.8	2.4
		(implicitly determined by $m = \varepsilon * \gamma$ with the base case value of γ)		
$\gamma = m/\varepsilon$	species colonisation-extinction ratio	4	8	12

574 Table 1: Parameterisation of the ecological-economic model and parameter variation.