

**Make-or-Buy: Should nature conservation agencies buy land
and perform conservation activities on their own behalves, or
should they pay farmers for voluntarily performing
conservation activities?**

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Abstract

Anthropogenic threat for the world's environment and ecosystems, especially biodiversity, make it necessary to implement policies, which conserve species and habitat effectively. Financial constraints necessitate a wise use of resources in the implementation of conservation measures. A cost-effective planning and governance of biodiversity conservation measures has to be done in order to achieve maximal ecological outcomes with given financial resources. Established conservation policies however, especially in terms of conservation governance, might not be able to achieve a cost-effective conservation implementation and governance. Hence, a better understanding of available modes of governance can help improve the cost-effectiveness of biodiversity conservation planning and implementation.

This thesis investigates the influence of governance mode choice on the cost-effectiveness for biodiversity conservation measure implementation in selected conceptual and empirical settings, and identifies ecological, economic and climatic factors of influence. Linking the cost-effectiveness analysis for biodiversity conservation measures to the *make-or-buy decision*, traditionally used to analyse production and supply chain implementations in firms, enables a novel and so far largely unexplored perspective on cost-effective biodiversity conservation. In this context, conservation agencies are assumed to have two main options to implement biodiversity conservation measures in principle: (1) the *buy alternative* in which land is purchased for the purpose of biodiversity conservation measure implementation, and (2) the *compensation alternative* where a conservation agency compensates landowner's opportunity costs to incentivise voluntary conservation measure implementation by landowners.

Against this background, two conceptual ecological-economic models were developed to analyse the cost-effective governance mode choice and their sensitivity to changing ecological, economic and climatic parameters. The conceptual analysis is accompanied by two empirical studies, which address the question of cost-effective governance mode choice for a specific conservation project in Schleswig-Holstein (Germany) and a federal state wide comparison of conservation activities and implementations in California (USA).

It was found in the conceptual analysis that the *buy alternative* is more cost-effective than the *compensation alternative* if conservation budgets and interest are high. In addition, not only the choice of governance mode is relevant with respect to cost-effectiveness, but also

the mechanism to spatially select conservation sites chosen within a selected governance mode. Against this background, changing climate change speed for example – among other factors – influences the cost-effectiveness of governance modes and site selection strategies. A key insight from the empirical studies was that the cost-effective governance mode choice crucially depends on the intended timeframe of a conservation project. For shorter timeframes, compensating landowners is the cost-effective governance mode, however for longer durations the buy alternative becomes more cost-effective. This effect arises specifically due high one-off costs for buying land, which however amortize over time, compared to high recurring costs for compensating landowners. In addition, findings suggest that these relations do also exist on different spatial scales of activity and governance of the active conservation agency, such as city, county, state level. Based on these findings, the policy recommendations are provided which aim to improve the cost-effectiveness of governance modes available for biodiversity conservation.

Zusammenfassung

Wachsender anthropogener Druck auf Umwelt und Ökosysteme weltweit verlangt die Umsetzung von Politikmaßnahmen, welche bedrohte Arten und Habitate effektiv und kosten-effizient schützen. Es ist notwendig, dass limitierte finanzielle Ressourcen in diesem Zusammenhang sinnvoll und zielführend eingesetzt werden. Dafür muss eine kosten-effiziente Planung und Steuerung (*governance*) von Artenschutzmaßnahmen erfolgen, um einen maximalen ökologischen Nutzen bei gegebenen zur Verfügung stehenden Budgets zu erzielen. Bereits verfügbare Naturschutzmaßnahmen und Politiken können womöglich keine kosten-effiziente Umsetzung und Steuerung garantieren, weshalb ein grundsätzlich vertieftes Verständnis der zur Verfügung stehenden Steuerungsalternativen eine Verbesserung der kosten-effizienten Naturschutzplanung und –umsetzung fördern kann.

Die vorliegende Arbeit untersucht vor diesem Hintergrund konzeptionell und empirisch, welche Steuerungsalternativen für die Umsetzung von Naturschutzmaßnahmen zur Verfügung stehen, sowie deren Einfluss auf die Kosten-Effizienz. Zudem wird die Sensitivität der Kosten-Effizienz auf Veränderungen bestimmter ökologischer, ökonomischer und klimatischer Faktoren analysiert. Die Analyse der Kosten-Effizienz ist verbunden mit der Entscheidung über Eigenfertigung oder Fremdbezug (*make-or-buy decision*), welche traditionell Produktions- und Zulieferprozesse im Umfeld von Unternehmen betrifft. Die Anwendung der *make-or-buy* Entscheidung auf den Kontext des Natur- und Artenschutzes erlaubt neue und bisher zum großen Teil unerforschte Erkenntnisse in Bezug auf Kosten-Effizienz und Steuerungsalternativen. In diesem Sinne wird in der Arbeit angenommen, dass Naturschutzorganisationen prinzipiell zwei unterschiedliche Steuerungsalternativen zur Verfügung stehen: (1) der Kauf von Flächen zum Zweck des Artenschutzes, durchgeführt durch die Naturschutzorganisation selbst (*buy alternative*), und (2) die Kosten-Kompensation von Landnutzern für von ihnen freiwillig durchgeführte Artenschutzmaßnahmen (*compensation alternative*).

Vor diesem Hintergrund wurden zwei konzeptionelle, ökologisch-ökonomische Modelle entwickelt, um die Kosten-Effizienz der verschiedenen Steuerungsalternativen sowie deren Anfälligkeit gegenüber Veränderungen von ökologischen, ökonomischen und klimatischen Modellparametern zu analysieren. Die konzeptionelle Analyse wird ergänzt durch zwei empirische Fallstudien, welche die Frage nach einer kosten-effizienten

Steuerungsalternative für ein Artenschutzprojekt in Schleswig-Holstein (Deutschland), sowie für einen bundestaatsweiten Vergleich von Naturschutzmaßnahmen in Kalifornien (USA) analysiert.

Eine zentrale Erkenntnis der konzeptionellen Analyse ist, dass die Kaufalternative eine höhere Kosten-Effizienz erzielt als die Kompensationsalternative, je höher die zugrundeliegenden Zinssätze und finanziellen Ressourcen der betreffenden Naturschutzorganisation sind. Zudem ist neben der Wahl der Steuerungsalternative auch der Mechanismus zur Auswahl geeigneter Naturschutzlokalitäten ausschlaggebend für die Kosten-Effizienz. Die Geschwindigkeit, mit welcher sich die klimatischen Bedingungen ändern, beeinflussen ebenfalls – neben anderen Faktoren – die Kosten-Effizienz der beiden Steuerungsalternativen innerhalb der konzeptionellen Analyse. Auf empirischer Ebene zeigt sich, dass die Kosten-Effizienz der Steuerungsalternativen stark vom Zeitrahmen der geplanten Artenschutzmaßnahme abhängt. Bei kurzen Projekten erzielt die Kompensation von Landnutzern eine höhere Kosten-Effizienz, während bei lang andauernden Projekten der Erwerb von Flächen die höhere Kosten-Effizienz erzielt. Dieser Effekt ergibt sich insbesondere durch die hohen Einmalkosten und Erwerbsnebenkosten des Landkaufs, welche sich hingegen über längere Zeiträume amortisieren. Demgegenüber stehen dauerhaft wiederkehrende Kosten in der Kompensationsalternative, welche über kurze Zeiträume in der Summe moderat, über lange Zeiträume hingegen hoch ausfallen. Die Ergebnisse der empirischen Studien legen nahe, dass diese Effekte sowohl auf unterschiedlicher räumlicher Skala (einzelnes Artenschutzprojekt vs. regionale Artenschutzstrategie), als auch bei Naturschutzorganisationen mit unterschiedlichen Organisationsformen vorliegen.

Basierend auf den Erkenntnissen der Arbeit konnten Politikempfehlungen entwickelt werden, welche die Verbesserung der Kosten-Effizienz im Zuge der Wahl von Steuerungsalternativen für Natur- und Artenschutz zum Ziel haben.

Table of contents

Abstract	v
Zusammenfassung	vii
Table of contents	ix
List of figures	xiv
List of tables	xxii
List of abbreviations	xxiv
1 Introduction	2
1.1 Background and motivation	3
1.2 Methods and research objectives	4
1.3 Literature overview	6
1.3.1 Transaction cost economics	6
1.3.2 Transaction cost economics in environmental economics and conservation ..	7
1.3.3 Environmental governance	8
1.3.4 Cost-effectiveness	10
1.4 Scope of the thesis	11
2 Land for biodiversity conservation — To buy or borrow?	16
Abstract	16
2.1 Introduction	16
2.2 The model	19
2.2.1 Landscape structure, conservation costs and landscape dynamics	19
2.2.2 The decision problem of private landowners	21
2.2.3 The decision problem faced by the nature conservation agency	22
2.2.4 Ecological benefit function	25
2.3 Analysis	27
2.4 Results	30
2.4.1 Effects of variations in B_0 and r	30

2.4.2	Effects of variations in γ and ε	31
2.4.3	Effects of f , P_{part} and σ	33
2.5	Summary and discussion	35
	Acknowledgements	38
3	Climate change and the cost-effective governance mode for biodiversity conservation	40
	Abstract.....	40
3.1	Introduction.....	40
3.2	The model	42
3.2.1	Landscape and conservation costs.....	42
3.2.2	Climate change	45
3.2.3	Ecological dynamics.....	45
3.2.4	Decision problem of the conservation agency.....	46
3.3	Analysis	52
3.4	Results.....	54
3.4.1	Climate change speed	55
3.4.2	Strength of climate prioritization.....	57
3.4.3	Climate suitability threshold.....	59
3.4.4	Shape of the climate suitability bell curve	61
3.5	Summary and discussion	61
	Acknowledgements	64
4	Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area.....	66
	Abstract.....	66
4.1	Introduction.....	66
4.2	Materials and methods	68
4.2.1	Case study description	68
4.2.2	Data collection.....	71

4.2.3	Costs of the buy option.....	72
4.2.4	Costs of the compensation option	74
4.2.5	Baseline and sensitivity analysis	76
4.3	Results	78
4.3.1	Results of the baseline.....	78
4.3.2	Sensitivity analysis.....	79
4.4	Summary and discussion	81
	Acknowledgements	86
5	Easement or public land? An economic analysis of different ownership modes for nature conservation measures in California	88
	Abstract	88
5.1	Introduction	88
5.2	Methods	90
5.2.1	Study system	90
5.2.2	Costing functions.....	92
5.2.3	Data description.....	94
5.3	Results	94
5.3.1	Development of total annual costs	94
5.3.2	Development of costs of conservation	95
5.3.3	Development of costs of management	97
5.3.4	Sensitivity analysis.....	98
5.4	Discussion.....	98
	Acknowledgements	100
6	Summary, discussion and conclusion.....	102
6.1	Summary of results.....	102
6.2	Research gaps, new research questions and outlook	105
6.3	Policy recommendations.....	107

6.4 Conclusion	108
Acknowledgements	110
References	112
A. Appendix to Chapter 2.....	xxvii
B. Appendix to Chapter 3.....	xxx
B.1 Distance calculation	xxx
B.2 Patch selection strategies.....	xxxi
B.3 Patch restriction effect.....	xxxiv
B.4 Influence of economic and ecological variables	xxxv
C. Appendix to Chapter 4.....	xli
C.1 Detailed overview about land purchases.....	xli
C.2 Agricultural land price, leasehold rent and crop price development	xli
C.3 Crop Mix.....	xliv
C.4 Hypothetical profit and costs	xlvi
C.5 Compensation payment.....	l
C.6 Sensitivity analysis: transaction costs.....	l
C.7 Sensitivity analysis: leasehold rent	lii
C.8 Sensitivity analysis: land value	liii
C.9 Sensitivity analysis: AES payment eligibility	liv
C.10 Sensitivity analysis: crop mix	lvi
C.11 Data overview from literature review	lviii
D. Appendix to Chapter 5.....	lix
D.1 Costing sub-functions for buying land	lix
D.2 Costing sub-functions for easements	lx
D.3 Parameter selection.....	lxi
D.4 Development of area and amount of conservation sites	lxiii
D.5 Sensitivity analysis	lxiii

D.6 Extended discussion	lxxiii
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List of figures

Figure 2.1: The percentage share of bought patches and patches receiving compensation payments, depending on the initial budget B_0 , shown for the two levels of interest rate $r = \{0.02, 0.06\}$, increases.....	29
Figure 2.2: Relative efficiency performance of the ecological benefit (mean metapopulation lifetime) of the borrow and buy alternatives, expressed in logarithms ($\log(T_{meta}^{borrow}) - \log(T_{meta}^{buy})$) and plotted as a function of interest rate r and available initial budget 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 2.2).....	30
Figure 2.3: 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. 2.2.....	32
Figure 2.4: Impact of changes in the species-specific local extinction ratio (ϵ) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2. 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.....	32
Figure 2.5: 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. 2.2.	33
Figure 2.6: Impact of changes in the participation probability (P_{part}) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2.	34
Figure 2.7: 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. 2.2.	34
Figure 3.1: (a) Spatially explicit landscape consisting of 10×20 patches including the climatically suitable zone (CSZ, shaded area) at time-steps $t = 0$ and $t = 100$, (b) climate suitability bell curves according to Eq. (1) in their respective base case parametrization (see Table 2) and climate suitability threshold $cs^{thr} = 0.5$, leading to the CSZ at the	

different time steps $t \in \{0,100\}$. The shaded area and the corresponding borders represents the CSZ at each given time-step.....	43
Figure 3.2: Visualization of the four different PSSs and the corresponding patch location.	
(a) ‘Price prioritization’ allows for patch selection in the complete CSZ, only depending on the purchase price or compensation costs. (b) ‘Species abundance prioritization’ only selects patches within the dispersal distance of already occupied patches. (c) ‘Climate change prioritization’, prefers patches with higher climate suitability over patches with lower climate suitability, and (d) ‘Climate change direction prioritization’, prefers patches at the northern end of the CSZ over patches at the southern end of the CSZ.	51
Figure 3.3: Changes in extinction rates due to changes in maximum length of the simulation timeframe T , (i.e. climate change speed decreases with increasing T). (a)-(d) represent the extinction probabilities for all four strategies in the buy and compensation alternative.	57
Figure 3.4: Influence of changes in κ on the extinction probability in the buy alternative and the compensation alternative for each of the four PSSs.	58
Figure 3.5: Extinction probabilities of the different GM-PSS pairs with changing climate threshold, resulting in changing CSZ sizes.	60
Figure 4.1: Lake Bültsee conservation area in Schleswig–Holstein, Germany, with its different terrain and grassland types. The map is based on ArcGIS map data Europe NUTS 1, management plan data for the FFH area “Großer Schnaaper See, Bültsee und anschließende Flächen”, and data from SNSH.	69
Figure 4.2: Cumulative discounted costs of the buy option (straight line) and the compensation option (dashed line) for the baseline value between 1980 and 2015. The values are discounted to the base year 1980.	78
Figure 4.3: Costs of the buy option (dashed line), and compensation option (straight line). The values are discounted to the base year 1980. (a) Bandwidth for compensation costs (grey shaded area) due to variations in transaction costs (high value is at the upper edge; low value is at the lower edge of the shaded area). (b) Bandwidth for buying costs (grey shaded area) due to variations in the leasehold rent (high value is at the lower edge; low value is barely visible as it is very close to the actual, already very low lease hold rents). (c) Bandwidth for buying costs (grey shaded area) due to variations in the land value factor θ (high value is at the lower edge; low value is at the upper edge of the shaded area).	79
Figure 4.4: Costs of the buy option (dashed line) and compensation option (straight line) with the corresponding bandwidths (grey shaded areas) due to a hypothetically non-	

existing energy crop boom and thus decreased land and produce prices. The values are discounted to the base year 1980..... 81

Figure 5.1: Conservation areas in the state of California: (a) extent of conservation areas (green) and easements (orange); (b) ownership of conservation parcels (legend colors represent governance levels responsible for management of land); (c) type of land management (yellow: national parks; purple: Bureau of Land Management; green: United States Forest Service; orange. state parks; grey: United States Department of Defense); (d) California Department of Parks and Recreation properties; (e) East Bay Regional Park District properties; and (f) City and County of San Diego. The California State Parks manages about 900 properties that were added to their portfolio since 1970, corresponding to 792 km² distributed throughout the state. The agency has an Office of Grants and Local Services that since 1964 has provided funding to 7,400 local parks to be created or improved. The County of San Diego currently manages about 200 properties, amounting to 207 km² of land managed since 1970. The county faces high rates of development and it is ambitious and determined to increase the area in Open Space, as stated in its strategic plan. The City of San Diego manages 430 properties with about 190 km² of land added since 1970. In California, cities decide upon land use regulations within their jurisdiction (Santos et al., 2014b), and San Diego manages most of the City Parks. The East Bay Regional Park District (EBRPD) is a Special District, that is, a limited purpose local government, separate from cities and counties. Special districts provide focused public services such as fire protection, water supply, parks, recreation, and so on. EBRPD manages 242 properties with an area of 222 km² added since 1970. Financing for Special Districts comes from property taxes, fees that users pay for services and special assessments. These types of districts may handle a revenue that varies between 10 and 30 billion USD, but only a small fraction of that budget goes into parks and recreation..... 91

Figure 5.2: Total annual costs ((a) and (b)), i.e. the total amount of costs which arise in each year of the analysis time frame, separated for each of the four conservation agencies, and total annual costs per hectare ((c) and (d)), i.e. the total amount of costs in each year of the conservation time frame, divided by the total area in hectare in that year, separated for each of the four conservation agencies..... 95

Figure 5.3: Total cumulative costs and total cumulative costs per hectare of implementing and managing easements or in-fee conservation areas..... 96

Figure 5.4: Development of cumulative management costs with easement management (a) and cumulative management costs with in-fee managed areas. Management costs are

calculated according to Eq. (D.2) (see Appendix D.5) separately for each area and cumulative over time for each agency. Development of per hectare management costs with easement management (c) and per hectare management costs with in-fee management (d). While all agencies do at least manage some easements, no data was available for the CDPR in the considered timeframe. 97

Figure 5.5: Annual management costs and annual management costs per hectare for each management option. 98

Figure A.1: Comparison of share of patches under conservation in the buy and borrow alternative determined by the analytical approach (Eqs. A.2 and A.4; thin lines) and the simulation approach (bold lines, identical with Fig. 2.1 in the main text) as a function of initial budget B_0 and interest rate r xxix

Figure B.1: Distance between two patches calculated by Eq. (B.1). The yellow-shaded and red-framed area represents the climatically suitable zone (CSZ). The blue- shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. Numbers indicate the distance of the respective patch to the highlighted blue patch. Note that the distance for all patches directly neighbouring the blue patch is 1..... xxx

Figure B.2: ‘Prize prioritization strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘prize prioritization strategy’. The blue-shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. xxxi

Figure B.3: ‘Species abundance targeting strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘species abundance targeting strategy’. The yellow-shaded areas represent patches, which are non- eligible for selection in this strategy. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species..... xxxii

Figure B.4: ‘Climate suitability prioritization strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘climate suitability prioritization strategy’. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch

selected for conservation; the star-symbol indicates that this patch is occupied by the target species..... xxxiii

Figure B.5: 'Climate change directional prioritization strategy'. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the 'climate change directional prioritization strategy'. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. xxxiv

Figure B.6: Patch restriction effect in the 'species abundance prioritization strategy' due to changes in CSZ sizes due to varying cs^{thr} . The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the 'species abundance prioritization strategy'. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. The red-shaded areas represent patches, which could have been selected by the respective strategy, if the CSZ was large enough, but in fact are restricted in eligibility by the patch restriction effect. xxxv

Figure B.7: Extinction probabilities of the different GM-PSS pairs with changing interest rates, resulting in changes in available budgets and discount rates. The red line represents results for the buy alternative, the green line for the compensation alternative. xxxvi

Figure B.8: Extinction probabilities of the different GM-PSS pairs with changing mean opportunity costs. The red line represents results for the buy alternative, the green line for the compensation alternative. xxxvii

Figure B.9: Extinction probabilities of the different GM-PSS pairs with changing emigration rates. The red line represents results for the buy alternative, the green line for the compensation alternative. xxxvii

Figure B.10: Extinction probabilities of the different GM-PSS pairs with changing dispersal distances. The red line represents results for the buy alternative, the green line for the compensation alternative. xxxviii

Figure B.11: Extinction probabilities of the different GM-PSS pairs with changing mean transaction costs. The red line represents results for the buy alternative, the green line for the compensation alternative. xxxviii

Figure B.12: Extinction probabilities of the different GM-PSS pairs with changing mean monitoring costs. The red line represents results for the buy alternative, the green line for the compensation alternative.	xxxix
Figure B.13: Extinction probabilities of the different GM-PSS pairs with changing climate suitability bell curvature parameter ρ . The red line represents results for the buy alternative, the green line for the compensation alternative.	xxxix
Figure B.14: Extinction probabilities of the different GM-PSS pairs with changing climate direction prioritization strength λ . The red line represents results for the buy alternative, the green line for the compensation alternative.	xl
Figure C.1: Agricultural land prices (per hectare) and leasehold rent (per hectare per year) development in Schleswig-Holstein from 1980 to 2015 (Statistikamt Nord 2015a).	xliii
Figure C.2: Development of crop prices in Schleswig-Holstein from 1980 to 2015 for the four most used field crops, measured in Euro per dezi-ton.	xliii
Figure C.3: Development of per ha yields of the four most used field crops in Schleswig-Holstein, measured in dezi-ton.	xliv
Figure C.4: Shares of cultivated field crops in Schleswig-Holstein between 1980 and 2015 (Statistikamt Nord 2015b). From 2006 to 2009, a transition phase is assumed in which the average field crop cultivation gradually changes from mixed agriculture to pure maize cultivation (mainly used as energy crop, personal communication). This results in pure maize cultivation from 2010 onwards.	xlv
Figure C.5: Potential profit of extensive grassland management.	xlvi
Figure C.6: Hypothetical compensation payment per ha and year necessary offset the assumed opportunity costs of a prescribed extensive measure.	l
Figure C.7: Compensation payment including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).	li
Figure C.8: Farmers' transaction costs including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).	li
Figure C.9: Farmers' transaction costs including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).	lii
Figure C.10: Leasehold fee for the case study project (dashed line) and actual leasehold average payments for agricultural areas in Schleswig-Holstein (dotted line). The difference between the baseline and the low value case is only visible after year 2008, as leasehold rents for the baseline before that date have been 0 €, the same as in the low case value, and only differ from 2009 onwards.	liii

Figure C.11: Land value of extensive grassland with bandwidth due to variations of θ (high value is at the upper edge; low value is at the lower edge of the shaded area).....	liv
Figure C.12: Costs for the buy option (dashed line), and compensation option (straight line) with the corresponding bandwidth for changing compensation expenses (grey shaded area) due to variations in potentially received payments for AES (high value is at the lower edge of the shaded area; moderate values are represented by the dotted line in the middle; the case study value is represented by the straight line at the upper edge of the shaded area).	lv
Figure C.13: Range of the assumed auxiliary subsidies from agri-environmental schemes utilised for the compensation option. The lower dotted line (identical with the x-axis) represents the baseline, as here zero subsidies are assumed. The dashed line stands for the moderate, and the upper, dotted line for the high value case.	lvi
Figure C.14: Alternative crop mix compared to phase out of wheat, rape seed and barley cultivation in favour of pure maize production after 2006, compare Figure C.4 (Statistikamt Nord 2015b).	lvii
Figure C.15: Changes in compensation costs due to alternative crop mix.	lvii
Figure D.1: Number of conservation properties in easement and in-fee for each of the 4 agencies, City of San Diego, County of San Diego, East Bay Regional Park District, and California Department of Parks and Recreation.....	lxiii
Figure D.2: The graphs show the development of total annual costs in either governance mode over time. The bold lines indicate cost developments for the base case parameter values. The shaded areas indicate the bandwidth for cost variations if higher or lower purchasing prices (and thus opportunity costs) are assumed. Higher purchasing prices generally cause an upwards shift of the cost curves, while lower prices cause a downwards shift.	lxiv
Figure D.3: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices.	lxiv
Figure D.4: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) opportunity costs.	lxv
Figure D.5: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) opportunity costs.	lxvi

Figure D.6: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices and opportunity costs.	lxvi
Figure D.7: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices and opportunity costs.	lxvii
Figure D.8: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).	lxviii
Figure D.9: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).	lxviii
Figure D.10: Development of annual management costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).	lxix
Figure D.11: Development of annual management costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).	lxix
Figure D.12: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) income generated on managed land.	lxx
Figure D.13: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) income generated on managed land.	lxxi
Figure D.14: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) transaction costs.	lxxii
Figure D.15: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) transaction costs.	lxxii

List of tables

Table 2.1: Description of economic and ecological model variables.	20
Table 2.2: Description of parameters and parameterisation of the ecological–economic model.	21
Table 3.1: Overview and description of model variables.	44
Table 3.2: Overview and description of parameters and parametrization values specified for computation of the Monte-Carlo-Simulation and the sensitivity analysis.	53
Table 4.1: Buying costs and side costs, i.e. notary fees, taxes, and measurement costs, of the implementation of Lake Bültsee conservation project (Euro values are given in nominal terms; deviations in the cumulative values due to rounding errors). In 1985, a marginal section of the conservation area was ceded for the implementation of an infrastructure project, resulting in a decrease in area, with the mentioned sale revenues and side costs. In 1990, parts of Lake Bültsee conservation area were sold, while in return a different conservation area was extended. This exchange resulted in the given positive buying and side costs.	70
Table 4.2: Annual management costs and rental income.	73
Table 4.3: Parameter values for different sensitivity analyses, with corresponding low scenario, baseline and high scenario values.	77
Table B.1: Overview about the parametrization value and value ranges specified for computation of the Monte-Carlo-Simulation and used in the sensitivity analysis for non-climatic factors.	xxxvi
Table C.1: Detailed overview about purchasing costs of individual land parcels around Lake Bültsee, containing administrative information, buying prices, area sizes and purchasing side expenses, such as property tax, notary fee, SHL-fee, land registration fee, and measurement costs.	xlii
Table C.2: Overview about the hypothetical revenue and costs of extensive grassland measures in the case study area.	xlvi
Table C.3: Hypothetical profit and costs for intensive agricultural land use for the four most used field crops in Schleswig-Holstein, measured in Euro per ha.	xlvi
Table C.4: Hypothetical costs for intensive agricultural land use for the four most used field crops in Schleswig-Holstein, measured in Euro per ha.	xlix
Table C.5: Overview about the different sources for the costing calculation.	lviii

Table D.1: Overview, description and sources of costing function parameters with corresponding low and high values chosen in the sensitivity analysis. lxi

Table D.2: Overview, description and sources of costing function parameters used in the parametrization of our costing calculations, and which are not further investigated in a sensitivity analysis. It is to be expected that the effect of changes in mentioned parameters is pretty straight forward (i.e. an increase (decrease) of management cost parameters increases (decreases) management costs and thus shifts the corresponding cost curve of easements upwards (downwards)). lxii

List of abbreviations

AES	Agri-environment scheme
BNatSchG	Bundesnaturschutzgesetz (i.e. Federal Nature Conservation Act)
BW	Bunde Wischen
CA	Conservation Agency
CCED	California Conservation Easements Datasets
CDPR	California Department of Parks and Recreation
CPAD	California Protected Areas Data Portal
CSZ	Climatically suitable zone
DGLG	Dauergrünlanderhaltungsgesetz (i.e. <i>Permanent Grassland Conservation Act</i>)
EBRPD	East Bay Regional Park District
FFH	Flora Fauna Habitat
GM	Governance mode
NGO	Non-governmental organization
NUTS 1	Nomenclature des unités territoriales statistiques (i.e. <i>Nomenclature of Territorial Units for Statistics</i>)
PES	Payments for ecosystem services
PSS	Patch selection strategy
SNSH	Stiftung Naturschutz Schleswig-Holstein (i.e. <i>Nature Conservation Foundation Schleswig-Holstein</i>)
UK	United Kingdom
USDA	United States Department for Agriculture
USD	United States Dollar

Chapter 1

Introduction

1 Introduction

The world's environment and ecosystems face an ever-growing threat from human alteration of environmental conditions, of which the two most pressing are anthropogenic climate change and biodiversity loss (IPBES, 2018; MEA, 2005). To tackle this issues, a wise mixture of mitigation, adaptation, conservation and restoration measures needs to be implemented, which by ecological and economic standards need to be as effective as possible (IPCC, 2014; Rands et al., 2010) to avoid further and potentially irreversible alteration of earth's systems (Biermann et al., 2010).

Habitat loss for endangered species due to anthropogenic causes such as land use and climate change are considered key drivers for the loss of biodiversity (Sala et al., 2000; Thomas et al., 2004; Urban, 2015). Biodiversity conservation and respective land use planning, e.g. by the help of agri-environment schemes (AES) or other policy instruments, comprises a common set of measures to improve environmental conditions on and the conservation of cultural land (cp. Ring et al., 2010).

Such measures can counteract some of the negative consequences of e.g. agricultural intensification (Kleijn et al., 2011; Poláková et al., 2011) by biodiversity enhancing land use practices and allow for agricultural extensification and reduction of direct ecological degradation through the conservation of cultural land (Fischer et al., 2008; Lin and Fuller, 2013; Phalan et al., 2011).

It is imperative to not only find appropriate conservation measures to effectively reach intended ecological goals, such as habitat preservation or species conservation (Poláková et al., 2011), but also to reach given goals at least costs (Wätzold and Schwerdtner, 2005). The combination of cost-effective implementation with novel or improved conservation measures, e.g. by newly designed AES or an improved understanding of governance structures and their implications for conservation outcomes, can help to maintain and restore ecosystems and environmental quality (Poschlod et al., 2005).

1.1 Background and motivation

A so far not well-researched field in the context of biodiversity conservation is the question for the cost-effective mode of governance for conservation measure implementation. The choice of a suitable and cost-effective governance structure depends on various economic and ecological characteristics. Governance structure refers to a managerial structure and ownership mode, a certain conservation area is organised in to reach given conservation targets at least costs. It is the purpose of this work to investigate these characteristics and analyse economic, ecological and climatic factors of influence on the optimal choice of governance modes for biodiversity conservation.

The question for optimal governance is linked to the more traditional *make-or-buy decision*, which originates from transaction-cost and new institutional economics and the assessment of a company's mode of governance (Coase, 1937; Williamson, 1975, 2002). Within this field of research, the internal and external organization of economic actors, e.g. firms, is assessed against the background of analysis criteria such as transaction frequency, uncertainty of transactions and outcomes, and the specificity of assets relevant for a transaction (Coase, 1937, 1960; Commons, 1934). A key research question asks for the cost savings potential of an economic actor by internalizing economic activities or transactions (i.e. to *make* products or services themselves) which otherwise would be organized externally (i.e. to buy products or services from third party suppliers; Coase, 1937; Williamson, 1971, 1981). This *make-or-buy decision* regards the integration of economic transactions and characterizes the choice between the integration of operations into its own governance structure, or the solicitation and procurement from external suppliers (Walker and Weber, 1984).

Linking the general question for cost-effective biodiversity conservation to the *make-or-buy decision* enables a novel and so far largely unexplored perspective on cost-effective biodiversity conservation. A translation of the organisational and production decision-themed *make-or-buy decision* into the domain of biodiversity and nature conservation is however necessary to fully capture the interrelation of governance and conservation outcomes. While literature generally transfers this decision into the governmental policy options of incentive-based instruments and command and control regulation (Richards, 2000), in this work a different approach is chosen.

Following Birner and Wittmer (2004, p. 668), the production decision in this context can be transferred to the “conservation of natural resources [...] such as wildlife, biodiversity or stream flow” by e.g. placing conservation areas. In this sense, the conservation of cultural landscapes can be assessed against the background of the transferred production decision of a conservation agency – i.e. the question, under which type of governance mode conservation activities are organized.

A commonly used practice to implement biodiversity conservation measures is to motivate landowners to voluntarily provide conservation activity by offering a compensation payment, which incentivises landowners to implement conservation measure on their areas (Uthes and Matzdorf, 2013; Wätzold et al., 2016). The purchase of areas for conservation purposes as a contrasting approach to landowner compensation is used by e.g. governmentally funded conservation actors, like e.g. conservation trusts in Germany (“Naturschutz Stiftungen”), non-governmental land trusts or governmentally controlled conservation bodies in the US.

While the ecological and cost-effectiveness of AES is well studied in many different facets, the cost-effective choice of governance modes for biodiversity conservation on cultural lands as well as influencing factors however remain largely under-investigated.

The research presented in this thesis asks whether it is cost-effective for a conservation agency to either (1) buy land to implement conservation activities, or (2) to offer compensation payments to land owners to incentivise voluntary implementation of conservation measures. It thus applies the analysis of the aforementioned *make-or-buy decision* on the field of optimal governance mode choice for biodiversity conservation. The aim of this thesis is to analyse the cost-effective governance modes for biodiversity conservation against the background of the *make-or-buy decision* in four different fields of research. Selected aspects of the *make-or-buy decision* were thus analysed from different perspectives.

1.2 Methods and research objectives

This thesis approaches the *make-or-buy decision* from a conceptual and an empirical perspective. Hence, different research methods had to be used to investigate the aspect of cost-effective governance mode choice for biodiversity conservation.

Chapter 2 analyses the *make-or-buy decision* conceptually to determine the cost-effective governance mode for biodiversity conservation. Ecological and economic factors influencing the cost-effectiveness of governance modes are analysed. Chapter 3 extends the conceptual understanding of the *make-or-buy decision* by analysing the specific influence of changing climatic conditions.

Conceptual ecological-economic models were used to understand the main ecological, economic and climatic drivers for the cost-effective choice of different governance modes for biodiversity conservation purposes. These models were used to calculate economic costs and ecological outcomes for specific parameter settings. For this purpose, a Monte-Carlo simulation approach was necessary due to the dynamic and probabilistic nature of the underlying integrated ecological-economic models which otherwise would not have allowed a straightforward analytical analysis. Within the Monte-Carlo-simulations of each model, the cost-effectiveness of governance modes were assessed on the averaged ecological and economic outcomes of the individual simulations within each specific parameter setting and then compared against each other. This allowed for an analysis of individual ecological, economic and climatic drivers of the cost-effective choice of governance modes.

The evaluation criterion of cost-effectiveness was identified as most useful to assess the combined ecological-economic quality of different governance modes used for biodiversity conservation. Within the conceptual and empirical analysis of this thesis, different governance modes and specific implementation strategies of biodiversity conservation were then analysed and compared against their cost-effectiveness.

Besides a conceptual understanding, an empirical basis of the cost-effective governance mode choice in biodiversity conservation was researched in Chapters 4 and 5. Within two separate case studies, empirical data on the specific costs of conservation through land purchase and landowner compensation was gathered. While Chapter 4 analyses the specific costs of land purchase versus hypothetical costs of a compensation payment based conservation scheme in a small-scale conservation project conducted in northern Germany, Chapter 5 investigates aggregated data on costs of land purchase versus landowner compensation on large-scale conservation activities in California, implemented by multiple organizational bodies.

To analyse the cost-effective choice of governance modes, a framework for cost-estimation was developed which allowed to compare the costs of land purchase with the costs of landowner compensation as they would arise to a conservation actor. An important part of the analysis was the possibility to compare the development of conservation costs in each governance mode over time which allowed to derive recommendations of optimal governance mode choices, depending on the intended time frame of a conservation project.

Chapter 4 presents the internal perspective of a conservation agency on one specific conservation project and the respective analysis of the cost-effective choice of governance modes, as internal data from the conservation actor on the costs of the specific conservation project was available. In contrast, Chapter 5 adopts an external perspective on different conservation agencies with different spatial extents and different internal governance structures, for which external data sources were used.

1.3 Literature overview

1.3.1 Transaction cost economics

The discussion of the governance of economic activities finds its origin in Ronald Coase's "The Nature of the Firm" (Coase, 1937), where the underlying question – i.e. whether a firm should buy e.g. intermediate product parts from other companies or decide to produce those parts itself – was posed initially. A firm's common interaction with the surrounding economy shapes the mode of governance in which it is optimally organized: be it in an internalized and aggregated form in which the firm is its own provider of goods and services; in a more market oriented and externalized form in which goods and services important for the firm's operation are procured externally from other economic entities; or as a hybrid mode in between (Joskow, 2008). Following this discussion, later developed transaction costs economics describes a firm not as a pure production function, as traditionally understood in classical economic thinking, which defines the relationship between its inputs and outputs, but rather as a governance structure organizing a firm's internal and external transactions (Coase, 1937; Commons, 1934; Williamson, 1975, 1985, 1989, 1998).

Within a firm, a transaction is the basic unit of economic analysis (Commons, 1934) and can be defined as the "elementary coordination problem" between economic actors

(Bougherara et al., 2009, p. 80). Each transaction can take place in a variation of different governance structures organizing the economic relationships and hence defining the limits of individual economic activity (Coase, 1937; Williamson, 1981).

The governance structure chosen to organize firm's activities, also referred to as the aforementioned *make-or-buy decision*, regards the level of so-called vertical integration a firm has to decide on (Walker and Weber, 1984; Williamson, 1981). Vertical integration describes an internal and external structure, supply chain and production processes of a firm. Different degrees of vertical integration thus refer to different governance structures of such processes.

Historically, the analysis of vertical integration originated in the field of industrial organization regarding the integration of production into the supply chain (Rindfleisch and Heide, 1997). Klein (2008, p. 437) considers the field of transaction costs economics (TCE) and the analysis of the *make-or-buy decision* in this context as “the study of alternative institutions of governance” regarding supply chain, production and procurement processes.

1.3.2 Transaction cost economics in environmental economics and conservation

Many disciplines, e.g. sociology, political science and contract law, have used TCE beyond its classical application in economics to understand internal organisation and decision making (Rindfleisch and Heide, 1997). Rindfleisch and Heide (1997) point out the necessity for empirical research in the field of transactions costs, as well as generally in the area of environmental governance.

Williamson (1985, p. 41) confidently states “any problem that can be posed directly or indirectly as a contracting problem is usefully investigated in transaction cost economizing terms”. On a more conceptual basis, Williamson (1998) calls for a closer and more formal investigation of non-profit forms of organizational governance, in which contractual relations are often complex, but which may very well be frequent in environmental settings.

Already early on, TCE was applied as an analysis framework for governance structures and internal organization characteristics within agricultural and environmental economics, where the nature of produced goods – having characteristics like perishability or high spatial specificity – allow for a straightforward application of the TCE analysis framework

(cp. Masten, 2000). In this domain, different aspects have been in focus of investigation. Besides the determination of transaction costs related to farming and agriculture (Falconer, 2000; Hagemann et al., 2015; McCann, 2013), the role of transaction costs in water markets and water market regulation (McCann and Garrick, 2014), repeated biodiversity conservation programs (Groth, 2008), carbon offset programs (Cacho et al., 2013; Phan et al., 2017a), or PES (Peterson et al., 2015; Phan et al., 2017b; Zanella et al., 2014) and AES (Mettepenningen et al., 2011).

McCann (2013) suggests that from an environmental economic perspective, in particular transaction costs are an important object of research as they are decisive in the implementation of policy instruments at farm level, e.g. AES. In reality, environmental regulation and implementation causes substantial amounts of transaction costs. Transaction costs cause impacts on actor's behaviour and the optimality of their actions. The more detailed a regulation and application process is, e.g. for an AES, the higher the associated transaction costs are, especially on the agent's side, i.e. the landowner (Falconer, 2000; McCann, 2013; McCann et al., 2005).

1.3.3 Environmental governance

Environmental governance can be understood as the “variety of institutional arrangements [for natural resource management], involving both state agencies and non-state actors” (Birner and Wittmer, 2004, p. 667). Besides a broader extension of the term environmental governance, this definition is in line with Paavola (2007) suggesting environmental governance to range from state to non-state involvement in the management of environmental resources.

So far, only few studies have investigated this field. E.g., Muradian and Rival (2012) conceptually analyse governance of conservation and argue that market-based policy instruments are limited in dealing with PES governance in particular, due to the complex nature and common good characteristics of PES. Thus, hybrid governance modes in the aforementioned Coasean or Williamsonian sense are preferable for the provision of ecosystem services. Various influential factors on the effectiveness and applicability of PES, such as equity issues or strategic behaviour of participants (Neuteleers and Engelen, 2015; Vatn, 2015) have been assessed to understand the usefulness of PES as a policy instrument and a governance structure for biodiversity conservation (Gómez-Baggethun and Muradian, 2015). Empirical research with respect to governance of conservation

implementation focuses on the cost-effectiveness of PES implementation vs. land purchase or easement (Curran et al., 2016), and cost differences of implementation permanent forest conservation from land purchase vs. temporal conservation on leased land in Finland (Juutinen et al., 2008).

Generally, within any given governance mode used for conservation purposes and in particular the two principle governance modes of land purchase and landowner compensation, various practical strategies of implementation of conservation measures are available. Conservation actors, besides choosing a governance mode for their conservation actions, also have to decide, among other things, on aspects like conservation site selection, conservation measure, or contract or payment design.

One common governance mode to implement biodiversity and nature conservation policies is the compensation of conservation costs of landowners. Thus, landowners are motivated to implement conservation measures voluntarily by covering the financial burden or even providing a profit margin for the respective land user due to the implementation of conservation measures on their lands (Uthes and Matzdorf, 2013). This method is used for example within the main pillar of integrating environmental aspects in the Common Agricultural Policy of the EU (Kuhfuss et al., 2019; Uthes and Matzdorf, 2013), in preserving erosion and destruction of different kinds of ecosystems in various programmes in the USA (Baylis et al., 2008), and more recently are seen as a viable option for conservation in emerging and developing countries (Zhu et al., 2018).

Many questions regarding the optimal design of such compensation payment schemes have been addressed and reviewed in length, e.g. ecological effectiveness and cost-effectiveness of payment design (e.g. Armsworth et al., 2012; Ekroos et al., 2014), spatial aspects (e.g. Arponen et al., 2013; Drechsler et al., 2016; Lewis et al., 2011; Phalan et al., 2011; Uthes et al., 2010; Wätzold and Drechsler, 2014), or temporal aspects (e.g. Ando and Chen, 2011; Drechsler et al., 2017; Johst et al., 2015; Juutinen et al., 2014; Makino et al., 2014).

As compensation payment schemes are implemented predominantly through temporarily limited contracts to incentivise landowners for voluntary provision of conservation activities, land purchase for conservation purposes can be seen as a more long-term alternative to govern conservation activities. Hence, land purchase can be considered as an alternative governance mode of conservation measures, which typically – in contrast to most compensation payment schemes – is of a more long-term nature (Knight et al., 2011).

This approach is already used in various countries in different forms. For example, in Germany federally funded nature conservation agencies (e.g. the “Stiftung Naturschutz Schleswig-Holstein”) purchase land to either actively implement conservation measures or to set aside areas as nature reserves permanently (e.g. Stiftung Naturschutz Schleswig Holstein, 2017). In-fee land purchase in the US for example allows land purchase on different governmental levels, i.e. national, state, county, etc., to then be utilized in the interest of the purchasing body, e.g. for conservation purposes (Santos et al., 2014b). Internationally, a wide array of different types of land purchase can be observed, ranging from privately financed land purchase (Kamal et al., 2015), over governmental purchase of partial land development rights, to full title governmental land purchase for conservation (cp. Nolte, 2018).

This work contributes to this debate of governance mode choice by analysing the *make-or-buy decision* in the context of biodiversity conservation from different perspectives to shed light on the cost-effective choice of governance modes and the respective ecological, economic and climatic factors influence. In particular, the two governance modes of land purchase and landowner compensation are investigated.

1.3.4 Cost-effectiveness

Financial resources are a scarce good in the political arena. Limited budgets have to be split between many competing ends, of which biodiversity conservation is just one. Hence, resources attributed to the cause of conservation should be used such that they maximize the intended ecological outcome of conservation activities with the financial resources available (Birner and Wittmer, 2004; Naidoo and Ricketts, 2006; Wätzold and Schwerdtner, 2005; Wilson et al., 2009).

The concept of cost-effectiveness addresses this issue. A formal definition of cost-effectiveness is the maximization of ecological goals for given financial resources (Wätzold and Schwerdtner, 2005). Alternatively, the minimization of necessary financial resources to reach a given ecological target is also considered a cost-effective solution (Ando et al., 1998; Ferraro and Pattanayak, 2006). Specifically, the criterion of cost-effectiveness in either perspective can be used to rank different conservation measures, implementation strategies or governance modes accordingly. Both approaches can be useful to measure cost-effectiveness in different conservation settings. Within this thesis,

either of the two cost-effectiveness interpretations are in the following used when appropriate.

Conservation research embodies the concept of cost-effectiveness to analyse the performance of various conservation policies. Studies exist – among many others – in the field of conservation planning (e.g. Polasky et al., 2008), design of conservation payments (e.g. Wätzold et al., 2016), contract length (e.g. Ando and Chen, 2011; Drechsler et al., 2017), and habitat type selection (e.g. Petersen et al., 2016).

The optimal choice of conservation measures and governance modes and their respective implementation, similarly faces the challenge to reach best possible outcomes in ecological terms, while at the same time facing limited available conservation budgets. The cost-effectiveness of different governance modes and strategies to implement conservation measures hence is analysed and used to inform about and to compare the relative performances of biodiversity conservation in various settings.

1.4 Scope of the thesis

This thesis includes three peer-reviewed and published scientific articles and one working paper presented in Chapters 2 to 5 as the main body of research following this introductory chapter.

Chapter 2 approaches the *make-or-buy decision* in a biodiversity conservation setting from a conceptual perspective and includes Schöttker et al. (2016) into this thesis. The work provides a conceptual understanding of different governance modes, i.e. land purchase for biodiversity conservation (i.e. the *buy alternative*) versus landowner compensation for their voluntary provision of conservation measures (i.e. the *compensation alternative*). It analyses a set of economic and ecological parameters and their influence on the cost-effectiveness of the two governance modes, which was simulated in an integrated ecological-economic model. A spatially implicit metapopulation model based on Drechsler and Johst (2010) was used to assess the impact of habitat turnover on species conservation and the respectively calculated mean metapopulation lifetime of a target species. In turn, the area selection decisions performed by a conservation agency and their respective costs, combined with the mean metapopulation lifetime generated by the specific area selection are used to derive an average cost-effectiveness of the different area selection dynamics

generated by the two governance modes. The model calculates the cost-effectiveness within a Monte-Carlo simulation, also taking into account parameter changes to allow for a conceptual analysis of their influences on the cost-effective governance mode choice. It was, among other things, found that the *buy alternative* is more cost-effective compared to the *compensation alternative*, if higher conservation budgets are available or if interest rates are high. The *compensation alternative* is e.g. more cost-effective compared to the *buy alternative* in case of low habitat turnover.

Chapter 3 builds on the conceptual analysis of governance mode choice for conservation purposes developed in Chapter 2 and extends the analysis towards the discussion of potential influences of climate change on this decision. This chapter was published as a working paper (Schöttker and Wätzold, 2020). A computational approach was chosen to simulate different governance modes for conservation (land purchase vs. landowner compensation) in combination with different conservation site selection strategies in a landscape with changing climatic conditions. This approach was used to calculate the costs of implementation and the ecological success of each governance mode and site selection strategy. The climate change model is based on Hily et al. (2017) and was adapted to fit the specific needs of the simulation model of this chapter. In contrast to Chapter 2, a spatially explicit ecological model based on Hanski (1999) was used to describe species dynamics and migration in the landscape. Based the results of a Monte-Carlo simulation a cost-effective governance mode choice was derived and factors influencing this decision from an economic, ecological and climatic perspective were analysed. It was found that for a cost-effective implementation of conservation, not only the governance mode is relevant, but also the site selection strategy by which relevant areas are selected for conservation. Changing model characteristics like climate change speed influence the cost-effectiveness of governance modes and implementation strategies.

Chapter 4 includes Schöttker and Wätzold (2018) into the thesis and approaches the question for a cost-effective governance mode on an empirical basis and evaluates a case study of a particular conservation project, i.e. the Lake Bültsee conservation area in Schleswig-Holstein, Germany. In this case study, the actually occurred costs of implementing a particular conservation project on areas, which have formerly been bought by the conservation agency, the Stiftung Naturschutz Schleswig-Holstein, were compared with the hypothetically occurring costs for setting the same land under conservation within a compensation scheme. Next to the provision of empirical insights and an ex-post

evaluation of a concrete conservation project, Chapter 4 provides a framework to assess costs in different governance modes. The framework allows for an easy and structured assessment of costs and a consequential comparison of costs and cost developments over time. A key finding is that the choice of a cost-effective governance mode depends on the intended timeframe of a conservation project, with land purchase being preferable in long-term projects and landowner compensation preferable in short-term projects. In addition, with factors like increasing transaction costs and decreasing land resale prices, land purchase increases in cost-effectiveness relative to the compensation payment based conservation scheme.

Chapter 5 includes Schöttker and Santos (2019) into this thesis. Here, the framework developed in Chapter 4 was extended to evaluate the *make-or-buy decision* to answer the question for a cost-effective governance mode for biodiversity conservation in California, USA. The principally available modes of governance for conservation implementation available in this setting are conservation easements and land purchase (i.e. “in-fee” managed land). The study investigates four differently sized conservation organizations operating in California with different regional areas of interest and different governance structures, i.e. governance on the level of state, county, city, and a Special District. It was found that for all measures evaluated, governance via conservation easements exceed conservation costs of in-fee managed land in the long run, suggesting preferability of land purchase over landowner compensation based conservation. Similar cost development patterns were found for all governance levels, i.e. state, county, city, and a Special District level.

Following the main body of the thesis, concluding remarks and a discussion are provided in Chapter 6. Besides summarizing the objectives and results of the previously presented chapters, additionally policy recommendations and research gaps are presented.

Chapter 2

Land for biodiversity conservation — To buy or borrow?

This chapter has previously been published as:

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The chapter includes a reprint of the published version,
with minor corrections in grammar and spelling, as well
as adapted cross-references and numbering to match the
format of the thesis.

2 Land for biodiversity conservation — To buy or borrow?

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 given 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.

2.1 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 (Naidoo et al., 2006). 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 (Henle et al., 2008; Metera et al., 2010; Young et al., 2005). Although extensive farming measures and the maintenance of farming in marginal areas are better for conservation, they are costly to farmers.¹ If in such a situation property rights are allocated in a way that land

¹ Considering ecosystem services generated by the measure (for example pollination) may reduce the costs to farmers (cp. Cong et al., 2014).

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 main 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 (Sears and Gilbert, 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, in several German federal states farmers are offered 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 (Wätzold et al., 2016).

Conservation organisations are faced with the question of budget efficiency, i.e. which of the two alternatives will lead to a higher level of biodiversity conservation for the budget available to an organisation (Armsworth et al., 2011; Wätzold and Drechsler, 2014; Wätzold and Schwerdtner, 2005)? 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 present a conceptual ecological–economic model, and apply the model to analyse how changes in ecological and economic parameters influence the relative efficiency performance of the two alternatives. The model is 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 is designed to capture the main features of conservation measures in such situations.

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. The only work we are aware of that is somewhat close to our research is Juutinen et al. (2008) who compare for a case study in Finland the costs of purchasing forest with the costs of paying landowners for forest conservation. We go beyond Juutinen et al. as we develop a general conceptual ecological–economic model that also takes into account the ecological effects of the two alternatives.

Our model is built based on knowledge and experience from different areas of research. We draw on conservation planning by combining costs and benefits of conservation measures for cost-effectiveness analyses (Armsworth, 2014; Naidoo and Ricketts, 2006; Shackelford et al., 2015) in a dynamic environment (Jantke and Schneider, 2011; Pressey et al., 2007). Our conceptual ecological–economic model is inspired by similar models applied to compare the budget efficiency of different conservation policy instruments (Drechsler et al., 2010; Wätzold and Drechsler, 2014). Regarding the behaviour of landowners in the borrow alternative, our model is related to analyses of optimising compensation payments for biodiversity-enhancing land-use measures in agricultural landscapes (Bamière et al., 2011; Cong et al., 2014; Mouysset et al., 2015). The ecological component of our model is based on conceptual research on the impact of habitat turnover on species conservation (Drechsler and Johst, 2010; Johst et al., 2011, 2012). Similar to the debate on the optimal length of contracts for payments to compensate landowners for conservation measures (Ando and Chen, 2011; Lennox and Armsworth, 2011), we consider that conservation measures are carried out for different periods of time.

2.2 The model

2.2.1 Landscape structure, conservation costs and landscape dynamics

We consider a landscape, which consists of $N = 100$ patches (Table 2.1 provides a brief explanation of all model variables and Table 2.2 of all parameters). Each patch can be managed either intensively or extensively. An intensively used patch generates maximum profit 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; but see the final section for a brief discussion of this assumption).

The profits π_i in the landscape are heterogeneous and vary in a range from $(\bar{\pi} - \sigma)$ to $(\bar{\pi} + \sigma)$ with $\bar{\pi}$ the average profit of the patches and σ the possible profit variation. 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. 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 extensively managed patches by multiplying the potential profit of these patches generated by intensive management with a landscape-wide factor $(1 - f) < 1$. To take into account the cost difference for extensive management between private landowners and the conservation agency (i.e. the borrow and buy alternatives) in a simple manner, we assume that extensive management in the buy alternative generates zero profits for the agency ($f = 1$). In contrast, for the borrow alternative we assume

positive profits for the landowners ($0 < f < 1$). For a discussion of these assumptions see Section 2.5.

Table 2.1: Description of economic and ecological model variables.

variable	description
economic model variables	
N	total number of patches in the landscape ($N = 100$), either managed intensively or extensively
π_i	maximum achievable profit for patch i
$\bar{\pi}$	average profit of all patches in the landscape ($\bar{\pi} = 1$)
T	model timeframe with $t \in [0, T]$ and $T = 49$ periods
$price_i$	price of patch i
B_t	budget available for the agency in period t , with B_0 the initial budget and B_T the budget in the models final period
$x_{buy,i}$	dummy variable: $x_{buy} = 1$ if patch i is bought, $x_{buy} = 0$ if patch i is not bought for conservation
$x_{comp,i,t}$	dummy variable: $x_{comp} = 1$ if patch i is compensated in period t , $x_{comp} = 0$ if patch i is not compensated for conservation activity in period t
cp_t	homogeneous compensation payment for every patch i in period t
cp_{total}	total amount of compensation payments in the landscape
ecological model variables	
N_{buy}	patches bought in the buying alternative
N_{borrow}	number of habitable patches
μ	average, landscape wide habitat destruction rate (proportion of patches destroyed per period) within the timeframe $[0, T]$, with μ_t the landscape wide habitat destruction rate in period t
λ	average, landscape wide patch creation rate (proportion of patches created per period) within the timeframe $[0, T]$, with λ_t the landscape wide patch creation rate in period t
T_{meta}	mean metapopulation lifetime as proxy for the ecological benefit
\bar{e}	geometric mean over local extinction rates e_i
$q = \bar{c}/\bar{e}$	aggregated colonisation extinction ratio
\bar{c}	power mean of local colonisation rates c_i
H	patch connectivity measure with $H = 1$ in our calculations (i.e. good connectivity is assumed)

Table 2.2: Description of parameters and parameterisation of the ecological–economic model.

parameter	description	values		
<i>economic parameters</i>		ranges		
		min		max
B_0	initial budget	50		150
r	interest rate	0.01		0.06
<i>further economic parameters</i>		low value	base value	high value
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
<i>ecological parameters</i>				
ε	species extinction rate	0.05	0.10	0.30
m	species colonisation rate	0.4	0.8	2.4
		(implicitly determined by $m = \varepsilon * \gamma$)		
$\gamma = m/\varepsilon$	species colonisation-extinction ratio	4	8	12

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.

2.2.2 The decision problem of private landowners

Landowners are assumed to be profit-maximising, 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; Unay Gailhard and Bojnec, 2015).

As this assumption is somewhat uncommon, it requires a brief motivation. Some landowners may want to sell their farm and if the buyer is bound in anyway 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 they often only have expected values about the costs of managing their land extensively. Landowners only receive the full information about costs through experience, which may then cause them to reverse their management decision (Fronzel et al., 2012). Also, non-economic factors like personal believe and education may influence a landowner's willingness to participate and change over time (Defrancesco et al., 2008; Howley 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.

2.2.3 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 in a simple manner 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 (cp. Eqs. (2.2) and (2.5)).

2.2.3.1 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.² The price of patch i therefore is equal to:

$$price_i = \sum_{t=0}^T \frac{\pi_i}{(1+r)^t} \quad (2.1)$$

in which $\pi_i \in [(\bar{\pi} - \sigma), (\bar{\pi} + \sigma)]$ is the randomly assigned profit value of patch i and r the interest rate. Formally, the resulting budget constraint of this alternative reads as follows:

$$B_{T,buy} = B_0 - \sum_{i=1}^N price_{buy,i} \times x_{buy,i} + \sum_{i=1}^N price_{sell,i} \times x_{buy,i} \quad (2.2)$$

with B_0 being the initial budget of the agency, B_T the agency's 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. 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 . This implies that the location of extensively and intensively managed patches remains unchanged and the resulting landscape is static.

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. The number of purchased patches N_{buy} is calculated by simulating the buying process of the agency according to Eq. (2.2). As an alternative solution, we provide an analytical approach to determine N_{buy} in the Appendix A.

2.2.3.2 Borrow alternative

We assume that the agency knows the average profit for the landscape π , 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

² The capitalization formula suggested by Burt (1986) provides a simple model of farmland prices for situations with a fixed farmland quantity. The farmland price is considered to be the discounted sum of possible future profits from the farmland.

landowners and offers a homogeneous compensation payment cp_t to landowners for extensive management. Hence, the payment equals the costs of extensive land management for the marginal landowner.

To cover the costs for the compensation payments for extensive management in each period, the agency generates revenue 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 timeframe 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} . Therefore, the resulting landscape is dynamic.

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

$$\max \sum_{i=1}^N x_{comp,i,t} \quad (2.3)$$

subject to the budget constraint:

$$\sum_{i=1}^N x_{comp,i,t} \times cp_t \leq B_t \quad (2.4)$$

Here, $x_{comp,i,t} = 1$ with probability P_{part} if the compensation payment cp_t 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:

$$B_{T,comp} = B_0 + \sum_{t=0}^T \frac{B_0 \times r - cp_{total}}{(1+r)^t} \quad (2.5)$$

where $cp_{total} = \sum_{i=1}^N x_{comp,i,t} * cp_t$ with $x_{comp,i,t} = 1$ if the corresponding landowner of patch i is compensated in period t , and 0 otherwise. 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 .

The number of extensively managed patches N_{borrow} in the borrow alternative is calculated by simulating the decisions of the landowners as described above. It thus depends on the initial budget B_0 , the interest rate r , the randomly drawn profits π_i of each patch and the participation willingness P_{part} of the landowners. In the Appendix A we provide an analytical approach to determine N_{borrow} reflecting these dependencies.

2.2.4 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 patch number and habitat dynamics.

The buy alternative generates a static landscape without habitat turnover and with habitat patch number N_{buy} (Section 2.2.3.1). In contrast, the borrow alternative generates a dynamic landscape with habitat patch number N_{borrow} (Section 2.2.3.2) and a certain habitat turnover described by a patch destruction rate μ (the rate at which extensively managed patches are transformed into intensively managed patches) and a patch creation rate λ (the rate at which intensively managed patches are transformed into extensively managed patches). As the habitat turnover in each period is driven by the randomly drawn outcome of the participation willingness of landowners, we calculate average patch numbers N_{borrow} and average rates $\mu = \sum_{t=0}^T \mu_t / 50$ and $\lambda = \sum_{t=0}^T \lambda_t / 50$, 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). This formula is designed to calculate the mean metapopulation lifetime T_{meta} of a certain species in static and dynamic landscapes in a

straightforward way. The mean metapopulation lifetime can be explained as “the expected lifetime, or mean time to extinction of a metapopulation” (Drechsler and Johst (2010), p. 1889), with metapopulations being a population of a species consisting of multiple local sub-populations (Hanski, 1999). The formula for T_{meta} reads as follows:

$$T_{meta} \approx \frac{1}{\tilde{e}} \sum_{i=1}^{N_{dyn}} \sum_{k=i}^{N_{dyn}} \frac{1}{k} \times \frac{(N_{dyn} - i)!}{(N_{dyn} - k)!} \times \frac{1}{(N_{dyn} - 1)^{k-i}} q^{k-i} \quad (2.6)$$

The stochastic metapopulation dynamics in a habitat network where local species populations can go extinct but empty habitat patches can be recolonised by neighbouring local populations is modelled as a so-called Markov process. Eq. (2.6) is basically a double sum over all patches (summation index i for the first sum and k for the second sum) and the faculties consider the many possible paths of local extinction and recolonization until the metapopulation as a whole goes extinct.

Quantity \tilde{e} in Eq. (2.6) is the geometric mean over the local extinction rates (the rates at which local populations go extinct from one period to another). These rates include the patch size dependent, species-specific local extinction rates e_i and the patch destruction rate μ as patch destruction results in population extinction as well Drechsler and Johst (2010). As we assume equal patches, all $e_i = \varepsilon$ and the geometric mean is:

$$\tilde{e} = \varepsilon + \mu \quad (2.7)$$

with the species specific parameter ε describing the extinction rate of a species on a patch.

The quantity q in Eq. (2.6) is an aggregated colonisation–extinction ratio defined as

$$q \approx \frac{\bar{c}}{\tilde{e}} H = \frac{m}{\varepsilon + \mu} \quad (2.8)$$

The patch connectivity measure H in Eq. (2.8) is set to $H = 1$ for our analysis implying that a species can reach all patches in the landscapes equally well. The colonisation rate c in Eq. (2.8) is the power mean of the patch size dependent, species-specific local colonisation rates c_i , which describe the rate at which a species can colonise new patches within one period. As we assume patches of equal size, all $c_i = m$ and the power mean is:

$$\bar{c} = m \quad (2.9)$$

We quantify species-specific colonisation rate m in relation to its extinction rate ε by

$$m = \gamma \times \varepsilon \quad (2.10)$$

where γ is called the species-specific colonisation-extinction ratio.

All ecological input parameters (species extinction and colonisation rate ε and m) and the patch turnover rates refer to the unit ‘per period’ thus resulting in metapopulation lifetimes in units of ‘periods’. Accordingly, a metapopulation lifetime of e.g. 100 periods may correspond to quite different times depending on the choice of period length.

Quantity N_{dyn} in Eq. (2.6) is the number of habitat patches depending on the management alternative, i.e. $N_{dyn} = N_{buy}$ and $N_{dyn} = N_{borrow}$, respectively (for their calculation see Sections 2.2.3.1 and 2.2.3.2).

2.3 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 , π , and σ influence the economic conditions which affect both the buy and borrow alternatives. The parameters f and P_{part} influence the number and turnover of habitat patches in the borrow alternative but have no impact on the buy alternative. Finally, the species-specific parameters ε and γ (Eq. 2.10) affect the ecological benefit resulting from the management alternatives. Variations in these parameters represent species with different ecological characteristics (see below). Table 2.2 shows the base case parameterisation and the possible parameter variations. We generally selected the parameters in a way that they resemble reasonable economic and ecological values were possible (see explanations below), and varied both ecological and economic parameters in a range showing the sensitivity of our results to parameter variations.

For the given base case parameters of ε , γ , $\bar{\pi}$, σ , f and P_{part} (compare Table 1) we vary the initial budget B_0 from values of 50 to 150 (in steps of 5 units) and the interest rates r from 0.01 to 0.06 (in steps of 0.0025). For each parameter combination we simulate the economic model 100 times to capture the stochastic variability of the economic outcome and calculate average values for habitat patch number and turnover (see Section 2.2.4).

Afterwards we numerically evaluate the ecological outcome for each of the two management alternatives with the mean metapopulation lifetime (Eq. 2.6). To measure the relative efficiency performance of the two alternatives as a function of ecological and

economic parameters we use the difference of the logarithmic mean metapopulation lifetime from the borrow alternative and the buy alternative ($\log(T_{meta}^{borrow}) - \log(T_{meta}^{buy})$).

Afterwards we individually vary the parameters f , P_{part} , σ , ε and γ and investigate each parameter combination as described above to analyse the possible effects of changes in these parameters on the relative efficiency performance. This is done by setting each of the parameters separately on a low and high level (see Table 2).

The 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) together determine the proportion of the landscape which is included in buy or borrow activities and thus the number of participating patches (Fig. 1). For example, for small budget and interest rate values (e.g. $B_0 = 50$, $r = 0.02$), this proportion is rather small, i.e. just 2% of the total available patches participate in the borrow alternative. For large values ($B_0 = 150$, $r = 0.06$), this proportion is high, i.e. 14% of available patches participate in the borrow alternative. These values represent a range of proportions of a landscape designated for conservation that are common in human-dominated landscapes. 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 patches in the borrow alternative for the same B_0 and r values (see Fig. 2.1).

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 $m = \gamma \times \varepsilon$ 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 local population lifetime of 10 periods which can be seen as an average value. The value is then varied to 0.05 and to 0.30. Lower (higher) values of ε suggest species with lower (higher) local extinction risks.

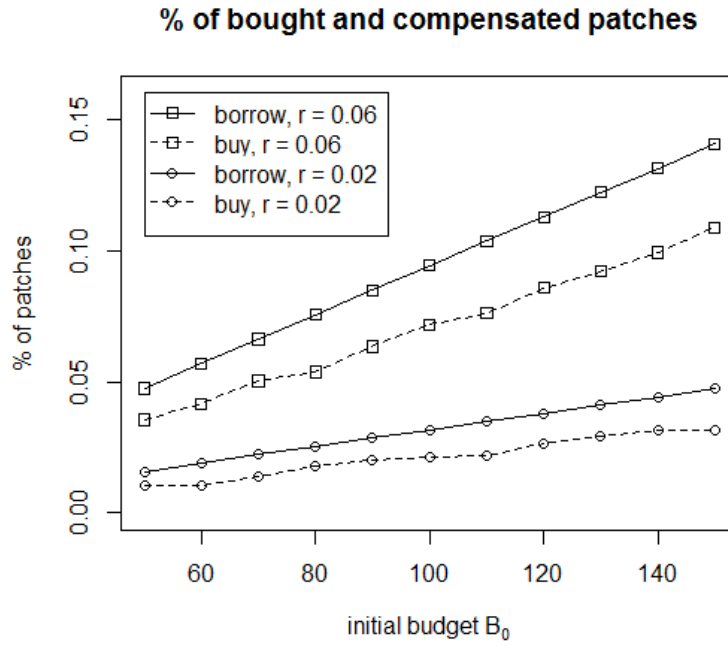


Figure 2.1: The percentage share of bought patches and patches receiving compensation payments, depending on the initial budget B_0 , shown for the two levels of interest rate $r = \{0.02, 0.06\}$, increases.

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. The value is then varied to 4 and to 12. Lower (higher) values of γ indicate species which can less (more) easily colonise new patches.

Identifying appropriate economic parameters faces the challenge that there is either no empirical information available to suggest possible parameter values as we define them in our model (participation willingness of landowners in the borrow alternative, P_{part}) or the values strongly differ in reality depending on the specific situation (forgone proportion of profit, f , and variation in profit, σ). Therefore, we selected the economic parameters in our calculations in such a way that they seem within a realistic range and easily visualise the results with respect to changes in the mean metapopulation lifetime of the buy and the borrow alternative.

By normalizing the average profit π to the value of 1, the variation in profits σ determines the range of profits in the landscape. We set a base value of $\sigma = 0.10$ and vary it to 0.05 and to 0.15, generating lower and higher profit variability in the landscape. The value for

the forgone proportion of profit is set to $f = 0.7$ in the base case scenario and varied to a low value of $f = 0.65$ and to a high value of $f = 0.75$. The value for the participation willingness of landowners in the borrow alternative is set to $P_{part} = 0.95$ in the base case scenario and varied to 0.925 and to 0.975. We selected a relatively high base case value of P_{part} as events that induce a landowner to change participation behaviour in the situation where the opportunity costs are compensated seem rare.

2.4 Results

2.4.1 Effects of variations in B_0 and r

For all scenarios we find that the budget efficiency of the two alternatives depends on the initial budget B_0 and the interest rate r . This dependency is shown in Fig. 2.2 for the base case scenario. Consider the initial budget first.

In Fig. 2.2 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

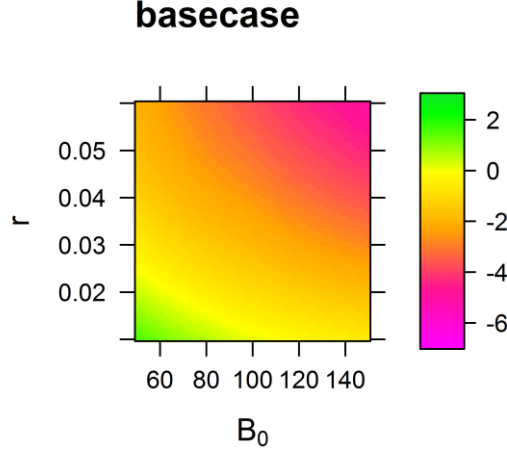


Figure 2.2: Relative efficiency performance of the ecological benefit (mean metapopulation lifetime) of the borrow and buy alternatives, expressed in logarithms ($\log(T_{meta}^{borrow}) - \log(T_{meta}^{buy})$) and plotted as a function of interest rate r and available initial budget B_0 . Increasing (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 2.2).

reason for this consider for both alternatives what happens in the case of an increasing budget. In the buy alternative costlier 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 a rising budget enables the agency to increase the amount of extensively managed patches by increasing the compensation payment. While on the one side more landowners participate, on the other side the already extensively managed low-cost patches also receive the higher compensation payment as payments are homogeneous. This is the reason why with an increasing initial budget the efficiency of the borrow alternative decreases in comparison to the buy alternative.

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. (2.1) 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. (2.1)) 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.

2.4.2 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 $\tilde{\varepsilon}$ in Eq. (2.7) and decreases q in Eq. (2.8), both decreasing metapopulation lifetime (see Eq. 2.6). Therefore, the borrow alternative can only perform better than the buy alternative if habitat turnover is compensated by a sufficiently larger habitat patch number $N_{borrow} > N_{buy}$ (see Fig. 2.1). At low B_0 and r the difference between N_{borrow} and N_{buy} 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. 2.2). 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 (red area in the right upper corner of Fig. 2.2).

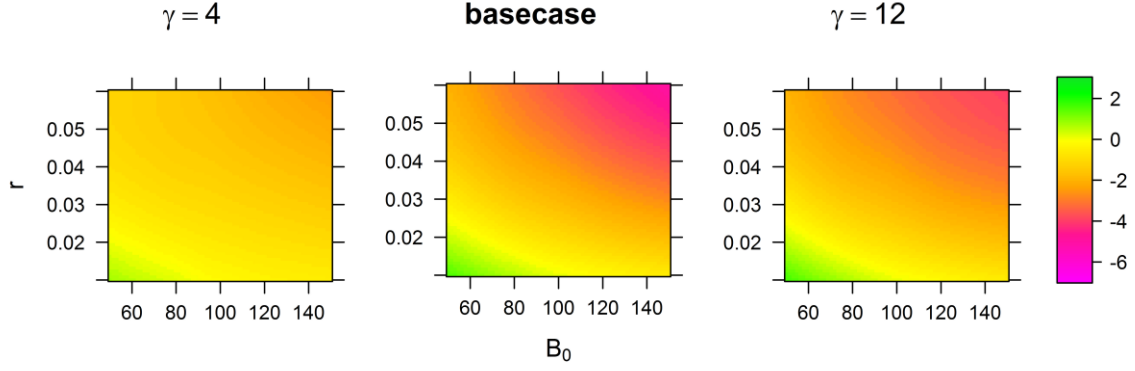


Figure 2.3: Impact of changes in species specific colonisation-extinction ratio (γ) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2.

A change in the species parameter γ (Fig. 2.3) describing the colonisation potential of a species does not reverse the efficiency performance pattern of the base case scenario of Fig. 2.2. Nevertheless, we can observe for species with low levels of γ (weaker dispersers; $\gamma = 4$ in Fig. 2.2) 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.

A change in the species parameter ε (Fig. 2.4), however, has a much larger effect. Increasing ε considerably decreases the performance of the buy alternative and may even reverse the relative efficiency performance. An increase in ε increases the total local extinction risk \tilde{e} , thereby decreasing the relative contribution of the habitat destruction rate

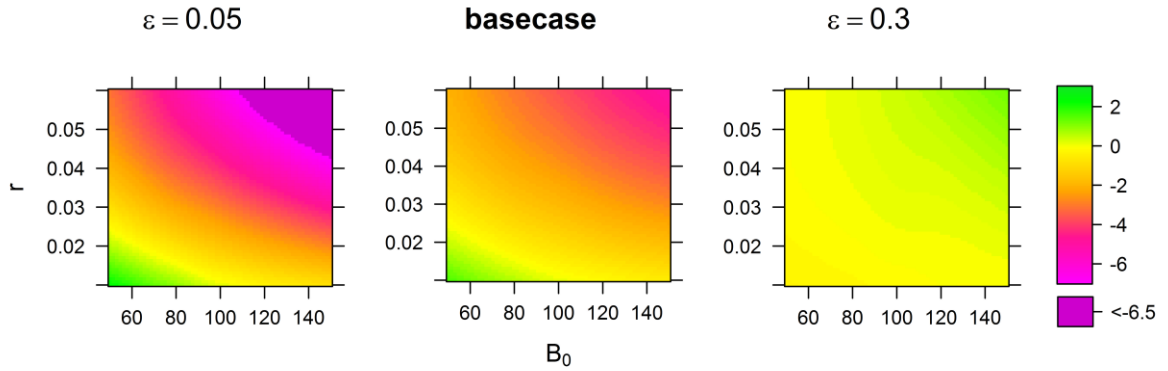


Figure 2.4: Impact of changes in the species-specific local extinction ratio (ε) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2. The dark magenta area indicates the difference of the logarithmic ecological benefits to be smaller than -6.5 , i.e. an even better relative efficiency performance of the buy alternative compared to the borrow alternative.

μ in both Eqs. (2.7) and (2.8). This in turn decreases the relative impact of habitat destruction and thus patch turnover on the metapopulation lifetime (see Eq. 2.6), 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 higher relative performance of the borrow alternative.

2.4.3 Effects of f , P_{part} and σ

We find that with an increasing proportion of profit f which landowners lose by using the land extensively the efficiency of the buy alternative increases compared to the borrow alternative (Fig. 2.5). 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).

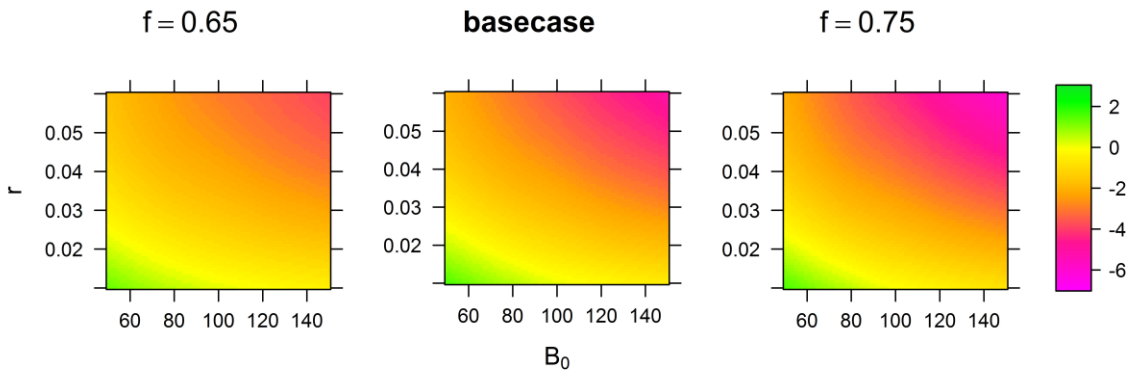


Figure 2.5: Impact of changes in the foregone proportion of profit (f) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2.

Conversely, increasing values of P_{part} , i.e. increasing probabilities of landowners to participate in compensation schemes for given payments, imply that the efficiency of the borrow alternative increases in comparison to the buy alternative (Fig. 2.6). There are two reasons for this result. The first is that increasing values of P_{part} imply a decreasing probability that landowners switch from participation in one period to non-participation in the next period and vice versa in the borrow alternative. This in turn leads to decreasing patch turnover, which is beneficial to biodiversity. The second reason is decreasing compensation costs. Lower compensation costs arise because a high willingness to participate means that a relative high share of low-cost patches is managed extensively. This implies that the agency is able to contract the same number of patches with lower compensation payments. Therefore, for a given budget a higher amount of patches can be managed extensively resulting in a higher ecological outcome of the borrow alternative.

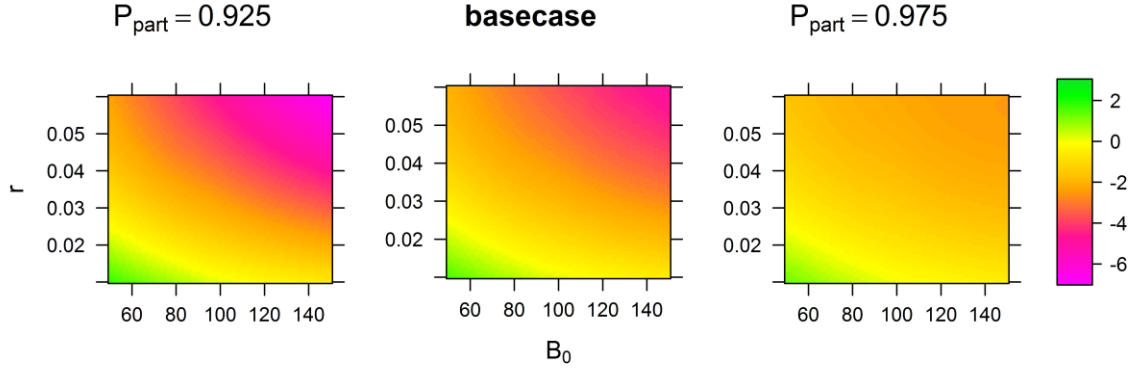


Figure 2.6: Impact of changes in the participation probability (P_{part}) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2.

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. 2.7). 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 lead 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.

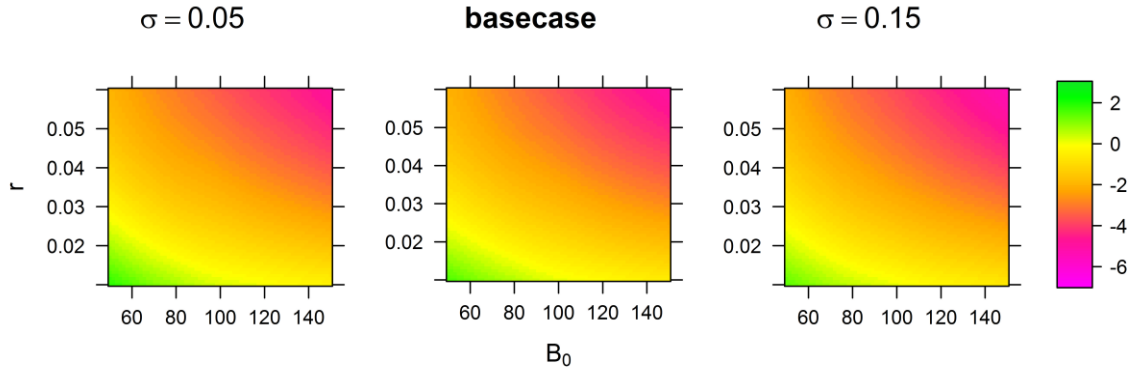


Figure 2.7: Impact of changes in the profit variation in the landscape (σ) on the relative efficiency performance of the buy alternative compared to the borrow alternative. Figure explanations as in Fig. 2.2.

A second consequence of increasing σ is that the purchasing price according to Eq. (2.1) and the compensation payment (see Section 2.2.3) 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 patch receiving compensation payments 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.

2.5 Summary and discussion

Given that landowners often cannot or should not be forced to carry out conservation measures on their land, conservation organisations have, in principle, two main options. The first alternative is to buy land and carry out the conservation measures themselves. 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 conservation measures. Here, no high initial payment is needed but landowners may not participate continuously in the compensation scheme leading to habitat turnover, which is disadvantageous for some species. We 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 as well as in the interest rate 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. We further 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 conservation measures are applied. Concerning the ecological parameters, the relative efficiency performance of the two alternatives is by and large not affected by the species colonisation rate but can be reversed at high species local extinction rates, which generally favour the borrow alternative (for details see Section 2.4).

Our model contains some restrictive assumptions, which require a brief discussion. First, we assume that in the buy alternative the conservation agency does not sell the land that it has purchased for conservation. We consider this a key feature and advantage of the buy alternative, but, of course, in reality reasons exist why the agency might want to sell the land. These could be ecological reasons (for example, a valuable habitat might become unsuitable for a species due to climate change) or economic reasons (for example a steep increase in land prices might make it worthwhile to sell conserved land and buy cheaper land elsewhere). However, the agency is not forced to sell the land and a rational conservation agency would do so only if the conservation benefits outweigh the conservation costs in terms of habitat turnover. Therefore, the possibility of the agency to sell the land does not alter the advantage of the buy alternative that arises from being able to keep the land for conservation as long as the agency wishes. It rather adds an additional argument in favour of the buy alternative. A further assumption is that the conservation agency does not make any profit from managing the land it buys and uses for conservation. We make 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.

We assume 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 make 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 or other restrictions. For example, by allowing landowners 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.

We also assume perfect knowledge about the future development of key economic parameters such as profit made from land management, interest rates and land prices (Eq.

2.1). 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 alternatives. 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 can conserve fewer patches than foreseen with the interest. However, at this stage it is too late to buy land because land prices have also increased (cf. Eq. 2.1). 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.

A further assumption that requires discussion is that the spatial location of the patches does not matter for conservation. This assumption is realistic for some species, but the spatial configuration of the landscape clearly matters for other species (Bamière et al., 2011; Drechsler et al., 2010; Hanski, 1999). Again, we leave it to further research to investigate how the efficiency performance of the two options changes if it is considered that the spatial allocation of patches matters.

In our model we considered with the buy and the borrow alternatives two ‘pure’ options. In reality, there are also other options. For example, the conservation agency might opt for a ‘mix’ and decide to spend part of the available financial resources to buy land and the other part to compensate landowners for conservation measures. This might be a promising option if the agency aims to conserve two species and the characteristic of one species is such that the borrow alternative is likely to be more favourable and the characteristic of the other species that the buy alternative is better. If the conservation agency is a large nature conservation foundation it might also decide to open up its own farms to gain experience in carrying out conservation measures. Then it might not lead to a profit loss if the agency does the measures by itself.

Our work was designed to capture typical conservation problems found in cultural landscapes, which require a certain active type of biodiversity-enhancing land use. However, the decision problem of whether to buy or borrow land for conservation 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 habitat 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 timeframe. Nevertheless, we think that our model provides a useful starting point for analyses and believe that further research in this field could be fruitful.

Obviously, our model is of a conceptual nature and its direct policy relevance is admittedly limited. 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 provides a framework for real world case studies in which the budget efficiency of an existing (or planned) borrow or buy alternative is compared with the respective other alternative. For such case studies economic data on interest rates r , land prices and cost differences 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. Moreover, the length of the time periods t has to be defined.

Such case studies might provide valuable decision support regarding the budget efficiency of the buy and the borrow alternatives, similar to other case studies where ecological and economic data have been fed into ecological– economic models to provide policy recommendations (cf. Armsworth et al., 2012; Bamière et al., 2011; Drechsler et al., 2010). Decision support is urgently needed as nature conservation foundations and other conservation organisations increasingly have funds available to finance conservation measures. Otherwise, less biodiversity than possible is conserved and the available funds are wasted (Ferraro and Pattanayak, 2006).

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Chapter 3

Climate change and the cost-effective governance mode for biodiversity conservation

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99049, posted 18 Mar 2020.

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The chapter includes a reprint of the working paper,
with minor corrections in grammar and spelling, as well
as adapted cross-references and numbering to match the
format of the thesis.

3 Climate change and the cost-effective governance mode for biodiversity conservation

Abstract

Optimal planning of biodiversity conservation and habitat location is paramount for the cost-effective implementation of nature and biodiversity conservation measures.

Established approaches for land use planning and conservation site selection however might not be optimal in a world with changing climatic conditions. Generally, conservation organizations can choose one of two main governance modes: (1) buy land to implement conservation measures themselves on their land, or (2) compensate landowners for their voluntary provision of conservation measures on their land. We analyse in a conceptual ecological-economic simulation four different conservation site selection strategies in either of the two governance modes. Afterwards, we investigate the ecological and economic effectiveness of each governance-mode-strategy combination in a climatically changing environment, and in particular the influence of climate change characteristics. We show that the choice of the two governance modes and four patch selection strategies influences the cost-effectiveness of the implementation, generally suggesting that buying land, combined with the a species targeting patch selection strategy generates the highest cost-effectiveness.

3.1 Introduction

Financial resources for biodiversity conservation projects are scarce. A cost-effective use of these resources – understood as maximising conservation goals for given financial resources or minimising financial resources to achieve given goals – is thus of utmost importance (Ando et al., 1998; Ferraro and Pattanayak, 2006). A growing field of research hence focuses on the cost-effectiveness analysis of biodiversity conservation policies (Ansell et al., 2016; Drechsler, 2017; Wätzold et al., 2016). Examples include studies on the cost-effective selection of habitat types (Petersen et al., 2016) and of land for

conservation in an uncertain environment (Armsworth, 2018), on the cost-effective design of conservation payments (Drechsler et al., 2016, 2017), and on the empirical assessment of conservation contracts (Hily et al., 2015; Schöttker and Santos, 2019).

A novel perspective regarding the cost-effective design of conservation measures is related to the question of governance (Schöttker et al., 2016; Wang et al., 2016). Applying Williamson's analysis of the firm (Williamson, 1998, 1989) to biodiversity conservation, it is of interest how the conservation agency chooses among several alternative governance modes (GMs) representing different levels of vertical integration of conservation measure provision into the agency's organizational structure. Following Schöttker et al. (2016), we assume that conservation agencies in principle have the choice between two GMs: (1) to buy land and implement biodiversity conservation measures on this land themselves, or through delegating the actual implementation to a contractor, e.g. a farmer (*buy alternative*), or (2) to compensate landowners for voluntarily implementing conservation measures on their own land by offsetting implementation costs with a compensation payment (*compensation alternative*).

Literature addresses aspects such as the conceptual analysis of optimal GM choice (Muradian and Rival, 2012), the development of ecological-economic models to assess the cost-effectiveness of different GM (Schöttker et al., 2016), specific conservation settings like forestry and corresponding GM options in developed (Juutinen et al., 2008) and developing countries (Curran et al., 2016), and cost assessments of specific GMs related to conservation projects (Schöttker and Santos, 2019; Schöttker and Wätzold, 2018). These studies suggest a substantial impact of GM choice on the cost-effective implementation of conservation policies.

A key threat to global biodiversity, which has not been discussed in the context of cost-effective GMs, is climate change. According to Thomas et al. (2004) between 15% and 37% of species face a high risk of extinction due to climate change in sampled regions worldwide. Araújo et al. (2011) state that by 2080 58% of currently protected species in Europe will lose suitable habitat. In order to conserve biodiversity, the development of climate change compatible conservation strategies and policies is important (Heller and Zavaleta, 2009; Jones et al., 2016; Reside et al., 2018). However, most research in this field considers the ecological effectiveness of conservation policies (e.g. Zomer et al., 2015), and only a few studies analyse conservation policies from an economic perspective

(Gerling and Wätzold, 2019; Hily et al., 2017; Lewis and Polasky, 2018; Mallory and Ando, 2014); and to our knowledge no study from the perspective of cost-effective GM.

The purpose of this work is to contribute filling this research gap. We analyse the effects of GM choices on biodiversity and conservation costs against the background of variations in climatic conditions. Our background is species conservation in cultural landscapes. This implies that a conservation agency has to provide land with appropriate climate characteristics for a species but also that it has to ensure that specific conservation measures are carried out on that land (for example specific mowing or grazing regimes for endangered grassland birds, Wätzold et al. 2016).

We develop a conceptual, spatially explicit ecological-economic model in a dynamic landscape. We calculate for the considered two GMs the cost-effectiveness of four different implementation strategies under climate change. These strategies include spatial targeting of conservation areas with respect to (a) implementation costs, (b) species abundance, (c) local climatic conditions and (d) climate change direction. The underlying ecological metapopulation model (Hanski, 1999) is used to determine the ecological benefit of the different GMs and site selection strategies.

In a Monte-Carlo simulation, we analyse the different GM options. The impact of varying model parameters is then assessed in sensitivity analysis, climatic characteristics such as spatial climate characteristics and climate change speed.

3.2 The model

3.2.1 Landscape and conservation costs

We assume a landscape with $10 \times 20 = 200$ equally sized, square patches i (Table 3.1 provides an overview of all conceptual variables used in the model and Table 3.2 of all parameter values used in the computation). The landscape has a size of 10 patches in the east-west dimension and 20 patches in the south-north dimension (Fig. 3.1a).

We assume Euclidean distances d_{ij} between the midpoints of patches i and j , i.e. the distance d_{ij} between patches (x_i, y_i) and (x_j, y_j) is $d_{ij} = \sqrt{(x_i - x_j)^2 + (y_i - y_j)^2}$. Without loss of generality, we assume for the eight nearest patches a distance of one, equalling the minimum dispersal distance of the target species.

Each patch in the landscape can potentially serve as a habitat for a target species under two conditions. First, each patch has a certain, time-dependent, climate suitability value, which determines to what degree the target species can find suitable habitat on the patch. Second, conservation measures need to be carried out on a patch i in a specific time-step t ($c_{i,t}^{cons} = 1$). This causes opportunity costs of conservation of OC_i which are assumed to be constant over all time steps. If no conservation measures are carried out ($c_{i,t}^{cons} = 0$) the patch may be used for economic purposes, e.g. intensive agricultural production, and no conservation costs arise.

Conservation costs are spatially heterogeneous and follow a random distribution within a range of $[\overline{OC} + \sigma_{OC}, \overline{OC} - \sigma_{OC}]$, where σ_{OC} is the standard variation and \overline{OC} the mean conservation costs which equals 1.

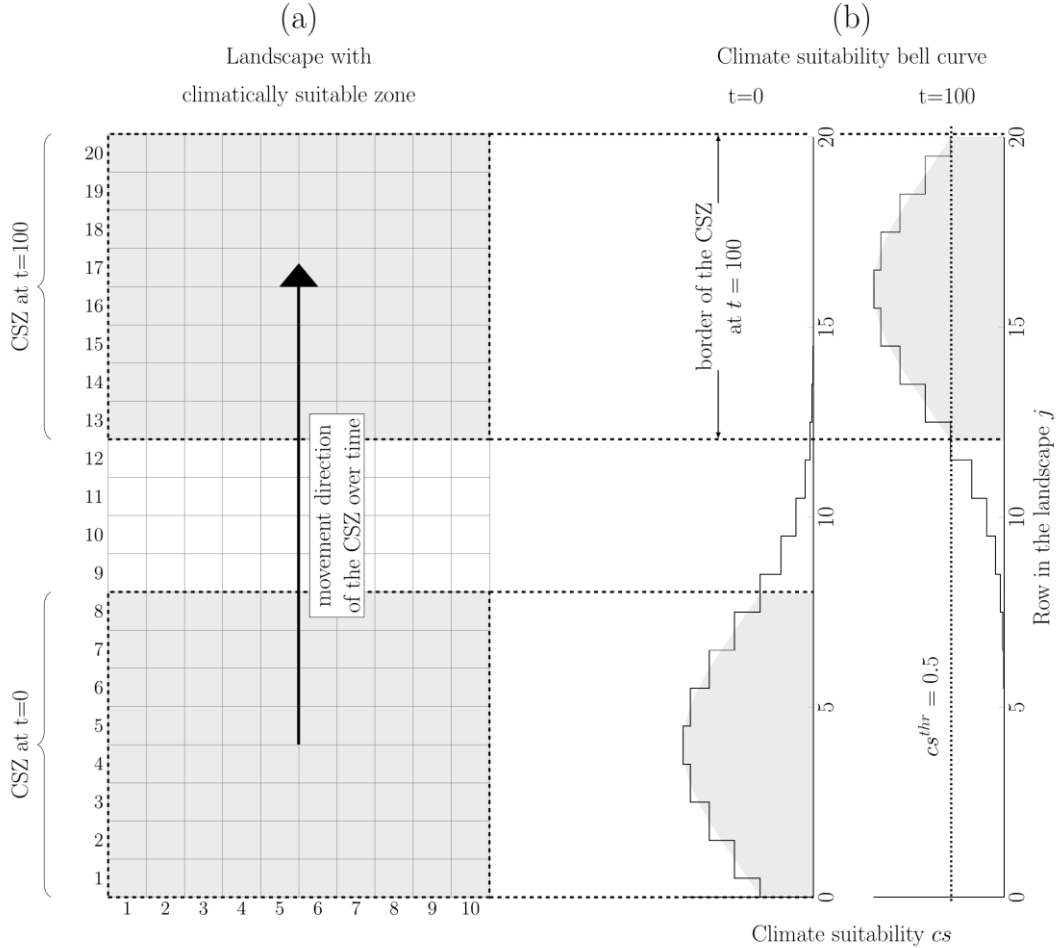


Figure 3.1: (a) Spatially explicit landscape consisting of 10×20 patches including the climatically suitable zone (CSZ, shaded area) at time-steps $t = 0$ and $t = 100$, (b) climate suitability bell curves according to Eq. (1) in their respective base case parametrization (see Table 2) and climate suitability threshold $cs^{thr} = 0.5$, leading to the CSZ at the different time steps $t \in \{0, 100\}$. The shaded area and the corresponding borders represents the CSZ at each given time-step.

Table 3.1: Overview and description of model variables.

<i>Variable name</i>	<i>Variable description</i>
B^{buy}	Budget for purchasing patches
B_t^{buy}	Budget to purchase land within a specific time-step t
B^{comp}	Budget to compensate landowners
B_t^{comp}	Budget to compensate landowners within a specific time-step t
$c_{i,t}^{cons}$	Conservation status of patch i
c_i^{comp}	Total expenses to compensate a single patch i for one time period
c_i^{buy}	Total expenses to buy a patch i
c_i^{sell}	Total amount of money received when selling a patch i
$cs_i(t)$	Climate suitability of patch i at time-step t
d_{ij}	Distance between patches i and j
ε	Residual budget in the <i>compensation alternative</i>
$h_{i,t}$	Dummy variable to indicate if a patch i is colonized at time-step t
$Im_{i,t}$	Immigration rate into patch i at time-step t
K	All patches within the climatically suitable zone
mc_i	Monitoring costs of patch i
OC_i	Opportunity costs of conservation of patch i
p_i^{buy}	Purchasing price of a patch i
$\overline{p^{buy}}$	Mean purchasing price of patches in the landscape
S	Number of all climatically suitable patches that can be reached by dispersal of the target species from already occupied patches
σ_{pbuy}	Standard deviation of purchasing prices
t	Time-step
$\tau_{i,t}$	Colonization probability of patch i at time-step t
tc_i^{buy}	Transaction costs of purchasing a patch i
tc_i^{comp}	Transaction costs to compensate the landowner of patch i
(x_i, y_i)	Coordinates of patch i

3.2.2 Climate change

The modelling of climate change is based on Hily et al. (2017) and we slightly adapted it to fit our simulation model. We assign a climate suitability value $cs_i(t) \in [0,1]$ to each patch in the landscape, representing the probability with which habitat is provided if that patch is under conservation. Over time, the climate suitability of a patch $cs_i(t)$ changes in every time-step t such that

$$cs_i(t) = \exp\left(\frac{-(j - \mu_t)^2}{2 \times \rho^2}\right) \quad (3.1)$$

with $\mu_t = \rho + t \times \frac{j-2 \times \rho}{T}$ being the centre of the climate suitability bell curve at time-step $t \in [1,100]$, ρ an indicator for the bell shapes curvature and j the y-coordinate of patch i . The bell-shaped climate suitability distribution in the landscape moves through the landscape from south to north (Fig. 3.1b).

A patch provides only suitable habitat for a target species, if the climate suitability of a patch at a specific point in time is larger than a threshold value ($cs_i(t) > cs^{thr}$). Due to the general bell shape nature of the climate suitability in the landscape, the introduction of a climate suitability threshold cs^{thr} generates a climatically suitable zone (CSZ), containing all patches in the landscape which are suitable for a target species' habitat. Smaller (larger) values of cs^{thr} generate a larger (smaller) CSZ by allowing the target species to colonize patches with lower (higher) climate suitability and the CA to set respective patches under conservation. The CSZ moves through the landscape from south to north over time, implying that the target species can only survive if it relocates northwards.

3.2.3 Ecological dynamics

We assume the target species to populate the landscape and colonize new patches according to metapopulation dynamics (Hanski, 1999). The occupation of a patch by the target species depends on an immigration rate $Im_{i,t}$ of the species into that patch, an immigration threshold necessary for successful colonization θ , and a resulting colonization probability

$$\tau_{i,t} = \frac{Im_{i,t}^2}{Im_{i,t}^2 + \theta^2} \quad \text{if } cs_i(t) \geq cs^{thr} \text{ and } c_{i,t}^{cons} = 1 \quad (3.2)$$

and 0 otherwise. The immigration rate is defined as

$$Im_{i,t} = \sum_{k=1}^K h_{k,t} v \frac{\exp(-d_{i,k}/\delta)}{S_t} \quad (3.3)$$

with K being the number of all patches within the CSZ in principle available for colonization, $h_{k,t}$ a dummy variable indicating if a patch k is occupied at time t , v the emigration rate from patch k , $d_{i,k}$ the distance between patches i and k , δ the dispersal distance of the target species, and S_t the number of climatically suitable patches in the neighborhood of patch k (the neighborhood of a patch consists of all patches within the dispersal distance of the target species). By migrating from an occupied patch i to an unoccupied patch j , the target species can colonize new habitat over time, while also facing the probability of extinction on already occupied patches. These colonisation and extinction processes generate dynamics in the metapopulation model.

Climatic conditions are updated for each patch in every time step. With a northward shift of CSZ the climate suitability of patches at the southern end of the CSZ falls below the climate suitability threshold cs_{thr} and these patches become unsuitable for the species.

We calculate the overall share of simulation runs in which the target species goes extinct as an indicator for the ecological outcome of our model. Hence, increasing (decreasing) extinction risks reduce (increase) the cost-effectiveness of a selected GM and implementation strategy.

3.2.4 Decision problem of the conservation agency

In order to reach a desired conservation outcome, a conservation agency (CA) implements certain conservation measures in the landscape. The CA chooses between two GMs: (1) buy land and implement conservation measures itself (*buy alternative*), or (2) pay landowners for their voluntary provision (*compensation alternative*) of equally designed conservation measures. For the implementation of conservation measures, the CA has to develop a patch selection strategy (PSS) to decide which patches to conserve. We consider four strategies resulting for each of the two GM resulting in eight GM-PSS pairs. In the following, we first introduce the budget available for covering conservation costs and its allocation over time. We then explain how we model the two GMs and the corresponding budget equations, before we finally describe the four PSS.

3.2.4.1 Budget comparability

The implementation of conservation measures within a certain GM-PSS combination causes costs, which are covered by the agency's budget. For all 8 GM-PSS pairs we assume equal available budgets at the beginning and the end of the simulation to allow comparability of the ecological outcomes and thus be able to assess the relative cost-effectiveness of the GM-PSS pairs.

As the two different GM alternatives generate different cost streams, with high initial costs for buying and relatively high recurring costs for compensation, we assume that the present value (PV) of the two cost-streams has to be equal. The available budgets in each GM-PSS pair and each time-step thus differ and the relation of present values of the respective budgets, $PV(\sum_{t=0}^T B_t^{buy}) = PV(\sum_{t=0}^T B_t^{comp})$, translates into:

$$B^{buy} = \sum_{t=0}^{t=T} B_t^{comp} \times d_t \quad (3.4)$$

$$B_t^{comp} = \frac{-r \times (B^{buy} \times r^T - \varepsilon)}{1 - r^{T+1}} \quad (3.5)$$

with B^{buy} being the budget available for patch purchase, T the length of the total timeframe (i.e. 100 time-steps), r the interest rate, and ε the residual budget at the end of period T (necessary to keep the budgets for the two GMs comparable over the complete timeframe). The whole budget is available at the beginning of time-step $t = 0$ for the *buy alternative*. For the *compensation alternative*, we assume that B_t^{comp} is set so that in each time-step t an equal monetary amount (*compensation annuity*) is available for the CA to be spend, i.e. B_t^{comp} of eq. 3.8 (for a detailed explanation, see Appendix B.4). The CA conserves as many patches as possible for a given budget in a certain period t . Any leftover budget at the end of a period is transferred to the next period and added to the respective budget, including interest.

3.2.4.2 Buy alternative

The *buy alternative* characterizes the CA's option to purchase and consecutively manage patches for conservation. The costs of an individual patch purchase are defined as

$$c_i^{buy} = p_i^{buy} + t c_i^{buy} \quad (3.6)$$

with $p_i^{buy} = \overline{p^{buy}} \pm \sigma_{pbuy}$ being the uniform randomly distributed purchasing price, $\overline{p^{buy}} = \frac{\overline{OC}}{r}$ the mean purchasing price of patches in the landscape, $\sigma_{pbuy} = \sigma_{OC} \times \overline{p^{buy}}$ the standard deviation of purchasing prices, \overline{OC} the mean conservation costs, r the interest rate, σ_{OC} the standard deviation of conservation costs. Transaction costs for purchasing a patch $tc_i^{buy} = \overline{tc^{buy}} \pm \sigma_{tc}$ (such as notary fees, contract negotiation costs, legal counsel) are uniform randomly distributed. For simplicity, we assume that patch prices do not change over time.

The CA is able to purchase new patches as long as the remaining budget is high enough. The CA is not allowed to have negative budgets, i.e. taking loans to fund patch purchase. We assume myopic spending behavior of the CA, thus strategically saving budget for later periods is not allowed. Purchased patches are managed in the prescribed conservation sense. Following Schöttker et al. (2016) we assume, that the costs of managing patches are equal to potential income generated from these measures, hence we need to consider only the costs of purchasing patches in the *buy alternative*.

Depending on the chosen PSS species monitoring costs might occur. These are recurring monitoring costs of $mc_i = \overline{mc} \pm \sigma_{mc}$ per patch in each time-step, with \overline{mc} the mean monitoring costs and σ_{mc} the variation bandwidth. Monitoring costs are initially drawn randomly, like transaction costs, from a uniform distribution (according to \overline{mc} and σ_{mc}) and do not change over time..

After a patch i is purchased it is set under conservation, resulting in habitat generation on this patch, if climatic conditions for the target species on that patch are good enough, i.e. $cs_i(t) \geq cs^{thr}$. Patch purchase then results in $c_{i,t}^{cons} = 1$.

We assume that in all four PSS the agency only purchases patches within the CSZ as $cs_i(t) < cs^{thr}$ for all patches outside the CSZ. We also assume that if an earlier purchased patch after some time falls out of the CSZ due to climate change, the CA sells the respective patch and receives the amount

$$c_i^{sell} = p_i^{sell} - tc_i^{sell}. \quad (3.7)$$

Following from the assumption that purchasing prices do not change over time, the CA receives the same amount from selling a patch as it paid for its acquisition ($p_i^{sell} = p_i^{buy}$).

However, it has to bear the transaction costs, which are assumed to be equal for patch purchase and sale ($tc_i^{sell} = tc_i^{buy}$).

3.2.4.3 Compensation alternative

In the *compensation alternative*, the CA does not purchase areas for conservation, but offers a compensation payment to landowners to incentivize them to implement conservation measures voluntarily (equivalent to the measures in the *buy alternative*) on their land. Compensation payments are spatially homogeneous and are selected such that they equal the opportunity costs oc_i of the landowner who has the highest conservation costs of the participating landowners.

For each patch under conservation, the CA has to pay

$$c_i^{comp} = oc_i + tc_i^{comp} \quad (3.8)$$

in every time period, resulting in a periodical payment subtracted from the budget in each time-step, with tc_i^{comp} the transaction cost for each time-step for setting up and implementing a conservation measure (such as patch finding costs, contract negotiation, etc.).

After a patch is set under conservation ($c_i^{cons} = 1$), it remains in that state for one time-step. In the next time-step, the CA renegotiates conservation contracts. Depending on the PSS, the CA might want to keep certain patches under conservation for more than one time-step, or wants to alter the conservation location according to its priorities (see Section 3.2.4.4).

Comparable to the *buy alternative*, the CA also chooses potential conservation areas only within the CSZ. Hence, $cs_i(t) \geq cs^{thr}$ for all patches under conservation. The periodically renewed conservation decision of the CA results in potentially varying locations of patches under conservation.

3.2.4.4 Patch selection strategies

To implement conservation measures, the CA has to identify suitable patches. We consider four different PSS for this purpose ('price prioritization', 'species abundance prioritization', 'climate suitability prioritization', 'climate change direction prioritization'). The first PSS is motivated purely by cost concerns, whereas PSS 2-4 follows the notion that prioritization of potential habitats based on natural processes and characteristics (here

species abundance and general climate-related suitability of potential habitats) most likely provides a cost-effective conservation strategy (Reside et al., 2019).

(1) ‘Price prioritization’ characterizes a PSS in which the CA prefers cheaper patches over more expensive ones. This translates for the CA, in case of the *buy alternative*, to buy the cheapest available patches in the CSZ. In case of the *compensation alternative*, the patches with the lowest compensation payment requests are added to the conserved patches (Fig. 3.2a). The resulting conservation patches do not necessarily consist of connected patches in which a target species can successfully migrate between patches under conservation, thus potentially inhibiting colonization. However, this PSS will generate the highest number of patches under conservation for a given budget.

(2) For the PSS ‘species abundance prioritization’ the CA only buys or compensates patches, which are within the dispersal distance of colonized patches (Fig. 3.2b). This generates a cluster of conserved patches around existing habitat and leads to connected areas for the target species to colonize. However, as not all patches are available for conservation, more expensive patches might have to be added leading to a lower number of conserved patches than with PSS ‘price prioritization’. Due to the need to identify colonized patches in this PSS, monitoring costs of $mc_i = \overline{mc} \pm \sigma_{mc}$ arise for the CA in each time-step.

(3) We assume that the CA has full information of the climate suitability of all patches in the landscape. The PSS ‘climate suitability prioritization’ prefers patches with a high climate suitability (Fig. 3.2c), specifically, patches in the center of the climate suitability bell curve, as here the climate suitability value is highest. However, if only sufficiently cheap, also more northern or southern patches can be selected, allowing for a spatial spread of the conserved patches over the CSZ. By introducing a scaling factor λ (Eq. 3.7), we are able to foster or loosen this prioritization and thus either allow the CA to almost exclusively focus on the most centered patches (high λ), or to allow a broader spread of patches as (for given climate suitability) less expensive but further away patches are selected (low λ). In order to include costs into this PSS, we introduce the “suitability price” of each patch, which is a non-homogeneous payment, depending on a combination of the climate suitability of a patch and its opportunity costs. The “suitability price” includes both the (normalized) price and the (normalized) climate suitability of that patch as follows:

$$p_i^{suit} = p_i^{norm} + cs_i^{norm}(t) \times \lambda, \quad (3.9)$$

with p_i^{norm} the price of patch i normalized on a scale of 0 to 1 (on which the cheapest patch price in the landscape is 0 and the most expensive price is 1), $cs_i^{norm}(t)$ the normalized climate suitability of patch i and λ the scaling factor. Instead of using only the price for patch selection (as in the PSS ‘price prioritization’), now the suitability price is used as a selection criterion. Obviously, we use the regular price with respect to budgetary calculations.

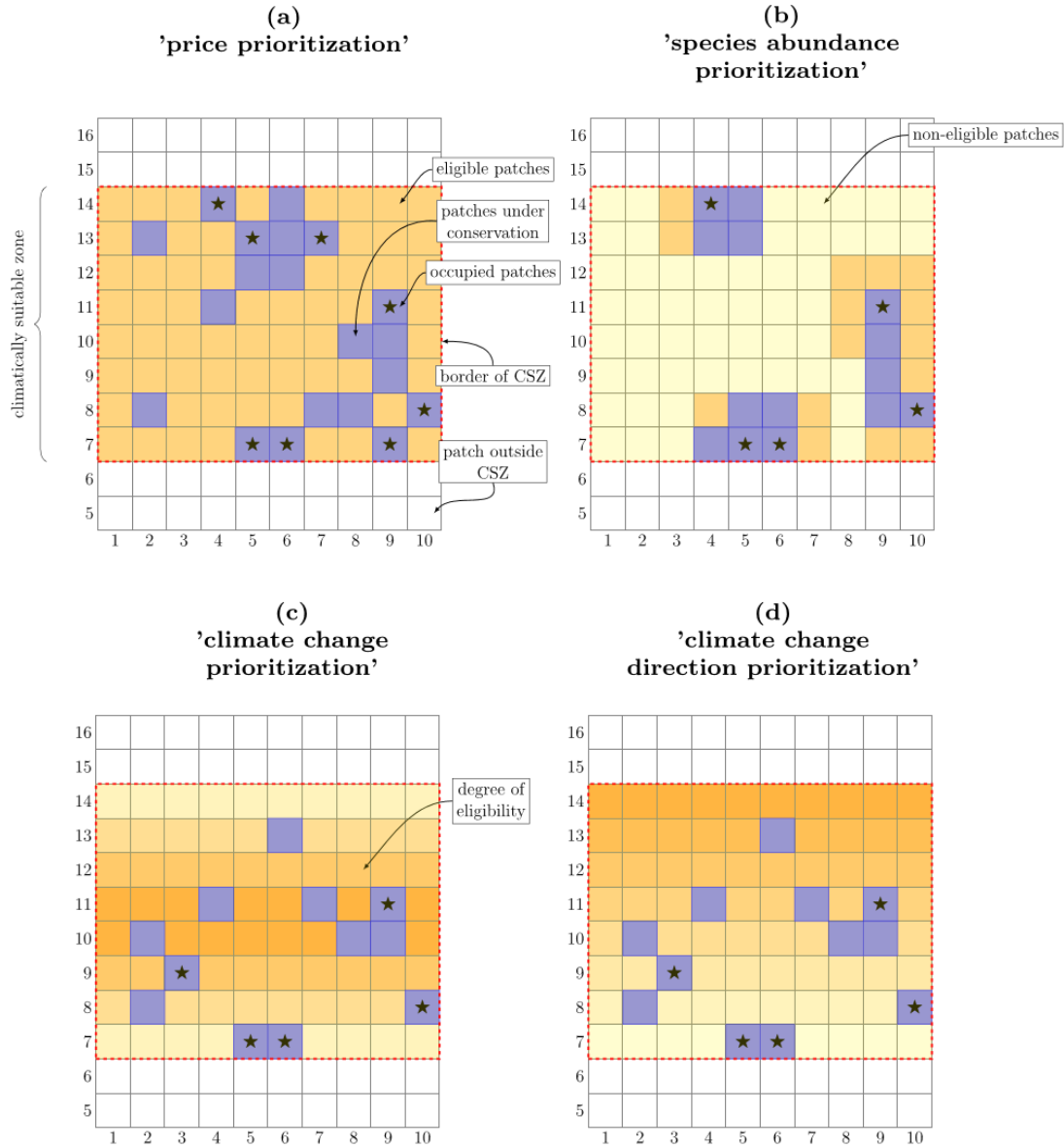


Figure 3.2: Visualization of the four different PSSs and the corresponding patch location. (a) ‘Price prioritization’ allows for patch selection in the complete CSZ, only depending on the purchase price or compensation costs. (b) ‘Species abundance prioritization’ only selects patches within the dispersal distance of already occupied patches. (c) ‘Climate change prioritization’, prefers patches with higher climate suitability over patches with lower climate suitability, and (d) ‘Climate change direction prioritization’, prefers patches at the northern end of the CSZ over patches at the southern end of the CSZ.

(4) Due to the CSZ's movement into the northern direction over time, already selected and colonized patches move to the southern edge of the CSZ. By assuming that the CA has full information on the direction of climate change, we can design a fourth PSS in which the CA prioritizes patches closer to the northern edge of the CSZ (Fig. 3.2d). These patches will, due to the northward movement of the CSZ, stay in the CSZ for a long time with a high possibility of being colonized. The resulting conserved patches are comparable to the ones under the 'climate suitability prioritization', but biased towards northern patches. By introducing a scaling factor κ into this PSS, we can vary the CA's prioritization strength and either allow for a more or less strict patch selection close to the northern edge of the CSZ. Similar to the PSS 'climate suitability prioritization', we calculate a "suitability price" for each patch, which includes both the (normalized) price and the (normalized) climate suitability of that patch and represents a non-homogeneous payment to the individual landowners:

$$p_i^{suit} = p_i^{norm} + csz_i^{row}(t) \times \kappa, \quad (3.10)$$

with p_i^{norm} the price of patch i normalized on a scale of 0 to 1 (on which the cheapest patch price in the landscape is 0 and the most expensive price is 1), and $csz_i^{row}(t)$ the normalized row number in which within the CSZ a certain patch i is located (more northern patches have higher row numbers and thus higher $csz_i^{row}(t)$ leading to the intended prioritization).

3.3 Analysis

For model analysis we apply a Monte-Carlo-simulation, in which each parameter set – i.e. selected combinations of parameters specified in Table 3.2 – is simulated 2000 times to allow an analysis of the whole bandwidth of potential outcomes and to avoid randomly extreme results resulting from the model inherent stochasticity. A simulation run refers to one single calculation of the model for one parameter set.

The parameters cs^{thr} , ρ , m_t , and θ influence the shape of the climate bell curve, and thus have potentially an effect on both GM and all PSS. In contrast, λ and κ affect the prioritization strength of the two climate sensitive PSS, and hence may only influence the outcome of these PSS. The economic parameters \overline{OC} , $\overline{tc^{buy}}$, and \overline{mc} impact the different cost measures, while the interest rate r is used for discounting and budget calculations in

Table 3.2: Overview and description of parameters and parametrization values specified for computation of the Monte-Carlo-Simulation and the sensitivity analysis.

<i>Parameter name</i>	<i>Parameter description</i>	<i>Parametrization Value</i>
i	Patch index	$\in [1,200]$
μ_t	Centre of the climate suitability bell curve at time-step t	1
σ_{OC}	Standard deviation of opportunity \overline{OC}	0.1
σ_{tc}	Standard deviation of transaction costs	0.01
σ_{mc}	Standard deviation of monitoring costs	0.01
θ	Immigration threshold for successful colonization	5

Economic parameters		
\overline{OC}	Mean opportunity costs in the landscape	1.0
$\overline{tc^{buy}}$	Mean transaction costs of purchasing a patch	1.0
\overline{mc}	Mean monitoring costs	0.1
r	Interest rate	0.03

Ecological Parameters		
ν	Emigration rate from any patch	100
δ	Dispersal distance of the target species	1

Climate Parameters		<i>Value Range</i>	<i>Base case</i>
T	Maximum number of time steps	$\in \{50,100,150\}$	100
cs^{thr}	Climate suitability threshold	$\in \{0.3,0.5,0.7\}$	0.5
ρ	Curvature of the climate suitability bell shape	$\in \{2,3,4\}$	2
λ	Scaling factor for PSS ‘climate suitability prioritization’	$\in \{1.5,2.0,4.0\}$	2.0
κ	Scaling factor for PSS ‘climate change direction prioritization’	$\in \{1.5,2.0,2.5\}$	2.0

all GM-PSS pairs. σ_{OC} , σ_{tcbuy} , and σ_{mc} determine the range of all randomly drawn cost parameters in the simulation. The ecological parameters ν and δ influence the dispersal ability of the target species affecting the ecological dynamics in all GM-PSS pairs.

We calculate a reference base case with a respective base case parametrization, which was selected to resemble economic, ecological, and climatic conditions, which allow the model to generate inherently consistent outcomes (see Table 3.2). Afterwards, we individually vary some parameters in specified ranges to values lower and higher than the base case value to identify the impact of each parameter on the cost-effectiveness of each GM-PSS pair (sensitivity analysis).

3.4 Results

We first present the results of the base case parametrization of the eight GM-PSS pairs as it already provides valuable and general insights into the choice of the cost-effective GM. To identify factors influencing the relative performance of the eight GM-PSS pairs, we then present results of a sensitivity analysis in which climatic model parameters are varied individually. The analysis of the results revealed four effects influencing the cost-effectiveness of each GM-PSS pair. Wätzold and Drechsler (2014) have identified already two of the effects – the patch restriction effect and the connectivity effect –, while the remaining two effects – the *climate prioritization effect* and the *flexibility effect* – are newly identified in this work. In particular, the effects are:

- (1) The *patch restriction effect*, which exists as due to the limitation of eligible patches, if connected habitat network requirements or certain climate suitability restrictions are to be met by a specific GM-PSS pair. In these cases, most likely more costly patches are to be selected compared to a situation in which the CA can freely choose patches in the whole CSZ. Therefore, a restriction of eligible patches tends to increase conservation cost and hence to reduce cost-effectiveness.
- (2) The *connectivity effect*, as with improved connectivity of conserved patches, the ecological outcome increases, and hence the cost-effectiveness increases.

- (3) The *climate prioritization effect*, which leads to improved ecological conditions of patches under conservation as they are chosen in climatically more suitable areas within the CSZ.
- (4) The *flexibility effect*, which exists as due to the selected GM, the adaptability of the conservation network (e.g. to changing climatic conditions) can be fast (for the *compensation alternative*) and slow (for the *buy alternative*). This adaptation possibility increases conservation costs, but allows for a flexible selection of suitable patches and hence increases ecological outcome. The net effect depends on the respective GM-PSS pair.

In the following, the influence of changes in climatic parameters on the cost-effectiveness and extinction probabilities of the different GM-PSS pairs is analysed. Results of the influence of ecological and economic parameters are found in Appendix B.4.

3.4.1 Climate change speed

The cost-effectiveness of three GM-PSS pairs was influenced by variations of climate change speed, i.e. variations of the overall simulation timeframe T . A short timeframe (small T) represents fast climate change as it takes less time steps for the climate suitability to vary and the CSZ to move across the landscape (Fig. 3.3).

Generally we find that patch selection in the *compensation alternative* is more flexible compared to the *buy alternative*. Patches can be reselected anew in every time step in the *compensation alternative*, depending on patch price, climate suitability and occupation status, while they are fixed for a longer time (until they are no more in the CSZ) in the *buy alternative* and thus cannot react to changing climatic conditions or occupation status. Hence, a strong *flexibility effect* exists which causes improved ecological outcome in the *compensation alternative* compared to the *buy alternative*.

For the ‘price prioritization’ strategy (Fig. 3.3a), we find that the cost-effectiveness of the *compensation alternative* decreases with increasing climate change speed whereas it remains constant for the *buy alternative*. We explain this result with the combination of a generally reduced ecological suitability of the landscape for the target species due to faster climate change, and the counteracting *flexibility effect*. In the *compensation alternative*, reduced ecological suitability and a strong *patch restriction effect* outperform the *flexibility effect* compared to the *buy alternative*, and hence lead to a reduced cost-effectiveness. In contrast, in the *buy alternative* the stability of the selected conservation network

compensates the negative ecological effects of fast climate change on the cost-effectiveness. Furthermore, prioritizing patches by price generally results in more patches under conservation as cheaper areas are selected, which at the same time are not necessarily well connected

Increasing climate change speed, however, increases the cost-effectiveness of the ‘species abundance prioritization’ strategy for the *buy alternative* in comparison to the *compensation alternative* (Fig. 3.3b). This result is somewhat surprising, as this strategy prioritizes patch selection around already existing habitat and hence allows for easy migration to new nearby habitat. Differences in climate change speed should not interfere with this effect. An explanation may be that the *connectivity effect* is increasingly relevant with increasing climate change speeds, which also would explain, why the other strategies result in increasing extinction rates, as there the connectivity effect is less pronounced. Against the background of more volatile conservation networks in the *compensation alternative*, it is however unclear why no cost-effectiveness reduction can be observed in the ‘species abundance prioritization’ strategy in the *compensation alternative*.

We do not observe any influence of changing climate change speed on the extinction probability and hence cost-effectiveness in the ‘climate suitability prioritization’ strategy in any of the two GMs (Fig 3.3c). A possible explanation is that conserved patches are located in well-functioning conservation networks in case of the *buy alternative*, or adapt quickly enough to location changes of the CSZ in case of the *compensation alternative*, so that eventually extinction rates are not affected.

Moreover, we find faster climate change speed increases extinction rates in the ‘climate change direction prioritization’ strategy for the *buy alternative*, and hence a decrease in cost-effectiveness of the GM-PSS pair compared to the *compensation alternative* (Fig 3.3d). This result is expected as the period when patches are located inside the CSZ is reduced with a shorter timeframe. This is especially true for this strategy, which prioritizes patches at the northern-edge of the CSZ that stay in the CSZ longer compared to other PSS. Within the *compensation alternative*, for every parameter setting the survival rates are at 100%, indicating a strong *flexibility effect*, which leads to increased survival rates.

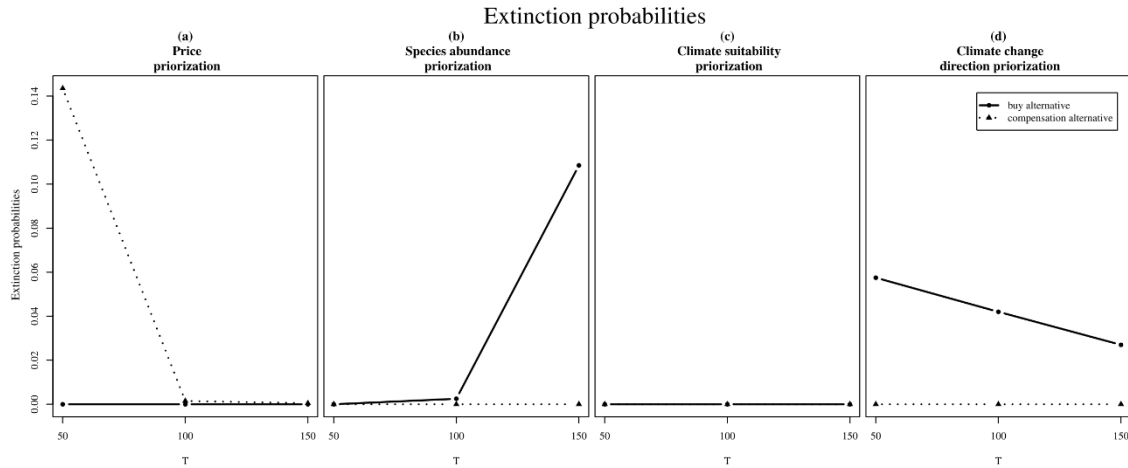


Figure 3.3: Changes in extinction rates due to changes in maximum length of the simulation timeframe T , (i.e. climate change speed decreases with increasing T). (a)-(d) represent the extinction probabilities for all four strategies in the buy and compensation alternative.

3.4.2 Strength of climate prioritization

Within the PSSs ‘climate suitability prioritization’ and ‘climate change direction prioritization’, patch selection takes place according to either climate suitability or climate change direction. We introduced a scaling factor λ for each strategy to define the strength of prioritization of respective patches. A higher λ (κ) results in a stronger prioritization for climate suitability (climate change direction) relative to patch prices. Thus, increases in either parameter generate a *patch restriction effect* and *climate prioritization effect* by narrowing the spatial extent of the conserved patches in the respective strategies. Changes in λ only affect the ‘climate suitability prioritization strategy’, and changes in κ only affect the ‘climate direction prioritization strategy’. The ‘prize prioritization strategy’ and the ‘species abundance prioritization strategy’ remain unaffected, as both parameters do not alter their respective patch selection mechanism.

We did not find any influence of the climate suitability scaling factor λ , neither in the *buy* nor the *compensation alternative* within our parameter range (compare graphical analysis in Appendix B.4). This is somewhat surprising as increasing values of λ cause a prioritization of patch selection in the center of the CSZ, and hence have a *patch restriction effect* and *climate prioritization effect*. Both effects are probably cancelling each other out in their influence on the cost-effectiveness.

However, changes in κ do show an influence on the cost-effectiveness of the ‘climate change direction prioritization strategy’ for the *buy alternative*, while the *compensation*

alternative remains unaffected (see Fig. 3.4d). Low values of κ (low prioritization for climate change direction) result in an increased cost-effectiveness compared to larger values of κ due to the high *connectivity effect*. With increasing κ newly added patches are predominantly located in the most northern part of the CSZ while large portions of the CSZ remain unconsidered for selection. Hence, conserved patches are spread far across the complete CSZ, resulting in large distances between conserved patches and leading to an increase in extinction probability with increasing κ , and hence a reduction of cost-effectiveness. For lower values of κ however, new patches are selected in a larger proportion of the landscape, hence are more likely located closer to already occupied patches, which results in better migration possibilities and increased cost-effectiveness. Also, with larger (smaller) parts of the CSZ eligible for patch selection with smaller (larger) values of κ , the *patch restriction effect* becomes weaker (stronger), hence also increasing (decreasing) the cost-effectiveness.

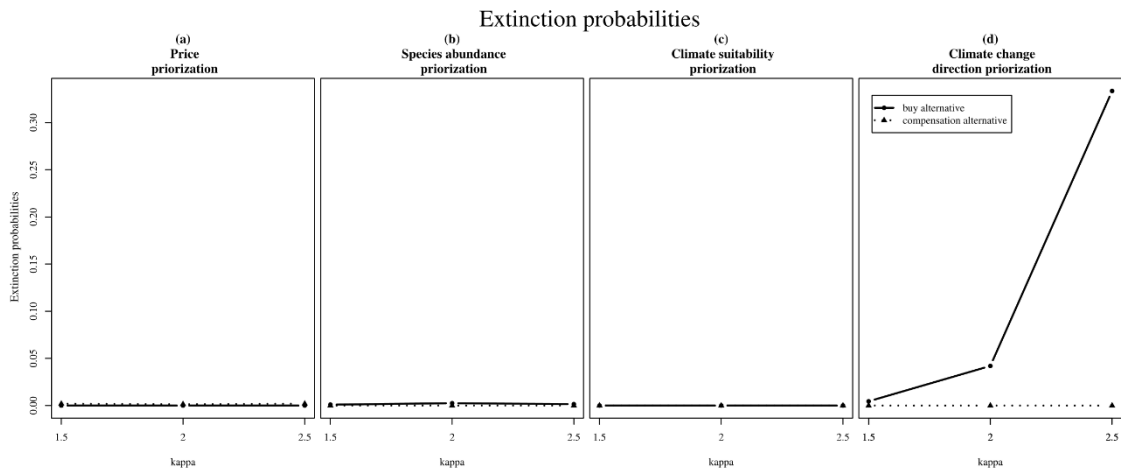


Figure 3.4: Influence of changes in κ on the extinction probability in the buy alternative and the compensation alternative for each of the four PSSs.

We did not find any influence on the *compensation alternative* strategies by variations in κ . A possible explanation is the interplay between *connectivity effect* and *flexibility effect* in either alternative. Due to repeated reselection of new patches in the prioritized area they are relatively well connected in the *compensation alternative*, compared to the wide spatial spread in the *buy alternative*, leading to relatively good migration possibilities and hence a better cost-effectiveness of the *compensation alternative*, compared to the *buy alternative*.

3.4.3 Climate suitability threshold

Changes in the climate suitability threshold value cs^{thr} influence the cost-effectiveness of four GM-PSS pairs (Fig. 3.5). The value of the climate suitability threshold cs^{thr} determines the width of the CSZ and hence has potentially an effect due to the *connectivity* and *patch restriction effects*. Generally speaking, with an increasing CSZ (low cs^{thr}) the *connectivity effect* weakens, while the *patch restriction effect* is decreasing for all GM-PSS pairs.

We find with increasing CSZ size (decreasing cs^{thr}) for both GMs in the ‘price prioritization strategy’ (Fig 5a) a decrease in cost-effectiveness suggesting that the *patch restriction effect* dominates the *connectivity effect*. However, the reduction of cost-effectiveness in the *compensation alternative* is stronger than in the *buy alternative*. This effect may be explained as the size of the CSZ and hence the number of eligible patches increases with decreasing cs^{thr} . For the *compensation alternative*, more volatile patch selection (compared to the *buy alternative*) causes frequent changes of habitat location (potentially every period), and hence reduces migration possibilities as selected patches are potentially far apart. This effect is especially prominent in the ‘price prioritization strategy’ as patches are purely selected based on compensation costs and hence will be selected randomly across the whole CSZ. In other strategies (see details below), patch selection is restricted to a more narrow area within the CSZ, leading to a more compact conservation network and hence decreased extinction probabilities, compared to the ‘price prioritization strategy’.

In the ‘species abundance prioritization strategy’ however, only the cost-effectiveness of the *buy alternative* decreases with increasing cs^{thr} (see Fig. 3.5b). A decreasing size of the CSZ due to increasing cs^{thr} limits the CA to purchase patches nearby already occupied patches. If a CA wants to select further patches within this strategy, it would be necessary to select patches outside of the dispersal distance of the target species and which thus could not be colonized in the current time step (though they would still be connected to the habitat network). Hence, a decreasing *connectivity effect* causes a reduction of the cost-effectiveness in the ‘species abundance prioritization strategy’ in the *buy alternative*. A higher *flexibility effect* in the *compensation alternative* positively contributes to the cost-effectiveness compared to the *buy alternative*. This impact is not present in the *buy alternative*.

In the ‘climate suitability prioritization strategy’, no negative effect occurs with a decreasing climate threshold on the cost-effectiveness of both GMs within the chosen parametrization range (Fig 3.5c). Our explanation is that the *patch restriction* and *connectivity effect* cancel each other out.

A strong negative effect on cost-effectiveness can be observed for small values of cs^{thr} (large CSZ), in the ‘climate change direction prioritization strategy’ (Fig. 3.5d) for the *buy alternative*, while no effect can be seen in the *compensation alternative*. This may again be explained by a combination of the *connectivity effect* and the *climate prioritization effect*. While patches remain under conservation in the *buy alternative* as long as they are located within the CSZ, this duration grows, as well as the resulting gaps between conserved areas and unoccupied patches, with a decrease in cs^{thr} , eventually leading to a lower cost-effectiveness. In the *compensation alternative*, patches under conservation in contrast might be reselected anew if they are unoccupied at the end of the time step. Reselection then happens in the northern part of the CSZ, automatically locating newly added patches close to other patches in the conservation network, hence leading to low extinction probabilities and high cost-effectiveness.

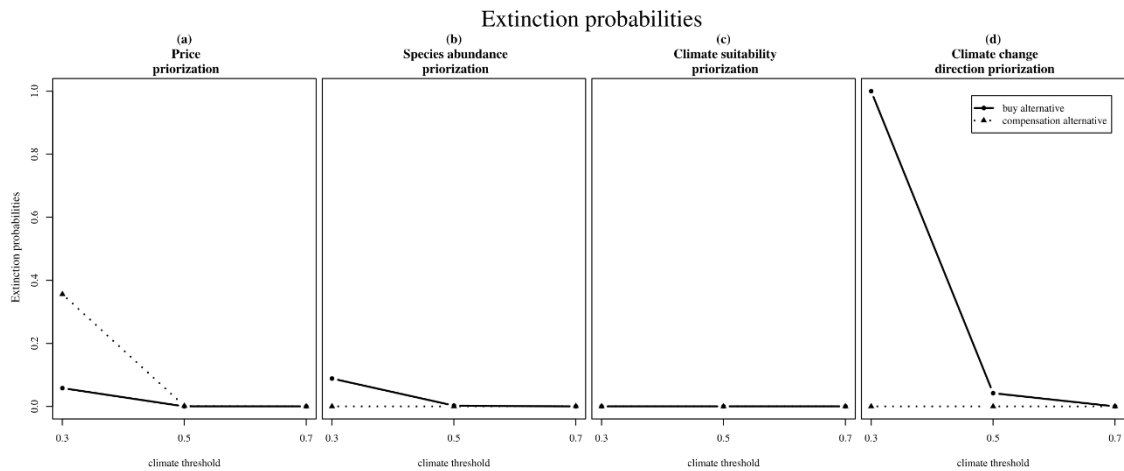


Figure 3.5: Extinction probabilities of the different GM-PSS pairs with changing climate threshold, resulting in changing CSZ sizes.

Generally, *compensation alternative* strategies perform well even with large CSZs because of the *flexibility effect* with the exception of the ‘price prioritization strategy’, in which the *flexibility effect* is counteracted by a small *connectivity effect*.

3.4.4 Shape of the climate suitability bell curve

We only find small effects of variations in ρ (influencing the curvature of the climate suitability bell shape) on the cost-effectiveness of the GM-PSS pairs. A possible reason might be that the climate suitability bell shape determines the climate suitability in the complete landscape, while only a relatively narrow strip around the center (which the CSZ covers) is actually eligible for patch selection. Because changes in the curvature of the bell shape are not necessarily very strong within the CSZ and only have marginal effects on CSZ size, the effects on GM-PSS pairs performances is negligible (compare graphical analysis in Appendix B.4).

3.5 Summary and discussion

The purpose of this paper was to analyse with a conceptual model the impact of changes in climate parameters on the cost-effectiveness of different governance modes (GM) and specific implementation strategies (PSS). We assume that conservation agencies (CA) have two alternative GM to select. (1) Buy conservation areas and implement conservation activities on this land (*buy alternative*), and compensate private landowners for their voluntary provision of conservation measures on their own land (*compensation alternative*). We further assume that the CA chooses from four PSS. (1) Select the cheapest patches in the landscape ('price prioritization'), (2) select patches close to areas already populated by a target species ('species abundance prioritization'), (3) select patches with highest climate suitability ('climate suitability prioritization'), and (4) select patches which remain climatically suitable for the longest time ('climate change direction prioritization').

We wish to highlight the following two general key insights. First, buying areas for conservation produces a relatively rigid spatial selection of conserved patches due to the long-term commitment for certain conservation areas within the landscape. While more rigid patch location improves the ecological effectiveness by e.g. reducing habitat turnover it does not allow swift adaptation to changing climatic conditions. In contrast, the *compensation alternative* is more flexible, i.e. patches are potentially changing their conservation status more often as compensation contracts are typically only valid for short time periods (cp. also Gerling and Wätzold, 2019). More specifically, differences in flexibility result in a higher possibility of the *compensation alternative* to adapt to

changing conditions and thus being a more robust choice against uncertain and changing climatic conditions than the *buy alternative*.

Second, we find that against the presence of changing climatic conditions, the cost-effectiveness of GM strongly depends on the choice of the PSS. In this context, to buy conservation areas yields a higher cost-effectiveness against changing climatic conditions when focusing on the cheapest available conservation sites (i.e. applying the ‘prize prioritization strategy’), while private landowner compensation seems to be more cost-effective with more specific PSS (i.e. the ‘species abundance prioritization’, ‘climate suitability prioritization’ or ‘climate change direction prioritization’ strategy). While purchasing areas for conservation typically generates high up-front and one-off costs, the resulting areas should stay under conservation for as long as possible. Prioritizing cheaper patches then allows for an increase in total conservation areas as more patches can be selected, which in turn improves the ecological outcome and increases cost-effectiveness. Given the advantages in terms of flexibility of the *compensation alternative*, a more specific site selection by prioritizing either ecological or climatic characteristics has a stronger influence than in the *buy alternative* and, hence, price prioritization is comparatively less relevant.

In designing the ecological-economic model, we made several simplifying assumptions, which deserve discussion. We only considered two GMs, which are polar types of governance structures and ignored hybrid GMs. For example, a CA might split its budget and spend part of it to buy areas and the rest on compensation contracts with landowners. By doing so, benefits of both GMs might be combined (e.g. fixed location of purchased patches with ecologically beneficial effects, and flexibility of compensated areas with fast adaptability to changing climatic conditions). However, to what extent this happens and what other effects occur is a matter of further research.

We further assumed that landowners are willing to sell their land or take part in compensation contracts as long as the monetary benefits from participation exceed the costs. Some authors question the assumption that landowners are always willing to sell their land and suggest strategies to optimally time the purchase of land for reserves in insecure ecological and economic conditions (Costello and Polasky, 2004; McDonald-Madden et al., 2008), and with changing land prices (Dissanayake and Önal, 2011). Moreover, literature suggests factors which influence the general willingness to participate

in compensation schemes (e.g. contract duration and flexibility, land productivity, and farm size; cp. Greiner (2016) and Unay-Gailhard and Bojnec (2016)), and indicate that the willingness to participate may also be reversed due to e.g. cost-related learning effects (Fronzel et al., 2012). A reduced willingness to participate would directly increase the costs of conservation projects, as more costly areas would have to be chosen. In addition, the ecological effectiveness might be reduced, as less suitable patches might have to be selected or due to increased habitat turnover (cp. Schöttker et al., 2016). Ultimately, both effects negatively influence the cost-effectiveness of the corresponding GM. However, more research is required to understand which GM is likely to suffer from higher cost-effectiveness losses of modified assumptions on landowners' behaviour.

We also assumed that conservation costs in the landscape are constant over the complete timeframe and unaffected by the CA's behaviour. By assuming constant costs we ignore any kind of strategic behaviour, for example from landowners by overstating conservation costs to achieve higher payments or a higher price if they intend to sell their land (Banerjee et al., 2016; Gerling and Wätzold, 2019; Kuhfuss et al., 2016). A strategic overstatement of conservation costs could increase patch prices in both GMs, in turn reducing their cost-effectiveness. Further research is necessary to understand which GM is more prone to strategic behaviour and how to design possible mechanisms to reduce it.

We further assumed that the CA is allowed to sell patches in the *buy alternative*, as patches which are no longer in the CSZ for a specific species do not provide any more suitable habitat for this species. Thus, the potentially regained budget by selling these patches can be utilized to purchase new patches at more suitable locations. It has to be mentioned however that selling conserved land may not be possible in reality for a CA due to legal restrictions regarding the permanence of conservation areas (Schöttker and Wätzold, 2018).

The conceptual nature of our model limits the possibility for direct policy implications of our results. Nevertheless, our model improves the general understanding of the influence of climate change on the cost-effective choice of GMs for biodiversity conservation. We show that the cost-effectiveness of GMs and PSS may be influenced by changing climatic conditions and thus policy makers are advised to explicitly include climate change concerns in their design. The availability of respective conservation strategies to allow for specific targeting of species or climatic conditions is important in this context.

In addition, the more flexible or more rigid character of conservation networks due to different GMs and the resulting implications on cost-effectiveness should be accounted for in the decision about the optimal GM choice. Similarly, dependent on climatic characteristics, the optimal choice for CA may vary, as may the optimal choice of PSS.

Further research may investigate the topic of this work with more empirical data in real landscapes. Climate models are able to provide precise estimations about future climate developments on a regional level, species-specific ecological models are able to assess the impacts of conservation measures in a changing climate and the development of scenarios about future costs is feasible. Such models and data may be combined in empirical climate-ecological-economic models providing policy makers with important recommendations about cost-effective GM and PSS choices. We hope our model motivates such future work and provides a useful basis for it.

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Chapter 4

Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area

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The chapter includes a reprint of the published version, with minor corrections in grammar and spelling, as well as adapted cross-references and numbering to match the format of the thesis.

4 Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area

Abstract

Cost-effective implementation of measures to conserve biodiversity is often a major target of conservation organisations, and choosing the correct mode of governance can be important in this context. Nature conservation organisations can, in principle, choose between two distinct modes of governance to implement conservation activities: they can (1) buy desired areas of interest and implement conservation measures themselves (buy option), or (2) offer payments to landowners to incentivize them to voluntarily preserve or create habitat on their land (compensation option). In this paper we analyse the cost-effectiveness of these two modes of governance in a case study on a conservation project in a Natura 2000 area in Schleswig–Holstein, Germany. The actual costs of the buy option are compared with the potential costs of implementing the compensation option. We developed a costing framework to compare the costs of both options over time, given they generate the same ecological results on an identical project area. We find that the cost-effective solution depends, among other things, on the conservation timeframe considered and on cost components such as transaction costs, leasehold rent and land prices.

4.1 Introduction

Conservation funds are scarce. Conservation agencies therefore need to use their funds cost-effectively, i.e., they must select conservation options which maximise the achievement of their conservation goals with the financial resources available (Birner and Wittmer, 2004; Ferraro and Pattanayak, 2006; Naidoo and Ricketts, 2006; Wätzold and Schwerdtner, 2005). Suggestions for improving the cost-effectiveness of conservation policy instruments have hitherto focussed mainly on improving the spatial allocation of conservation measures. Examples of this type of research include Polasky et al. (2008), Van Teeffelen et al. (2012), and Duke et al. (2013), for conservation planning, and

Armsworth et al. (2012); Wätzold et al. (2016); Wätzold and Drechsler (2014) for conservation payments. Other research compared the cost-effectiveness of a proactive conservation policy with a policy that only sets in when a species is nearly extinct (Drechsler et al., 2011), the cost-effectiveness of integrating borrowing and budget carry-over in land acquisition strategies by conservation organisations (Lennox et al., 2017), the cost-effectiveness of private versus public conservation policies (Gordon et al., 2011), the participation of private versus public landowners in conservation contracts (Hily et al., 2015), the impacts of land acquisition on represented land-cover types (Santos et al., 2014a), and the influence of land tenure security on conservation outcomes (Robinson et al., 2017).

A hitherto neglected area of research to enhance the cost-effectiveness of conservation policy instruments is the choice of the relevant mode of governance for an area of conservation interest and of the management options implemented on that area (Juutinen et al., 2008; Muradian and Rival, 2012; Schöttker et al., 2016). In this context it is important to ask whether it is more cost-effective for nature conservation agencies to (1) buy desired areas of interest and implement conservation measures themselves or through closely monitored organisation (buy option), or (2) offer payments to landowners to incentivize them to voluntarily preserve or create habitat on their land (compensation option) (Curran et al., 2016; Schöttker et al., 2016).

This question is related to the “make-or-buy decision” originally investigated in the context of the theory of the firm and the field of transaction cost economics (Coase, 1937; Williamson, 1975). The general make-or-buy decision addresses the question of whether firms should produce some of their (intermediate) products themselves, i.e., internal provision, or instead purchase the same product from another company, i.e., external provision. In the context of nature conservation, internal provision can be seen as the management of nature conservation sites through nature conservation agencies on land that was originally bought by the agency and by measures performed by the agency itself or closely monitored by it. External provision on the other hand is equal to a provision of the same conservation outcome by the same conservation measures only performed by a firm or an individual outside the conservation agency but contracted and paid by the agency (compare Klein, 2008). Research related to the make-or-buy decision in nature conservation is rare. Schöttker et al. (2016) apply an ecological-economic model to analyse how ecological and economic parameters of the decision problem influence the cost-

effectiveness ranking of the two options. Juutinen et al. (2008) compare costs for forest conservation through conservation contracts and land purchase in Finland, and Curran et al. (2016) compare costs of hypothetical land purchase and costs of a payments for ecosystem services (PES) scheme for forest conservation in Central Kenya.

Here, we aim to contribute to this small but growing research area and present a further empirical study on the make-or-buy decision in nature conservation. We analyse the cost-effectiveness of the buy option compared with the compensation option for the conservation of an oligotrophic lake in a Natura 2000 area in Schleswig–Holstein, Germany. The decision between the two governance modes of buying and compensation is equivalent to the make-or-buy decision context as the agency has to choose between internal provision – buying, equivalent to internal provision of products in the make-or-buy decision context – or external provision – i.e. compensation, equivalent to external provision. We calculate the actual costs incurred for buying land at the conservation site and managing it for the purpose of reducing nutrient input from the surrounding agricultural areas from 1980 until 2015. We then compare these costs with the hypothetical costs of compensating farmers for applying the same management, which enables us to derive the cost-effective solution. Our study differs from those of Juutinen et al. (2008) and Curran et al. (2016) in several ways. We analyse conservation on agricultural land whereas they focus on forest conservation. This implies for example that forest management leads to a more long-term costing framework due to the long-term growth of forest ecosystems (Zabel et al., 2018), while our research considers a rather short-term costing framework due to the annual decision of farmers how to manage their grassland. Moreover, we carry out an ex post analysis whereas their studies are of a prospective nature. This enables us to assess the impact of real world events on the cost-effectiveness of the two options such as the shift to the production of energy crops in Germany (Dauber and Miyake, 2016).

4.2 Materials and methods

4.2.1 Case study description

4.2.1.1 Conservation problem

The conserved area around Lake Bültsee covers approximately 71 ha in the federal state of Schleswig–Holstein, Germany (Fig. 4.1). It consists of the approximately 20 ha sized lake

and a surrounding grassland area of approximately 51 ha (Kämmer, 2002). The area is located about 35 km northwest of the federal state's capital, Kiel, and close to Eckernförde Bay. It was purchased gradually from 1980 to 2011 to establish a conservation site at and around the lake (for a detailed overview of the land purchases, see Table 1 and Appendix C.1). Lake Bültsee and the surrounding littoral zone was declared a nature conservation site in 1982. It is embedded in the German Natura 2000 network “Naturpark Schlei” and the FFH area “Großer Schnaaper See, Bültsee und anschließende Flächen” (FFH DE 1524-391).



Figure 4.1: Lake Bültsee conservation area in Schleswig-Holstein, Germany, with its different terrain and grassland types. The map is based on ArcGIS map data Europe NUTS 1, management plan data for the FFH area “Großer Schnaaper See, Bültsee und anschließende Flächen”, and data from SNSH.

Lake Bültsee is a kettle hole, i.e., a shallow waterbody formed by retreating glaciers during the Weichselian glacial period and filled with sediment. It is an oligotrophic, i.e., nutrient-poor, lake. The surrounding area consists of sandy soils, which strongly determine the nutritional supplement and supply of the lake. Past farming activities, especially intensive farming in the twentieth century, led to a strong increase in the nutrient supply—i.e., the concentration of NO_x and other fertilizer-induced nutrients. This transformed the lake from poor to medium nutrient levels (mesotrophic). The lake is surrounded by fields used for agriculture, which add a strong nutrient supply to the lake by surface water drainage and pollution of groundwater by fertilizers.

The main conservation target is to re-establish an oligotrophic lake with a corresponding surrounding ecosystem of specially adapted fauna, e.g. the water lobelia (*Lobelia dortmanna*), quillworts (*Isoetes lacustris*) or the European shore-weed (*Littorella uniflora*),

Table 4.1: Buying costs and side costs, i.e. notary fees, taxes, and measurement costs, of the implementation of Lake Bültsee conservation project (Euro values are given in nominal terms; deviations in the cumulative values due to rounding errors). In 1985, a marginal section of the conservation area was ceded for the implementation of an infrastructure project, resulting in a decrease in area, with the mentioned sale revenues and side costs. In 1990, parts of Lake Bültsee conservation area were sold, while in return a different conservation area was extended. This exchange resulted in the given positive buying and side costs.

	Buying costs	Side costs	Area in ha
1980	207,073 €	56,314 €	17.6
1981	229,600 €	49,873 €	23.7
1985	-1,239 €	8,028 €	-0.05
1990	9,514 €	1,520 €	-1.2
2000	68,002 €	8,591 €	6.3
2011	83,101 €	19,042 €	4.3
	596,051 €	143,365 €	
	739,416 €		50.65 ha

and a generally nutrient-poor regime of the surrounding dry grassland. The first two species are protected under the German Federal Nature Conservation Act (BNatSchG) as endangered native species, while the third species is not protected but considered endangered. Since 1996, an extensive cattle grazing regime was implemented on the area under conservation. A herd of Galloway cattle grazes on the grassland and littoral zone, thereby effectively reducing unwanted growth of vegetation around the lake and thus improving the growth of target species in the littoral zone through reduced nutrient intake into the lake. The cattle graze throughout the year, without being fed additionally and without additional external supply of fertilizers. In 1995, as an initial conservation measure, willow and alder trees were removed. In 1996, this was repeated and additionally, any excessive growth of vegetation along the southern shoreline was mowed or kept low with the help of sheep grazing to allow the target fauna to grow unimpeded. In 1996, the management of the littoral grassland was changed from extensive sheep grazing to extensive cattle grazing (Kämmer, 2002).

4.2.1.2 Conservation actors

The conservation project is implemented by a government-funded but independent nature conservation foundation, Stiftung Naturschutz Schleswig-Holstein (SNSH), which buys and administratively manages land for nature conservation purposes in the federal state of

Schleswig–Holstein, Germany. SNSH was founded in 1978 with the goal of managing areas used for agriculture or forestry in order to establish environmental or biodiversity protection. For this purpose, SNSH leases or buys areas at the public land-market. These areas are then withdrawn from their original use and transferred into (permanent) conservation sites. SNSH owns a total of over 35,000 ha in Schleswig–Holstein, of which our case study area represents only a small, but one of the oldest fractions (Stiftung Naturschutz Schleswig Holstein, 2015).

The extensive grazing management is operated by a contractor, Bunde Wischen e.V. (BW). BW is a registered association founded in 1986 in the context of a local project for orchid protection through the implementation of extensive grassland measures. BW manages over 700 ha of agricultural land in Schleswig–Holstein for the purpose of organic farming and nature conservation (Kämmer, 2002). BW leased the land around Lake Bültsee from SNSH and implements the measures prescribed by SNSH. BW does not receive compensation payments from SNSH for implementing the prescribed measure, however it qualifies for AES funding for extensive grassland measures. BW keeps the economic profit generated on the area. Only in recent years, leasehold payments are paid from BW to SNSH.

4.2.2 Data collection

Together with SNSH and BW, data were gathered on the actual costs and financial outlays regarding the purchase and management of the conservation area at Lake Bültsee conservation site. The data contains detailed information on buying costs and side costs, i.e., notary fees, taxes, and measurement costs, from 1980 to 2011. Third party data was used to estimate the costs of the hypothetical compensation option, i.e., the profitability of agricultural land under intensive and extensive management and the resulting compensation payment. A detailed overview about internal and third party data can be found in Appendices C.1 and C.4. A literature research revealed further data on transaction costs, land prices and interest and discount rate estimates. Some data gaps, especially in the profitability datasets, were filled using German consumer price index-based interpolation. An overview of the relevant data sources can be found in Appendix C.11 in Table C.5. Data was easily accessible and all data sources are reliable.

4.2.3 Costs of the buy option

4.2.3.1 Cost function

The total costs of the project from year t_{start} to a given end-year T , expressed in values of the year t_{start} , can be calculated with Eqs. (3.1a) and (3.1b):

$$C_{t_{start}}^{buy} = \left(\sum_{t=t_{start}}^T d_t (p_t + \phi_t + m_t + a_t - l_t) \right) + (-v_T^{ext} + s_T) \times d_t \quad (4.1a)$$

with

$$d = \begin{cases} 1 & \text{for } t = t_{start} \\ (1 + i_{t-1})^{-1} \times d_{t-1} & \forall t_{start} < t < T \end{cases} \quad (4.1b)$$

where d_t is the discount factor for the year t , p_t the sum of the purchasing prices of all parcels of land bought, ϕ_t the side costs of purchasing the parcels, m_t the annual agricultural costs of managing the purchased parcels in a desired way, a_t the administrative management costs, l_t the leasehold income gathered from third party contractors, all in year t . The value of conserved land is v_T^{ext} , $s_T = \sum_{t=t_{start}}^T \phi_t \times d_t$ the side costs of selling the land, both in year T , and i_t the real market interest rate (based on the yield of German government bonds with longer than 4 years maturity).

We assume that the extensive grassland value v_T^{ext} is proportional to the intensive agricultural land value v_T^{int} (Ciaian et al., 2010; USDA, 2015) and thus

$$v_T^{ext} = \theta \times v_T^{int} \quad \text{with } \theta \leq 1 \quad (4.2)$$

To calculate the costs of the buy option, we subtract the value of the extensive grassland in year T from the cumulative costs of buying the land in the respective timeframe and add selling side costs s_T (see Eq. 4.1a). This step is necessary to establish comparability with the hypothetical compensation scheme in which land is reused for possible intensive agriculture after a conservation contract terminates.

4.2.3.2 Actual costs of land acquisition and management

The costs of land acquisition including side costs from 1980 until 2011 are provided in Table 4.1. The annual administrative costs of managing the conservation site are relatively small. Furthermore, they cannot be accounted for directly, as the relevant employees are responsible for multiple projects. SNSH estimates annual expenses for personnel of 970 € and travel costs of 72 €.

By giving the area as a leasehold to a third party contractor, SNSH was able to generate a leasehold income. From 1980 to 2008, the annual rent was zero, as it was considered that a rent would make the extensive management of the area unprofitable (cp. Mewes et al. (2015) for costs of extensive grassland management). From 2009 onwards a rent of 40 € per ha and year was charged (Table 4.2).

Table 4.2: Annual management costs and rental income.

	Administrative costs	Leasehold rent	
	per year	total	per ha
1980-2008	1,042 €	0 €	0 €
2009-2010	1,042 €	1,853 €	40 €
2011-2015	1,042 €	2,026 €	40 €

From 1980 to 2015, the price of agricultural land has fluctuated strongly not only in Schleswig–Holstein but all over Germany. While in 1980 one hectare of agricultural land in the study region cost 14,240 €, the price dropped to 7770 € in 1993 (Statistikamt Nord, 2015a). Since 2006, a sharp increase in prices for agricultural land in Schleswig–Holstein can be observed which is due in particular to the increasing cultivation of energy crops such as maize (Dauber and Miyake, 2016; Lupp et al., 2014). This has resulted in an increase in prices for agricultural land to 27,500 € per hectare in 2015 (additional data are given in Appendix C.2).

Based on the average agricultural land prices in Schleswig–Holstein, we estimate the land value of the conservation site around Lake Bültsee for each year. We assume that the land can be sold and reused for agricultural purposes to recoup the initial monetary outlay.³

Following Eqs. (4.1a) and (4.1b), the resulting overall costs of the buy option are calculated by summing up the discounted buying and side costs for the individual purchasing transactions and subtracting the discounted leasehold income and the discounted value of acquired land. The values are calculated for each year during the case study timeframe and discounted from the respective year to the base year 1980. This

³ Federal and European law, however, regulate withdrawal of land from extensive land use and consequential re-intensification (e.g. DGLG of 2013 in Schleswig–Holstein). Additionally, SNSH does not plan to sell or re-intensify any of the area around Lake Bültsee. See “Summary and discussion” section for a discussion of the influence of greening regulations.

allows us to evaluate the costs of the project from the beginning up to each year during that timeframe retrospectively.

4.2.4 Costs of the compensation option

4.2.4.1 Cost function

The costs of buying relevant agricultural area and managing it internally are to be compared with the hypothetical costs of implementing a compensation scheme with an identical conservation outcome. We assume that a certain conservation target can be achieved through identical conservation measures independently of the mode of governance, in turn causing only different cost patterns (cp. Muradian and Rival, 2012; Schöttker et al., 2016). Furthermore, we assume that each landowner is, in principle, willing to participate in a scheme, if offered a compensation payment, which at least covers the costs of participation, i.e., the foregone profit due to extensive management as prescribed by the scheme plus additional transaction costs due to participation and implementation (Defrancesco et al., 2008; Franzén et al., 2016; Greiner, 2016). For simplicity, we ignore the possibility of landowners to use the bargaining power they obtain if their participation is crucial for the success of the conservation measure (cp. “Summary and discussion” section). In addition to the costs for compensation, the agency faces a certain amount of transaction costs including personnel expenses, travel expenses, and monitoring and enforcement costs (McCann, 2013).

The total costs for the hypothetical compensation scheme from year t_{start} to a given end-year T , expressed in values of the year t_{start} , can thus be calculated with Eq. (4.3) with f_t the area in hectares receiving compensation, \overline{cp}_t the homogeneous compensation payment, and t_t^{agency} the transaction costs borne by the agency, all in year t .

$$C_{t_{start}}^{borrow} = \sum_{t=t_{start}}^T d_t \times f_t \times \overline{cp}_t \times (1 + t_t^{agency}) \quad (4.3)$$

The compensation payment \overline{cp}_t paid in the scheme is determined with Eq. (4.4) where π_t^{int} is the potential profit per hectare agricultural land under intensive management, π_t^{ext} the hypothetical profit with (the prescribed) extensive grassland management, and t_t^{farmer} the transaction costs borne by the farmer for participating in the scheme.

$$\overline{cp}_t = (\pi_t^{int} - \pi_t^{ext}) \times (1 + t_t^{farmer}) \quad (4.4)$$

According to Falconer (2000) and McCann (2013) the agency level transaction costs as well as the farm level transaction costs are measured as a proportion of the compensation payment \overline{cp}_t .

4.2.4.2 *Hypothetical costs of compensating farmers for conservation*

For the profit-maximising intensive land use we assume that land in the conservation area around Lake Bültsee is cultivated with the average land use pattern for crop production in Schleswig–Holstein. Thus, we assume the land is cultivated with a mixture of field crops according to the four most common field crops grown in Schleswig–Holstein—i.e., wheat, barley, maize and rapeseed—during the study case timeframe. According to our interviews, we assume that from 2009 onwards the crop cultivation pattern transitions towards the more common and more profitable maize production. This is at least true in the study area, where most of the agricultural fields, and especially the fields surrounding the conservation area are cultivated with maize as an energy crop. Thus, from 2009 to 2015 maize is assumed to be the sole crop cultivated in the conservation area under profit-maximising land use. From 2006 to 2008, we assume a transition period in which the share of maize cultivation increases and the shares of the other three crops decrease (additional data given in Appendix C.3).

Based on data on average yield per hectare, crop prices and cultivation costs (Hydro Agri Dülmen GmbH, 1993; KTBL, 2014, 2009, 2005, Ruhr-Stickstoff Aktiengesellschaft, 1988, 1974) we estimate the average profit and cost from intensive agriculture with the corresponding field crops (see Appendices C.4 and C.11).

We estimate for the potential profit in the hypothetical compensation scheme a revenue between 90 € and 160 € from the sale of cattle and/or dairy products per ha extensive grassland, costs of between around 310 € to 500 € per ha, and a total profit per hectare in the range from – 200 € to – 340 € (see Appendix C.4 for details). Our calculations show that over time, the corresponding profitability of extensive grassland measures varies (compare Appendix C.4) but is negative throughout the whole study case timeframe, meaning that extensive grassland measures are, from the farmer’s perspective, not economically attractive and need subsidies (cp. Mewes et al., 2015).

Additional to production and management costs, transaction costs contribute a substantial share to the overall cost of the compensation option. We take average values based on literature (Falconer, 2000; McCann, 2013), and assume that the farmers’ transaction costs

amount for 10% of the actual compensation payment, and the agencies transaction costs contribute 15% of the compensation payment.

4.2.5 Baseline and sensitivity analysis

To analyse the costs of the case study we define an initial baseline with a combination of average cost parameters. We then modify each parameter individually to assess possible causes for payment changes and show the management alternative's sensitivity towards parameter variation. This is done by creating high and low values for the parameters of transaction costs, leasehold payments and land value factor θ (Table 4.3 and Section 4.3). To analyse the impact of the unprecedented increase in land prices on the cost-effectiveness of the two options—driven by the increase in energy crop production in the study region since 2006—we estimate the hypothetical outcome of the management options with and without this boom in the energy crop sector.

Sensitivity analysis 1 considers variations of the baseline value for transaction costs. The transaction costs are included into our calculations in Eqs. (4.3) and (4.4), either being borne by the farmer or the agency. Literature suggests that overall transaction costs for agri-environment schemes vary between 14 and 40% of the compensation payment (Falconer, 2000; Kersten, 2008; McCann, 2013; McCann and Easter, 2000; Thomas et al., 2009). We thus assume low and high values in this range and individually vary farmer's and agency's transaction costs to 0.05 above and below the baseline values of 0.1 for the farmers and 0.15 for the agency.

The actual leasehold rent that BW pays to SNSH is only a small fraction of what is paid on average for agricultural land or even extensive grassland in Schleswig–Holstein. We therefore also calculate the sensitivity of our results to changes in the leasehold rent, which is considered in Eq. (4.1a). For the low value, we assume zero rent to be paid as was the case prior to 2008, while the high value is assumed to be the average values for leasehold rents in Schleswig–Holstein (see Appendix C.7).

We assume that the value of extensive grassland is proportional to the value of intensive agricultural land (see Eq. 4.2). Ciaian et al. (2010) suggest for this proportion a value of $\theta = 0.7$ in Central Europe. We take this as the value for the baseline case and vary it to 0.4 and 1.0 in sensitivity analysis 3 as estimates for weaker and stronger proportional

Table 4.3: Parameter values for different sensitivity analyses, with corresponding low scenario, baseline and high scenario values.

Sensitivity analysis	Parameter	Description	Low scenario value	Baseline value	High scenario value
1	Transaction costs farmers transaction costs agency	<i>As a fraction of the offered compensation payment</i>	0.05	0.1	0.15
			0.1	0.15	0.2
2	Leasehold payment		0 € per hectare and year	0 € per hectare and year from 1980 to 2008, and 40 € per hectare and year from 2009 to 2015	Average leasehold payment for agricultural land in Schleswig-Holstein
3	Land value factor θ	<i>Grassland/intensive cropland price ratio</i>	0.4	0.7	1.0
4	Energy crop boom	<i>Assuming, the development of energy crop sector expansion did not happen</i>		Transition in cultivated crops from crop mix to full maize cultivation	No transition in cultivated crops to full maize cultivation; average, federal state level cultivation pattern

relations. This is equivalent for SNSH being able to sell the extensive grassland either to the full market price for intensive agricultural land ($\theta = 1.0$), or to a reduced price ($\theta = 0.4$) (cf. Eq. 4.2).

In our baseline, the expansion of the energy crop sector with resulting increases in agricultural land prices and product prices is considered. Since 2006, land prices in Schleswig–Holstein have more than tripled and income from intensive agricultural land use has risen steeply as well (see Appendix C.2). As this development could not have been anticipated when the conservation project was initially set up in 1980, we analyse the sensitivity towards ignoring the effects of the energy crop boom and thus keep land prices and product prices fixed after the year 2006. As a result, we get a comparison between the actual development and a potential uninfluenced economic situation. As this is highly

speculative, we do not implement any further price adaptation (e.g., due to inflation) and thus leave all cost parameters except product, land prices and leasehold rents unchanged.

4.3 Results

4.3.1 Results of the baseline

We find, for the baseline, that between 1980 and 2004, the costs of the buy option exceeded the potential costs of the compensation option (Fig. 4.2), whereas from 2004 onwards the costs of the compensation option were higher than the costs of the buy option. This result can be easily explained, as for long-term investments one-time transaction costs (e.g., side costs) are lower on a per annum basis than for short-term projects. In the short-run, high one-time transaction costs make the buy option relatively unattractive. In the case study, this result is additionally driven by the development of agricultural land prices, which decreased significantly between 1980 and 1993 (see Appendix C.2) and thus would have caused high losses, if land had been sold during or shortly after this period.

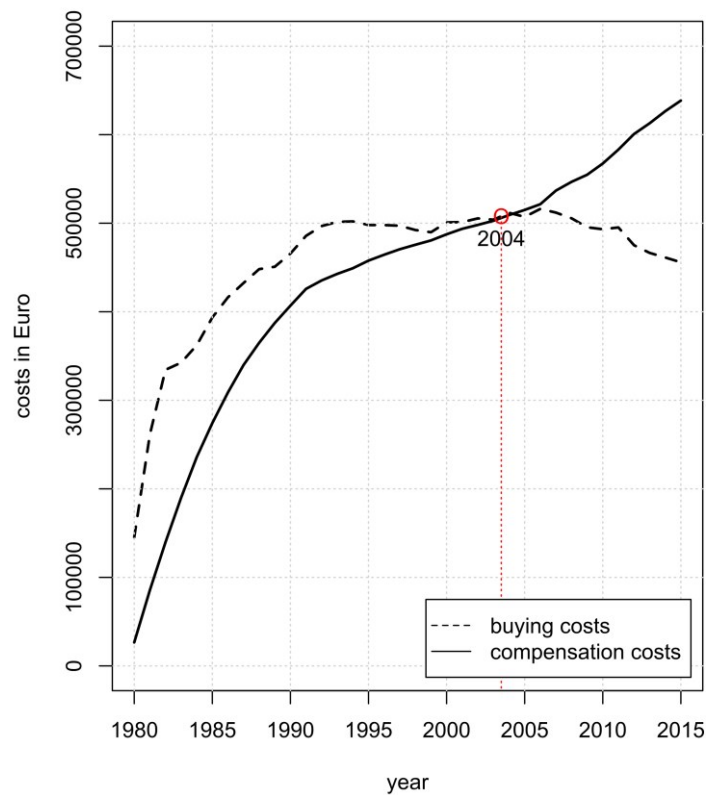


Figure 4.2: Cumulative discounted costs of the buy option (straight line) and the compensation option (dashed line) for the baseline value between 1980 and 2015. The values are discounted to the base year 1980.

4.3.2 Sensitivity analysis

4.3.2.1 Transaction costs

The sensitivity analysis for the transaction costs results in a cost range for the compensation option. This is caused by a change in the transaction costs (borne by the conservation agency) and the compensation payments (which include compensation for the transaction costs of the farmer) [see Eqs. 4.3 and 4.4]. Although transaction costs make up only a small fraction of the total costs of the compensation option, changes in those costs have a substantial impact on predating (from 2004 to 1995) or postponing (from 2004 to 2008) the point where the buy option becomes more cost-effective than the compensation option, henceforth referred to as critical point (see Fig. 4.3a). For a detailed analysis of the influences of transaction costs on compensation payment and on the development of farmer's and agency's transaction costs over time see Appendix C.6.

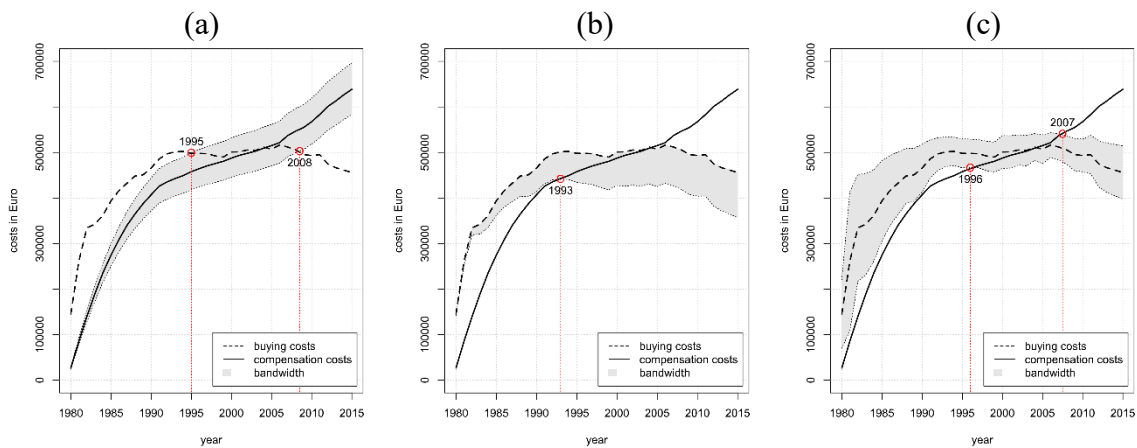


Figure 4.3: Costs of the buy option (dashed line), and compensation option (straight line). The values are discounted to the base year 1980. (a) Bandwidth for compensation costs (grey shaded area) due to variations in transaction costs (high value is at the upper edge; low value is at the lower edge of the shaded area). (b) Bandwidth for buying costs (grey shaded area) due to variations in the leasehold rent (high value is at the lower edge; low value is barely visible as it is very close to the actual, already very low lease hold rents). (c) Bandwidth for buying costs (grey shaded area) due to variations in the land value factor θ (high value is at the lower edge; low value is at the upper edge of the shaded area).

4.3.2.2 Leasehold rent

Unsurprisingly, if we assume high values for leasehold rent, the buy option gets more attractive as the agency can generate higher income from giving the land as a leasehold to a contractor (compare Appendix C.2). Consequently, the timeframe in which the compensation option outperforms the buy option is shortened, shifting the critical point

forward to the year 1993 (Fig. 4.3b). The low value scenario with no leasehold payment has hardly any effect. Differences in the baseline only start in 2008 and due to the small differences between the two scenarios, the advantage of the buy option is only marginally reduced between 2009 and 2015 compared to the baseline.

4.3.2.3 *Land value factor*

Over the whole project timeframe, the land value varies significantly (Fig. 4.3c). This variation is amplified in the high value case and dampened in the low value case. It shows that with a decreasing value of θ the costs for the buy option rise (upper bound of the grey shaded area) as the land value decreases and thus can only be sold at a low price at the end of the conservation project (see Eqs. 4.1a and 4.1b). Hence, the critical point is postponed to the year 2007. For increasing values of θ , which implies an increase in the land value and thus a decrease in the costs of the buy option, this critical point is already reached in 1996 (see lower bound of the grey shaded area). Beyond that, in the preceding project run time between 1980 and 1996, the difference between both options is marginal with higher levels of θ . For sufficiently high values of θ , the buy option would always be preferable over the compensation option. However, even higher values of $\theta = 1$ are less reasonable, as they would represent a situation in which formerly extensive grassland could be sold for higher prices as intensive agriculture land, thus being essentially overvalued.

Rising land prices explain the growing advantage of the buy option over the compensation option in the last years of the project (since 2004). However, it is more likely that θ is in the range of 0.4– 0.7 thus resulting in postponement of the critical point.

4.3.2.4 *No energy crop boom*

Figure 3.4 shows that under the assumption that an energy crop boom did not occur and, as a result, land and product prices have been more stable in the study area since 2006 (the start of the energy crop boom), the cost-effectiveness advantage of the buy option compared to the compensation option is strongly reduced in comparison to the baseline. This is plausible under this scenario, because land prices do not rise as strongly as in the baseline, which reduces the potential revenue from selling land (see Eq. 4.1a). This in turn increases the overall costs of the buy option. A further effect in this sensitivity analysis is that the potential revenue from extensive and intensive agricultural measures is reduced by lower agricultural product prices (see Eq. 4.1a). This causes a reduction in the costs of the compensation option as compensation payments decrease (see Eq. 4.4). If agricultural

product prices fall, intensive agriculture is less profitable, hence, the opportunity costs of an extensive management scheme decrease, causing the compensation payments to decrease as well. Consequentially, both cost developments are closer to each other from 2006 onwards indicating a decreased superiority of the buy option.

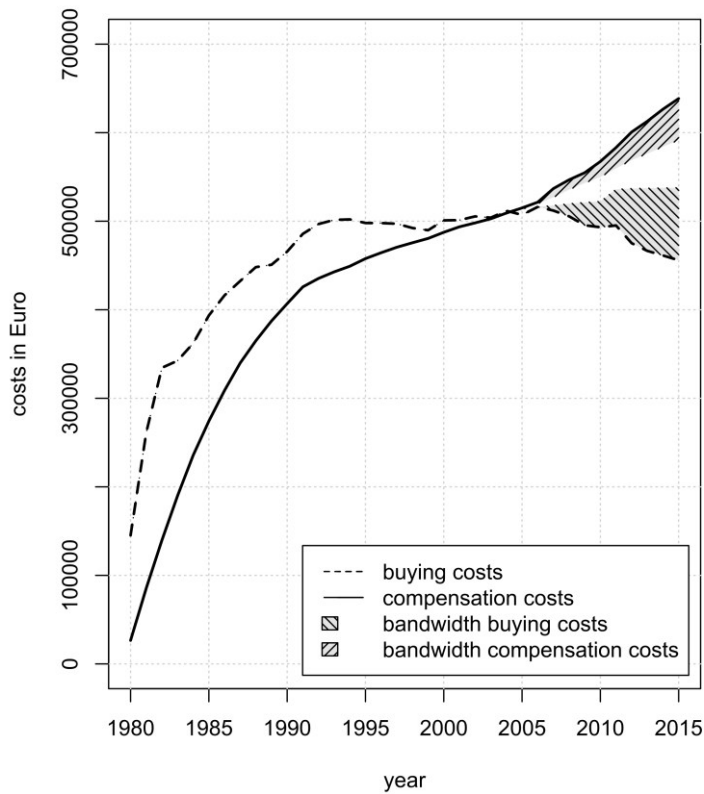


Figure 4.4: Costs of the buy option (dashed line) and compensation option (straight line) with the corresponding bandwidths (grey shaded areas) due to a hypothetically non-existing energy crop boom and thus decreased land and produce prices. The values are discounted to the base year 1980.

4.4 Summary and discussion

Our research objective was to assess the cost-effectiveness of two modes of governance for an area of conservation interest and the resulting management options implemented on this area for a case study. We considered as modes of governance that (1) a conservation agency buys desired areas of interest and implements conservation measures itself or through closely monitored firms (buy option), and (2) an agency offers payments to landowners to incentivise them to voluntarily preserve or create habitat on the areas of interest (compensation option). Our case study looks at the implementation of extensive

grassland measures around the oligotrophic Lake Bültsee in a Natura 2000 area in Northern Germany from 1980 to 2015 aimed at reducing the nutrient inflow from the surrounding agricultural areas.

We find that the buy option outperforms the compensation option in the long run from 2004 onwards. In the short run however, the compensation option is the superior mode of governance. This result is driven by the fact that the one-time costs of buying land, i.e., the buying and selling side costs, dominate the cost structure in the short run and obviously can only be offset after a significantly long project runtime. Land price fluctuations were a further key factor. Extensive parts of the conserved area were bought at relatively high prices in the 1980s, while prices dropped steeply in the 1990s. The low value of land during the 1990s implies that acquired land can only be resold at a lower price, which increases the costs of the buy option. The annual costs of the compensation option during the same timeframe however decrease, as the conservation agency faces lower compensation payments due to lower revenue from intensive land use. This trend, however, was reversed after land prices increased substantially from 2006 onwards. In our study, the prices for land varied mainly due to underlying changes in the profitability of agriculture in the region during the 1990s, which affected the costs of both options. However, price changes can also be driven by other factors solely influencing the costs of only one governance mode (Gordon et al., 2011).

We made a few assumptions in our case study that require discussion. First, we assumed that the mode of governance in practice does not change the possible conservation results. We assume that, *ceteris paribus*, the different modes of governance only result in different cost structures and thus total project costs, while still being able to result in the same ecological outcome. This assumption is necessary to allow a proper comparison between the two alternative governance choices in terms of their cost-effectiveness. Whether this assumption holds in reality is however an open question. Conservation organisations that buy land tend to keep and manage their properties for long time periods, and thus have a corresponding planning horizon for their conservation targets and the resulting implementation of measures (Groves et al., 2002; Theobald et al., 2000). Compensation-based projects, on the other hand, require more flexibility as potential participants can decide on a short-term basis whether to participate in a programme or not. Moreover, conservation funding, if spent annually, is more easily cut if the political or economic circumstances become less favourable for biodiversity conservation. This leads to a

different planning horizon and may consequently result in different conservation targets and measures for such projects with different ecological outcomes.

We assumed further that the landowners are willing to participate in conservation measures with certainty, if it is profit maximising for them. We made this assumption to ensure that the intended conservation target can be reached. However, this might be different in reality. The willingness to participate in nature conservation schemes depends, among other factors, on former experience with conservation projects, programme objectives, and environmental attitude, but also on participants' expectations of potentially permanent changes in the usability of their land due to implementation of measures (Greiner, 2016; Knight et al., 2011; MEA, 2005; Moon and Cocklin, 2011; Unay-Gailhard and Bojnec, 2016; Yeboah et al., 2015) and certainly varies between conservation projects, timing and location (Vanslebrouck et al., 2002). Therefore, landowners of potential conservation areas might not at all or only temporarily be willing to participate in a compensation scheme with resulting negative ecological impacts (Schöttker et al., 2016; Van Teeffelen et al., 2012).

In this context, habitat turnover and a changing spatial configuration of habitats may occur due to changes in landowner's participation in conservation measures. The spatial and temporal aspects of the optimal choice of governance modes thus become relevant. They may also become relevant in other contexts, for example if climate change leads to a change in the spatial suitability of habitat over time. We leave it to further research to address such spatio-temporal issues and here only refer to Johst et al. (2011) as an example of research that addresses the topic of how habitat turnover can be counterbalanced by a change in habitat connectivity and increase in habitat size. A different situation arises if landowners are, in principle, willing to participate in a conservation project, and realise that their land is crucial for the realisation of the project. In this case farmers may act strategically and try to capture so-called information rents (Ferraro, 2008) by overstating the opportunity costs of participation and requesting higher compensation payments or purchasing prices for their land (Banerjee et al., 2016; Kuhfuss et al., 2016; Prazan and Theesfeld, 2014; Vergamini et al., 2015). To what extent this farmer-side strategic behaviour is prevalent in the study area and to what extent it can be counteracted by agency-side behaviour such as risk reduction through trust-building remains an open question.

We compared two governance modes (buying and compensation) that can be considered the ‘pure’ implementation options. However, also other mixed forms are possible, e.g., a conservation agency may decide to buy a part of the relevant land to be conserved, while it may compensate landowners on the other part. Also in terms of compensation payments, different options are possible such as the introduction of tax-breaks. Instead of compensating landowners directly in monetary terms for their effort of implementing conservation activities, governing bodies could offer such more indirect incentives. In how far the use of different compensation methods has an influence on the costs of the compensation option, or if different methods of compensation have impacts on e.g., landowner’s compliance or participation willingness is an open issue and may also be a matter of further research.

German and European legislation allows intensification of extensively used grassland only under certain conditions (referred to as greening), e.g., the provision of appropriate compensation areas (see DGLG of 2013 for Schleswig–Holstein). A complete intensification prohibition is only in place in Natura 2000 and FFH-areas, both of which apply to the Bültsee area. Therefore, our assumption that the area used in the project can easily be sold and used intensively only reflects the real life situation to a limited extent. However, according to its bylaws, SNSH is, in principle, allowed to sell every part of its conservation areas if compensation areas are developed in return. If, therefore, it was necessary or opportune to sell the land at Lake Bültsee for ecological, environmental or economic reasons, this would be possible as long as compensation areas are provided. Against this background, a re-intensification of the project area is at least difficult under existing law; however, a potential sale is possible. This obviously might lead to reduced land prices. We account for this in our analysis by introducing the land value factor $\theta = 0.7$ for the case study calculations, representing the price ratio of extensive and intensive agricultural land prices.

When selecting between the buy option and the compensation option in the real world, aspects other than those considered in this paper play a role as well. For example, monitoring and enforcement of compliance with the prescribed conservation measure seem to be easier in the buy option. There is no need for compliance monitoring if the conservation agency carries out the conservation activities by itself and it seems rather easy if the agency selects a firm with an intrinsic interest in conservation, as with Bunde Wischen e.V. in our case study. In contrast, if landowners without an intrinsic motivation

carry out the conservation measures, the importance and hence the costs of monitoring and enforcement measures increase (Wätzold and Schwerdtner, 2005). Another relevant aspect is the landowners' acceptance of the two options. German farmers seem to have a strong preference for the compensation option as selling the land to a conservation agency implies that it is taken away “irreversibly” from agricultural use (Beer, 2016).

Generalising insights from a case study is always only possible to a limited extent, and the assessment of the cost-effective mode of governance of conservation projects comparable to our case study is an empirical issue and requires a detailed examination of the specific economic and ecological conditions of the case study. However, we are able to identify a few patterns that allow some careful generalisations. One-time transaction costs of land purchase and sale are high which suggests that—*ceteris paribus*—the cost-effectiveness of the buy option increases in comparison to the compensation option with the planned duration of the conservation project. Note that although, in principle, long-term conservation is desired from an ecological perspective, conservation takes place in a dynamic world where aspects such as climate change and changing socio-economic conditions may call for a re-allocation of conservation areas (Van Teeffelen et al., 2012, 2014) implying the need to sell conserved land. Our case study also draws attention to the general importance of changes in land prices (cp. Carwardine et al. (2010) as an example of research on future cost uncertainty) and their impact on the cost-effectiveness comparison of the buy option and the compensation option. The drop in agricultural land prices in the 1990s substantially extended the cost-effectiveness advantage of the compensation option in those years and if the increase in land price due to the boom in energy crops had not occurred, the cost-effectiveness advantage of the buy option from 2004 onwards would have been much smaller.

We consider the presented costing framework as generally transferrable to similar situations, namely, biodiversity conservation in agricultural landscapes in developed countries. An extension of the framework to capture costs in a variety of further governance modes (e.g. land easements or covenants, or privately vs. publicly managed land, cp. Gordon et al. (2011)) and landscapes (e.g. forests that would require more long-term considerations, cp. Zabel et al. (2018)) is possible. However, an adaptation of the relevant costing relationships is needed. Another extension of the costing framework is the inclusion of conservation measures implemented in perpetuity as that suits the general governance background in many actual conservation situation (e.g., easements or

covenants). Although technically already allowing for such types of analysis, a reasonable long term cost relationship needs to be developed. Data availability in this context is crucial and clearly depends on the actual case. Further research in this field is certainly desirable, as it would provide a better understanding of optimal governance mode choices in different conditions and landscapes.

Acknowledgements

We are grateful to Christian Dolnik, Erich Ruser and Walter Hemmerling from the Stiftung Naturschutz Schleswig–Holstein for helping us to collect data and for discussing the conservation strategies with us. We also thank the Bunde Wischen e.V. and Gerd Kämmer for supporting our work with further insights and data.

Chapter 5

Easement or public land? An economic analysis of different ownership modes for nature conservation measures in California

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5 Easement or public land? An economic analysis of different ownership modes for nature conservation measures in California

Abstract

Biodiversity conservation requires space where conservation measures are implemented for a desired purpose. Setting land aside for conservation has been widely applied, while novel conservation modes (private–public partnerships, private multipurpose land management) may be fundamental to achieve conservation goals. We perform an economic analysis of the cost development for two conservation options in California, in-fee and easements, from 1970 to today. We find that in-fee options have lower costs than easements in the long run. While there are high costs of purchase for in-fee, ultimately they even-out or generate profits. Costs of easements continue growing exponentially overtaking costs of purchase. Sensitivity analysis shows increases in purchasing prices and opportunity costs positively influence conservation costs, while increasing interest rates negatively influencing them. The results suggest that easements are not yet an economically viable alternative for in-fee conservation purchases. Our analysis is a first step to assess economic viability of choosing easements.

5.1 Introduction

Conservation of biodiversity as a basic need requires space, on which conservation measures are implemented. Systematic conservation planning has defined conservation goals and optimal solutions to reach such goals (Pressey et al., 2007). Goals include representation of biodiversity and its processes and functions; solutions involve identifying an optimal set of lands that best meets the defined conservation goals, strategizing when and how to add them to a conservation network, and ultimately acquiring and managing land (Lovejoy, 2006; Pressey et al., 2007). While setting goals is relatively straightforward, implementing them can be challenging, and among other things, determines land

governance, i.e., ownership. Ownership of this space is an important factor influencing the costs of implementation (Adams et al., 2010; Naidoo et al., 2006), the duration of conservation measures, and the ecological and economic success of conservation projects (Balmford et al., 2003). However, few studies have assessed the economics of conservation, in particular when the options are to purchase public land or to lease private land as easements or covenants (Cross et al., 2011; Iftekhhar et al., 2014). Here, we assess which is the best option from an economic perspective: to purchase as public land or to lease as easements? To maximize ecological outcomes, while not placing an ever-growing burden on taxpayers or relying on donations for purchases to be achieved by NGO's, a cost-effective implementation of conservation goals is desirable. Such implementation would either maximize ecological benefits at predefined costs or minimize costs for a given and desired ecological outcome (Wätzold & Schwerdtner, 2005). The optimal design of conservation measures can be achieved by, for example, the design of agri-environmental schemes (Armsworth, 2014; Naidoo et al., 2006), spatial and temporal allocation (Drechsler et al., 2017, Mouysset et al., 2011; Polasky et al., 2008), optimal length of conservation contracts (Ando and Chen, 2011; Lennox and Armsworth, 2011), and efficiency gains from variable payment structures (Armsworth et al., 2012). A so far neglected area of research is the influence of the mode of governance on the optimality of conservation measures. The question, whether conservation agencies should either buy land and manage it themselves, or monetarily compensate landowners that voluntarily provide conservation measures is an important issue (Muradian and Rival, 2012; Schöttker et al., 2016). Studies so far formalized the cost-relation of buying land versus compensating landowners and identified influencing economic factors (Schöttker and Wätzold, 2018), highlighted the effects of land markets and property value fluctuations on opportunity costs (Curran et al., 2016), and provided a general framework for comparing conservation contracts with different modes of governance (Juutinen et al., 2008). Nonetheless, the exact implementation of monetary compensations might have a large effect on the costs of conservation (Engel et al., 2008; Wätzold and Drechsler, 2005).

In this work, we provide an overview of different modes of governance for conservation relevant in the state of California, and discuss the costs and costs-structure of a selection of conservation areas. In principle, in-fee land causes relatively high upfront one-time costs, while easements cause relatively small, but recurring costs. By calculating the present value of both cost streams, the upfront one-time costs are (depending on the discount

schedule) valued differently than the recurring costs and thus cause different present values and cost developments. We expect land purchase to have a different present value of cost per hectare than easements because of lower vulnerability to volatile costs and because of differences in recurring and one-time costs. We chose California because of the wealth of conservation action over the past 100 years resulting in about one third of the state being conserved in both public and private ownerships (Santos et al., 2014b). Land in the state has a positive value, as each purchase is weighted against development claims, and prices are growing exponentially. Recent efforts for conservation include easements, which allow private landowners to offset the easement investment against tax liabilities that result in foregone tax payments for the government, and relief government and NGOs from expensive land purchases. This decision is currently ongoing in California but also in many other regions of the world.

Easements are also expected to increase stewardship (Merenlender et al., 2004; Sorice et al., 2013; von Hase et al., 2010). We chose to analyse only buying or leasing alternatives because they represent the most extreme conservation decisions, therefore likely provide a wide range of costs of conservation. In our calculation, we included acquisition costs, land management costs, transaction costs, and potential income. We present results in a way that makes the alternatives more comparable (e.g., total and per hectare costs over some finite time period). We discuss our findings in light of current economic choices and limitations unveiled by a sensitivity analysis, and then tackle the unobservable, heterogeneous nature of opportunity costs to private land managers of switching to conservation easements (hidden information creating opportunities for rent seeking).

5.2 Methods

5.2.1 Study system

California is a biodiversity hotspot (Myers et al., 2000) and while facing pressures to develop, one third of the state area is under conservation (Figure 5.1). Most conservation area is public land but a part is easements. We selected four conservation organizational bodies to account for the diversity of governance levels on decisions of land acquisition, which we assume to generate equal ecological benefits when implementing conservation measures. We chose the California Department of Parks and Recreation as a representative of state level governance, the County of San Diego as a representative of county level

governance, the City of San Diego to represent city level governance, and the East Bay Regional Park District (EBRPD) as a representative of a Special District. Currently it is possible to negotiate a contract with landowners for a property to become a conservation easement. Conservation easements in the state currently amount to 8000 km².

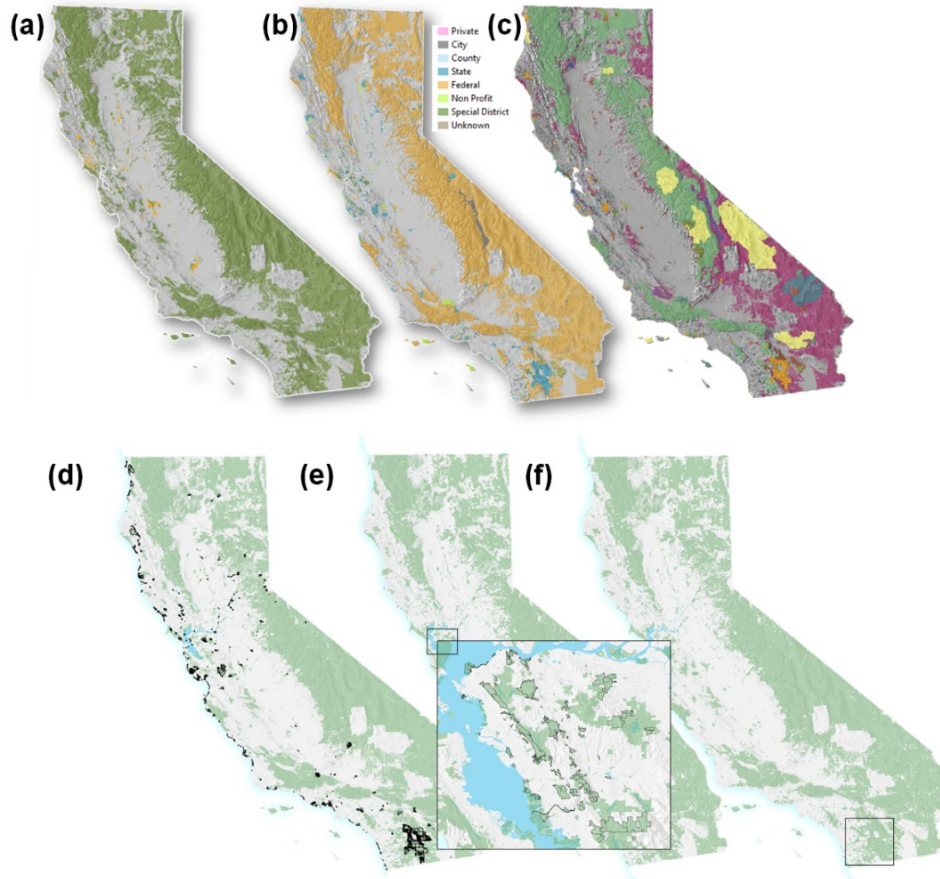


Figure 5.1: Conservation areas in the state of California: (a) extent of conservation areas (green) and easements (orange); (b) ownership of conservation parcels (legend colors represent governance levels responsible for management of land); (c) type of land management (yellow: national parks; purple: Bureau of Land Management; green: United States Forest Service; orange: state parks; grey: United States Department of Defense); (d) California Department of Parks and Recreation properties; (e) East Bay Regional Park District properties; and (f) City and County of San Diego. The California State Parks manages about 900 properties that were added to their portfolio since 1970, corresponding to 792 km² distributed throughout the state. The agency has an Office of Grants and Local Services that since 1964 has provided funding to 7,400 local parks to be created or improved. The County of San Diego currently manages about 200 properties, amounting to 207 km² of land managed since 1970. The county faces high rates of development and it is ambitious and determined to increase the area in Open Space, as stated in its strategic plan. The City of San Diego manages 430 properties with about 190 km² of land added since 1970. In California, cities decide upon land use regulations within their jurisdiction (Santos et al., 2014b), and San Diego manages most of the City Parks. The East Bay Regional Park District (EBRPD) is a Special District, that is, a limited purpose local government, separate from cities and counties. Special districts provide focused public services such as fire protection, water supply, parks, recreation, and so on. EBRPD manages 242 properties with an area of 222 km² added since 1970. Financing for Special Districts comes from property taxes, fees that users pay for services and special assessments. These types of districts may handle a revenue that varies between 10 and 30 billion USD, but only a small fraction of that budget goes into parks and recreation.

Unfortunately, it was not possible to include NGOs such as The Nature Conservancy in our analysis because of the lack of available data on land purchase prices and other costs; however, we do acknowledge that this would have been a valuable exercise as these NGOs are major actors in easement conservation in the United States. Nonetheless, we were interested in the choice for purchase or easement, irrespectively on whether agencies are NGOs, so we believe our analysis is still interesting and provides valuable insights beyond the current perception that easements are NGO-only options. Further, there is no reason to expect that the buying processes would differ between NGOs and other agencies, only varying the funding sources and the mechanisms to bring in participants to easement schemes.

5.2.2 Costing functions

We adapt the cost functions of Schöttker and Wätzold (2018), who provide a framework of costing relations, relevant in the assessment and estimation of costs of conservation implementation. Under this framework, the general nature of costs differs according to governance mode and provides a functional relationship for the costs of conservation, if the land purchased versus land owned privately, and is compensated monetarily for the voluntary provision of conservation.

To assess the costs of a conservation project, the opportunity costs of implementation have to be calculated. Generally relevant cost components are one-time costs—e.g., purchasing expenses, contract negotiation costs, transaction costs—and recurring costs—e.g., monitoring costs, land management costs, contract renegotiation costs. Depending on when the different costs arise, they have to be discounted and brought into a common metric to make them comparable over time (one Dollar in 1950 has a different value as one Dollar in 2018). After discounting, all one-time and recurring costs can be accumulated to calculate the overall costs of a project.

We can thus simplify the general structure of the opportunity costs as follows:

$$C = \sum_{t=t_{start}}^{t_{end}} C_t \times d_t \quad (5.1)$$

with C_t the general costs within a project at time t , and d_t the relevant discount factor at time t defined as follow, with i_{t-1} the real interest rate in year $t - 1$:

$$d_t = (1 + i_{t-1})^{-1} \times d_{t-1} \quad \text{with } d_{t_{start}} = 1 \quad (5.2)$$

In general, we consider only economic costs to be discounted and neglect ecological benefit discounting, as no detailed information is available on when and which scale ecological benefits arise, and because of the permanent nature of conservation areas generating ecological benefits (Armsworth, 2018). This structure can then be used to reflect the different cost structures of land that has been bought by an agency and henceforth managed by themselves (“in-fee”), or land for which private owners are compensated for conservation (“easement”).

5.2.2.1 Costs of “in-fee” land management

Land purchase causes a mixture of one-time expenditure and recurring cost components, together resulting in overall costs of implementation and execution of a conservation project. We calculate the costs $C^{in\ fee}$ of “in-fee” conservation projects as follows:

$$C^{in\ fee} = \sum_{t=t_{start}}^{t_{end}} (P_t + M_t + S_t + T_t^{in\ fee} - I_t) \times d_t \quad (5.3)$$

with P_t , the purchasing costs of land; S_t , the purchasing side costs (e.g., contract negotiation, notary fees, taxes, etc.); M_t , the regularly recurring management costs; $T_t^{in\ fee}$, the transaction costs of the purchase; and I_t , the potential income generated from managing the conservation measures. Forgone agricultural profits are included in the income calculation (Appendix D.2). The income is calculated as a fraction of the opportunity cost (by a scaling factor l) and thus is interpreted as a reduced income (compared to non-conservation use).

5.2.2.2 Costs of “easement” land management

We calculate the costs $C^{easement}$ caused by a compensation scheme as follows:

$$C^{easement} = \sum_{t=t_{start}}^{t_{end}} (OC_t + M_t + T_t^{easement,a} + T_t^{easement,l} - I_t) \times d_t \quad (5.4)$$

with OC_t , the opportunity costs for the landowner by managing land for conservation and thus not profit-maximizing; M_t , the management costs; the agency side transaction costs $T_t^{easement,a} = t_a^{easement} \times OC_t$; and $T_t^{easement,l} = t_l^{easement} \times OC_t$, the land user side transaction costs (see Appendix D).

5.2.3 Data description

Due to constraints on the data, as well as to historical reasons, we decided to only focus on areas set under conservation after 1969. After this year, state allowed public–private partnerships for the first time, a necessary step for conservation easements to be possible. This is also the time frame for which there is reliable data available on land values, interest rates, and inflation rates, which are required to calculate the costs of either management option. We estimated the actual land purchasing prices according to Equation (5.3) based on average county level housing price data (CAR, 2018; Davis and Heathcote, 2007). The discount factor is based on the 10-year treasury constant maturity, non-seasonally adjusted rate (FRED, 2018) and the consumer price index for all urban US consumers (USBLS, 2018). Data on conservation areas were provided by the California Protected Areas Data Portal (CPAD, GreenInfo Network 2014, <http://www.calands.org>) and the California Conservation Easements Datasets (CCED, GreenInfo Network, 2014).

5.3 Results

Overall, costs of implementation, total annual costs, costs of management, and total annual management costs all show that easements surpass the costs of in-fee properties in the long run, showing an exponential growth while costs of in-fee are more volatile, decreasing substantially after a decade. We found very similar patterns across agencies; however, they became more different as we estimated costs per unit of area.

5.3.1 Development of total annual costs

When looking at the development of total annual costs and total annual costs per hectare (Figure 5.2), the described cost characteristics are supported. Easement costs start late and increase over time, while in-fee costs start early and fluctuate strongly, even generating negative costs (i.e., income). Per hectare, easement, and in-fee total costs are of comparable size suggesting that although conservation was implemented in heavily different economic and ecological conditions, eventual costs per hectare are not influenced by potential governance differences. We believe this assumption is transferable elsewhere because land costs are market driven and independent of conservation agencies, their goals, and jurisdiction.

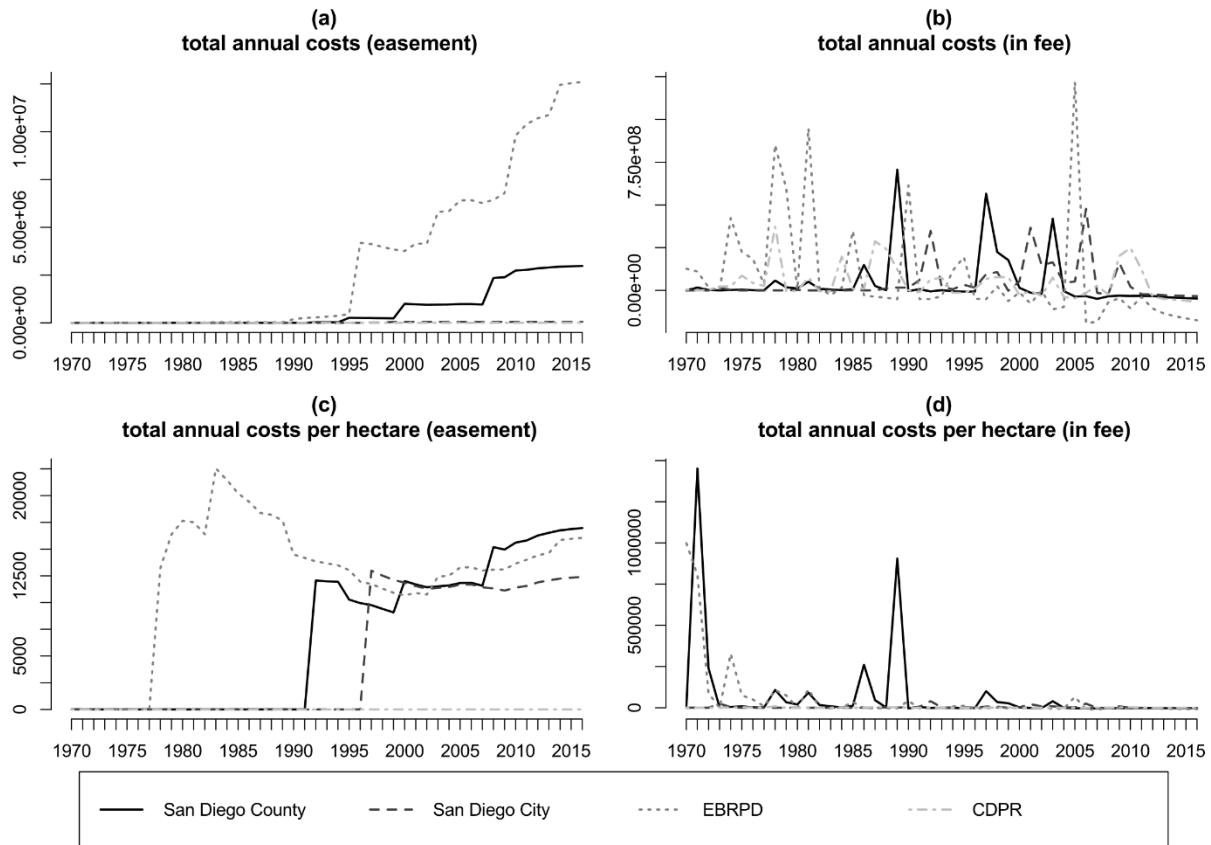


Figure 5.2: Total annual costs ((a) and (b)), i.e. the total amount of costs which arise in each year of the analysis time frame, separated for each of the four conservation agencies, and total annual costs per hectare ((c) and (d)), i.e. the total amount of costs in each year of the conservation time frame, divided by the total area in hectare in that year, separated for each of the four conservation agencies.

With easement, management total annual costs are constantly positive (although income of the same relative amount is considered as with “in-fee”), and even increase, while with “in-fee”, annual costs are highly volatile and fluctuate from positive to negative (i.e., income generating) over the analysed time frame, on a total and per hectare basis.

5.3.2 Development of costs of conservation

The total cumulative costs of conservation and their temporal development for both options are different (Figure 5.3). While the total cumulative costs of easements show an exponential development, those for in-fee areas are more volatile. This volatility is shown as costs of implementation for in-fee properties can increase drastically when new areas are purchased or even decrease due to relatively low maintenance costs and relatively high potential-income generated. We found a consistent pattern across agencies, except for agencies without easements.

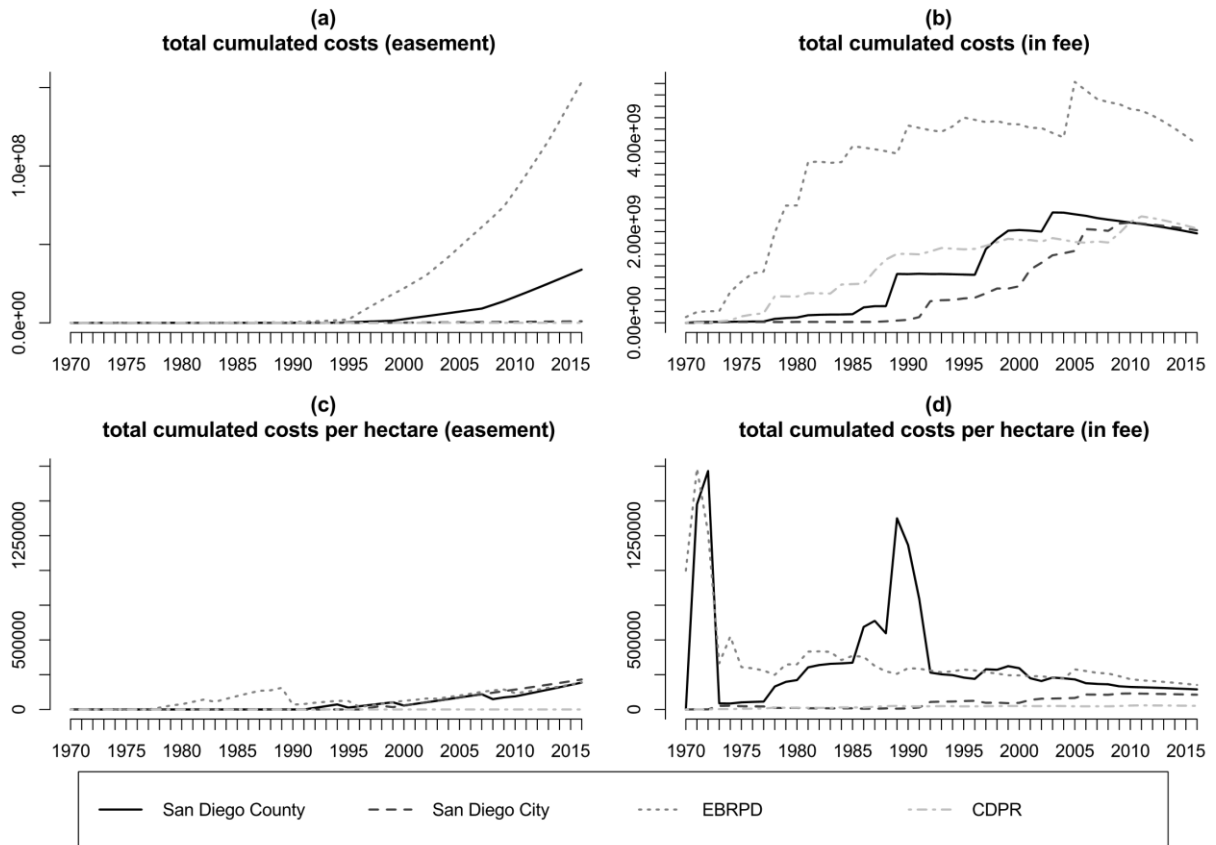


Figure 5.3: Total cumulative costs and total cumulative costs per hectare of implementing and managing easements or in-fee conservation areas.

As both options, however, are based on hugely different amounts of conservation area—i.e., 965 ha in easements and 136,198 ha in-fee in 2016—a comparison of total costs delivers an incomplete picture. On a per hectare basis, it can be seen that while easements start generating relatively low costs that are increasing over time, the in-fee start relatively high (even when the huge fluctuations in early years are neglected) and decrease over time. Surprisingly the total cumulative costs per hectare for easements increases over time for all agencies, while in-fee total cumulative costs per hectare consolidate in later years, after being relatively volatile in early years.

There are two governance modes that are outstanding in the development of their costs per hectare, EBRPD for easements and San Diego county for in-fee alternatives. EBRPD follows two exponential periods, the first until 1992 and the later still ongoing. San Diego county shows two peaks of investment, first in the 1970s and the second in the 1990s.

5.3.3 Development of costs of management

Substantial fluctuations in management costs (i.e., costs for implementing and running conservation measures) only arise in early years of conservation on a per hectare basis (Figure 5.4). At the same time, the total amount of management costs is marginal and thus seems negligible for three of the agencies, with the exception of EBRPD easement and the California Department of Parks and Recreation in-fee. Although management costs are lower on a per hectare basis for in-fee than for easements, total costs of conservation are not majorly driven in either case by the management costs.

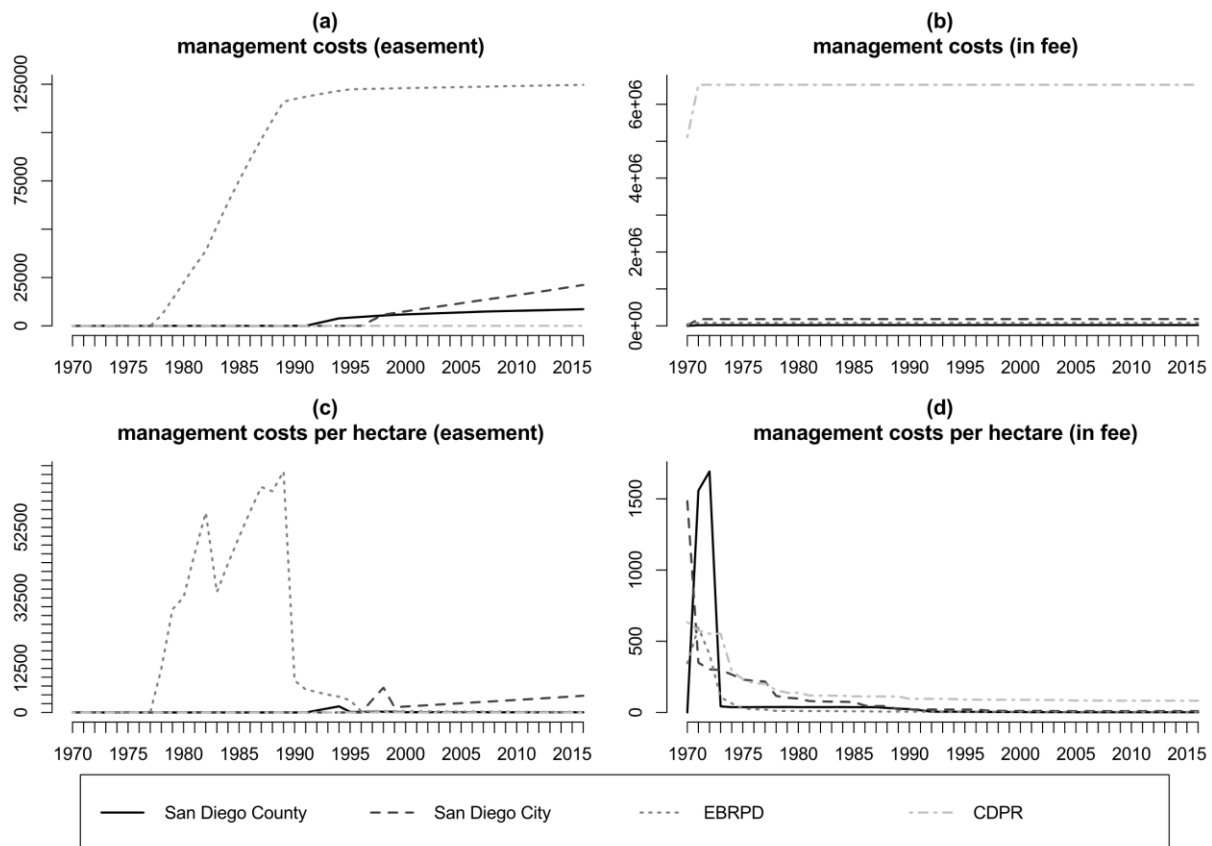


Figure 5.4: Development of cumulative management costs with easement management (a) and cumulative management costs with in-fee managed areas. Management costs are calculated according to Eq. (D.2) (see Appendix D.5) separately for each area and cumulative over time for each agency. Development of per hectare management costs with easement management (c) and per hectare management costs with in-fee management (d). While all agencies do at least manage some easements, no data was available for the CDPR in the considered timeframe.

Annual management costs (Figure 5.5) also show a generally non-comparable development over time. While, on a per hectare basis, management costs of easement in 2016 were between 100 and 900 \$ per ha, with in-fee these ranged between 1 and 25\$.

Also, the maximum values, in a total and per hectare, show substantially different general development patterns.

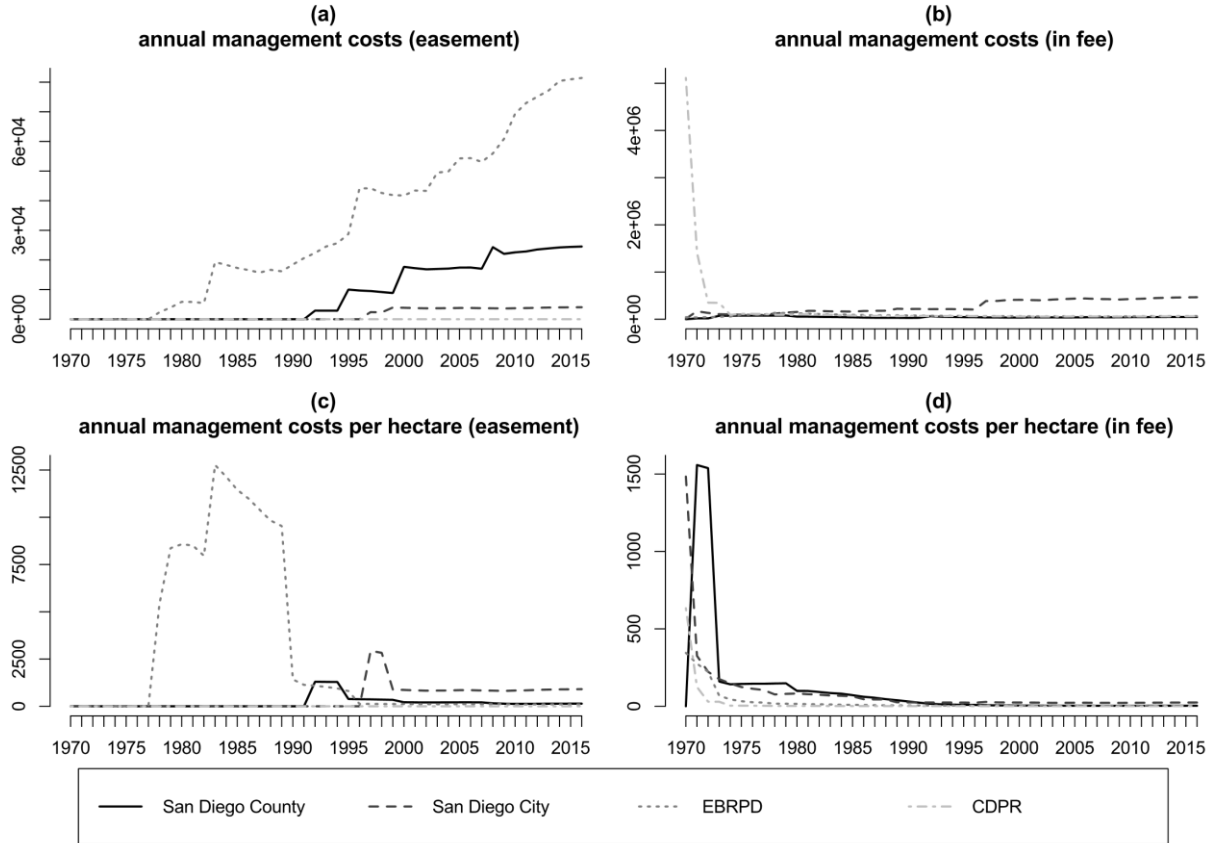


Figure 5.5: Annual management costs and annual management costs per hectare for each management option.

5.3.4 Sensitivity analysis

We find that variation in purchasing prices and opportunity costs have a rather large influence on the different cost measures and especially total and per hectare costs (both annually and cumulative), while the impact from variations in interest and discount rates, income, and transaction costs is rather small and even negligible (see Appendix D.5).

5.4 Discussion

We present an economic framework to analyse the costs of acquisition of land for conservation on private versus public land and how they are influencing decisions. Overall, total annual costs, costs of management, and total annual management costs all show that

easements surpass the costs of in-fee properties in the long run, with an exponential growth while costs of in-fee are and higher for purchase but decrease substantially after a decade. This suggests that from an economic point of view, easements are not a preferable option, mainly due to the high share of recurrent costs. We, furthermore, find that management costs are not a major discriminating factor between both governance modes as they are of relatively low importance in relation to other cost components.

Previous studies have shown a diversification of land acquisition options since 1990s. Easements or other public–private partnerships have been suggested because some ecosystems only occur in current private land not available for sale (Nolte, 2018). However, more information is needed on this incentive-based strategy to invest in acquiring partial interest in private land for conservation purposes (Merenlender et al., 2004). We find that from an economic point of view, easements are not a good option in the long run mainly due to the high share of recurrent costs, suggesting that, depending on economic factors like interest and inflation rates, buying land for conservation is cheaper than compensating landowners (Schöttker et al., 2016). These increased costs of easements could be because of their purchases at a time of rapid increase in land costs (Abraham and Hendershott, 1992). In our analysis, we only accounted for costs of purchase and management in conservation easements. However, if easements are also productive land and result in private purchase, these factors could be counted in the cost to better reflect the economics of these lands. Further, it could be that easements are economically viable when land prices are stable or increasing at a slower pace than in California. California housing market has plummeted since 1980s and with it the value of land (Quigley and Raphael, 2005). The easement option could also work if there are market controls on land prices aimed at lowering conservation land costs in comparison to productive and development land.

The economic options are surprisingly similar across management agencies; however, they differ per unit of area. Each of the governance modes have different costs and costs structure, and the framework is only limitedly applicable to compare cost developments between agencies, given heterogeneity in conservation conditions (Santos et al., 2014b). This is because different agencies have different missions and goals, and also target different types of land fee or geographical regions, and are able to access different funding sources. Different starting points of individual conservation areas also make cost comparison difficult, for example, due to a different total conservation time frame or

discounting of historical cost components. However, this is the reality of most conservation land acquisition, as it is often not possible to acquire all land at the same time. This also complicates the comparison of costs on an annual basis, which is additionally driven by economies of scale and maybe other scaling factors. It must be noted that landowners might engage in rent seeking behaviour by overstating their opportunity costs and thus causing increasing purchasing prices and compensation requests (Ferraro, 2008). How far this behaviour is prevalent in the study area or how agencies counteract is an open question.

Our description of costs and their development over time for conservation in different governance modes and conservation agency settings in California highly depends on available data. The quality of data on easements hampered a deeper quantitative analysis, while still unveiling the general development of costs. Increased data availability can improve cost estimation and reveal governance mode dependencies. Upon that, the conceptual character of the used cost estimation framework allows for an understanding of cost components and development for a subset of conservation actions, while having the potential for improvement by better information on cost characteristics. The costing framework is generally applicable to describe the individual cost patterns of one agency at a time and to compare management modes.

Easement implementation, although potentially beneficial from an ecological and social perspective, is relatively costly. Investing the same monetary amounts in land purchases can increase ecological benefits, by keeping the economic costs constant and increase efficiency. Local legislation to allow and incentivize long-term investments into land acquisition would also reduce long-term budgetary burden on national and state finances, let alone the many economic reasons for and spill-overs from nature conservation (Balmford et al., 2002). In how far implementation into legislation or conservation planning is happening, in general or in special in California, is however an open issue.

Acknowledgements

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Chapter 6

Summary, discussion and conclusion

6 Summary, discussion and conclusion

The loss of biodiversity is a threat to human wellbeing (MEA, 2005). Key drivers for biodiversity loss are anthropogenic climate change and human land appropriation, both of which threaten and destroy valuable habitats for endangered species (IPBES, 2018). In particular, threats to biodiversity due to land use change and habitat fragmentation pose a problem. Changing climatic conditions and subsequent influences on species and habitats aggravate these issues.

To address the problems, various policy instruments have been introduced to protect biodiversity and habitats in different ways. Among those instruments, land purchase and landowner compensation for biodiversity conservation are selected policy options for conservation agencies and governments to choose from. Both approaches also represent different governance modes for implementation of biodiversity conservation. A key issue for conservation actors however is to choose the cost-effective governance mode, i.e. to select the policy instrument, which provide the best possible ecological outcome for a given budget (Ando et al., 1998; Ferraro and Pattanayak, 2006).

This thesis addresses the question in how far the cost-effectiveness of the two governance modes (1) land purchase for the implementation of conservation (*buy alternative*) and (2) compensation payments to land owners for voluntary implementation of conservation measures (*compensation alternative*) is influenced by economic, ecological and climatic factors, and tries to answer this question conceptually and empirically.

In the following, Section 6.1 summarizes key results of this work. In Section 6.2 existing research gaps, newly generated research questions, and scientific outlook are discussed, followed by policy recommendations in section 6.3. Section 6.4 presents the conclusion of this work.

6.1 Summary of results

A conceptual investigation of the cost-effective governance mode choice for biodiversity conservation and respective factors of influence on the governance mode choice was conducted in Chapters 2 and 3 of this work. This analyses can be considered the first studies addressing the question of “land purchase” vs. “landowner compensation” on a

theoretical basis. The analysis broadens the conceptual understanding of the make-or-buy decision provided by Chapter 2, and extends it into the field of climate change adaptation in Chapter 3.

Selected insights from the conceptual analysis were analysed in empirical case studies in Chapters 4 and 5, which eventually support specific theoretical insights. By conducting two separate case studies in different economic, cultural and historical settings, the question for a cost-effective governance mode choice was addressed from different perspectives. Chapter 4 analyses the cost-effectiveness of governance modes in a small and relatively short-lasting conservation project in Schleswig-Holstein, Germany, and develops an analytical framework to estimate and compare the costs developments and cost-effectiveness of both governance modes empirically. Chapter 5 assesses the historic development of costs and cost-effectiveness for state-wide conservation efforts in California, USA, and applies the underlying framework to accommodate data from multiple conservation actors.

Based on the mentioned approaches, this thesis generates a number of key insights into the optimal choice of governance modes for biodiversity conservation and the factors of influence on the cost-effectiveness of different governance modes in the implementation of conservation measures.

1. Governance alternatives for conservation agencies, when implementing biodiversity conservation in cultural landscapes, can in principle be described by two main options: (1) land purchase and implementation of conservation measures on that land by themselves or through third party contractors (*buy alternative*), and (2) compensation of landowners for their voluntary provision of conservation activities on their land with a compensation payment (*compensation alternative*).
2. Land purchase generates high upfront, one-off costs (e.g. in form of purchasing expenses) and relatively low recurring costs, while landowner compensation generates relatively high recurring costs (e.g. repeated compensation payments and transaction costs) with hardly any upfront or one-off costs.
3. Land purchase is more cost-effective compared to landowner compensation in conservation projects with a longer time frame, while for shorter projects compensation payment based conservation is preferable. Both findings go against common findings in literature, which suggest strictly higher costs of land purchase.

compared to landowner compensation also in the long run (cp. Adams et al., 2010; Knight et al., 2011).

4. Costs-effectiveness of implementing conservation activities is strongly influenced by land prices; hence, the timing of land purchase and the start of conservation activities is an important factor to consider against the background of variations in land market prices.
5. High interest rates render land purchase more cost-effective compared to landowner compensation due to their influence on discounting of future cost streams and land prices.
6. The distinction of governance options into the *buy alternative* and the *compensation alternative*, which originate in the analysis of a mainly European conservation environment, can be transferred to different legislative frameworks, which lead to similar results and cost development patterns, e.g. when looking at in-fee managed land vs. easement conservation in the United States.
7. Land purchase generates a relatively rigid spatial network of conservation areas in the landscape, while landowner compensation causes spatially more flexible conservation area networks. An advantage of a rigid conservation network can e.g. be a decreased habitat turnover rate and low recurring conservation costs if areas are kept under conservation for long periods.
8. In presence of climate change, a flexible conservation network generated by landowner compensation is advantageous in terms of cost-effectiveness compared to a more rigid conservation network caused by land purchase due to the easier adaptability of conservation locations to changing climatic conditions.
9. Against the background of climate change, besides the choice of governance mode also the choice of an actual implementation and spatial selection of conservation locations has a noticeable influence on cost-effectiveness.

These key insights hold beyond the individual chapters and publications contained in this work and can be used to serve as a basis for further research and actual policy recommendations. In this sense, the empirical findings in particular support the theoretical and conceptual insights.

6.2 Research gaps, new research questions and outlook

Although further research is still necessary to understand potential governance modes for conservation better, this work addressed a variety of objectives in the domain of cost-effective choice of governance modes for biodiversity conservation and corresponding policy design. Hence, it is a matter of further research to address and answer research questions related to the general theme of this work.

Literature states in research on transaction cost economics and the make-or-buy decision that not only pure modes of governance of internal provision and external procurement are valid choices (e.g. Williamson, 1991, 1989). However, hybrid modes of governance laying in between those pure implementations are possible and are potentially more cost-effective governance mode choices (cp. Lemos and Agrawal, 2006; Muradian and Rival, 2012). The current literature on governance mode choice in biodiversity conservation does not adequately address this issue. It is an open question, which hybrid modes of governance (beyond a simple case-wise combination of both approaches on a landscape scale) exist, and how they can be utilized for conservation purposes, and what the implications for cost-effectiveness are.

Internationally, a huge variety of different governance modes as well as institutional and legislative frameworks to organize and implement conservation activities exists, both publicly and privately. A proper comparison of these legislative and institutional frameworks, in particular with respect to cost-effectiveness and ecological performance might foster the understanding of the optimal choice of governance modes for conservation (cp. Nolte et al., 2019).

Another aspect of the analysis of the governance modes for conservation regards equity issues. The utilization of alternative decision criteria (e.g. the producer surplus generated by different governance mode; the influence on landowner income distribution, or others measures) could be used to derive more equitable or fairer governance mode choices and implementation strategies.

This work conceptually showed that the presence of climate change indeed could influence the absolute and relative cost-effectiveness of different governance modes. However, an open question still is how big the influence of climate change on the cost-effectiveness of different governance modes is – quantitatively in terms of monetary savings potential or potential ecological performance. A challenging aspect also of this question is the proper

definition of relevant governance modes to choose from, which might depend on location, targeted species or ecosystem or other factors.

Beyond the domain of governance mode choice, further interesting research questions emerged in particular with respect to ecological economic modelling.

In ecological-economic modelling – used to research topics like cost-effectiveness of conservation measure design conceptually – it is commonly assumed for opportunity costs to be of heterogeneous nature with no underlying structural cost differences on the landscape scale. In real life situations opportunity cost heterogeneity might very well be accompanied by large-scale structural cost differences, e.g. through the presence of differently fertile and thus profitable agricultural land or the existence of human agglomeration in form of smaller or larger settlements. In how far such opportunity cost differences influence the cost-effective choice of conservation measures, their governance mode, or the respective conservation site selection strategy is an open question.

In addition, the influence of model design decisions, such as the spatial interpretation of landscapes in potentially different geometric forms (e.g. regular triangular or hexagonal grids, or irregular Voronoi partitions), is not researched. A potential bias of landscape representation geometry on the outcome of spatial ecological models (e.g. species movement patterns; cp. Dunn, 2010; Holland et al., 2007), might very well have influences on the cost-effective choice of governance modes. An appropriate model design and geometry selection might be important depending on e.g. specific model requirements such as species characteristics, or the necessity to represent real life landscapes appropriately.

In particular, the simulation software, which was developed and used throughout the research for this work, especially underlying Chapters 2 and 3, can serve as a blueprint and framework for future ecological-economic modelling. The simulation software already contains a variety of integrated ecological, economic and climatic models particularly developed for research conducted in this thesis. An extension or adaptation in terms of e.g. ecological or climatic models, landscape representation geometries, or conservation implementation strategies however can provide fertile grounds for the development of novel conservation policy design or cost-effectiveness assessments. Further development and research however is necessary.

6.3 Policy recommendations

Building on the findings of this thesis, a set of recommendations for political decision makers and conservation actors can be derived.

Against the insight that especially land purchase is more cost-effective compared to landowner compensation in long-term projects, land purchase should be considered a more viable option in the implementation of biodiversity conservation, especially if long-term project targets are to be achieved. This implies that on a policy level appropriate funding should be provided to conservation actors, i.e. conservation agencies, NGOs, or governmental conservation bodies (cp. Waldron et al., 2013). Additionally, legislation should be adapted to enable conservation actors in the first place to purchase land for conservation purposes (cp. Doremus, 2003; Lausche et al., 2013). On conservation actor level, awareness should be raised to the fact that land purchase can be a cost-effective alternative to landowner compensation and thus should be considered in the decision making process. It must however be noted, that land purchase for conservation purposes is seen critically by landowners, especially farmers which rely on the economic usability of their land for agricultural purposes for them or later generations (De Schutter, 2011; Hall et al., 2015). Thus, landowners might not be in favour of policies suggesting an increased purchase of land.

An alternative recommendation is to promote more long-term contract forms for landowner compensation. Such contracts, which essentially can be interpreted as a hybrid governance between landowner compensation and land purchase, could be the basis for spatially rigid conservation networks, leading to similar results in cost-effectiveness as land purchase, while still granting some of the flexibility generated by landowner compensation (Gerling and Wätzold, 2019). Such contracts might be a reasonable alternative for land purchase, especially given the potentially limited willingness of landowners to sell their land to conservation actors as mentioned before. Long term contracts can then provide increased economic security and plannability for landowners compared to shorter term compensation contracts, while at the same time delivering the long-term stability and network rigidity of land purchase for the conservation actor. However, more research is necessary in this field, as it for now remains unclear, in how far extended contract length influences conservation costs and thus cost-effectiveness.

This work showed beyond the general aspects regarding the cost-effective choice of governance modes of biodiversity conservation the existence and relevance of various factors of influence on this decision. Various ecological, economic and climatic factors interfere with the cost-effectiveness of either governance mode. Hence, policy recommendations regarding those factors can be made to enable the design of novel, biodiversity conservation policies.

It was shown that climate change speed influences the cost-effectiveness of the governance modes for conservation. In particular, the cost-effectiveness of different selection strategies for the location of conservation areas (compare Chapter 3) was influenced by climate change speed. In this context, landowner compensation was found to be favourable over land purchase if climatic conditions change rapidly, due to the flexibility of the conservation network. Novel policies could include, besides ecological goal definitions within a certain governance framework, the definition of climatic characteristics to ensure appropriate conservation measure implementation and governance mode selection in the presence of climate change, specifically focussing on compensation based governance modes.

Implementation strategies which included climatic characteristics for conservation location selection (i.e. the suitability of a certain conservation area for a target species, and its change over time), were found to be most favourable in terms of cost-effectiveness when implemented based on landowner compensation. It is thus recommendable to include climate characteristics into the eligibility criteria for participation in AES. Most current AES in principle allow every landowner to participate and to be eligible for receiving compensation payments. In future policy design, a predefined climate suitability measure could be introduced to prioritize areas with higher suitability for a given conservation target over areas with lower suitability in order to increase cost-effectiveness or to allow increased payments or bonuses to owners of particularly suitable land.

6.4 Conclusion

Based on the presented research it is evident that the study of governance of biodiversity conservation is an important endeavour, which brings along a deeper understanding of the integration of cost-effective governance mode choice in the context of biodiversity

conservation and thus provides grounds for improvements in conservation practices, conservation governance and policy design.

A conceptual understanding of the *make-or-buy decision* in the setting of biodiversity conservation was shown to be important, as it allows the identification and translation of relevant governance modes into the field of biodiversity conservation. In addition, the understanding of factors of influence on the cost-effective choice of governance mode alternatives is important for transferring governance mode choice into practice.

This work helped to understand these relationships, in particular the actually available governance modes for biodiversity conservation, the effects of ecological, economic and climatic factors of influence and the cost-effective choice of governance modes in selected situations, analysed conceptually as well as empirically. However, as mentioned before, more research is necessary to understand the interplay between governance mode choice and a cost-effective implementation of biodiversity conservation even better. For this, the underlying work has provided a starting point.

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Appendix

A. Appendix to Chapter 2

This chapter has previously been published as part of:

Schöttker, O., Johst, K., Drechsler, M., Wätzold, F., 2016. Land for biodiversity conservation — To buy or borrow? Ecological Economics 129, 94–103.

In our model, the number of purchased patches is determined by simulating the buying process of the agency (see Section 2.2.3.1). The patch number depends on the randomly drawn profits π_i respectively prices of the patches (Eq. 2.1), and the available budget B_0 . Alternatively, the number of purchased patches can be determined analytically.

Assume an initial budget B_0 and equidistantly separated profits π_j in the interval $(\bar{\pi} - \sigma, \bar{\pi} + \sigma)$ — in contrast to the randomly distributed profits π_j in our simulation model. Moreover, assume that the agency offers a price π_j/r to buy a certain patch j and all landowners with an annual profit equal or less than π_j will accept that offer. Then we can write the budget constraint as follows:

$$B_0 = j_{buy} \times \pi_j \times \frac{1}{r} = j_{buy} \times \left(\bar{\pi} - \sigma + \frac{2\sigma}{N} j_{buy} \right) \frac{1}{r} \quad (\text{A.1})$$

with j_{buy} the number of purchased patches and $\left(\bar{\pi} - \sigma + \frac{2\sigma}{N} j_{buy} \right) \frac{1}{r}$ The price (proportional to the periodical profit of patch j , Eq. (2.1)) of patch j . By solving Eq. (A1) for j_{buy} , the number of purchased patches, is

$$j_{buy} = \frac{[(\bar{\pi} - \sigma)^2 + (8B_0 r \sigma / N)]^{\frac{1}{2}} - (\bar{\pi} - \sigma)}{4\sigma / N} \quad (\text{A.2})$$

Similarly, to the buying decision, in our model the number of extensively managed patches in the borrow alternative is determined by simulating the borrowing process (see Section 2.2.3.2). However, we can also calculate the number of these patches analytically assuming equidistantly separated profits π_i in the interval $(\bar{\pi} - \sigma, \bar{\pi} + \sigma)$. Considering that landowners are able to keep a share f of the generated profits on their land, and that only a part of the landowners P_{part} for whom it would be profitable is actually willing to participate in a payment scheme, the budget constraint reads:

$$B_{borrow} = j_{borrow} \times \pi_j \times f \times P_{part} \times \frac{1}{r} \quad (A.3)$$

By solving Eq. (A3) for j_{borrow} , we obtain the number of extensively managed patches in the borrow alternative:

$$j_{borrow} = \frac{\left[(\bar{\pi} - \sigma)^2 + (8B_0 r \sigma / N) / (f P_{part}) \right]^{\frac{1}{2}} - (\bar{\pi} - \sigma)}{4\sigma / N} \quad (A.4)$$

A comparison of the results from the simulation and analytical approach shows that both methods result on average in the same or a very similar number of extensively managed patches for both the borrow and buy alternatives (see Fig. A.1). The slight deviations are due to our assumption of randomly drawn profits compared to assuming equidistantly separated profits in the analytical model. By providing this comparison, implementation errors for the economic model can be effectively ruled out. The analytical approach (Eqs. A.2 and A.4) has the advantage of providing deeper and more general theoretical insights into the economic part of our model, e.g. in being more explicit in the functional relationships between extensively managed (borrowed or bought) patches and budget, interest rate, participation willingness and profit. However, our work links the economic analysis to a numerical ecological model (Section 2.2.4). This model requires two input parameters from the economic analysis (Sections 2.2.3.2 and 2.2.3.3): (1) the number of extensively managed patches, and (2) the amount of habitat patch turnover.

For determining the first input parameter both the analytical approach and the simulation approach can be used as they generate very similar numbers (Fig. A.1). The simulation approach is nevertheless necessary because explicitly simulating the random decisions of the landowners from period to period (see Section 2.2.3.2) is necessary to quantify the second input parameter. Habitat patch turnover in our model is affected by the participation probability and the integer number of habitat patches which cannot perfectly fit the available budget generating a certain budget leftover in each period. This budget leftover is additionally available in the next time period, thereby slightly increasing the budget in this particular period and impacting patch turnover (note that only at high habitat patch numbers this impact becomes smaller and turnover is increasingly determined by the participation probability).

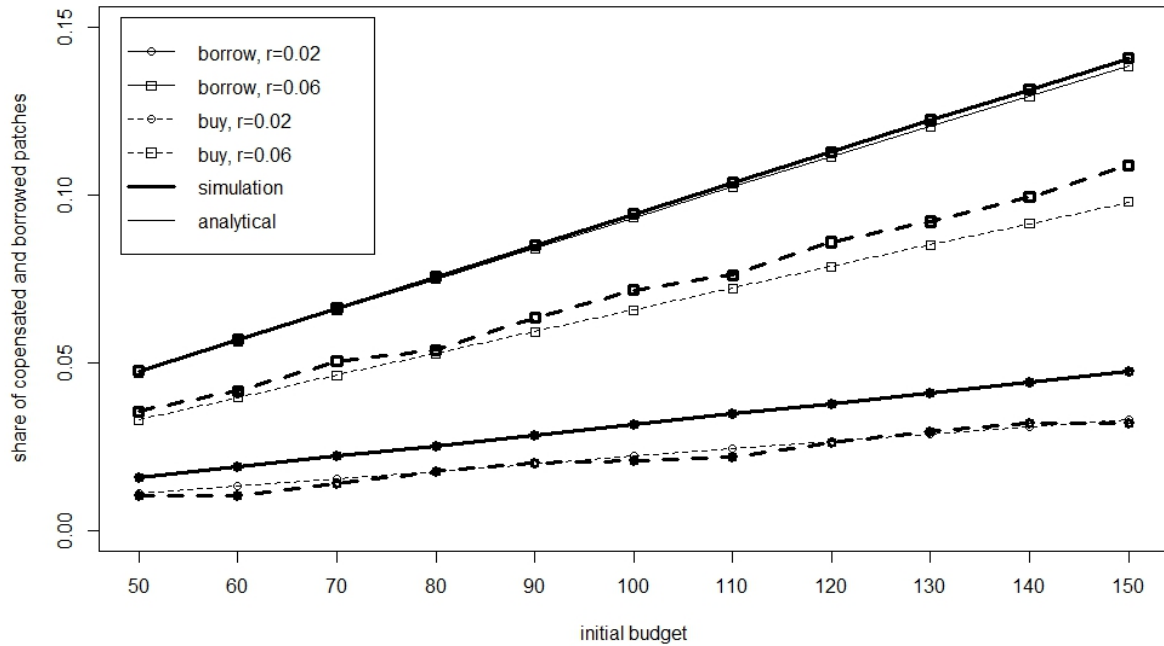


Figure A.1: Comparison of share of patches under conservation in the buy and borrow alternative determined by the analytical approach (Eqs. A.2 and A.4; thin lines) and the simulation approach (bold lines, identical with Fig. 2.1 in the main text) as a function of initial budget B_0 and interest rate r .

Importantly, even if we determined both patch number and patch turnover analytically an analytical calculation would not benefit our main modelling goal: understanding the combined ecological economic model. The reason is that the ecological part of the model, especially Eq. (2.6), can be solved only numerically (the double sum in Eq. (2.6) cannot be interpreted without numerics). This means that the behaviour of the combined ecological–economic model can be analysed only numerically — even though a part of it can be solved analytically.

B. Appendix to Chapter 3

This chapter is published as part of the working paper:

Schöttker, O., Wätzold, F., 2020. Climate change and the cost-effective governance mode for biodiversity conservation. MPRA Paper No. 99049, posted 18 Mar 2020.

<https://mpra.ub.uni-muenchen.de/99049/>

B.1 Distance calculation

We define the distance between the midpoints of any two patches i and j as follows:

$$d_{ij} = \begin{cases} 1 & \text{if: } |x_i - x_j| = 1 \text{ and } |y_i - y_j| = 1 \\ \sqrt{(x_i - x_j)^2 + (y_i - y_j)^2} & \text{for all other } i \text{ and } j \end{cases} \quad (\text{B.1})$$

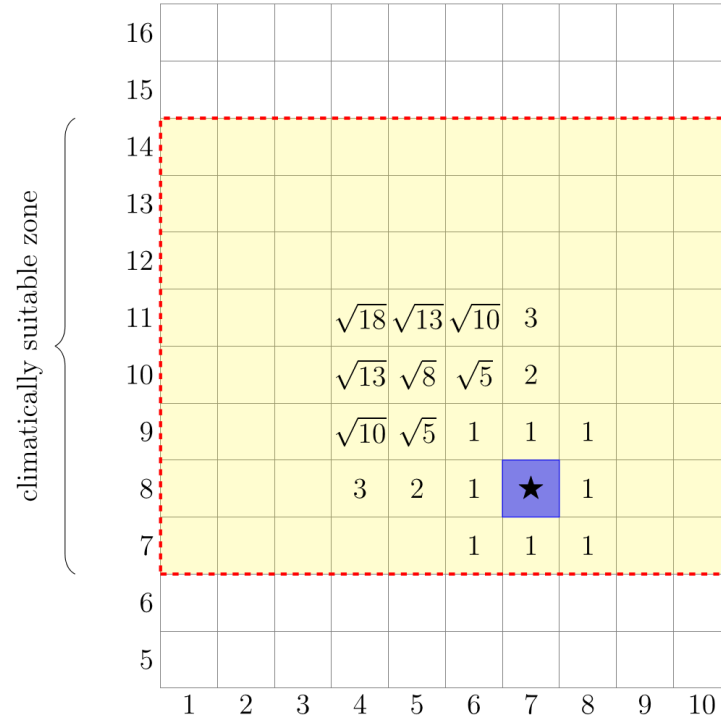


Figure B.1: Distance between two patches calculated by Eq. (B.1). The yellow-shaded and red-framed area represents the climatically suitable zone (CSZ). The blue-shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. Numbers indicate the distance of the respective patch to the highlighted blue patch. Note that the distance for all patches directly neighbouring the blue patch is 1.

We chose this method of distance calculation, as it seems agreed upon in the literature and is relatively easy to handle in the implementation of the model. The exception made for the distance of diagonally neighbouring patches to calculate as 1 instead of $\sqrt{2}$ results in an overestimation of species dispersal, especially if the dispersal distance of a species is only 1. Without this exception, in this case a dispersal would only be possible to vertically and horizontally neighbouring patches but not to the diagonally neighbouring ones, causing distortions and model artefacts.

B.2 Patch selection strategies

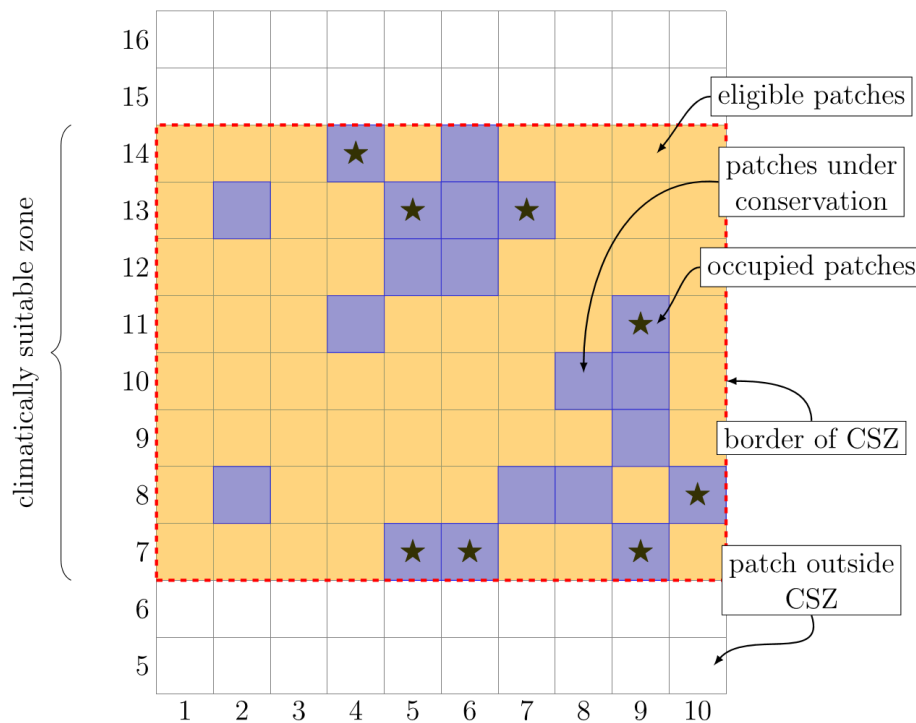


Figure B.2: ‘Prize prioritization strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘prize prioritization strategy’. The blue-shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species.

Figure B.2 illustrates a conservation network in the model landscape, created by the ‘prize prioritization strategy’. Within the CSZ potential conservation areas are located. Patches marked with a star are patches occupied by the target species.

Figure B.3 illustrates a potential conservation network generated by a ‘species abundance targeting strategy’. Conservation areas are clustered together around occupied patches. Patches eligible for future extension (i.e. newly bought or compensated areas) represent all patches within the dispersal distance of the target species. All yellow shaded areas, although within the CSZ, are outside the dispersal distance of the target species and thus not eligible for conservation.

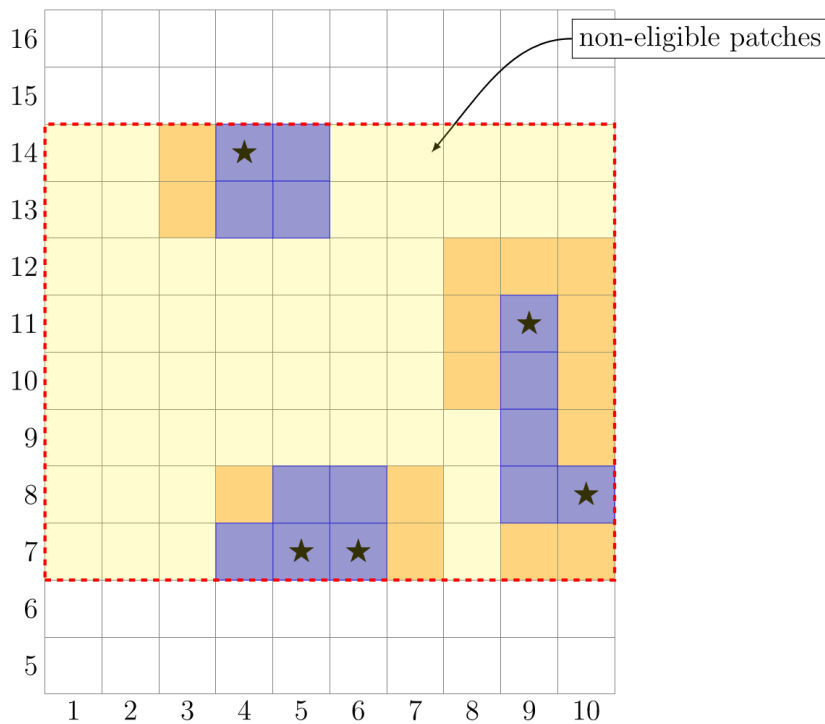


Figure B.3: ‘Species abundance targeting strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘species abundance targeting strategy’. The yellow-shaded areas represent patches, which are non-eligible for selection in this strategy. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species.

Figure B.4 visualizes a habitat network created by a ‘climate suitability prioritization strategy’. Patches cluster around the centre of the CSZ, representing the area with highest climate suitability for the target species. Due to the closer proximity of conservation area location, the complete network has a higher degree of connectedness, and the target species is more likely to be able to migrate to other conservation areas in the network, compared to the price prioritization strategy (Fig. B.2). For simplicity, we ignored the eligibility differentiation made in combination of climate suitability and conservation opportunity

costs per patch as described and used in the simulation model, and only depicted the climate differentiation aspect here.

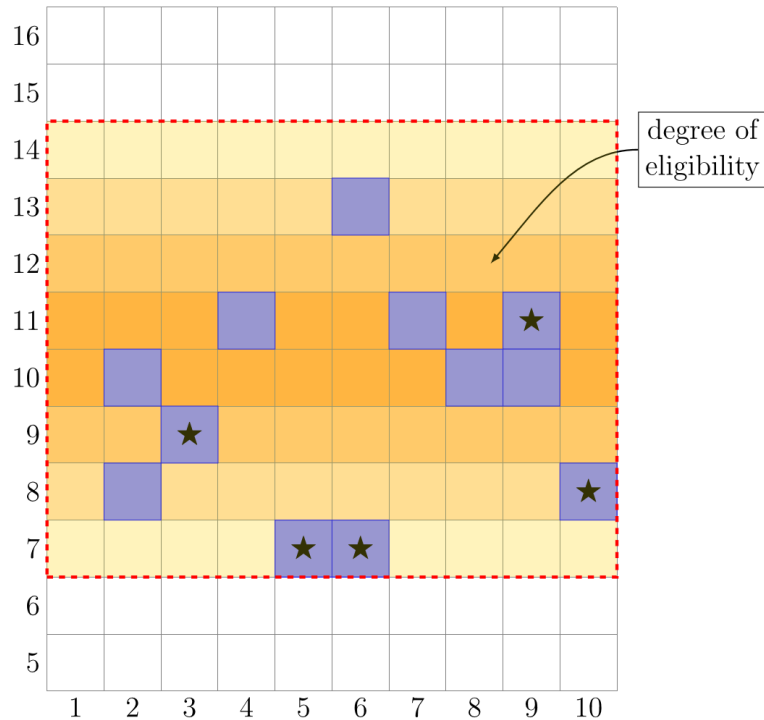


Figure B.4: ‘Climate suitability prioritization strategy’. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘climate suitability prioritization strategy’. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species.

Figure B.5 illustrates the ‘climate change direction prioritization strategy’. This strategy locates newly generated patches in the more northern range of the CSZ compared to the ‘climate suitability prioritization strategy’. Patches selected closer to the northern border of the CSZ are located within the CSZ for the longest time. This is due to the northwards propagation of the CSZ through the landscape as a result of climate change. If a patch close to the northern border is selected for conservation, it takes longer for the CSZ to move across this patch and to eventually drop out of the CSZ, compared to a patch closer to the southern border, which drops out of the CSZ earlier. This results in a generally more stretched out conservation network as patches can potentially be located throughout the whole CSZ, while being added most likely at the northern edge, compared to the climate suitability prioritization strategy.

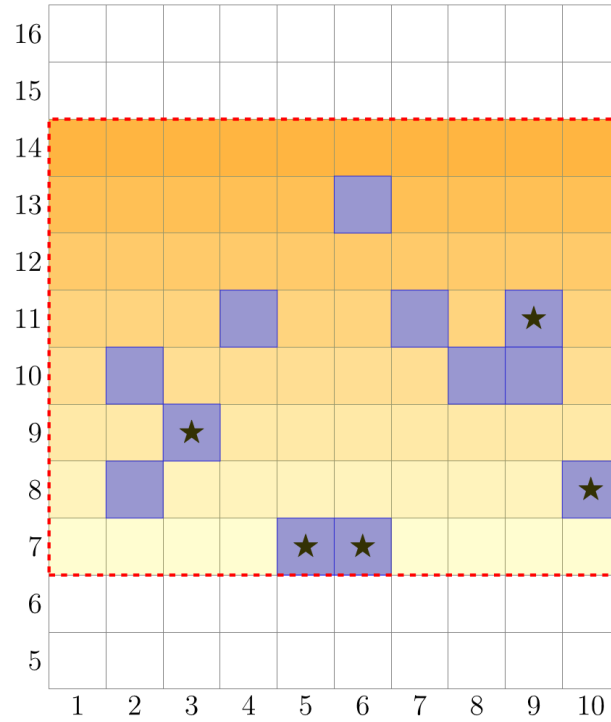


Figure B.5: 'Climate change directional prioritization strategy'. The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the 'climate change directional prioritization strategy'. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species.

B.3 Patch restriction effect

Decreasing the climate suitability threshold parameter cs^{thr} leads a decreasing extend of the CSZ (see Fig B.6.a; visualized for the 'species abundance prioritization strategy'). In any strategy, this can lead to an exclusion of otherwise potentially eligible patches from the selection mechanism. The result is a patch restriction effect (see main paper, Section 3.4) leading to an increased necessity to select patches in the remaining (smaller) CSZ, which in consequence are likely to be more expensive. Additionally, a connectivity effect can be observed, as the selected patches are closer together in case of a smaller CSZ and thus more likely to be well connected.

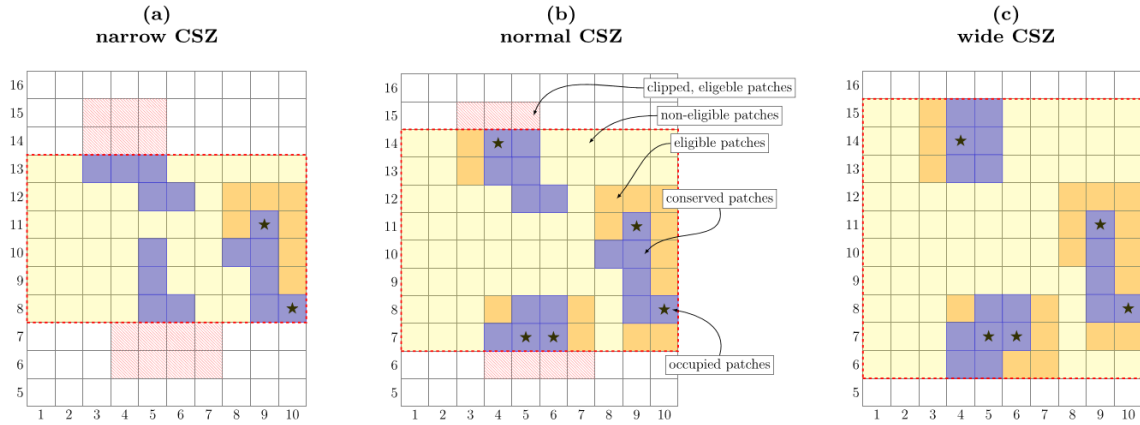


Figure B.6: Patch restriction effect in the ‘species abundance prioritization strategy’ due to changes in CSZ sizes due to varying cs^{thr} . The red-framed area represents the climatically suitable zone (CSZ), within which the orange-shaded area represents patches eligible for patch selection in the ‘species abundance prioritization strategy’. The degree of orange depicts the level of eligibility of a particular patch; darker-shaded areas have a higher eligibility than lighter-shaded areas. The blue shaded area represents a patch selected for conservation; the star-symbol indicates that this patch is occupied by the target species. The red-shaded areas represent patches, which could have been selected by the respective strategy, if the CSZ was large enough, but in fact are restricted in eligibility by the patch restriction effect.

B.4 Influence of economic and ecological variables

Additional to the sensitivity analysis for changes in climatic model parametrization presented in Section 3.4 of the main paper, we performed a sensitivity analysis with respect to changes in ecological and economic parameters, presented in the following. The corresponding parameter values can be seen in Table B.1.

Regarding the impact of interest rates on the cost-effectiveness of the different GM-PSS pairs we find that with decreasing interest rates, the cost-effectiveness is reduced in all GM-PSS pairs. These result is expectable, as reductions in the parameter eventually decreases the CA’s possibility to buy or compensate new patches, either by reducing their available budgets or by increasing patch prices or compensation requirements through increases in the discount factor (compare Schöttker et al. 2016). A graphical analysis can be found in Figure B.7.

Table B.1: Overview about the parametrization value and value ranges specified for computation of the Monte-Carlo-Simulation and used in the sensitivity analysis for non-climatic factors.

Parameter name	Parameter description	Parametrization Values Range	Base case
Economic parameters			
\overline{OC}	Mean opportunity costs in the landscape	$\in \{0.8, 1.0, 1.2\}$	1.0
\overline{tc}^{buy}	Mean transaction costs of purchasing a patch	$\in \{0.8, 1.0, 1.2\}$	1.0
\overline{mc}	Mean monitoring costs	$\in \{0.08, 0.10, 0.12\}$	0.1
r	Interest rate	$\in \{0.01, 0.015, 0.02, 0.025, 0.03, 0.035, 0.04, 0.045, 0.05\}$	0.03
Ecological Parameters			
ν	Emigration rate from any patch	$\in \{90, 100, 110\}$	100
δ	Dispersal distance of the target species	$\in \{1, 2, 3\}$	1

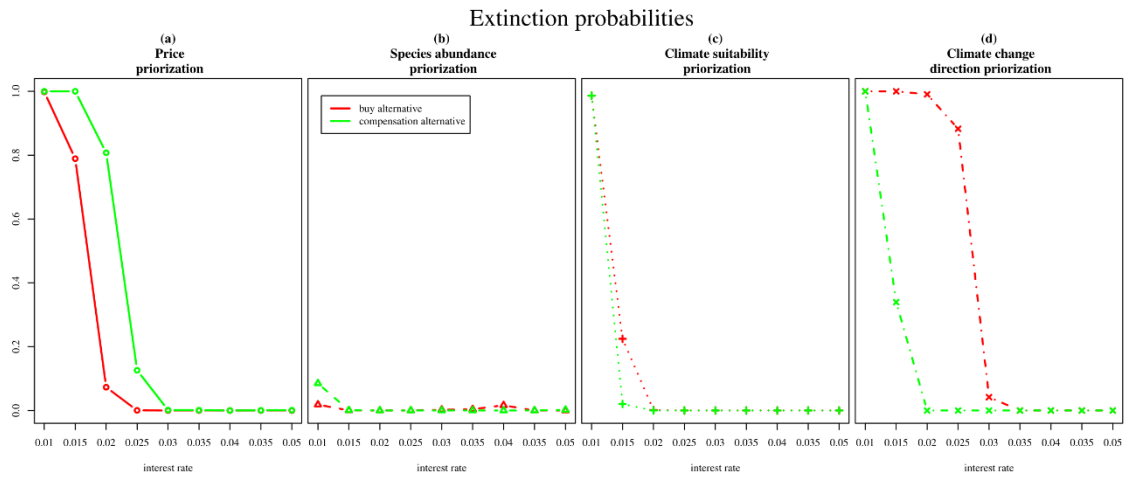


Figure B.7: Extinction probabilities of the different GM-PSS pairs with changing interest rates, resulting in changes in available budgets and discount rates. The red line represents results for the buy alternative, the green line for the compensation alternative.

A direct increase of patch prices (by increasing \overline{OC}) has a negative effect on the cost-effectiveness of the GM-PSS pairs (see Figure B.8).

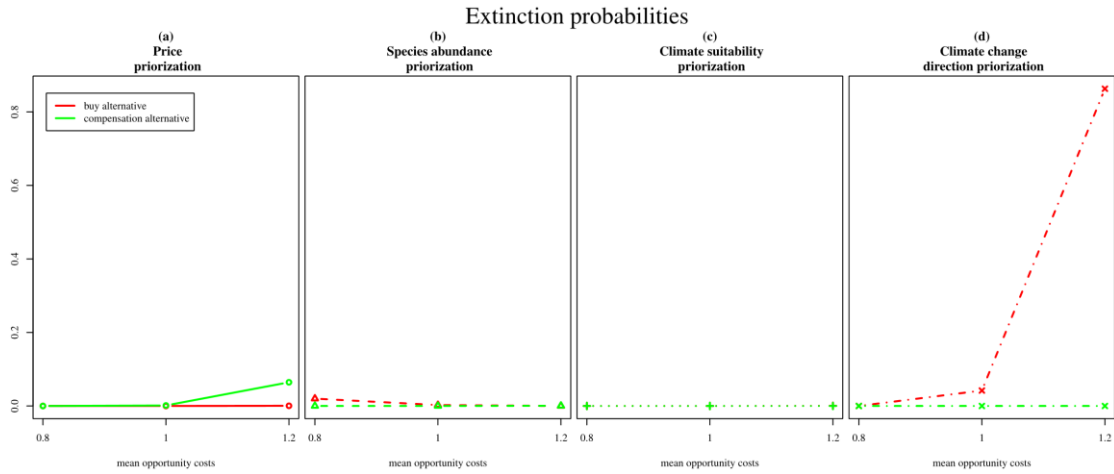


Figure B.8: Extinction probabilities of the different GM-PSS pairs with changing mean opportunity costs. The red line represents results for the buy alternative, the green line for the compensation alternative.

Variations in the emigration rate (v) did not result in observable changes of the extinction rates of the GM-PSS pairs (Fig. B.9), and increasing the dispersal distance (δ) slightly reduced the extinction rate of the *buy alternative*'s 'climate change direction prioritization strategy', while the other GM-PSS pairs remained unaffected (Fig. B.10).

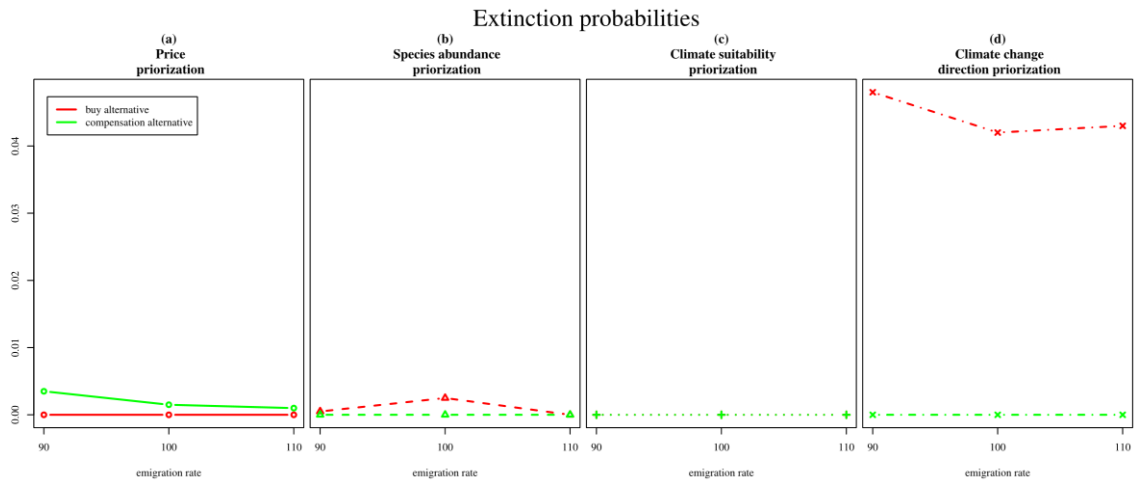


Figure B.9: Extinction probabilities of the different GM-PSS pairs with changing emigration rates. The red line represents results for the buy alternative, the green line for the compensation alternative.

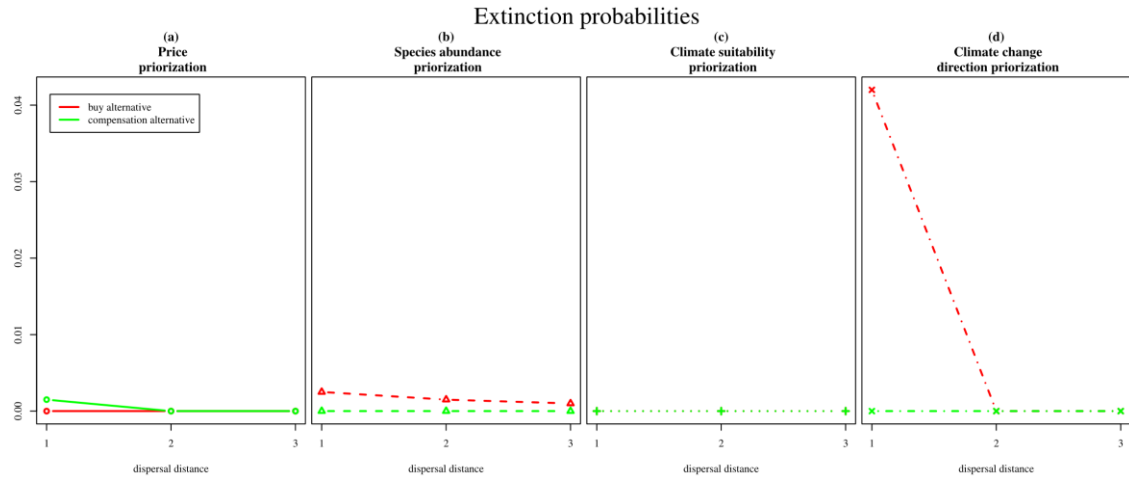


Figure B.10: Extinction probabilities of the different GM-PSS pairs with changing dispersal distances. The red line represents results for the buy alternative, the green line for the compensation alternative.

Decreasing land purchase related mean transaction costs \overline{tc}^{buy} only showed an influence in the climate direction prioritization strategy where cost-effectiveness increases (see Fig. B.11). Other strategies were not influenced by changes in mean transaction costs as the underlying model parametrization already resulted in complete species survival and no changes in cost-effectiveness were observable.

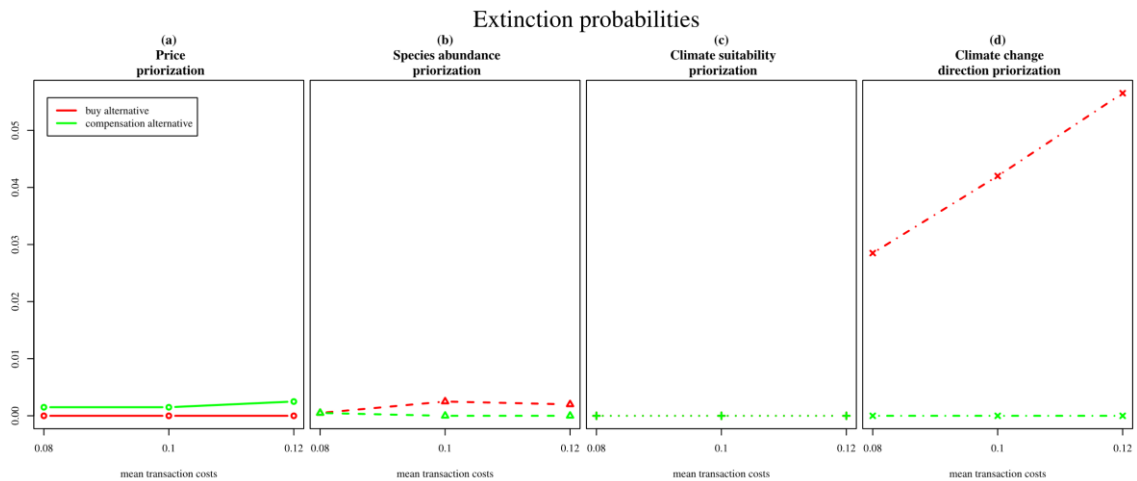


Figure B.11: Extinction probabilities of the different GM-PSS pairs with changing mean transaction costs. The red line represents results for the buy alternative, the green line for the compensation alternative.

Similarly, a decrease in mean monitoring costs resulted in an increase in cost-effectiveness as general conservation costs were reduced (Fig. B.12).

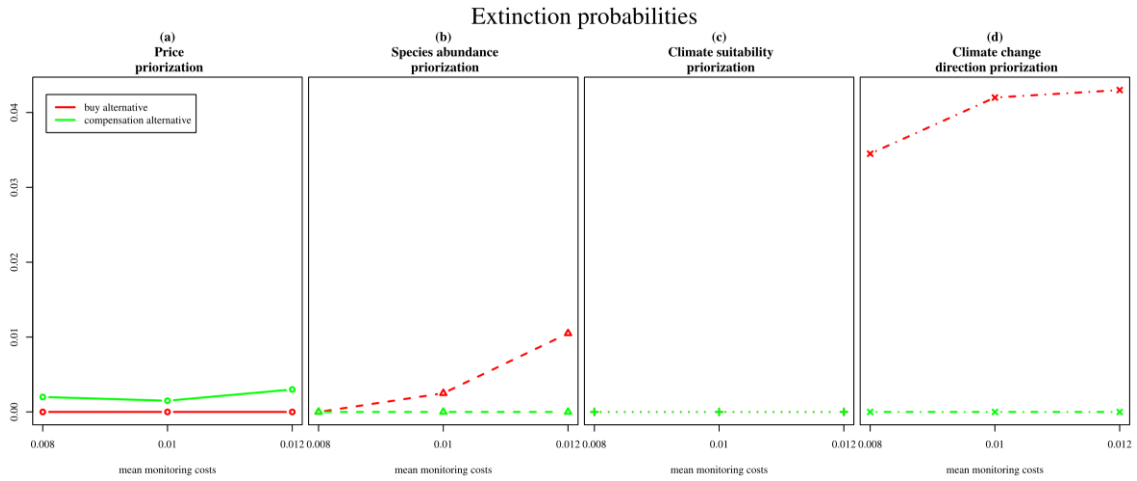


Figure B.12: Extinction probabilities of the different GM-PSS pairs with changing mean monitoring costs. The red line represents results for the buy alternative, the green line for the compensation alternative.

As discussed in the main part, changes in the parameter ρ , influencing the curvature of the climate suitability bell shape, can be considered negligible (see Fig B.13). Variation in ρ only influences the size of the CSZ and the absolute values of patch level climate suitability within the CSZ. These effects however are only marginal.

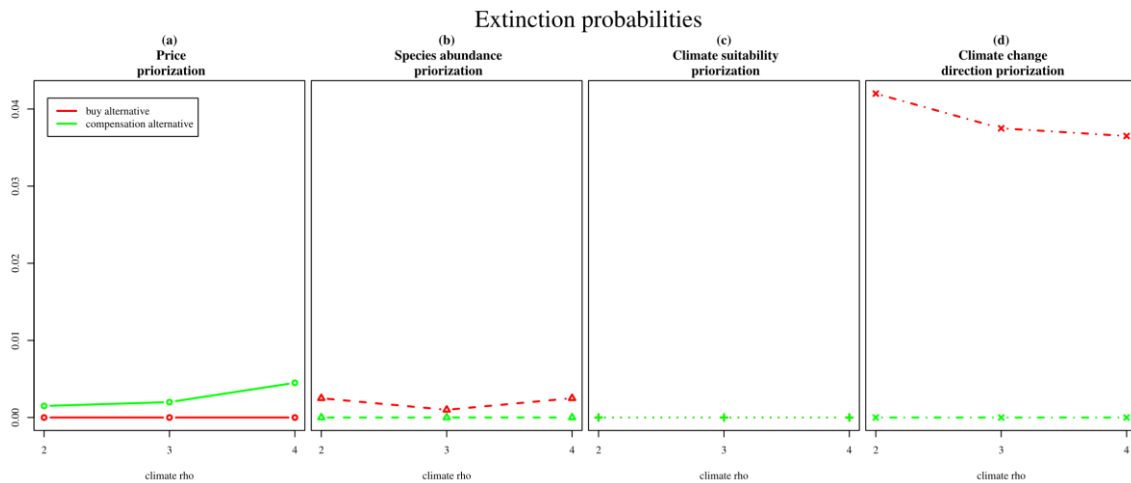


Figure B.13: Extinction probabilities of the different GM-PSS pairs with changing climate suitability bell curvature parameter ρ . The red line represents results for the buy alternative, the green line for the compensation alternative.

Variations in the climate direction prioritization strength parameter λ only have an effect on the respective PSS (compare Figure B.14). In particular, a marginal increasing effect on

the cost-effectiveness of the buy alternative due to decreases in λ can be observed. The direction of this effect is reasonable, as a decreasing value of λ results in a less restrictive and thus less costly patch selection within the CSZ. This in turn increases the cost-effectiveness of the corresponding GM-PSS pair.

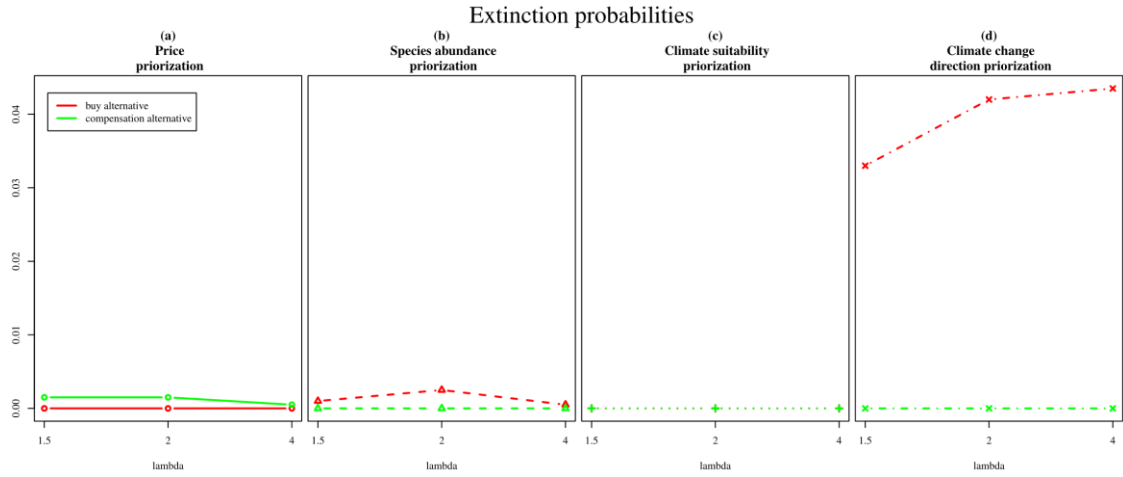


Figure B.14: Extinction probabilities of the different GM-PSS pairs with changing climate direction prioritization strength λ . The red line represents results for the buy alternative, the green line for the compensation alternative.

C. Appendix to Chapter 4

This chapter has previously been published as part of:

Schöttker, O., Wätzold, F., 2018. Buy or lease land? Cost-effective conservation of an oligotrophic lake in a Natura 2000 area. Biodiversity and Conservation 27, 1327–1345.

C.1 Detailed overview about land purchases

Initial purchases of the conservation area at Lake Bültsee of approximately 17 ha have been made in 1980. Together with a major purchase of approximately 27 ha in 1981, they built the foundation of the present conservation site. With some minor corrections of the conservation site (due to infrastructure projects and exchange of conservation area for other conservation projects in 1985 and 1990) the Lake Bültsee conservation site was extended further in 2000 and 2011. In total, today 50.64 ha of agricultural land are included into the conservation project plus additional 21 ha of lake surface.

See Table C.1 for a detailed information on buying costs for the individual land parcels.

C.2 Agricultural land price, leasehold rent and crop price development

During the period under review, land prices and leasehold rents, as well as agricultural crop prices, yields and thus profitability substantially fluctuated. While during the 1980s a period of low land prices, leasehold rents and agricultural profitability could be observed (see Figures C.1, C.2 and C.3), with even further decreasing buying prices for agricultural land in the 1990, land prices, rents and profitability increased drastically since 2006.

Table C.1: Detailed overview about purchasing costs of individual land parcels around Lake Biltsee, containing administrative information, buying prices, area sizes and purchasing side expenses, such as property tax, notary fee, SHL-fee, land registration fee, and measurement costs.

year	comment	area lot nr.	contract date	area transferration date	ha	buying price	SHL-fee	property tax	notary fee	land registry office fee	measurement costs
1980		1980-019	22.07.1980	01.07.1980	17,611	207.073,21 €	6.615,99 €	14.495,12 €	4.141,46 €		31.060,98 €
1981		1981-013	16.04.1981	01.05.1981	3,7515	30.689,78 €	985,14 €	2.148,28 €	613,80 €		4.843,90 €
1981		1981-014	17.04.1981	01.04.1981	2,4803	31.700,10 €	1.012,82 €	2.219,01 €	173,01 €		4.755,01 €
1981		1981-023	21.05.1981	01.10.1981	6,4747	52.220,80 €	1.668,45 €	3.655,46 €	405,22 €		1.122,24 €
1981		1981-024	18.06.1981	01.10.1981	6,8725	70.813,93 €	2.615,03 €	2.986,71 €	197,58 €		9.843,50 €
1981		1981-025	18.06.1981	01.10.1981	4,1172	44.175,62 €	1.411,41 €	2.241,91 €	347,69 €		6.626,34 €
1985		1885-022	14.10.1985	01.01.1986				7.317,10 €	711,10 €		
	infrastructure related										
1985	area exchange	1985-022	30.10.1985	02.01.1986	-0,0472	-1.238,93 €					
	area exchange with conservation site										
1990	Holmer See	1990-090	30.08.1990	01.10.1990		9.513,61 €	547,08 €	190,27 €	190,27 €		592,80 €
	area exchange for the conservation site										
1990	Biltsee				-1,1983	0,00 €					
2000		2000-027	10.03.2000	01.05.2000	6,2731	68.001,82 €	2.380,06 €	2.490,50 €	536,87 €	23,01 €	3.160,25 €
	50% financing by										
2011	ELER	2011-182	21.11.2011	30.12.2011	4,3079	83.101,30 €	2.908,55 €	2.908,00 €	737,90 €	22,61 €	12.465,20 €



Figure C.1: Agricultural land prices (per hectare) and leasehold rent (per hectare per year) development in Schleswig-Holstein from 1980 to 2015 (Statistikamt Nord 2015a).

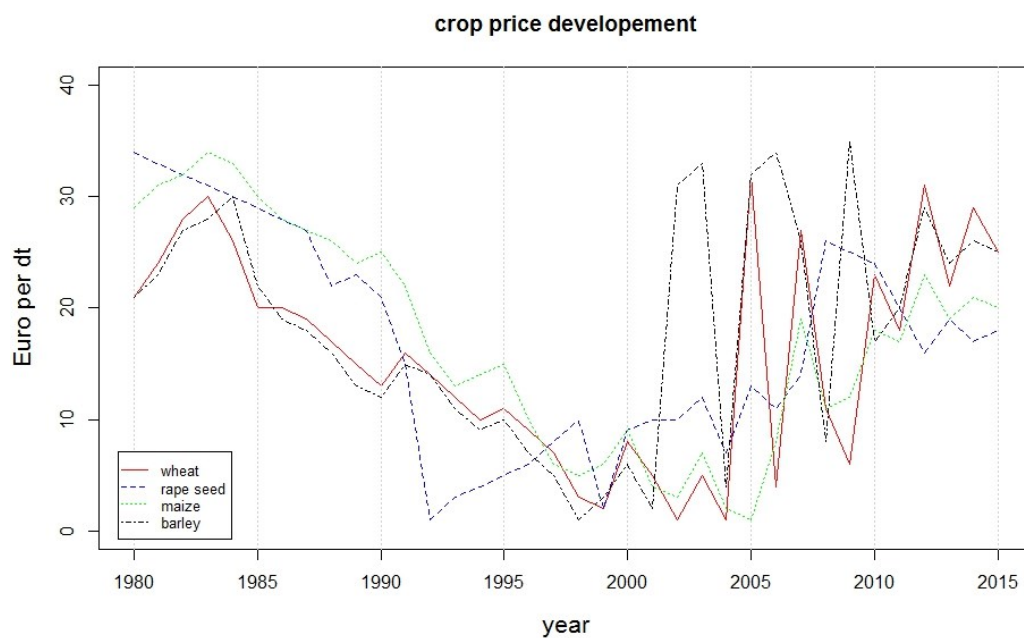


Figure C.2: Development of crop prices in Schleswig-Holstein from 1980 to 2015 for the four most used field crops, measured in Euro per dezi-ton.

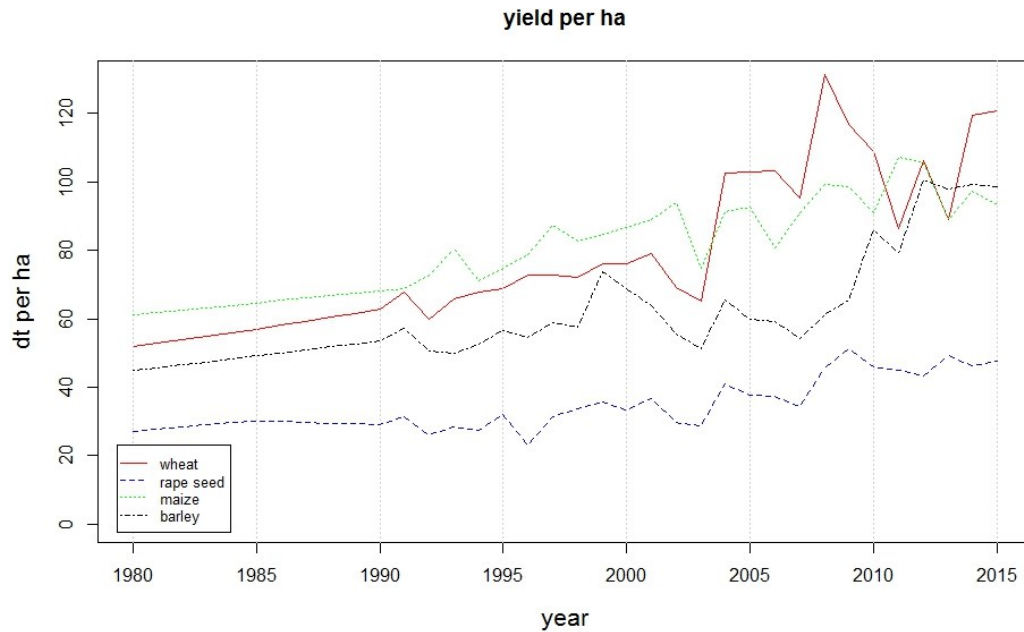


Figure C.3: Development of per ha yields of the four most used field crops in Schleswig-Holstein, measured in dezi-ton.

C.3 Crop Mix

For the calculation of the hypothetical compensation payment we estimated the potential profitability of extensively used grassland. For this, we estimated the difference between potential profit from intensively used agricultural land (as we assume it to be the profit maximizing alternative, if no conservation efforts were undertaken) and extensively used grassland. The potential profit for intensively used agricultural land is calculated as the weighted average of profits of the four most commonly used field crops in Schleswig-Holstein (see Figure C.4).

Based on assessments made from our study partners, we assumed that today, if no conservation efforts were made, the agricultural land in the study area was exclusively used for maize production (due to the high profitability). We therefore assumed that from 2006 to 2009 a transition period between a regional-average crop-mix towards exclusive maize production takes place, in which the wheat, rape seed and barley would be gradually faded out.

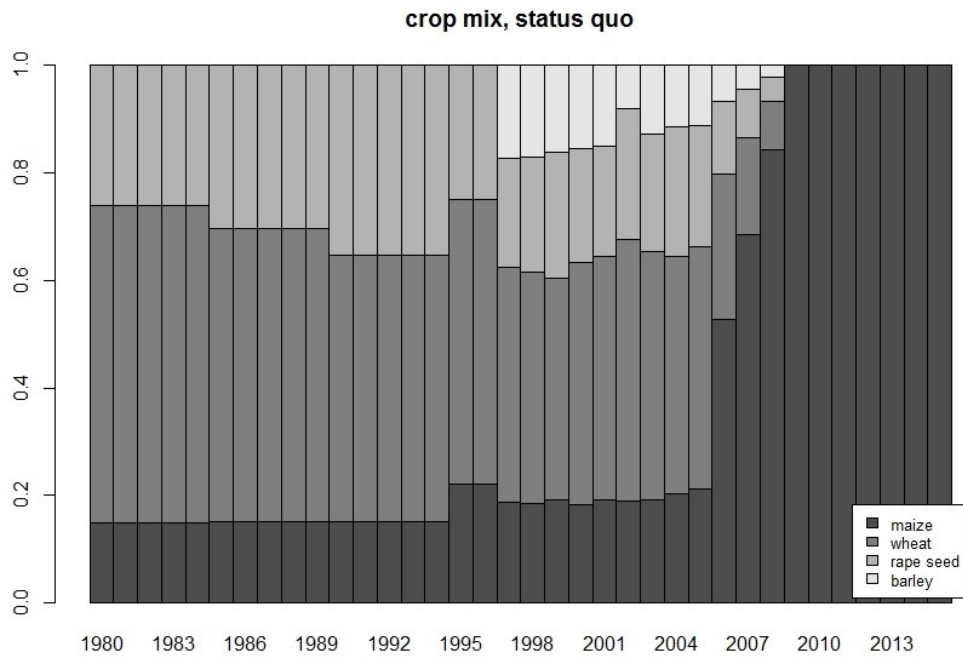


Figure C.4: Shares of cultivated field crops in Schleswig-Holstein between 1980 and 2015 (Statistikamt Nord 2015b). From 2006 to 2009, a transition phase is assumed in which the average field crop cultivation gradually changes from mixed agriculture to pure maize cultivation (mainly used as energy crop, personal communication). This results in pure maize cultivation from 2010 onwards.

C.4 Hypothetical profit and costs

To calculate the profitability of agricultural land in the study region, we calculated separately the revenue (see Table C.2) and production costs (see Table C.3) for the four most commonly used field crops based on literature data (Ruhr-Stickstoff Aktiengesellschaft 1974; Ruhr-Stickstoff Aktiengesellschaft 1988; Hydro Agri Dülmen GmbH 1993; KTBL 2005, 2009, 2014).

We additionally estimated the potential profit a land-user can realize through extensive grassland use, based on statistical data on grassland productivity in Germany (Statistisches Bundesamt 2014a, 2014b).

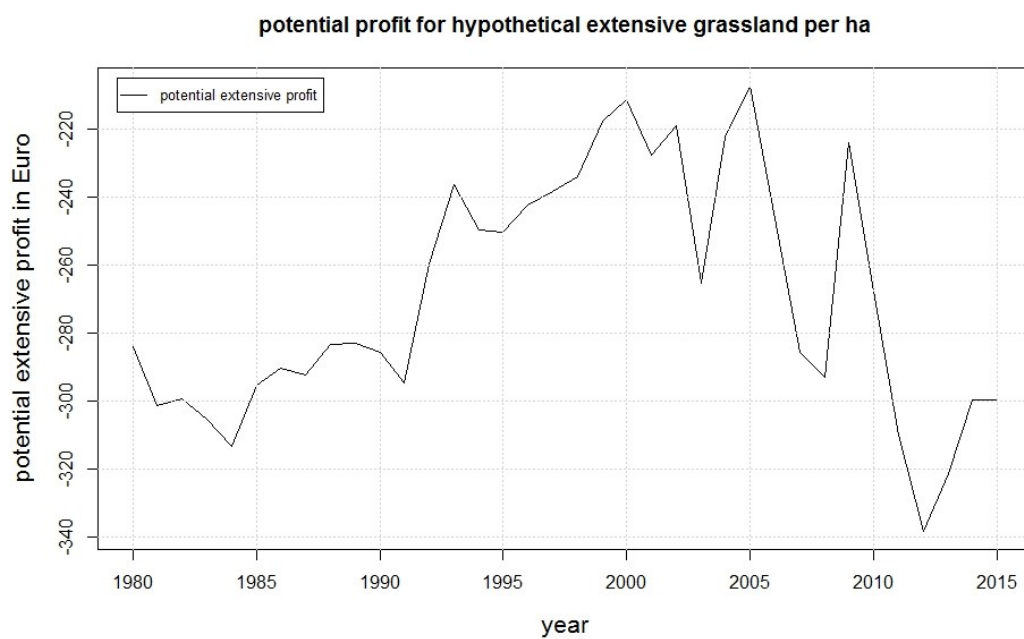


Figure C.5: Potential profit of extensive grassland management.

Table C.2: Overview about the hypothetical revenue and costs of extensive grassland measures in the case study area.

Year	Revenue per ha	Costs per ha	Sum
1980	134.52 €	-418.50 €	-283.99 €
1981	142.76 €	-444.14 €	-301.38 €
1982	141.74 €	-440.99 €	-299.24 €
1983	144.66 €	-450.06 €	-305.40 €
1984	148.46 €	-461.89 €	-313.43 €
1985	139.97 €	-435.47 €	-295.50 €
1986	137.56 €	-427.97 €	-290.41 €
1987	138.45 €	-430.73 €	-292.28 €
1988	134.26 €	-417.72 €	-283.45 €
1989	134.01 €	-416.93 €	-282.92 €
1990	131.33 €	-416.93 €	-285.59 €
1991	132.77 €	-427.58 €	-294.81 €
1992	113.62 €	-373.14 €	-259.52 €
1993	113.50 €	-349.87 €	-236.37 €
1994	117.98 €	-367.62 €	-249.64 €
1995	118.74 €	-369.20 €	-250.46 €
1996	110.97 €	-353.42 €	-242.45 €
1997	111.54 €	-349.87 €	-238.33 €
1998	116.02 €	-349.87 €	-233.85 €
1999	106.44 €	-324.23 €	-217.79 €
2000	110.69 €	-321.87 €	-211.17 €
2001	117.27 €	-344.74 €	-227.48 €
2002	114.80 €	-333.70 €	-218.89 €
2003	92.81 €	-358.15 €	-265.35 €
2004	109.30 €	-331.73 €	-222.42 €
2005	105.14 €	-312.40 €	-207.26 €
2006	111.84 €	-358.15 €	-246.31 €
2007	142.39 €	-427.97 €	-285.58 €
2008	136.01 €	-429.15 €	-293.15 €
2009	107.10 €	-330.94 €	-223.84 €
2010	126.78 €	-394.44 €	-267.66 €
2011	146.82 €	-456.77 €	-309.95 €
2012	160.25 €	-498.58 €	-338.32 €
2013	152.27 €	-473.73 €	-321.46 €
2014	133.76 €	-433.28 €	-299.52 €
2015	133.76 €	-433.28 €	-299.52 €

Table C.3: Hypothetical profit and costs for intensive agricultural land use for the four most used field crops in Schleswig-Holstein, measured in Euro per ha.

year	revenue in Euro per ha				weighted sum
	wheat	rape seed	maize	barley	
1980	1,158.01 €	1,508.65 €	1,852.16 €	891.98 €	1,353.30 €
1981	1,202.80 €	1,495.95 €	1,964.79 €	920.86 €	1,393.32 €
1982	1,275.27 €	1,481.15 €	2,050.02 €	988.81 €	1,445.01 €
1983	1,308.18 €	1,464.22 €	2,178.66 €	1,021.50 €	1,479.32 €
1984	1,310.14 €	1,445.19 €	2,169.92 €	1,070.52 €	1,474.22 €
1985	1,208.01 €	1,424.05 €	2,008.11 €	989.09 €	1,394.70 €
1986	1,232.17 €	1,361.70 €	1,913.68 €	967.44 €	1,374.68 €
1987	1,224.12 €	1,247.39 €	1,901.65 €	939.01 €	1,333.83 €
1988	1,143.69 €	1,071.62 €	1,693.03 €	899.86 €	1,205.08 €
1989	1,103.94 €	1,065.03 €	1,631.33 €	852.00 €	1,172.05 €
1990	1,071.38 €	1,048.53 €	1,694.99 €	833.94 €	1,157.79 €
1991	1,261.25 €	1,005.27 €	1,576.01 €	976.13 €	1,218.44 €
1992	1,066.20 €	332.76 €	1,422.98 €	834.38 €	860.96 €
1993	980.02 €	421.41 €	1,240.25 €	658.54 €	821.96 €
1994	914.86 €	462.67 €	1,130.89 €	653.96 €	787.73 €
1995	999.05 €	573.59 €	1,245.82 €	738.84 €	946.97 €
1996	947.48 €	441.21 €	1,100.40 €	644.65 €	854.56 €
1997	893.07 €	643.70 €	1,055.12 €	665.24 €	833.38 €
1998	802.52 €	719.04 €	949.90 €	577.28 €	773.39 €
1999	844.27 €	526.26 €	1,024.87 €	753.27 €	789.25 €
2000	960.12 €	686.96 €	1,137.08 €	784.32 €	907.57 €
2001	907.35 €	783.24 €	1,013.46 €	645.39 €	862.72 €
2002	739.37 €	635.58 €	1,032.90 €	505.05 €	750.66 €
2003	747.50 €	668.71 €	956.16 €	495.67 €	738.17 €
2004	1,097.82 €	805.56 €	976.91 €	675.68 €	954.95 €
2005	988.80 €	936.29 €	964.08 €	555.21 €	922.95 €
2006	1,177.62 €	857.90 €	1,041.03 €	579.18 €	1,022.01 €
2007	2,249.08 €	1,077.36 €	2,017.98 €	1,149.04 €	1,935.82 €
2008	1,903.85 €	1,819.77 €	1,436.95 €	750.30 €	1,480.75 €
2009	1,391.11 €	2,002.94 €	1,449.42 €	647.46 €	1,449.42 €
2010	2,472.03 €	1,790.78 €	1,999.80 €	1,573.80 €	1,999.80 €
2011	1,732.62 €	1,536.03 €	2,229.76 €	1,530.66 €	2,229.76 €
2012	2,650.00 €	1,432.94 €	2,489.80 €	2,190.90 €	2,489.80 €
2013	2,004.70 €	1,645.74 €	1,978.02 €	2,014.68 €	1,978.02 €
2014	2,838.74 €	1,538.92 €	2,228.17 €	2,101.98 €	2,228.17 €
2015	2,796.87 €	1,592.23 €	2,101.66 €	2,058.13 €	2,101.66 €

Table C.4: Hypothetical costs for intensive agricultural land use for the four most used field crops in Schleswig-Holstein, measured in Euro per ha

year	costs in Euro per ha				weighted sum
	wheat	rape seed	maize	barley	
1980	439.26 €	480.74 €	420.86 €	433.68 €	447.29 €
1981	454.63 €	495.67 €	429.54 €	441.71 €	461.54 €
1982	470.01 €	510.60 €	438.23 €	449.73 €	475.79 €
1983	485.38 €	525.53 €	446.91 €	457.76 €	490.05 €
1984	500.75 €	540.45 €	455.60 €	465.79 €	504.30 €
1985	516.12 €	555.38 €	464.29 €	473.81 €	520.16 €
1986	531.49 €	570.31 €	472.97 €	481.84 €	534.39 €
1987	546.86 €	585.24 €	481.66 €	489.87 €	548.61 €
1988	562.23 €	600.17 €	490.34 €	497.90 €	562.83 €
1989	577.60 €	615.09 €	499.03 €	505.92 €	577.06 €
1990	592.97 €	630.02 €	507.71 €	513.95 €	593.15 €
1991	608.34 €	644.95 €	516.40 €	521.98 €	607.35 €
1992	623.71 €	659.88 €	525.09 €	530.01 €	621.55 €
1993	639.08 €	674.80 €	533.77 €	538.03 €	635.76 €
1994	654.45 €	689.73 €	542.46 €	546.06 €	649.96 €
1995	652.87 €	674.39 €	551.14 €	544.37 €	635.87 €
1996	651.28 €	659.05 €	559.83 €	542.69 €	633.10 €
1997	649.69 €	643.71 €	568.51 €	541.00 €	614.28 €
1998	648.11 €	628.38 €	577.20 €	539.31 €	612.06 €
1999	646.52 €	613.04 €	585.89 €	537.62 €	609.43 €
2000	644.93 €	597.70 €	594.57 €	535.94 €	608.85 €
2001	643.35 €	582.36 €	603.26 €	534.25 €	606.69 €
2002	641.76 €	567.02 €	611.94 €	532.56 €	609.08 €
2003	640.17 €	551.68 €	620.63 €	530.87 €	603.11 €
2004	638.59 €	536.34 €	629.31 €	529.19 €	599.65 €
2005	637.00 €	521.00 €	638.00 €	527.50 €	598.79 €
2006	657.38 €	557.70 €	680.03 €	546.84 €	648.41 €
2007	677.76 €	594.40 €	722.06 €	566.17 €	695.58 €
2008	698.13 €	631.10 €	764.09 €	585.51 €	748.15 €
2009	718.51 €	667.80 €	806.12 €	604.84 €	806.12 €
2010	738.89 €	704.49 €	848.15 €	624.18 €	848.15 €
2011	759.27 €	741.19 €	890.18 €	643.51 €	890.18 €
2012	779.64 €	777.89 €	932.21 €	662.85 €	932.21 €
2013	800.02 €	814.59 €	974.24 €	682.18 €	974.24 €
2014	820.40 €	851.29 €	1,016.27 €	701.52 €	1,016.27 €
2015	840.78 €	887.99 €	1,058.30 €	720.85 €	1,058.30 €

C.5 Compensation payment

Against the background of estimated profitability with intensive agriculture and extensive grassland use, we calculated the necessary compensation payment a land-user would potentially request to implement extensive grassland measures on an area otherwise used for intensive maize and crop production (see Figure C.6).

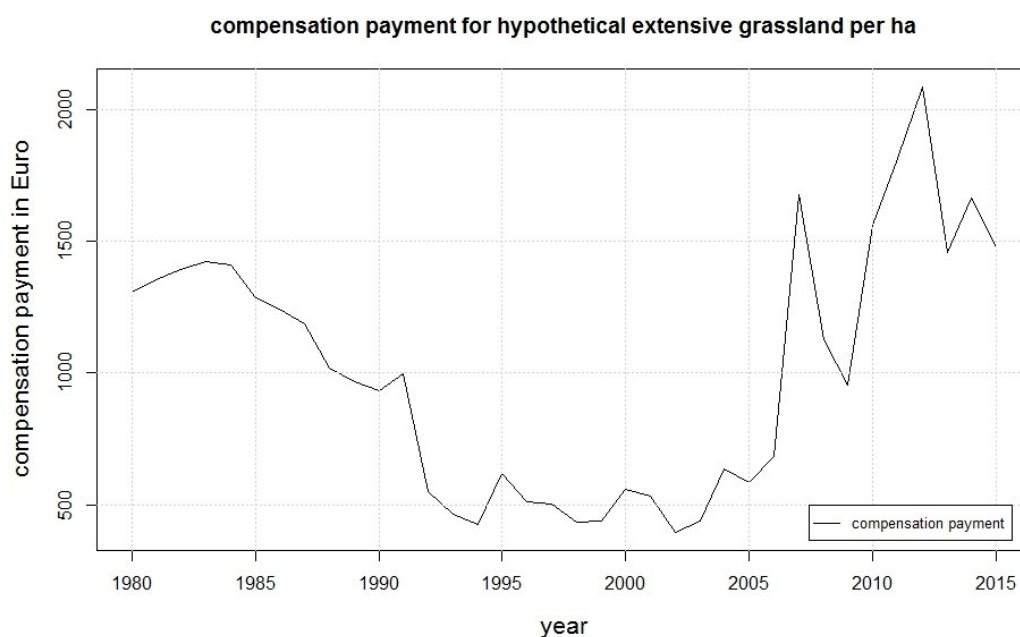


Figure C.6: Hypothetical compensation payment per ha and year necessary offset the assumed opportunity costs of a prescribed extensive measure.

C.6 Sensitivity analysis: transaction costs

The introduction of variability on transaction costs, as it has been assumed in Section 4.2.1 of the main paper causes a variation in the required compensation payment (see Appendix C.5 for details). Figure C.7 shows this variation in relation to the fluctuating compensation payment, while Figure C.8 and C.9 show the fluctuation transaction costs in the compensation option on the side of the conservation agency and the farmer.

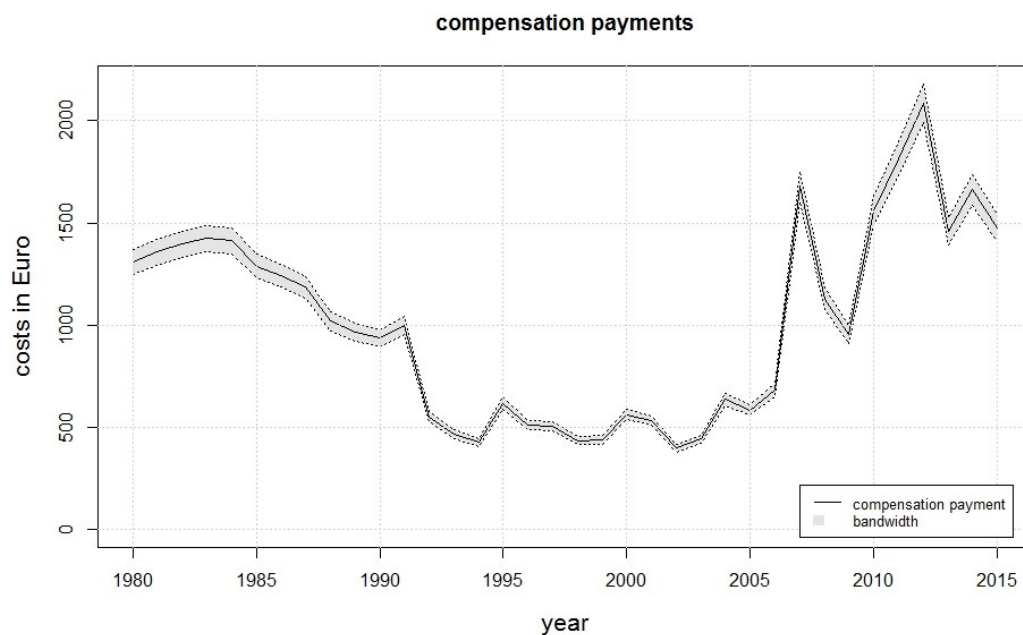


Figure C.7: Compensation payment including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).

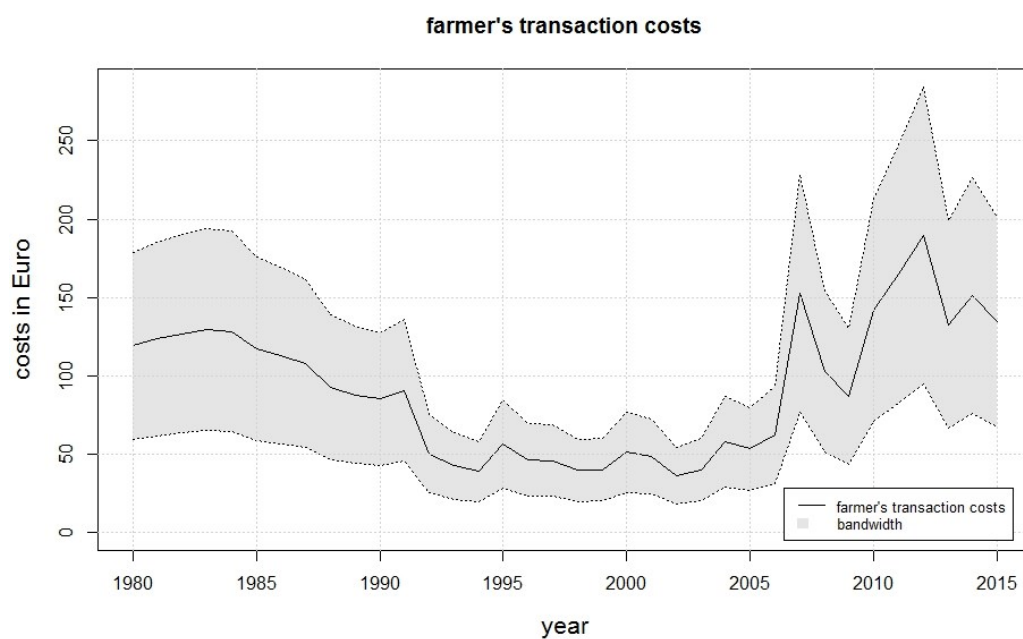


Figure C.8: Farmers' transaction costs including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).

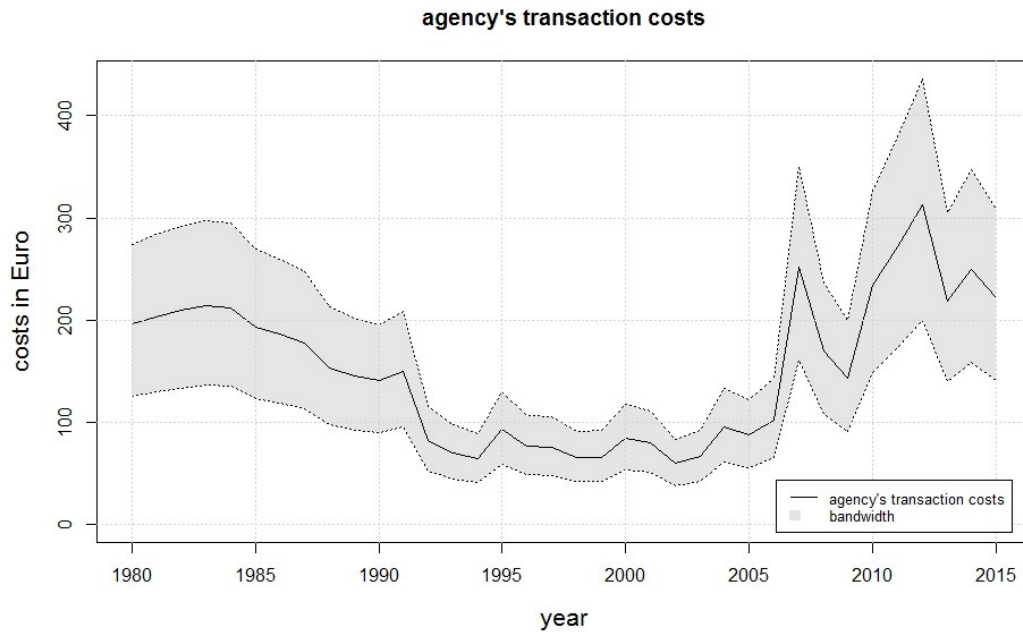


Figure C.9: Farmers' transaction costs including bandwidth for low and high values (high value is at the upper edge; low value is at the lower edge of the shaded area).

C.7 Sensitivity analysis: leasehold rent

The introduction of variability in the leasehold income, acquired by the conservation agency by giving the formerly bought land as a leasehold to a contractor, who in turn implements the prescribed conservation measures on the land (see Section 4.2.2), causes a variation in the costs of the buying option.

We assumed that an upper bound for leasehold payments to be paid for the case study area would be the regional average leasehold for agricultural land in Schleswig-Holstein (see Figure C.10, upper dotted line). As a lower bound we assumed zero leasehold payments to be paid by the contractor (see Figure C.10, lower dotted line), as was the case for the conservation project at Lake Bültsee between 1980 and 2008)

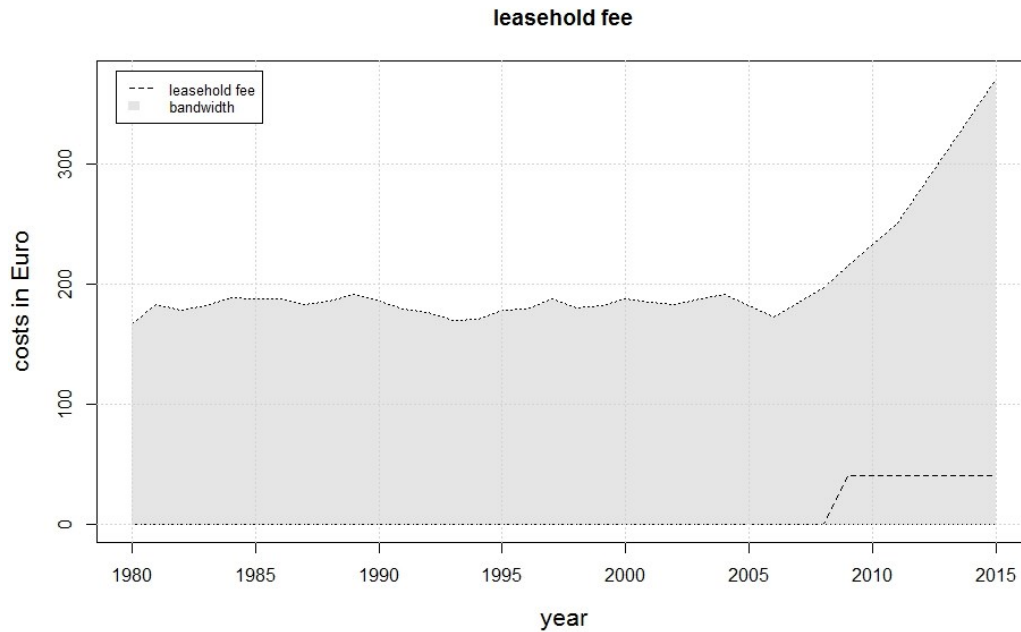


Figure C.10: Leasehold fee for the case study project (dashed line) and actual leasehold average payments for agricultural areas in Schleswig-Holstein (dotted line). The difference between the baseline and the low value case is only visible after year 2008, as leasehold rents for the baseline before that date have been 0 €, the same as in the low case value, and only differ from 2009 onwards.

C.8 Sensitivity analysis: land value

The introduction of variability in the market value of the purchased land (see Section 4.2.3), also causes a variation in the costs of the buying option as the potential resale price of land incorporated in the calculation of the costs of the buying option.

For our baseline calculations we assume that market prices for extensively used grassland in Schleswig-Holstein can be estimated as 70% of the prices of intensively used agricultural land (see Figure B.11, dashed line). We further assume that an upper bound for the potential resale value of extensively used grassland is the market value of intensively used agricultural land (see Figure B.11, upper dotted line), while as a lower bound we assume a potential resale value of 40% of the value of intensively used agricultural land. The actual development of land prices displays statistical data on average agricultural land prices in Schleswig-Holstein (Statistikamt Nord 2015a).

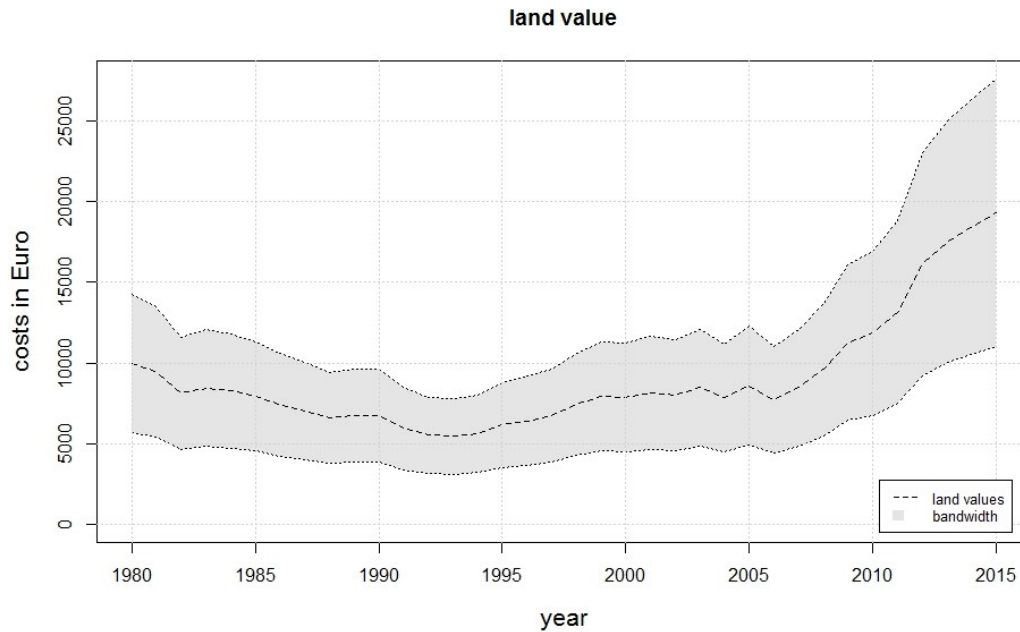


Figure C.11: Land value of extensive grassland with bandwidth due to variations of θ (high value is at the upper edge; low value is at the lower edge of the shaded area).

C.9 Sensitivity analysis: AES payment eligibility

In our baseline calculation, no further payments for agri-environment schemes are gathered neither from the SNSH nor the BW. We can however assume that to a certain extend further auxiliary payments for a compensation scheme could be utilized either by the governing body (the SNSH) or the contractor (the BW). We thus assume that maximally a total of 750 € of subsidies per hectare and year can be reached in the study area and for the case study project. The potential subsidies for the implementation of extensive grassland measures are rough estimates for reasonable minimal and maximal amounts, based on legislation for the federal state of Schleswig-Holstein (Deutscher Bauernverband 2016; Landesamt für Landwirtschaft 2016; Rebhann et al. 2016). A more moderate auxiliary payment level would be assumed to be at 150 € per hectare and year. The low value (corresponding to our case study values) is set to 0 €. For simplicity, we assume corresponding discounted values for subsidies for earlier years, based on estimated subsidy levels for 2015, and due to lack of more precise data discounted to all earlier years with a discount factor (cp. Eq. 4.1b).

With the possibility to attract further funding through AES and thus increase the landowner's income in the *compensation option*, the point in time when the *compensation*

option becomes less cost-effective is postponed. If we assume moderate values for potential subsidies, the break-even point shifts to the year 2009. With a further increase in subsidy levels, the break-even point eventually disappears completely, rendering the compensation option overall superior to the buy option (Figure C.12). This of course is reasonable as additional income, generated by attracting subsidies for conservation measure implementation counteracts the costs, and eventually leading to an increased economic profit generated by the landowner.

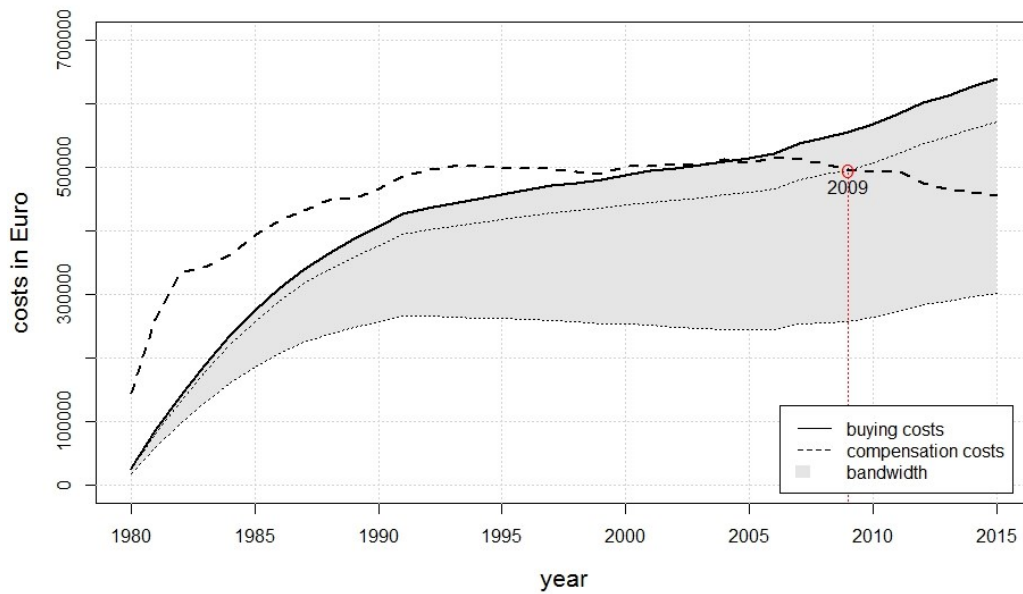


Figure C.12: Costs for the buy option (dashed line), and compensation option (straight line) with the corresponding bandwidth for changing compensation expenses (grey shaded area) due to variations in potentially received payments for AES (high value is at the lower edge of the shaded area; moderate values are represented by the dotted line in the middle; the case study value is represented by the straight line at the upper edge of the shaded area).

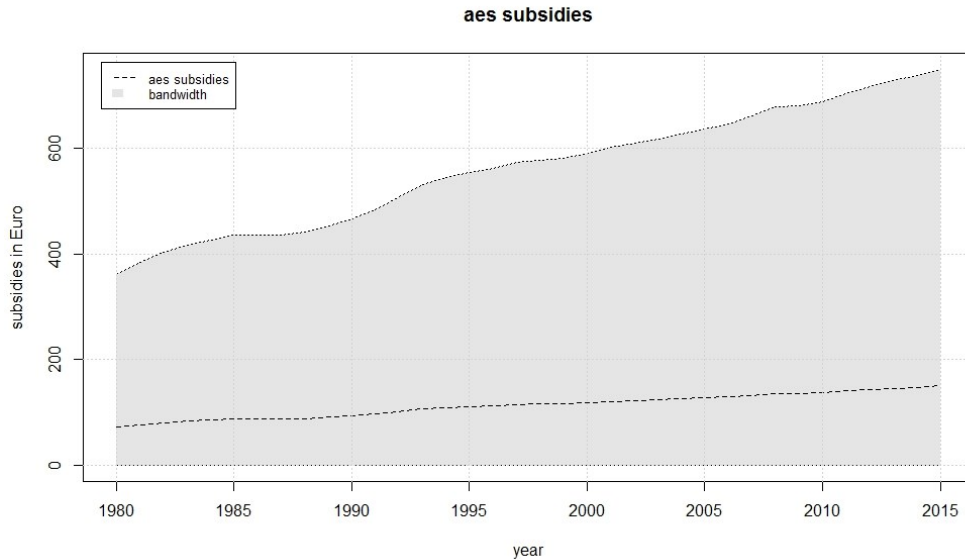


Figure C.13: Range of the assumed auxiliary subsidies from agri-environmental schemes utilised for the compensation option. The lower dotted line (identical with the x-axis) represents the baseline, as here zero subsidies are assumed. The dashed line stands for the moderate, and the upper, dotted line for the high value case.

C.10 Sensitivity analysis: crop mix

To calculate the costs and benefits of both governance modes we assume in Section 4.4.2 that after the year 2009, the only crop cultivated in the area if no biodiversity conservation project was implemented, would be maize. This assessment is based on the personal communication of our study partners¹, which described the cultivation of maize, especially as an energy crop due to the energy crop boom since 2006, is getting increasingly abundant in the case study region. Furthermore, the surrounding agricultural areas are today mainly used for maize production.

We can show in a sensitivity analysis that this assumption seems to be not crucial for the outcome determination. Had, instead, the land been used for a federal state average cultivation mix (i.e. a mix of the 4 most common field crops cultivated on average in the federal state of Schleswig-Holstein; compare Figure C.4 and C.14), the overall costs of the conservation project would have been altered only slightly, and the cost-effective solution, i.e. a critical point in the year 2004, remained unchanged (see Figure C.15). Also, no structural differences in the time after 2004 could have been observed.

However, data suggests that in recent years the share of maize cultivation in Schleswig-Holstein increased (compare Figure C.4 and C.14), backing the assessment of our interview partners who suggested that if the conservation area would not have been intensified, today most likely intensive maize production would be present at the area, as it is also the case at the surrounding agricultural areas.

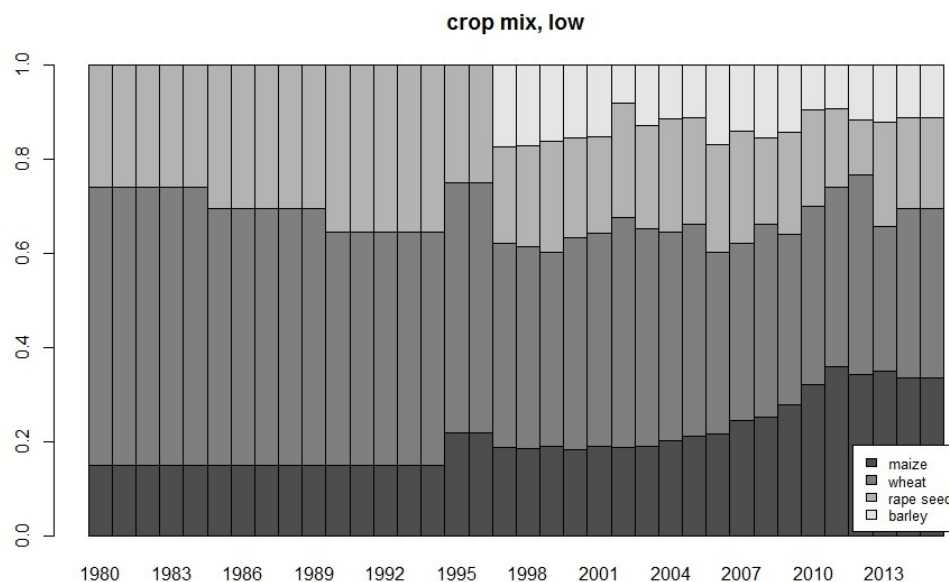


Figure C.14: Alternative crop mix compared to phase out of wheat, rape seed and barley cultivation in favour of pure maize production after 2006, compare Figure C.4 (Statistikamt Nord 2015b).

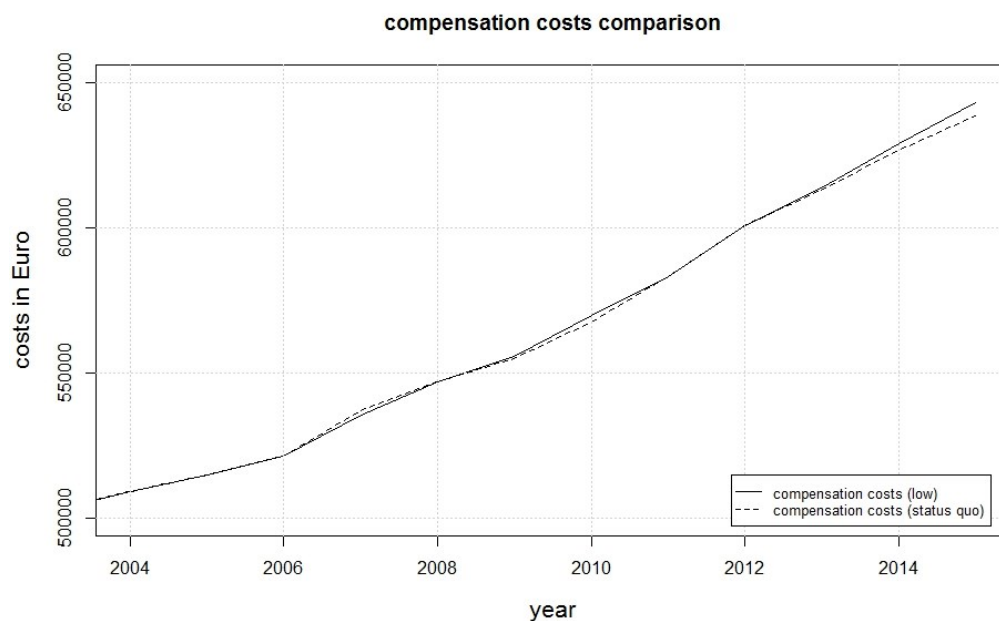


Figure C.15: Changes in compensation costs due to alternative crop mix.

C.11 Data overview from literature review

To calculate the costs for both modes of governance, a detailed literature review was conducted to gather necessary information on transaction costs, land prices for intensively and extensively managed grassland, as well as interest and discount rates. To close some data gaps, the German Consumer Price Index was utilized for interpolation of data. The following Table C.5 shows the data sources.

Table C.5: Overview about the different sources for the costing calculation.

<i>Data</i>	<i>Source</i>
Transaction costs	Falconer (2000), McCann and Easter (2000), Kersten (2008), Thomas et al. (2009), McCann (2013)
Land prices for intensively and extensively managed grassland	Statistikamt Nord (2015a)
Interest rates and German Consumer Price Index	Statistisches Bundesamt (2014c)
Discount rates	Conversion of the nominal interest rate from Statistisches Bundesamt (2014c), in combination with the values on the Consumer Price Index, according to Boardman (2006)
Revenue and management costs from intensively managed agricultural land	Ruhr-Stickstoff Aktiengesellschaft (1974, 1988), Hydro Agri Dülmen GmbH (1993), KTBL (2005, 2009, 2014), Bundesministerium für Ernährung und Landwirtschaft (2001, 2005, 2007), Landwirtschaftskammer Schleswig Holstein (2016)

D. Appendix to Chapter 5

This chapter has previously been published as part of:

Schöttker, O., Santos, M.J., 2019. Easement or public land? An economic analysis of different ownership modes for nature conservation measures in California. Conservation Letters, e12647.

In this manuscript we provide additional information and explanations for research presented in above-mentioned work. This includes (1) a more detailed explanation of the costing sub-functions for land purchase, (2) a detailed explanation of the costing sub-functions of land managed as easements, (3) an overview of the parametrization of the costing calculations and the data used, (4) a visualization of the development of number of conservation areas and total area under conservation, and (5) a sensitivity analysis of conservation costs for several economic parameters.

D.1 Costing sub-functions for buying land

We assume that the purchasing price of an area can be estimated by the size and the average price of land at the time of purchase or acquisition in the county the area lies in, scaled by $k^{in\ fee}$ to correct for price overstatement as we used data on median sold prices for land including structures as an price and price development indicator. Hence

$$P_t = a \times k^{in\ fee} \times price_t^{area} \quad (D.1)$$

We assume that the management costs of an area of purchased land are composed of a fixed costs component m_{fix} (which are actual expenses for actively managing and maintaining land) for each conservation area, a size depended component m_{var} for each area, and a scaling factor \underline{n}/N_t representing a scaling factor divided by the overall number of conservation areas a conservation organization manages at any given time. Hence the management costs are

$$M_t = m_{fix} + (m_{var} \times a \times \underline{n}) / N_t \quad (D.2)$$

With N_t the total number of areas under conservation by an organization at time t , and a the size of the area under conservation.

We further assume that the purchasing side costs S_t are defined as a fixed fraction s of the purchasing costs P_t . Hence

$$S_t = s \times P_t \quad (D.3)$$

We also assume that the transaction costs $T_t^{in\ fee}$ are defined as a fixed fraction $t_a^{in\ fee}$ of the purchasing costs P_t . Hence the agency side transaction costs are

$$T_t^{in\ fee} = t_a^{in\ fee} \times P_t \quad (D.4)$$

Transaction costs here include contract negotiation and set-up costs, personnel expenses, travel expenses, monitoring and enforcement cost, and tax breaks due incentive programs for easement owners.

D.2 Costing sub-functions for easements

OC_t is defined as a fixed fraction of the (potential) purchasing prices in that year, i.e. $OC_t = k^{easement} \times P_t$. A more precise interpretation of the opportunity costs would define OC as the difference in income between profit maximizing land use and actual nature conservation land use income. However, we simplify the estimation, as no reliable data on intensive or extensive income are available, and thus the conceptually more precise definition would not add further insight into our analysis.

We further assume that the income to be generated on an area under conservation as an easement, $I_t^{easement} = I_t^{in\ fee}$ is a fixed fraction l of the opportunity costs OC_t . Hence

$$I_t^{easement} = l \times OC_t = l \times k^{easement} \times P_t = I_t^{in\ fee} \quad (D.5)$$

Equivalent to the opportunity costs calculation, a more conceptually precise estimation of potential income on an area (either in-fee or easement) would be to use the generally applicable capitalization approach. The purchasing price of an area would thus be interpreted as the discounted sum of all future income to be generated on a piece of land; an equation which can be solved for income. We simplify this, for the same reasons given for the simplification of opportunity costs, so that the income now is estimated to be a fixed fraction of the opportunity costs, which in turn are a fixed fraction of the time dependent potential purchasing price. This simplification includes the agricultural profits foregone due to implementation of conservation measures, compared to a non-conservation setting.

D.3 Parameter selection

Table D.1: Overview, description and sources of costing function parameters with corresponding low and high values chosen in the sensitivity analysis.

<i>Parameter</i>	<i>Description</i>	<i>Low</i>	<i>Base case</i>	<i>High</i>	<i>Source</i>
$k^{in\ fee}$	Scaling factor for purchasing prices relative to land value (extrapolated from data on median sold prices for land including structures)	0.20	0.45	0.70	Schöttker and Wätzold (2018)
$t_a^{easement}$	Agency side transaction costs of “easement” conservation contracts	0.10	0.15	0.20	Falconer (2000), McCann (2013), Kersten (2008), Thomas et al. (2009), Fährmann B., Grajewski R. (2013)
$t_l^{easement}$	Land user side transaction costs of “easement” conservation contracts	0.05	0.10	0.15	
$t_a^{in\ fee}$	Agency side transaction costs of “in-fee” conservation contracts	0.10	0.15	0.20	
l	Scaling factor for the income generated from used land as a share of the opportunity costs of an area put under conservation	0.05	0.10	0.15	Own estimate
$k^{easement}$	Scaling factor regarding the opportunity costs regarding easements	0.05	0.10	0.15	Own estimate
	Absolute value added to / subtracted from the actual nominal interest rate for each year (changes in nominal interest rates do change discount rates accordingly)	- 0.01	0.00	+ 0.01	Own estimate

Table D.2: Overview, description and sources of costing function parameters used in the parametrization of our costing calculations, and which are not further investigated in a sensitivity analysis. It is to be expected that the effect of changes in mentioned parameters is pretty straight forward (i.e. an increase (decrease) of management cost parameters increases (decreases) management costs and thus shifts the corresponding cost curve of easements upwards (downwards)).

<i>Parameter</i>	<i>Description</i>	<i>Base case</i>	<i>Source</i>
s	Scaling regarding the purchasing side expenses (i.e. notary fees, measurement, etc.) are charged for in case of land purchase, as a share of the purchasing price	0.25	Own estimate
m_{fix}	Fixed share of management costs per conservation area (in Dollars)	1000	https://conservationtools.org/guides/86-costs-of-conservation-easement-stewardship: Costs of Conservation Easement Stewardship (accessed 5. April. 2018)
m_{var}	Variable share of costs per ha of conservation area (in Dollars)	350	https://conservationtools.org/guides/86-costs-of-conservation-easement-stewardship: Costs of Conservation Easement Stewardship (accessed 5. April. 2018) and https://static1.squarespace.com/static/52260563e4b0e56a47d7efa6/t/5947f020bebafb91bef60f91/1497886752963/Conservation+Easement+Costs.pdf (accessed 5. April. 2018)
\underline{n}	Scaling factor regarding the total size of a conservation organization (in terms of number of areas under conservation)	2	Own estimate

D.4 Development of area and amount of conservation sites

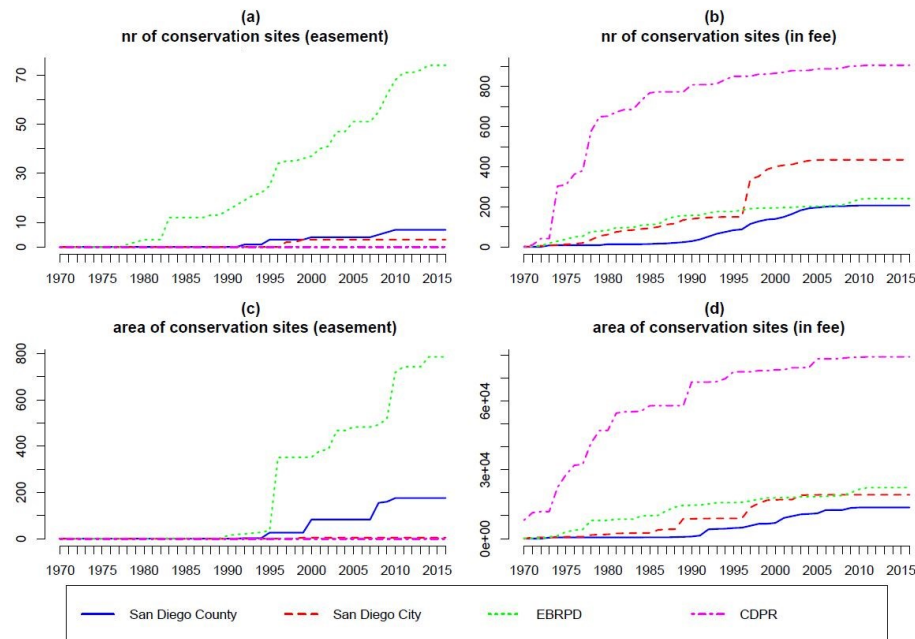


Figure D.1: Number of conservation properties in easement and in-fee for each of the 4 agencies, City of San Diego, County of San Diego, East Bay Regional Park District, and California Department of Parks and Recreation.

D.5 Sensitivity analysis

D.5.1 Purchasing prices

The impact of purchasing prices for land on costs for easements and in-fee managed areas is substantial. If, compare the base case value, we assume higher or lower values (see Tab. D.1 for a detailed parameter overview), a bandwidth of costs for easement and in-fee managed areas arises. Purchasing prices do have an impact on easement as well as in-fee costs, as they do directly influence the costs of land acquisition (in-fee), and indirectly also the opportunity costs (easements). Thus cost curves in both governance modes are shifted compared to the base case. Worth mentioning is that the total annual per hectare costs are still comparable in size across agencies and governance modes (Fig. D.2c and D.2d). From a cumulative perspective (Fig. D.3) the sensitivity analysis also shows, as would be expected, an increase in conservation costs (for easements and in-fee managed land) with increasing purchasing prices and a reduction in costs with decreasing purchasing prices. Effects here are as well consistent across agencies and governance modes.

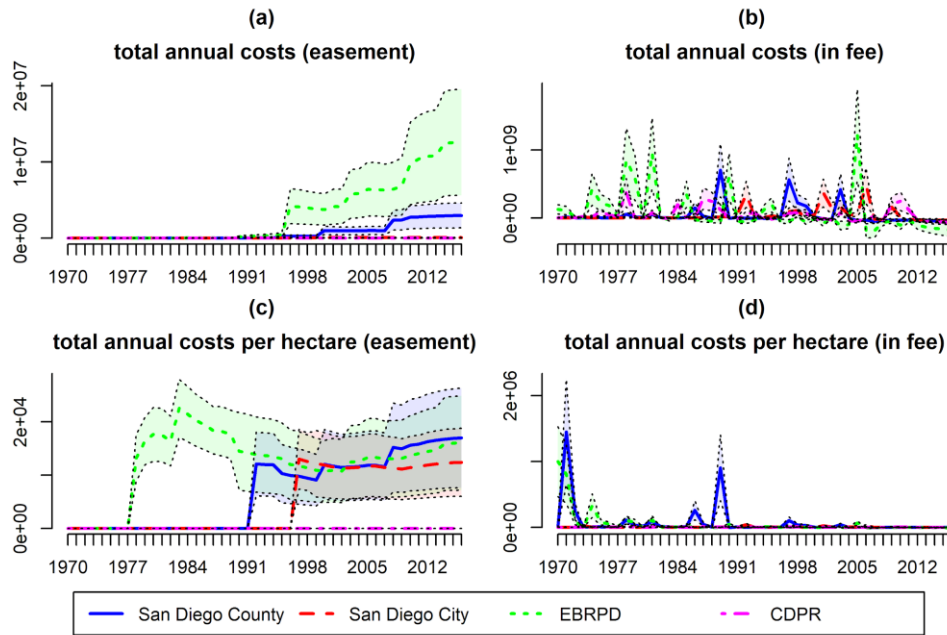


Figure D.2: The graphs show the development of total annual costs in either governance mode over time. The bold lines indicate cost developments for the base case parameter values. The shaded areas indicate the bandwidth for cost variations if higher or lower purchasing prices (and thus opportunity costs) are assumed. Higher purchasing prices generally cause an upwards shift of the cost curves, while lower prices cause a downwards shift.

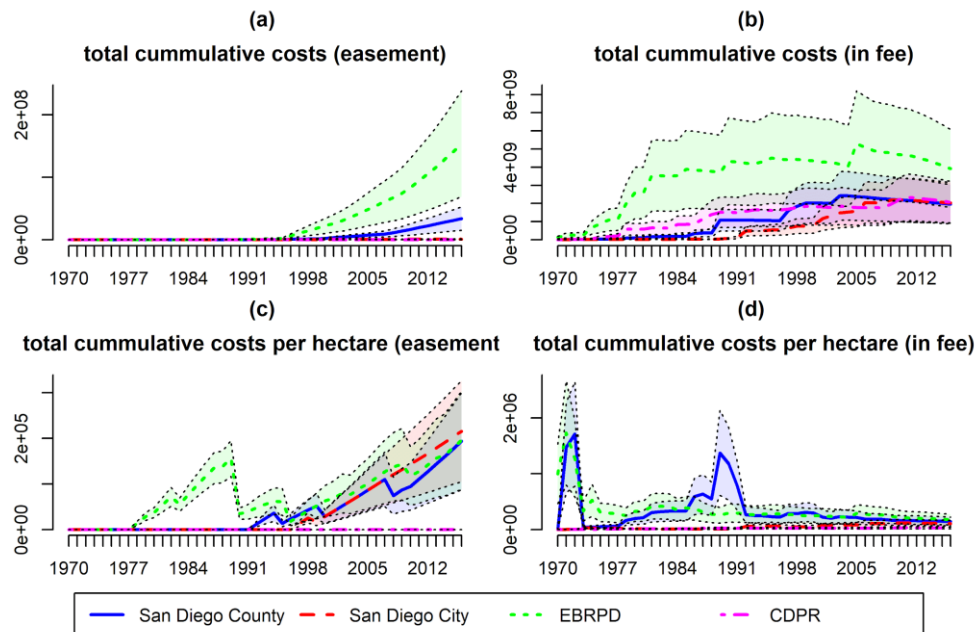


Figure D.3: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices.

D.5.2 Opportunity costs

Similar to the effects of changing purchasing prices, different amounts of opportunity costs change the costs of both governance modes. However, as opportunity costs are relatively high for easements in contrast to in-fee managed land, the effects of such costs changes are more substantial for easements as well (Fig. D.4). Only over time an increased level of opportunity costs also substantially increases the costs of in-fee managed lands. However, due to the overall stronger effect on easement costs, over time easements only become even less attractive with higher assumed opportunity costs. Only if opportunity costs are assumed low, total cumulative costs for easements over time can keep up with costs for in-fee managed lands. If such an assumption (low opportunity costs) however is reasonable is debatable.

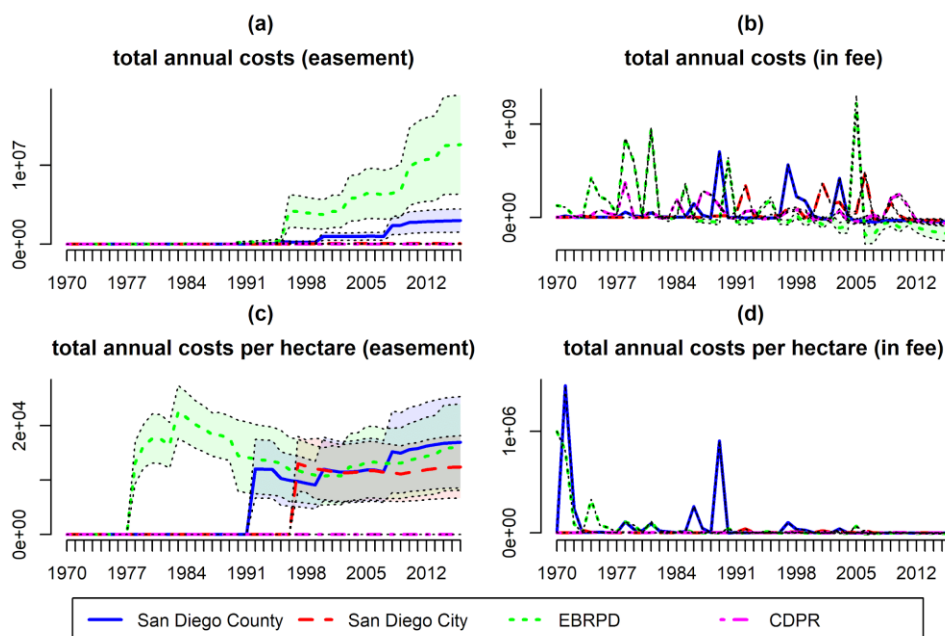


Figure D.4: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) opportunity costs.

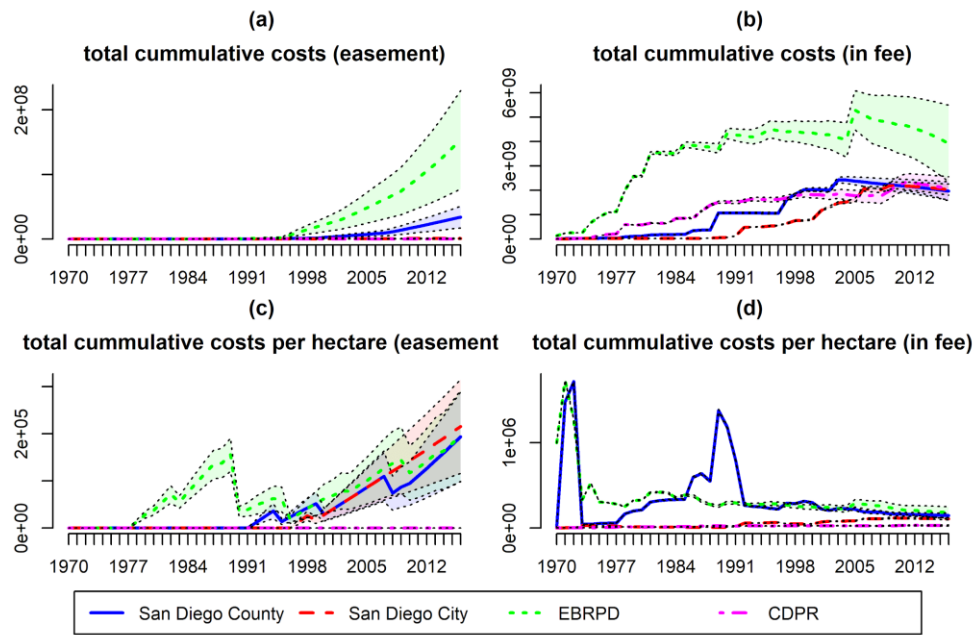


Figure D.5: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) opportunity costs.

D.5.3 Combined purchasing prices and opportunity costs

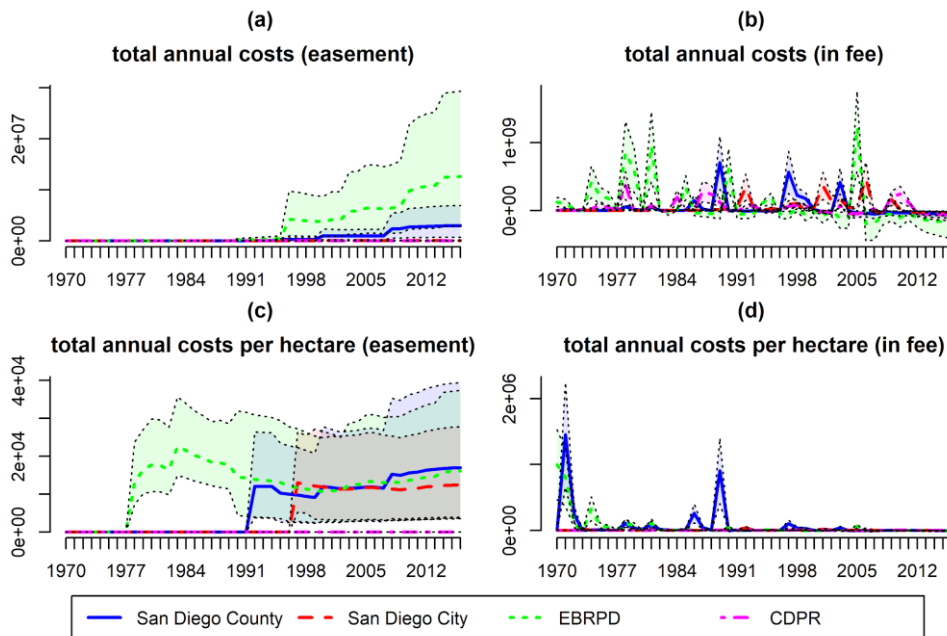


Figure D.6: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices and opportunity costs.

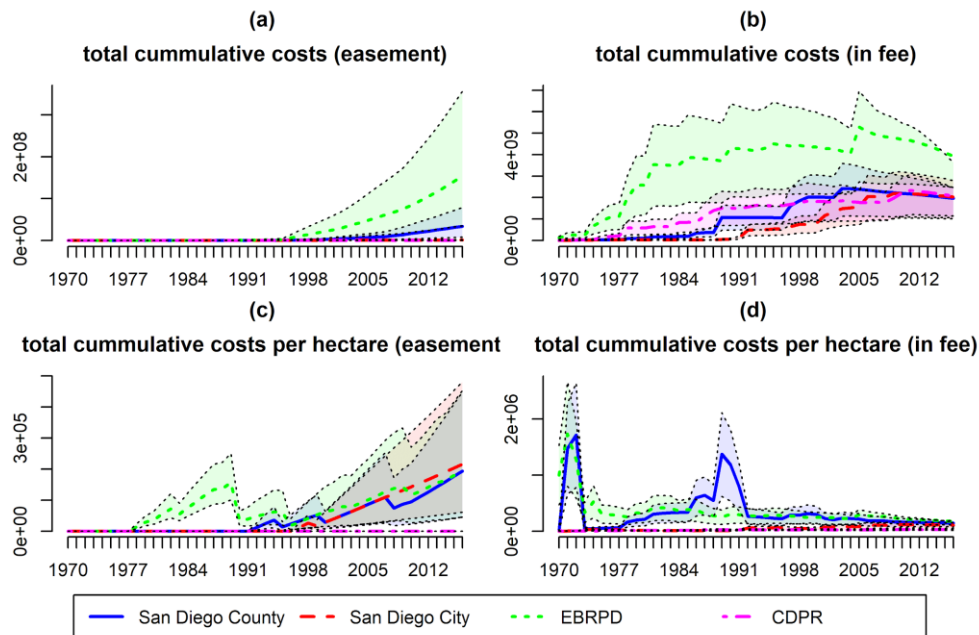


Figure D.7: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) purchasing prices and opportunity costs.

D.5.4 Interest rates

The assumption of higher (lower) interest rates also changes the development of costs in either governance mode. Early investments in easements or in-fee managed land are less (more) impactful with higher (lower) interest rates due to resulting lower (higher) discount rates. Thus, on an annual basis the costs bandwidth is relatively wide in early years, while it becomes more narrow over time (Figs. D.8 to D.11). Unlike in the former sensitivity analyses, interest rate changes, and thus changes in discount rates, also affect management costs.

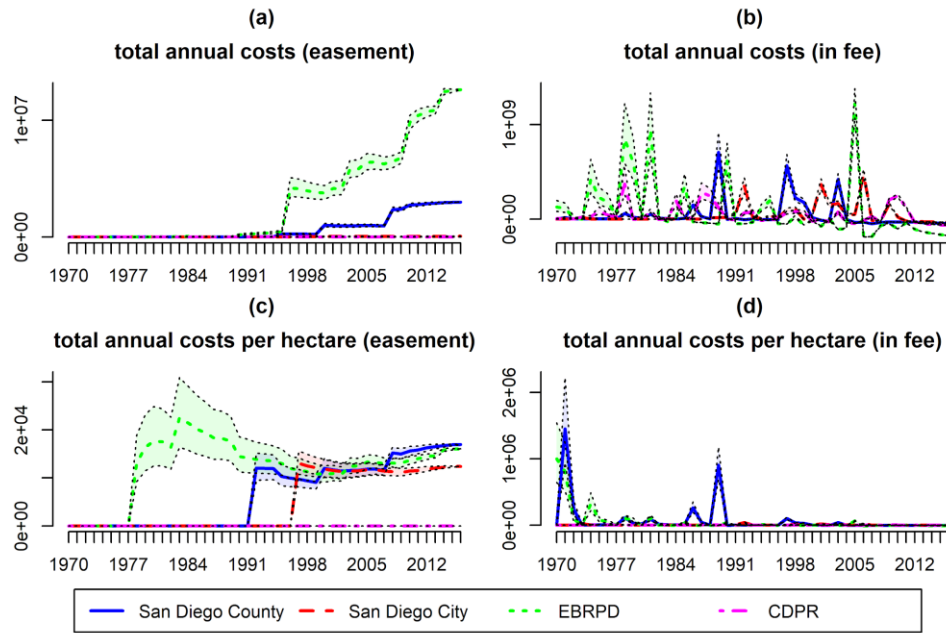


Figure D.8: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).

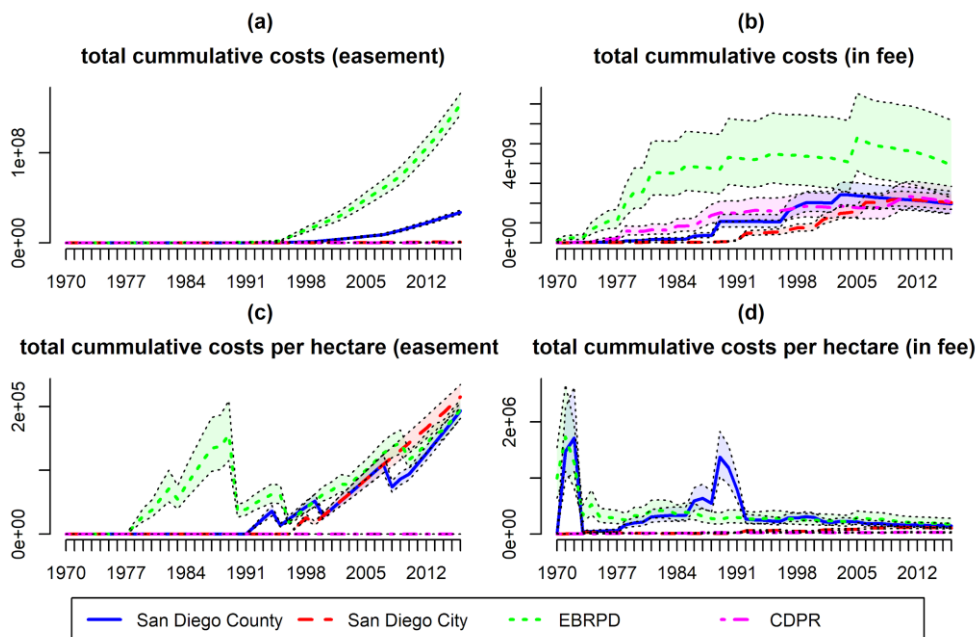


Figure D.9: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).

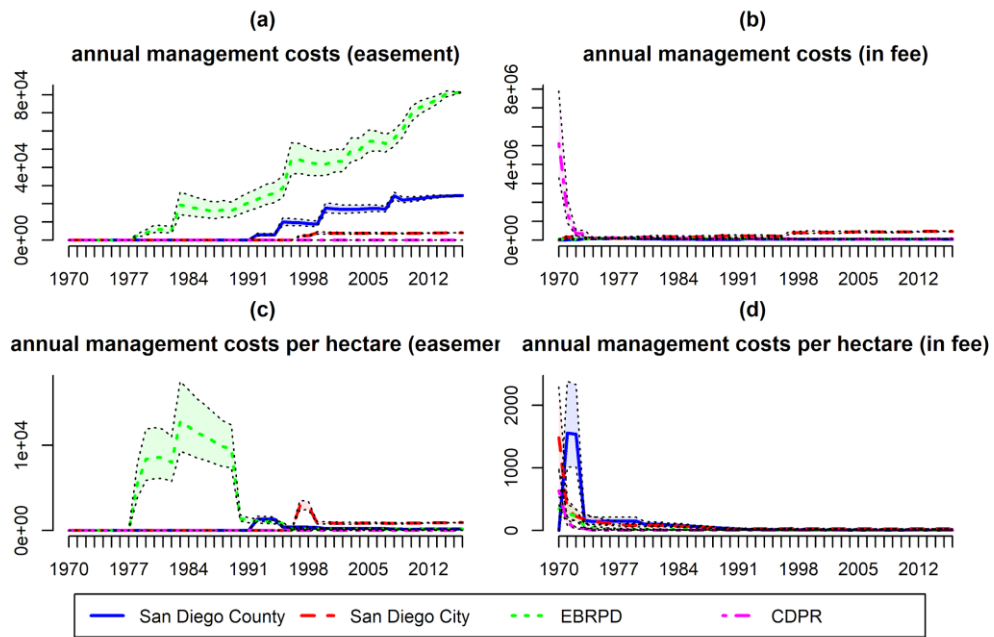


Figure D.10: Development of annual management costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).

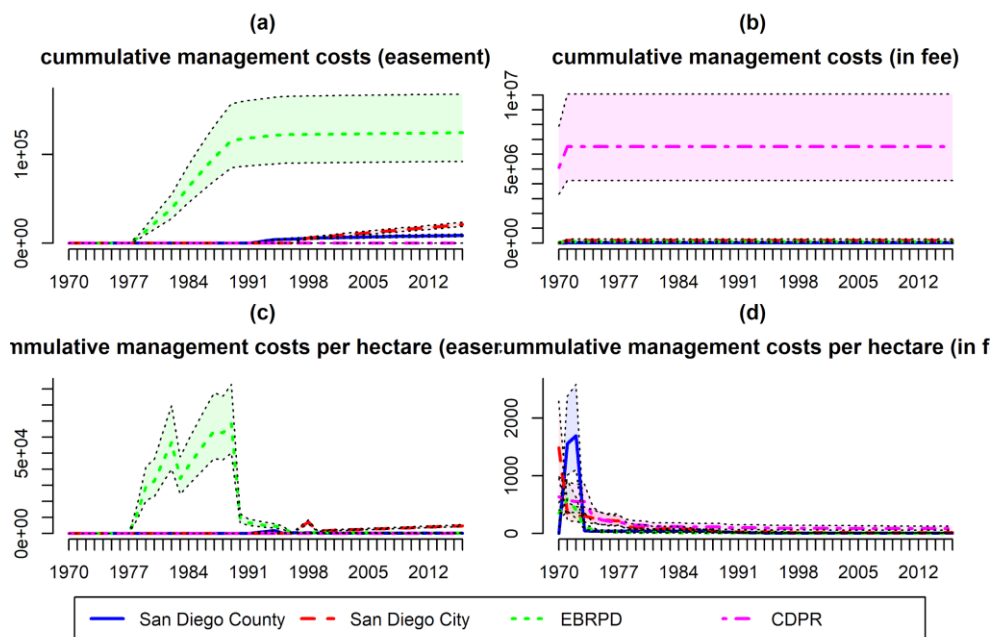


Figure D.11: Development of annual management costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) interest rates (and consequently discount rates).

D.5.6 Income

Variations in income do influence both management options in a comparable amount. Hence, the bandwidth introduced for both options is of comparable size. Due to the relative small general impact income has onto the total costs (relative to expenses), the introduced bandwidth is rather narrow and only marginally increases (with decreasing income) or decreases (with increasing income) costs.

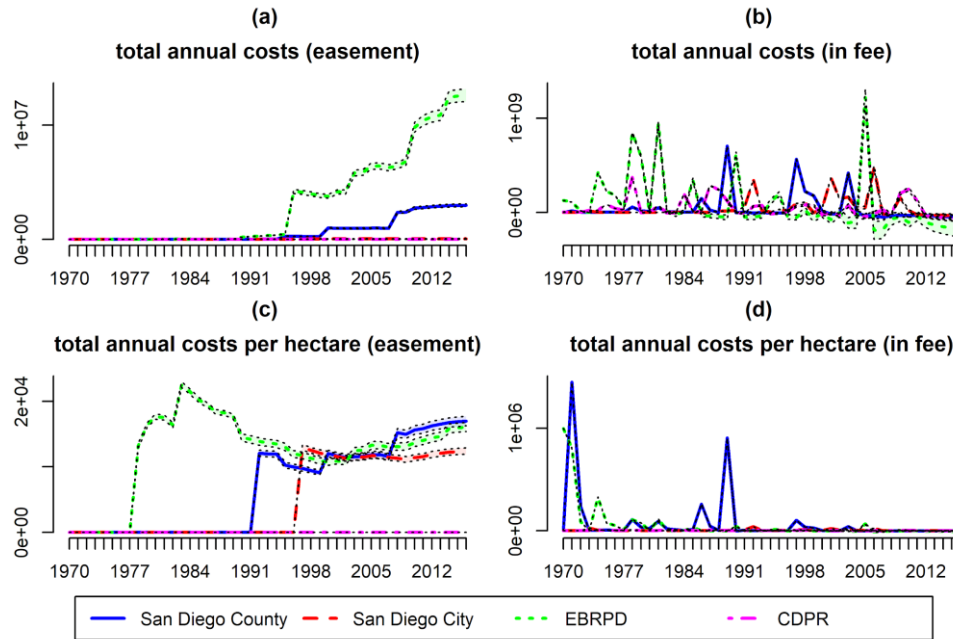


Figure D.12: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) income generated on managed land.

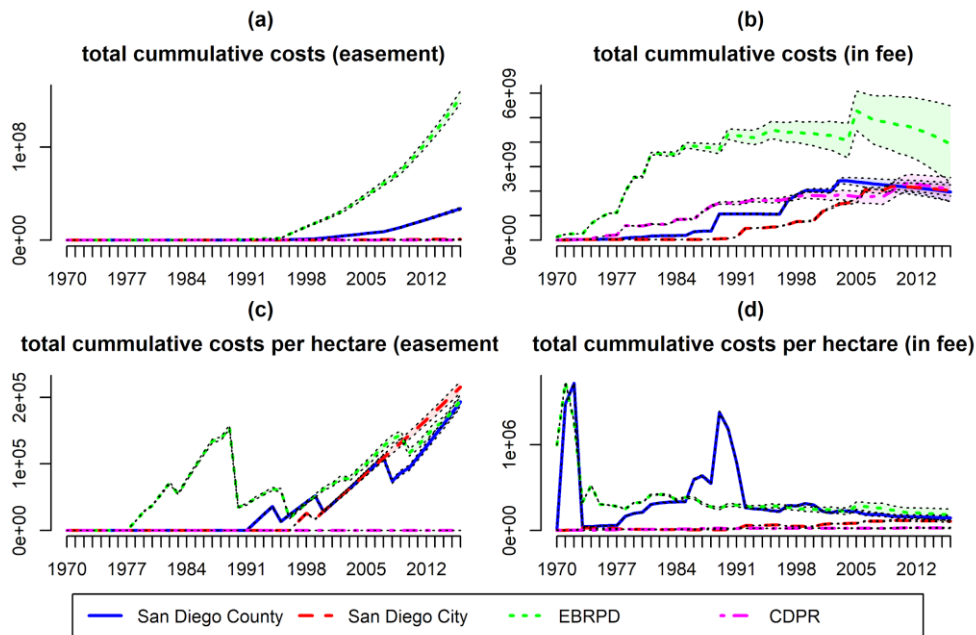


Figure D.13: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (lower edge) and lower (upper edge) income generated on managed land.

D.5.7 Transaction costs

Changes in transaction costs as well influence the cost curves of both management options and hence shift curves respectively. Mainly caused due the large area under conservation in total with in-fee managed land, and a consequent low average cost level per hectare the total annual costs per hectare and the total cumulative costs per hectare vary less with in-fee managed land compared to easements. Otherwise, the introduced bandwidths are of expectable size for both management options.

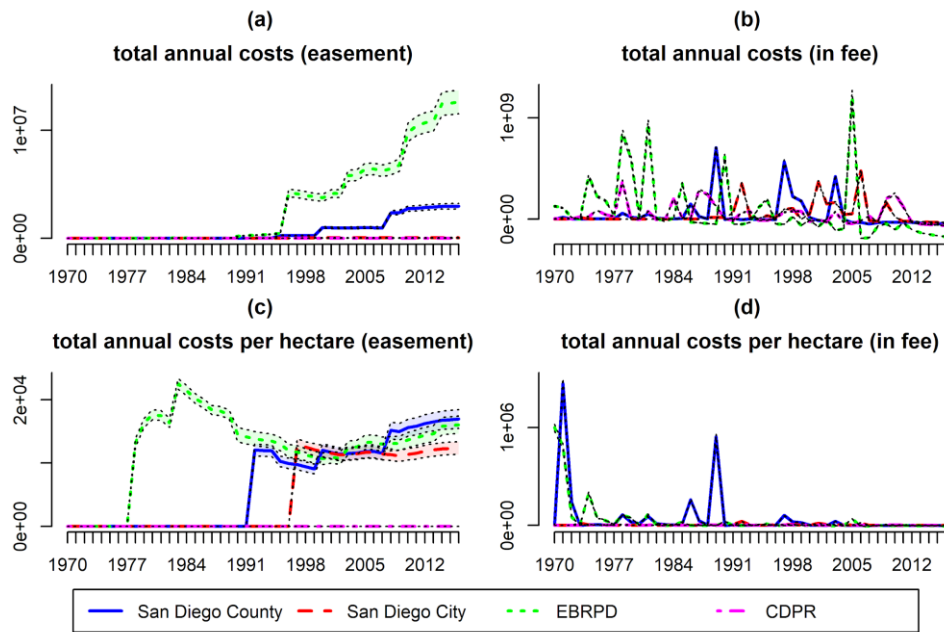


Figure D.14: Development of total annual costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) transaction costs.

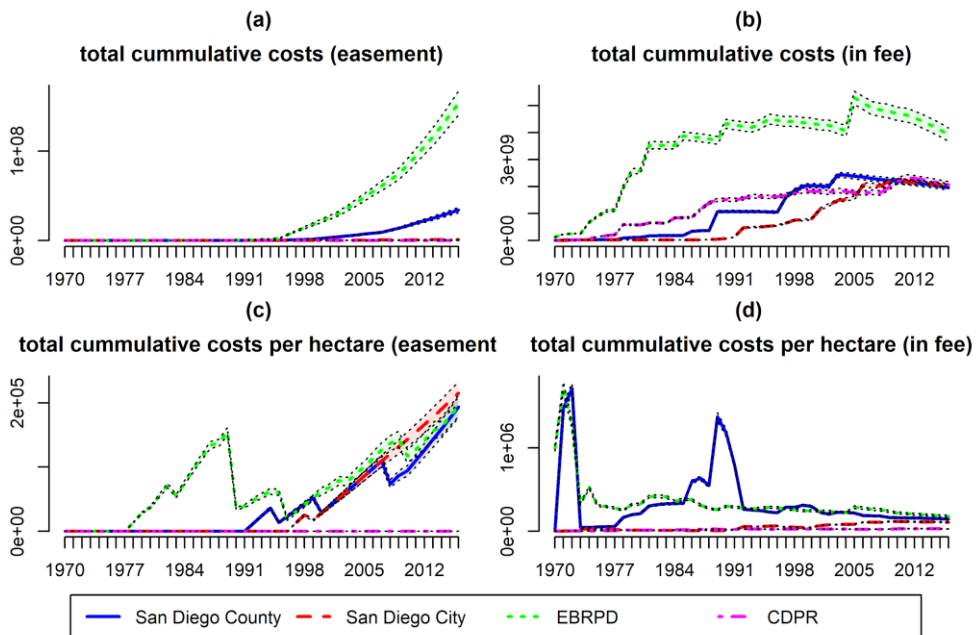


Figure D.15: Development of total cumulative costs in either governance mode over time with shaded areas representing the bandwidth of costs resulting from higher (upper edge) and lower (lower edge) transaction costs.

D.6 Extended discussion

Augmenting the analysis and considerations of conservation implementation costs presented in the main paper, a further, more political and institutional aspects of conservation and governance decisions needs to be made.

The finding that easements are more expensive than in-fee managed land goes in line with the economic rationale behind our analysis. By approaching expenses and income from different time periods, the necessity of discounting monetary values arises to generate a common monetary metric for all values. Discounting however, causes front-end loaded projects to be favoured over back-end loaded or evenly loaded projects. Against this background it is safe to say that due to the high amount of initial costs (i.e. purchasing expenses) with in-fee managed land, in contrast to mainly recurring costs with easements (i.e. regular tax reliefs), easements tend to be more expensive and thus less favourable from a cost-effectiveness point of view (i.e. reaching a given ecological target at lowest costs). However, it must be stated that easements might be superior to in-fee managed land from a purely ecological perspective. A multitude of reasons can cause ecological easement superiority, e.g. the fact that the state might not as good in running a conservation area from an ecological perspective as private landowner due to higher specialization, specific knowledge and experience of landowners in managing their own land. Another aspect in this regard might be that the implementation of privately and independently managed easements might have a higher robustness against politically induced change of preferences towards environmental protection. While the political regime might change and reduces environmental conservation on governmentally owned land, privately managed easements would still continue the provision ecological benefits.

On the other hand, if preferences for conservation within a state change and the government wants to change the conservation regime, management purchased land will not entail any additional cost or renegotiation. The owner of the land (i.e. the state) can immediately put into practice what the democratic elected government has decided. Easement negotiations have to be set up with private landowners to convince them and eventually implement newly the decided measures. This is true not only for private individuals managing their own land as easements, but also easements managed by larger organizations and NGOs (e.g. The Nature Conservancy). Furthermore, should private individuals or conservation organizations change their intentions or statutes, and

consequentially change the actual implementation of conservation activities, further negative ecological implications as well as socially not desirable consequences can arise. In short, with easements, the conservation elbowroom of the government is much smaller, making it even a less (and somehow undemocratic) institutional solution.

We find that beyond the analytical and empirical analysis of costs of conservation implementation, further research is necessary to look into the institutional and political dimension of conservation, conservation planning and consequential effects on conservation costs and benefits.