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RESEARCH ARTICLE

Economic assessment of rewilding versus agri-environmental nature management

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Abstract Policies aiming at improving biodiversity often consist of costly agri-environmental schemes, i.e. subsidized grazing or mowing of semi-natural areas. However, these practices have widely been found to be insufficient to mitigate biodiversity loss. Rewilding, i.e. restoring natural processes in self-sustaining biodiverse ecosystems, has been proposed as an alternative and is hypothesized to be a more cost-efficient approach to promote biodiversity conservation. Rewilding requires the availability of large natural areas which are not allocated for farming, forestry, and infrastructure to avoid potential conflicts over the use of the area. We perform an ex-ante private cost–benefit analysis of the establishment of four large nature reserves for rewilding in Denmark. We analyse the economic effects of changing from summer grazing in nature areas in combination with cultivated fields and forestry to the establishment of nature reserves in four case areas. We consider two scenarios involving conversion of agriculture and forestry areas into natural areas in combination with either extensive year-round cattle grazing or rewilding with wild large herbivores. In two case areas, it appears possible to establish large nature areas without incurring extra costs. Additionally, rewilding further reduces costs compared to year-round cattle grazing. Two opposing effects were dominant: increased economic rent occurred from the shift from summer grazing to year-round grazing or rewilding, while cessation of agriculture and forestry caused opportunity costs.

Keywords Biodiversity · Conservation grazing · Economic effects · Ecosystem restoration · Land sparing · Nature management

INTRODUCTION

Human impacts on global biodiversity are pervasive, with global extinction rates at more than 1000 times the natural background rate, mainly due to lack of appropriate habitats (Ceballos et al. 2017; Sánchez-Bayo and Wyckhuys 2019). Regarding terrestrial and freshwater ecosystems, the predominant driving force is the conversion and degradation of natural habitat for farming and forestry (Brondizio et al. 2019). To counteract habitat loss due to land use change, restoration and rewilding are increasingly being used as remedial actions. In 2019, the UN General Assembly officially adopted a resolution declaring 2021–2030 the UN Decade of Ecosystem Restoration.

Ecosystem restoration implies ambitious biodiversity targets and a firm commitment to long-term land designation for conservation, both of which align with scientific consensus regarding land sparing as the most cost-effective approach to halting or reversing biodiversity loss (Phalan et al. 2011). Ecosystem restoration involves reversing habitat deterioration from factors such as drainage (Zedler 2000), eutrophication (Bakker and Berendse 1999), coastal protection (Martínez et al. 2013) and defaunation (Dirzo et al. 2014). While all of these are important, we have focused the current analysis on the establishment of large nature reserves with reintroduction of large keystone herbivores, which is often referred to as rewilding (Svenning et al. 2019). However, we stress that such large reserves may also serve the purposes of hydrological, coastal and nutrient restoration as well as legal protection.

Large herds of wild animals once roamed in Europe. Aurochs, wild horse, wild boar, moose, red deer, bison and even elephants and rhinos razed, trampled and browsed through ecosystems, providing habitats and resources for a wealth of other species (Stuart 2015). Large mammals,

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along with other natural and dynamic processes, such as flooding, coastal erosion, and fire, characterized the evolutionary cradle for millions of years. Since then, severe defaunation has taken place, with notable extinctions of these large-bodied species (Dirzo et al. 2014). Today, wild mammals only make up 4% of the total mammal biomass on Earth, while humans and livestock make up 96% (Bar-On et al. 2018). In historical times, livestock still grazed the meadows, heathlands, grasslands, and woodlands, but agricultural improvements led to the replacement of semi-natural vegetation with reseeded and fertilized grass leys; horses were replaced by machines, and cattle are now mostly managed in cowsheds. More recently, grazing of nature areas, also deemed 'conservation grazing', is subsidized through agri-environmental schemes (AES) (Batáry et al. 2015) in the EU, and AES constitutes the largest expenditure on conservation in Europe (Batáry et al. 2015). Despite these efforts and despite an increased focus on biodiversity in AES (e.g., Matzdorf et al. 2008; Burton and Schwarz 2013), member states continue to report that large proportions of protected habitats have unfavourable conservation statuses and non-secure assessments of species (EEA 2015), indicating that biodiversity loss has not been halted.

Rewilding, often mentioned as a new or alternative conservation strategy, consists of restoring natural processes to promote more self-sustaining biodiverse ecosystems (Svenning et al. 2019). In practice, rewilding is often focused on the reintroduction of large mammals, notably large herbivores but also their predators, which are considered key species because of the diverse ecosystem processes they facilitate (Johnson 2009; Bakker et al. 2016). Restoring populations of large, wild-living herbivores requires large, contiguous, and often fenced nature reserves, both for ecosystem function and to avoid conflicts with agriculture and forestry (Hayward and Kerley 2009; Bull et al. 2018).

Although rewilding hints at reduced costs for landowners using words such as *self-sustaining* ecosystems and *reduced human interference*, rewilding also involves economic costs. Dedicated land sparing entails giving up or reducing privileges, such as hunting, logging, beekeeping, crop production, and meat production, potentially reducing the earnings of landowners. To date, actual economic analyses of typical AES-subsidized nature management versus rewilding strategies are very scarce. Examples are found in Brown et al. (2011), Ceauşu et al. (2015), and Sandom et al. (2018), but these studies are characterized by the use of various qualitative economic indicators. Ceauşu et al. (2015) presented a framework for analysing the interaction between socio-economic and ecological drivers, whereas Brown et al. (2011) and Sandom et al. (2018) discussed the use of rewilding in Scottish and English nature policies and the possible environmental, social, and economic benefits thereof. Given the limited resources in nature policies—as with any other policy, knowledge of the costs of different management strategies is important for policy decisions. In particular, studies comparing the costs of different management strategies may contribute insights into formulating cost-effective nature policies.

Building on the spatial analysis of Fløjgaard et al. (2017), which identified potential large contiguous nature and forest areas in Denmark, we chose four case areas for which we formulated and described a number of management scenarios and performed a full private cost–benefit analysis. The four case areas are selected based on variation in open habitat types and agricultural and forestry activities. To perform a consistent analysis, we compare a reference scenario (business as usual) and two alternative management scenarios. In the reference scenario, we assumed that common agri-environmental practices were applied on the open semi-natural and natural habitat types, primarily in the form of summer grazing and haymaking, while activities on cultivated fields and forests remained unchanged. Then, we performed an ex-ante analysis of the economic effects of shifting from the reference scenario to a nature reserve management scheme, in which the entire area is designated for natural purposes, exploring two strategies: (a) year-round cattle grazing and meat production and (b) rewilding with wild large herbivores. In both alternative scenarios, the production activities in cultivated agricultural and forest areas were abandoned.

MATERIALS AND METHODS

The steps of the analysis entailed (1) identification of the potential nature reserves using GIS analysis, (2) selection of four case areas and specification of their current land use and management practices, (3) private economic CBA (cost–benefit analysis), and (4) sensitivity analysis.

Spatial analysis

Fløjgaard et al. (2017) identified areas potentially suitable as large, contiguous nature reserves with the aim of meeting management needs in open semi-natural and natural habitat types (Fløjgaard et al. 2017). We collated not only the open habitat types protected by the Habitats Directive and included in Natura 2000 but also semi-natural and natural areas (bogs, meadows, saltmarsh, heathland, and dry grassland) that are protected by the Danish Nature Protection Act and that often qualify for AES. These habitat types are found more frequently along watercourses and the coast, and are, thus, not evenly distributed in the Danish landscape. In most cases, they are

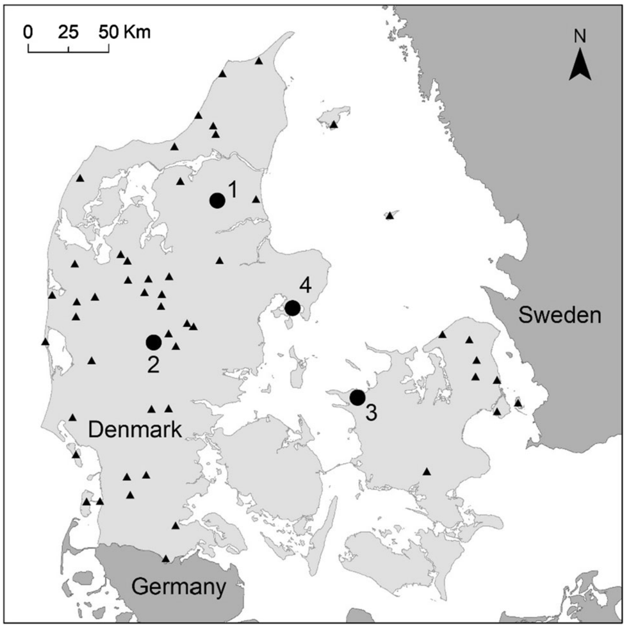

Fig. 1 An overview of 59 large nature areas identified by Fløjgaard et al. (2017). The four case areas are marked with large dots: (1) Rebild forest, (2) Nørlund plantation and heath, (3) Saltbæk Bay, and (4) Mols Hills

managed as small patches by different owners and may be divided by hedgerows, drainage canals, small arable fields, or roads. For the purpose of identifying potential large nature reserves, it was assumed that smaller management units could be merged and managed as a whole if the distance between them was less than 20 m, regardless of the dividing land use (e.g. a hedgerow or a small road). We fused neighbouring areas of open habitat types across different owners and management practices. Today, there is abrupt delineation between open land and forests that is primarily rooted in historical administrative barriers but that is today partly sustained by AES solely supporting initiatives in open habitat types. However, this delineation is meaningless in ecosystem restoration; as a result, we further merged open habitat types with adjacent state-owned forests. To increase connectivity and remove smaller interruptions in the nature reserves, we also included arable fields mainly surrounded by open habitat types and forest (> 60% semi-natural or natural habitat within a surrounding 200 m buffer of the field). In this way, we identified 55 potential nature reserves (>1000 ha, but up to 270 km^2) in Denmark.

Case areas

The four case areas were chosen from the 55 potential nature reserves to include variation in the cover of certain habitat types, e.g. from a high proportion of forest cover in the first two case areas and more open habitats in the last two case areas; see Fig. 1 and Table 1. The first case area, Rebild forest, is a hilly and sandy area formed by melt-water canyons from the last ice age. Although famous for its heathlands, Rebild forest is currently mostly forested (mostly coniferous plantation but also old-growth deciduous forest) and is owned and managed by the Danish Nature Agency. It largely overlaps with a Natura 2000 area (EEA site code DK00FX126) selected for valuable habitats (e.g. Lindenborg River valley) and protected bird species. The second case area, Nørlund plantation and heath, is also a sandy area and includes the Nørlund plantation, which has mostly coniferous trees, and Harrild heath. The heathlands are part of Natura 2000 area DK00BY165, which was appointed due to its valuable heaths, inland dunes, and raised bog. The area is mostly owned and managed by the Danish Nature Agency. The third case area, Saltbæk Bay, Western Zealand, is a part of Natura 2000 area DK005X221 and includes the terrestrial habitat types surrounding the bay, i.e. a large proportion of meadows, salt marshes, and fens that formed behind an embanked lagoon. The area is managed by both private landowners and the Danish Nature Agency. The fourth case area, Mols Hills, is a hilly and sandy area in Natura 2000 area DK00DX300, which was appointed because of its grasslands, heathlands, shrubs, and forests. Most of the grasslands are owned and managed by the Danish Nature Agency.

Table 1 Distribution of land use in the four case areas. Source: Fløjgaard et al. (2017)

Case areas	Total area (ha)	Natura 2000 (%)	Meadow (%)	Bog (%)	Heath (%)	Grassland (%)	Salt marsh (%)	Forest (%)	(%) (%)
1. Rebild forest	3941	76	22	15	3	7	0	45	9
2. Nørlund plantation and heath	3736	55	1	8	42	0	0	42	6
3. Saltbæk Bay	2354	82	27	22	7	4	21	7	10
4. Mols hills	1070	98	4	3	19	46	1	10	17

Table 2 Overview of management practices in the different scenarios

	Reference scenario	Year-round grazing with cattle	Rewilding
Summer grazing, cattle	+	–	–
Year-round grazing, cattle	+	+	–
Haymaking	+	–	–
Fencing	Small parcels	Around the perimeter	Around the perimeter
Forest production	+	–	–
Crop production	+	–	–
Agri-environmental subsidies	–	–	–

Reference scenario and alternative scenarios

The costs of the scenarios representing different alternative management schemes were measured against a reference scenario, in which the open habitat types were assumed to be managed by summer grazing, and agricultural and forest areas were subject to intensive production in rotation. The reference scenario corresponded to a business-as-usual management scheme in Denmark. The two alternative scenarios were based on the conversion of production areas to nature and on management of the areas as nature reserves, either with year-round grazing with cattle or with large wild herbivores, such as roe deer, red deer, or fallow deer. Although both alternative scenarios represented rewilding to some degree, we referred to the latter as the "rewilding scenario". Although the case areas were relatively remote and rural, the mere size of the areas entailed that existing infrastructure, such as roads, railways, and buildings would be included in the fenced area. However, the effect of infrastructural changes on form and function was not included in the analysis. In Table 2, the different assumptions regarding management practice were outlined for the scenarios.

Economic analysis

The private CBA was performed by comparing the net present value of the revenue and costs of the different economic activities, as shown in formula 1.

$$\mathrm{NPV} = \sum_{t=0}^{T} \frac{B_t - C_t}{(1+r)^t} \tag{1}$$

where NPV is the net present value, B_t is the sum of the revenue (benefits) in period t, C_t is the sum of the costs in period t, T is the time horizon of the project, and r is the discount rate.

Then, the net present value was transformed into a yearly amount (the annual economic rent, R) using the annuity factor, enabling us to compare the economic effects of economic activities with various time horizons. We used a discount rate of 4% when calculating the NPV, and all values were in 2016 prices. A discount rate of 2% was applied in the calculations of the expected values for the forest rotations according to Jacobsen and Meilby (2018). All economic figures were converted from DKK to USD using the 2018 average exchange rate of 633 according to the Danish National Bank (2020).

Only revenue and costs from marketed goods were included in our private CBA. This implies, for example, that the revenues from agricultural activities include the value of grains and meat and the costs consisted of input (e.g. fodder, fencing and fertilizers), capital (e.g. machines and land), and labour, whereas the derived effects on public goods and externalities (e.g. recreational activities, eutrophication, and GHG emissions) were not included in the analysis.[1] Last, EU-funded AES subsidies were not included because the future payment schemes within the Common Agricultural Policy are uncertain.

Change of and use are the determining factors for the reallocation of activities within the case areas. Denoting the area allocated to activity i in scenario j as $A_{i,j}$ and the annual economic rent of activity i as R_i, we can determine the aggregated economic rent in scenario j (R_j^T) as follows in formula 2:

$$R_j^T = \sum_{i,j}^{I,J} R_i A_{i,j} \tag{2}$$

Now, the aggregated economic effect of reallocating land use from the reference scenario to an alternative scenario can be found as the difference between the aggregated economic rent from the reference scenario and the alternative scenario.

We operated with six economic activities representing the different types of land use relevant for the analysis, and for each activity, we estimated the economic rent or profit, i.e. the remuneration of land after costs of all inputs and capital had been covered. The opportunity costs of

[1] Public goods and externalities would be included in a social CBA; See Freeman et al. (2014) for elaboration on private CBAs and social CBAs.

agricultural activities on rotational land were estimated as the average economic rent from winter wheat, which is the most common crop in Denmark. The data were derived from Farm Account Statistics (Farmtal Online 2020) and the methodology followed, e.g. Schou and Abildtrup (2005). The opportunity costs of forest production were derived from Jacobsen and Meilby (2018) and varied between the case areas due to variations in the tree species and the ages of the stock. Opportunity costs increased the more hardwood species there were and the older the stock. Further, we carried out a sensitivity analysis on the opportunity costs of the forest, in which only 200 m^3/ha of standing volume had to remain. This made it possible to remove the most valuable part of the standing volume in forest areas where after, the forests are left completely untouched. The costs of fencing were included in the economic rent of summer grazing as these were concentrated in small parcels. The cost of fencing in the year-round grazing and rewilding scenarios did not fully reflect the geographical/GIS-derived areas but were rather an approximation based on the cost of fencing a rectangular area of similar size using a solid wire fence. The delineation of the case areas was a result of an automatic grouping of neighbouring land parcels. This method are not intended for direct implementation, as detailed project projection would be required to clarify the exact borders (Fløjgaard et al. 2017). We used an investment cost of 40 USD/m of fence based on our own survey of market data, corresponding to an annualized cost of 1.7 USD/m with an expected durability of 40 years. Other investment costs related to the scenarios, such as restoring hydrology (e.g. actively closing drains) and adjustment of infrastructure, were not included in the analysis. Note, that management through rewilding does not include veterinary costs, regulation of the number of animals, securing public access to the area, or other types of more specific management actions.

RESULTS

In Table 3, the key economic figures used in the scenario analysis are shown.

Nature management generally results in negative economic rent (i.e. cost), as the revenue from cattle grazing and haymaking does not cover the production costs (Table 3). However, the economic rent increases with the productivity of the habitat types, and therefore, we saw that

Table 3 Economic rent of the agricultural and nature management activities (USD/ha/year) specified for common open habitat types and forests. Source: Own calculations based on Thomsen et al. (2018)

Economic rent of the agricultural and nature management activities, USD/ha per year				
Crop prod. in rotation	632	–	–	–
Nature management	Meadow	Bog and Heath	Grassland	Salt march
Summer grazing, cattle	− 1216	− 268	− 774	− 505
Haymaking	− 189	− 205	− 205	− 205
Year-round grazing, cattle	− 253	− 32	− 126	− 79

Economic rent of the forest activities, USD/ha per year		
Case areas	All standing volume is left	200 m^3/ha of the standing volume is left
1. Rebild forest	347	284
2. Nørlund plantation and heath	142	126
3. Saltbæk Bay	268	268
4. Mols hills	221	205

Perimeter and cost of fencing in the year-round grazing and rewilding scenarios		
Case areas	Perimeter, km	Fencing cost, 1000 USD/year
1. Rebild forest	25.2	44
2. Nørlund plantation and heath	24.3	42
3. Saltbæk Bay	19.2	33
4. Mols hills	13.1	23

The cost of fencing is based on an approximation based on the cost of fencing a rectangular area of similar size and the actual geographical areas (polygons)

Table 4 Overview of the case areas in terms of the area designated for nature management and the economic rent (USD/ha/year) of the scenarios measured per hectare of land designated for nature management

	1. Rebild forest	2. Nørlund plantation and heath	3. Saltbæk Bay	4. Mols hills
Area designated for nature management, ha				
Reference scenario	1895	1938	1908	781
Year-round grazing and rewilding scenarios	3941	3736	2354	1070
Economic rent, USD/ha of land designated for nature management				
Reference				
	− 68	− 72	− 341	− 250
Year-round grazing with cattle				
No logging	− 288	− 130	− 194	− 225
Limited logging leaving 200 m^3/ha	− 263	− 123	− 194	− 224
Rewilding (grazing with wild large natural herbivores)				
No logging	− 217	− 109	− 95	− 150
Limited logging leaving 200 m^3/ha	− 192	− 101	− 95	− 148

managing productive meadows resulted in a higher (but still negative) economic rent than the less productive heathlands. Additionally, the cost of natural management decreases with extensification of the grazing regime, as the negative economic rent was almost constant per animal. The fields that were interspersed with crop production and forested areas resulted in positive economic rent. The economic rent of forest activities per case area depended on the value of the standing crop and showed the high value of hardwood trees in Rebild forest compared to the lower value of coniferous plantation on the sandy soils in the Nørlund plantation and heath (Table 3). The difference in economic rent between leaving all standing volume and logging to 200 m^3/ha standing volume remaining was minor.

Turning to the results from the scenario analysis, shifting from the reference scenario to the nature reserve scenarios increases the land designated for nature management. This was most pronounced in Rebild forest and Nørlund plantation and heath, where the area designated for nature doubled, and less pronounced in Saltbæk Bay and Mols Hills, which already had a large proportion of nature areas in the reference scenario (Table 4).

The changes in aggregated economic rent stemmed from *a.* the loss of economic rent due to termination of intensive agricultural and forest production; *b.* the negative economic rent from implementing nature management in a larger area; and *c.* increased economic rent due to shifting from summer grazing to year-round grazing or rewilding in existing nature areas.

The alternative scenarios led to decreased aggregated economic rent for Rebild forest and Nørlund plantation and heath, reflecting the opportunity costs of transforming production areas—notably the large forests included in the case areas—to nature areas and thereby a significant de facto increase in the nature area. The opposite is seen for the case areas Saltbæk Bay and Mols Hills, as they already included a large share of open semi-natural and natural areas in the reference scenario and therefore also a smaller proportion of land to be converted from intensive production activities to nature. In these case areas, shifting from the reference scenario to year-round grazing increased the aggregated economic rent, and rewilding further increased the aggregated economic rent (Table 4).

Last, the results showed negative aggregated economic rent within the case areas in all scenarios. This goes even for the reference scenario despite the positive economic rent from the land parcels allocated for intensive agricultural and forest production and shows that the aggregate negative economic rent from the current nature areas managed with summer grazing outweighs the benefits from the more productive agricultural and forest areas unless EAS subsidies are included.

DISCUSSION

Implications for biodiversity conservation

The four case areas analysed were largely covered by Natura 2000 areas and designated for their species and habitat contributions to European biodiversity. By far, most of the funding for conservation initiatives in these areas comes from AES. The efficiency of such initiatives for biodiversity conservation in Denmark has never been directly evaluated, but national assessments of biodiversity have shown that biodiversity is declining and that the majority of the protected habitats and species designated by the Habitats Directive have unfavourable conservation statuses (Ejrnæs et al. 2011; European Environment Agency, 2015). Overall, this indicates that AES is ineffective and/or inadequate as a conservation tool: This realization has fostered an increasing interest in re-establishing natural dynamic processes by applying rewilding strategies, e.g. Jepson (2016), Jepson et al. (2018), Navarro and Pereira (2015). A further argument for pursuing rewilding strategies is that most species existing in Europe today did not evolve in a landscape characterized by production and maintained by agricultural practices—most species evolved in the Pleistocene or earlier (Kurtén 1968; Lang 1994; Coope 2004) and in ecosystems markedly influenced by dynamic natural processes, including grazing (see, e.g. MacFadden 1997; Sandom et al., 2014; Galetti et al. 2019; Gardner et al., 2019). Large herbivores contribute complex and diverse processes to ecosystems, benefitting biodiversity through seed dispersal, diversification of carbon sources, and nutrient cycling (Doughty et al., 2016; Malhi et al., 2016; Pires et al., 2017). These processes are impossible to fully substitute with agricultural practices. With this baseline for European ecosystems and biodiversity, restoring natural processes by reintroducing large herbivores is our best suggestion for maintaining biodiverse, self-sustaining ecosystems.

In this context, our finding that rewilding is not necessarily less economically attractive than traditional AES delivers important input to policy discussions. Agricultural activities seem to be inherent in the EU Common Agricultural Policy—at least in the current design—and it is therefore promising that natural grazing processes can also be partly restored by year-round grazing with cattle and horses at densities determined by available forage and without supplementary feeding (Vermeulen 2015). Dedomestication of these species can restore the ecosystem functions of aurochs and wild horses, as often seen in rewilding projects in Europe in combination with other ungulates, i.e. Oostvaardersplassen with cattle, horses, and red deer (Vera 2009) and Kraansvlak with cattle, horses, and European bison in the Netherlands (Cromsigt et al. 2018).

Large nature reserves present many benefits to biodiversity related to the sheer size and connectivity of habitats within reserves. Large reserves increase the persistence and survival of rare species (Higgs 1981). Large reserves may support minimum viable populations (Shaffer 1981) of large herbivores, enabling rewilding with minimal population management. Other natural processes benefitting biodiversity, such as fire and flooding, may also be restored in large reserves without conflict with other land uses.

Policy perspectives

The cost-efficiency of the current AES for biodiversity conservation has been questioned in several economic studies, e.g. Merckx and Pereira (2015). The critiques have included that farmers lack knowledge and motivation to succeed in managing nature areas for conservation, and several case studies and projects have attempted to develop and test result-based agri-environmental schemes with a focus on biodiversity as a management output (Matzdorf and Lorenz 2010; de Snoo et al. 2013). Various policy evaluations and case studies have documented these factors, e.g. Beharry-Borg et al. (2013), de Sainte Marie (2014), and Droste et al. (2018). The results from Beharry-Borg et al. (2013) showed that farmers' willingness to accept subsidies for changing land use to improve water quality were dependent on their farming practice, whereas de Sainte Marie (2014) and Droste et al. (2018) provided insights into how to design AES to enhance the protection of biodiversity and improve the effectiveness of conservation policies.

In our analysis, we found that applying comprehensive management schemes for biodiversity purposes led to negative economic rents if we disregarded the possibility of receiving payments for agri-environmental management schemes under pillar II of the EU Common Agricultural Policy (CAP). In a practical policy setting, farmers will most likely receive subsidies under the CAP to compensate for the negative economic outcome from extensive grazing. The current Danish subsidy for extensive grazing regimes under the CAP amount to 260 USD per hectare per year (The Danish Agricultural Agency 2020). If we compare this to the economic rent of summer and year-round grazing management, as shown in Table 3, this subsidy seems sufficient to compensate for the negative economic rent from these activities on all nature types.

However, eligibility to receive the EU agri-environment is conditional on maintaining some degree of agricultural practices on the land. Under the current EU regulations, biodiversity management through rewilding does not necessarily comply with these conditions. However, year-round grazing with domestic animals can also entail some degree of rewilding while being eligible to receive subsidies, making it a favourable rewilding strategy if future EU agri-environmental subsidies require maintaining agricultural use of the land. This point was also raised by Sandom et al. (2018) and becomes even more evident if the right to receive basic payments under the basic payment scheme under pillar 1 of the CAP are lost when shifting from domestic animals to rewilding with natural herbivores.

This points to an important policy failure. The CAP in itself makes AES more economically advantageous for both farmers and public policy-makers compared to rewilding or other non-agricultural conservation activities regardless of their effects on biodiversity. From the landowners' point of view, the choice is simple, as the all-else-equal situation favours receiving subsidies over not receiving subsidies. This goes even for national policy-makers because approximately 50% of the subsidies payed under the AES were financed over the EU budget (The Danish Agricultural Agency 2020). This transfer does not affect the contributions from the member states to the EU budget, as this is based on the gross national income (European Commission, 2020); therefore, the part of the subsidy financed by the EU can be considered a direct transfer from the EU to the member states.

The policy failure becomes even more obvious when considering the results from our economic analysis with the expected outcome for biodiversity under the management schemes discussed in the previous section. On the one hand, we find that rewilding in certain situations is economically advantageous to AES if EU subsidies are disregarded. Furthermore, as shown in the previous section, a large body of literature on the biological effects of different management schemes points to rewilding being the preferred management option, especially when it is possible to establish large coherent nature areas. Thus, the rules of compliance for being eligible for payment under the EU AES in combination with the mechanism for financing the EU budget do in some situations counteract the scientifically and socio-economic preferable management scheme.

CONCLUSIONS

In this paper, we performed an economic scenario analysis of three management schemes for nature conservation. We compared the economic rent in a business-as-usual scenario involving a combination of crop production, forestry, and summer grazing in protected nature areas and two biodiversity scenarios in which the whole area was designated as nature reserves by conversion of areas previously used for agriculture and forestry to biodiversity purposes, with extensive year-round cattle grazing or rewilding with wild large herbivores. By comparing the aggregate economic rent in each scenario, we found that shifting from summer grazing to year-round grazing was economically advantageous and switching to rewilding was even more economically attractive. However, since abandoning agriculture and forestry leads to a loss of economic rent, the economic effects of shifting from the reference scenario to one of the biodiversity scenarios depend highly on the size of the areas shifting from forest and agricultural activities to winter grazing or rewilding. In case areas with large nature areas in the reference scenario, the savings from shifting from summer grazing to year-round grazing and rewilding were larger and may lead to increased aggregate economic rent. However, in the case of areas with high shares of land allocated to intensive forest and agricultural production in the reference scenario, designating the area for biodiversity purposes led to a loss in aggregate economic rent.

To our knowledge, there have been no previous assessments of the cost-efficiency of nature management, including comparisons with rewilding scenarios. The results of our scenario analyses were ambiguous but still provided some general insights to be considered when deciding upon future nature management schemes. Considering the EU 2030 biodiversity strategy's goal to reserve 30% of terrestrial land for nature in the EU, the importance of opportunity costs should be stressed. Moreover, our analysis showed that the grazing strategy of the area designated for natural purposes was essential for economic effects. Grazing regimes based on summer grazing were the least advantageous, followed by year-round grazing and rewilding. Furthermore, shifting from the management of small fragmented nature areas to a larger coherent area reduces the per hectare management costs. However, the establishment of large natural areas may also result in additional costs because of the need to adjust infrastructure to circumvent the larger contiguous reserves or to actively remove drainage to promote natural hydrology in the area.

The negative aggregated economic rent of the scenarios also implied that some kind of financial transfer or subsidy would be required if the scenarios were to be implemented. This means that the economic effects of various nature management schemes were clearly affected by the possibility of receiving subsidies from the EU Common Agricultural Policy (CAP). In the current analysis, we left the option for receiving AES out as the EU currently is moving to reform the CAP. The reform will most likely lead to changes in the general subsidies and the subsidies targeted at integrating nature objectives in agricultural practices. Additionally, it is to be expected that the current cap on the total payments will remain, and given Brexit and interests from recent member countries, this cap may even be subject to reduction for high-income countries such as Denmark. For this reason, it is relatively uncertain how subsidies will develop in the years to come. Thus, if future AES subsidies remain unchanged and only agricultural areas are eligible for payments, year-round grazing with domestic animals will be clearly favourable to a rewilding strategy.

Finally, though the primary objective of establishing large nature areas is to provide socio-economic benefits related to biodiversity, these areas are also likely to provide recreational benefits in terms of nature tourism (Tisdell 2004), thus inducing reluctance in some visitors to areas with the risk of meeting large animals. Both positive and negative socio-economic effects, such as impacts on the local community, recreation, and the public attitude, are important to manage if one wants to pursue rewilding-based nature policies as discussed in, e.g. Bauer and von Altzigen (2019). In summary, our analysis showed that rewilding and year-round grazing may in some cases be both an economically preferable management strategy for increasing biodiversity in large nature areas even though the current AES under the CAP favours year-round cattle grazing.

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