Governing dual objectives within single policy mixes: an empirical analysis of large carnivore policies in six European countries

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Routledge
Taylor & Francis Group

Journal of Environmental Policy & Planning

ISSN: (Print) (Online) Journal homepage: https://www.tandfonline.com/loi/cjoe20

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To cite this article: Auvikki de Boon , Camilla Sandström , Ugo Arbieu , Inger Hansen , Lisa Lehnen , Agnese Marino , Mari Pohja-Mykrä , Camilla Risvoll , Geir-Harald Strand & Katrina Rønningen (2020): Governing dual objectives within single policy mixes: an empirical analysis of large carnivore policies in six European countries, Journal of Environmental Policy & Planning, DOI: 10.1080/1523908X.2020.1841614

To link to this article: https://doi.org/10.1080/1523908X.2020.1841614

JOURNAL OF ENVIRONMENTAL POLICY & PLANNING
https://doi.org/10.1080/1523908X.2020.1841614

Governing dual objectives within single policy mixes: an empirical analysis of large carnivore policies in six European countries

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ABSTRACT

Policy mixes (i.e. the total structure of policy processes, strategies, and instruments) are complex constructs that can quickly become incoherent, inconsistent, and incomprehensive. This is amplified when the policy mix strives to meet multiple objectives simultaneously, such as in the case of large carnivore policy mixes. Building on Rogge and Reichardt's analytical framework for the analysis of policy mixes, we compare the policy mixes of Norway, Sweden, Finland, the Netherlands, Germany (specifically Saxony and Bavaria), and Spain (specifically Castilla y León). The study shows that the large carnivore policy mixes in the case countries show signs of lacking vertical and horizontal coherence in the design of policy processes, weak consistency between objectives and designated policy instruments, and, as a consequence, lacking comprehensiveness. We conclude that creating consistent, coherent, and comprehensive policy mixes that build on multiple objectives requires stepping away from sectorized policy development, toward a holistic, systemic approach, strong collaborative structures across policy boundaries and regions, the inclusion of diverse stakeholders, and constant care and attention to address all objectives simultaneously rather than in isolation.

ARTICLE HISTORY
Received 28 March 2020
Accepted 19 October 2020

KEYWORDS
Large carnivore management; policy instruments; human-carnivore conflict; institutional and systemic failure

Introduction

Public policies are complex constructs, often including a variety of instruments, targeted actors, and implementation processes to steer society towards certain agreed objectives (Schneider & Ingram, 2005). In the case of policies for sustainable development, the complexity is amplified by the need to meet multiple objectives simultaneously (Baker, 2012). Policies regularly evolve in an incremental process of adding and altering elements whenever the original policy structure seems incapable of effectively addressing societal and/or ecological challenges that preclude achievement of the set objectives (Kern & Howlett, 2009). Hence, designing what Rogge and Reichardt (2016) define as policy mixes, i.e. the total structure of policy processes, strategies, and instruments, with multiple objectives in a unidirectional way, can be eminently

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Supplemental data for this article can be accessed https://doi.org/10.1080/1523908X.2020.1841614

challenging. The overarching overview may become lost and, as a result, the policy mix can lose its coherence and consistency (Cunningham et al., 2013, pp. 2–3) and thereby its effectiveness (Kern & Howlett, 2009).

Large carnivore policy is a case where multiple objectives are often combined within a single policy mix. As some large carnivore species are increasing their range and numbers in areas where they have been absent or declining for decades (Chapron et al., 2014), countries are striving to find ways in which humans and large carnivores can thrive together. For these efforts to become effective and sustainable, we argue that it is important that the policy mix in place is internally consistent and coherent.

Despite numerous studies on the impact of policy instruments used to mitigate human-carnivore conflicts (e.g. Bautista et al., 2019; Eklund, 2019; Treves et al., 2016; Trouwborst, 2018), there has been little comprehensive research considering the policy mix as a whole (exemptions are e.g. Hansson-Forman et al., 2018; Stöhr & Coimbra, 2013). After years of evolution of, and amendments to, the policy mix in many European countries, it is unclear whether policy mixes currently in use are internally coherent and consistent. The objective of this article is to address this knowledge gap by comparing the large carnivore policy mix of various European countries or regions, with the aim of contributing to cross-national learning and to provide guidance for policy design. We see large carnivore policy as representative for contentious environmental issues including strong socioeconomic and environmental values (Redpath et al., 2017). Our analysis includes the following species: wolf (*Canis lupus*), bear (*Ursus arctos*), lynx (*Lynx lynx; Lynx pardinus*), wolverine (*Gulo gulo*), and golden eagle (*Aquila chrysaetos).* By further operationalising and applying an analytical framework for policy mix analysis in a new setting, our article also contributes to the discussion on environmental policy mixes and on the challenge to create coherent, consistent, and comprehensive policy mixes.

The policy mix framework

The analytical framework builds upon the work of Rogge and Reichardt (2013, 2016). It allows for a systematic analysis of the components of a policy mix; here defined as processes, elements, and their interrelation (Bahn-Walkowiak & Wilts, 2017; Rogge & Reichardt, 2016) (Figure 1).

Policy processes

Following Rogge and Reichardt (2016), we focus on two steps of policy processes: policy making and policy implementation. Although policy making and policy implementation are often closely entwined, here we define the policy making process as those activities that bring forth the highest level of laws, regulations, and management plans within a country. Any other activity is categorised as policy implementation. We focus on the distinction between various process styles regarding the distribution of power across governance levels (i.e. local, regional, and national) and actors and their respective accountabilities. We differentiate between a centralised and a decentralised process style. A centralised process style implies that power mostly

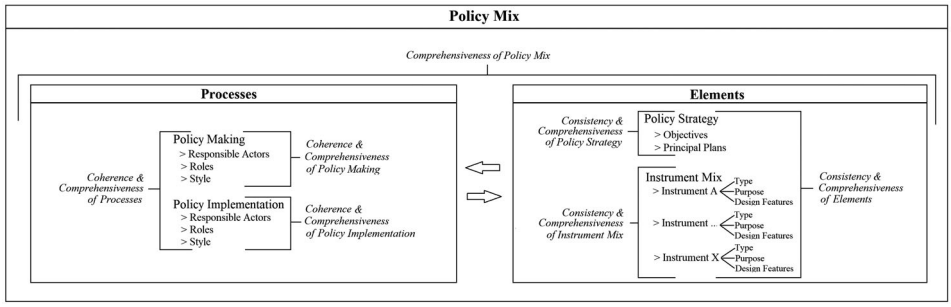

Figure 1. Policy mix framework, adapted from Rogge and Reichardt (2016).

remains at the national level whereas in a decentralised process, (some) power is transferred to lower levels of governance (Hansson-Forman et al., 2018).

The merits of a centralised policy process style reside in clarity regarding where the power and responsibility lie, plus it limits the need to coordinate policy objectives and principal plans as typically only a few actors will be involved in the process (Lange et al., 2013). However, centralised policy processes often lack legitimacy due to the limited influence of those who are affected by the process outcomes (Papadopoulos & Warin, 2007, p. 449). As a consequence, many multilateral environmental agreements, as well as Agenda 2030, specify the use of decentralised, participatory, policy processes since they may improve the effectiveness, legitimacy, and degree of democracy of the policy processes (Redpath et al., 2017).

Decentralised processes may take place either through deconcentration, where there is a limited transfer of power to local, upwardly accountable administrative units such as regional offices or through delegation, which refers to a process where part of the decision-making power is transferred to downwardly accountable regional or local representative bodies (Ribot, 2002). We treat these two decentralised policy process styles as a continuum, in which delegation yields the highest involvement of stakeholders. However, deconcentration and delegation are often mixed in practice, as administrative units and representative bodies may be both upwardly and downwardly accountable. Including local level actors in the policy processes is assumed to improve equity and justice due to their closer proximity to local realities, knowledge, and needs. Consequently, it can increase the acceptability, and thus success, of the resulting policy elements. However, at the same time, especially when accountabilities are unclear and the devolution of power is only temporary, decentralisation can lead to (additional) conflicts (Ribot, 2002; Sandström et al., 2018).

Elements

The elements of the policy mix include the overarching policy strategy and the associated instrument mix. The policy strategy comprises objectives and principal plans that stipulate the main proposed path towards reaching the objectives (Rogge & Reichardt, 2016). Here, we categorise the overarching plans into two main types: land-sharing and land-sparing (e.g. von Wehrden et al., 2014). Land-sparing refers to plans that focus on spatially separating large carnivores from human activity such as livestock farming (providing for both carnivores and livestock within one country by separating them), whereas land-sharing plans focus on spatial coexistence. The latter is often highlighted as an ideal situation, but it is under debate to what extent this is possible to achieve, partly due to disagreements over the meaning of the word 'coexistence' (Carter & Linnell, 2016). However, the focus of this article is not to dig deeper into the meaning of coexistence but to explore how this concept is applied within policy mixes. The policy strategy is critical as it can direct all decisions and actions within the policy processes (Rogge & Reichardt, 2016, p. 1623). The policy strategy may be determined at various levels of governance (Rogge & Reichardt, 2013, pp. 19–20), meaning that the local, regional, national, and international level might each have their own strategy within the same policy mix.

The instrument mix consists of a set of policy instruments, often clustered into three broad categories, regulative, economic (dis)incentive, or informative (Bemelmans-Videc et al., 1998), that work towards achieving the objectives defined in the policy strategy. Each of these instruments can be classified based on their primary type, where the regulative instruments are the backbone of the policy mix while the economic and informative instruments may complement laws to prevent, promote, or enable stakeholders to carry out actions that they would not otherwise do. They can also be classified based on their primary purpose (we distinguish between prevention of damage, fostering carnivore acceptance, or creating systemic change). In addition, instruments have a subset of descriptive and abstract design features such as their geographical scope and flexibility (Rogge & Reichardt, 2016, pp. 1623–1625).

Characteristics of the policy mix

Finally, a policy mix and its components can be analysed with respect to their characteristics: coherence, consistency, and comprehensiveness. Coherence relates to the processes of the policy mix. It addresses the

existence or absence of trade-offs with other policies (i.e. horizontal coherence between authorities at the same governance level) and across governance levels (i.e. vertical coherence). Coherence largely depends on sound two-way information exchange and collaboration between all involved authorities and levels (Lange et al., 2013). Consistency relates to the elements of the policy mix and their interplay. It illustrates whether the set policy objectives can be achieved simultaneously, whether the instruments reinforce one another, and whether the policy strategy and instruments work toward the same objectives. Comprehensiveness denotes the extent to which the policy mix addresses all relevant institutional and systemic failures (Rogge & Reichardt, 2016, pp. 1626–1627). An institutional failure can relate to a failure in laws and policies, organisational functioning, market dynamics, or norms (Young & Stokke, 2020). In environmental policy, it may arise when there is a misfit between institutions and the social-ecological context, e.g. when policy instruments are ineffective or missing, when the institutions adapt too slowly to changes in the social-ecological context, or when the temporal and/or spatial scale of the institutions is de-aligned with the temporal and/or spatial scale of the socio-ecological dynamics (Dressel, 2020). System failure originates from deeper underlying structural flaws. It arises, for example, when there are multiple institutional failures present within one policy mix and often requires not only smaller adjustments, but policy reforms to be fixed (Bergsten, 2013).

Cases

This article focuses on European countries since they have to fit their policy mix within the overarching European legislation and agreements including the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention) and in the case of European Union countries also the Habitats Directive (Kuijper et al., 2019, p. 107; Trouwborst, 2018, p. 307). At the same time, however, they differ regarding their experiences with the presence of large carnivores, because in some cases carnivores have only recently returned after being completely absent for decades (e.g. Germany, the Netherlands), while in others they have been present continuously (e.g. Finland, Spain) (Boitani & Linnell, 2015; Chapron et al., 2014). Differences also exist in polity traditions and climatic, ecological, and economic context. This combination of similarities and differences creates the opportunity for countries to learn from one another and to move beyond their own traditional policy repertoire (Schneider & Ingram, 1988). Our choice of study cases was motivated by (a) the availability of literature and (b) the opportunity to capture these differences in human-carnivore shared history. Hence, we focus on Norway, Sweden, Finland, Germany (specifically Bavaria and Saxony), the Netherlands, and Spain (specifically Castilla y León). The three Nordic countries are the only European countries that are currently home to all five large carnivore species (Chapron et al., 2014), Castilla y León is home to wolf, bear, and golden eagle, Bavaria is home to wolf, lynx, and golden eagle, and Saxony and the Netherlands are home to the wolf. However, we are not focussing on the large carnivore species per se, but treat them as a representative case for contentious environmental issues and focus on the overarching policy mix.

Materials and methods

The analysis is based on material collected from the scientific literature, policy documents, law texts, and expert knowledge. Our point of departure for the data collection was the overarching strategic intent of the policy mix. This allowed us to identify elements and processes that are directly corresponding to the large carnivore policy mix (Ossenbrink et al., 2019) and include all objectives as stated in the policy documents. First, we conducted a literature review based on material extracted by searching the Web of Science database (for the search strings, see online appendix A). An examination of the titles and abstracts of scientific papers allowed us to select relevant articles that focused on policy or management practices ($n = 60$). We did not set a timeframe for this search. When articles included outdated information, we excluded that information rather than excluding the article from the study. Articles that were not identified through this search, but that were known to the authors to be relevant were also included. The purpose of the literature review was to collect information on policy processes and the use of policy instruments and their functioning in practice. Second,

in cases where there were relatively few scientific papers to rely on, or if details were missing, we used original policy documents and law texts. They had a complementary function to the identified literature and helped to somewhat counterbalance the fact that some of the case countries and large carnivore species have to date received more attention in the scientific literature than others. We only included policy documents with large carnivores as their main focus, hence omitting material with a focus on livestock or reindeer husbandry, for example. For the analysis of the policy processes we focused on those actors involved in the policy making and implementation process, thereby leaving out those responsible for monitoring and data provision. However, this information is included in online appendix B as well as references to the law texts and policy documents that were included in the analysis. We conducted a qualitative content analysis of the material collected and used a deductive approach to sort the information in a spreadsheet (see online appendix B). Third, experts (i.e. researchers with multiple years of experience researching on large carnivores and their management) completed the dataset to the best of their knowledge, based on information from the grey literature and policy documents. Due to our comparative case study design many country-specific details that are included in online appendix B are not presented in this article. However, these details did inform our understanding and analysis of the policy mixes.

Results

Processes of the policy mix

Policy making

We found both centralised and decentralised policy making process styles across the case countries (see Table 1), and the authorities or task groups involved have their main roots in environmental- and agricultural policy (see online appendix C).

In Sweden, Norway, and Finland, the primary policy making process style can be defined as centralised, despite political ambitions to decentralise responsibility to the regional level. The national level is responsible for setting the overarching framing of the policy mix by determining national population goals and/or management plans and deciding on policy instruments. Within this framework, the regional level may decide upon management measures and create regional management plans if they do not jeopardise the national objectives. Hence the room for manoeuvre at the regional level is limited to implementation of the national policy and characterised by upward accountability (e.g. Hansson-Forman et al., 2018; Pohja-Mykrä & Kurki, 2014; Sandström et al., 2018). The local level plays (formally) at most a very limited role in the policy making process. This also adheres to the constitutionally protected Indigenous People, the Sámi, who use their traditional

	Sweden	Norway	Finland	Spain	The Netherlands	Germany	
				Castilla y León		Saxony	Bavaria
Policy making process style	↓	↓	↓	↕	↕	↕	
Policy implementation process style	↕	↕	↕	↑	↕	↑	
						↓	↕

↓ Centralised ↕ Deconcentrated
↑ Decentralised ↕ Hybrid

land, covering 30–50% of Finland, Norway, and Sweden, for semi-domesticated reindeer husbandry (in Finland also by Finns) (Hansson-Forman et al., 2018).

In Germany, the Netherlands, and Spain, the policy making process style can be classified as deconcentrated. The national level is only responsible for setting up international agreements and establishing national laws, whereas the regional level, i.e. the provinces in the Netherlands, the federal states in Germany, and the autonomous regions in Spain, is responsible for the creation of policies, including deciding on policy instruments (European Commission, 2020; Interprovinciaal Overleg, 2019; Köck & Kuchta, 2017). In the Netherlands, the provinces have decided to collaborate to create one joint management plan that applies to the country as a whole (Trouwborst, 2014). In all three countries, the local level is only involved in the policy making process through limited stakeholder involvement (e.g. advisory meetings) without decision-making power. In Saxony, the Netherlands, and Castilla y León, the responsibility for policy making is concentrated within one authority, whereas in Bavaria it is split between multiple authorities.

Policy implementation

Regarding the policy implementation process, all case countries except Saxony make use of some form of decentralisation (see Table 1). In Saxony, all responsibilities are combined and coordinated by the Saxon State Office for the Environment, Agriculture and Geology. Therefore, we classify the implementation process style as centralised. In the Netherlands and Bavaria, the implementation process is conducted primarily by upwardly accountable bodies and thus deconcentrated. Saxony and the Netherlands are distinct from the other case countries due to their limited involvement of stakeholder representatives and local level actors and Bavaria stands out because of the large number of governmental bodies and collaborative groups that are included in the implementation process (see online appendix C).

In Norway, Sweden, Finland, and Spain, the form of decentralisation is not obvious. At the regional level, all four countries have set up collaborative governance arrangements in the form of delegations, committees or working groups where political party representatives, stakeholder representatives, and, in the cases of Norway and Sweden, Sámi representatives come together (Risvoll et al., 2016). To what extent the arrangement has power and influence over policy and implementation vary considerably depending on process and implementation style in the respective country. Hansson-Forman et al. (2018) found that the Norwegian Large Carnivore Committees had substantially more influence over the implementation process compared to its Swedish and Finnish counterparts, while the influence over the overarching policy process are limited in all three countries. These arrangements are formally upwardly accountable. However, because they are comprised of (elected) regional or local representatives in Norway and Sweden, they can, in practice, be downwardly accountable as well. In addition, in the Swedish, Norwegian, and Finnish cases, the national level also plays a large role in the implementation process, resulting in a mixture of a centralised and decentralised implementation process styles (Hansson-Forman et al., 2018; Sandström et al., 2018). Finland stands out from the other case countries due to its high level of involvement of stakeholder representatives and local level actors (e.g. Hansson-Forman et al., 2018; Pellikka & Hiedanpää, 2017; Ratamäki, 2008) and Norway and Sweden deviate from the other case countries because they are the only countries in which responsibilities and power shift from the regional to the national level when set carnivore population targets are not being met (e.g. Risvoll et al., 2016; Sandström et al., 2018).

Overall, most actors involved in the implementation process have a background in agriculture, forestry, or environmental protection, with Sweden and Castilla y León being the exception by also including representatives from the e.g. tourism industry in their regional large carnivore stakeholder groups (Hansson-Forman et al., 2018).

Elements of the policy mix

The Bern convention and the EU Habitats Directive set the overarching framework for the elements of the policy mix. However, as will be shown below, there exists leeway for multiple interpretations and differences in the details of the policy mixes of our case countries can be identified.

Policy strategy

At the national level, all case countries have the same overarching objective for large carnivores: attaining or maintaining a viable population. However, they differ in how specific they are about their objective(s) and whether or not they connect them with objectives in other policies. Norway and Sweden state specific target numbers for each of the carnivore species either in the form of number of annual reproductions or in overall population numbers (Bostedt & Grahn, 2008; Gervasi et al., 2015). The other case countries' objectives remain more general (i.e. achieving or maintaining a favourable conservation status, protection of biodiversity, ecosystem stability, and ecosystem services). All case countries, except the Netherlands, have dual objectives as they connect their large carnivore objective with an objective for livestock by aiming to maintain traditional (pasture-based) livestock/reindeer husbandry with a minimum level of conflict even when large carnivores are present (see online appendix B).

Strict protection of carnivore species is included in each country's principal plan of how to achieve the objectives. However, depending on the population status of the respective species and under which Annex of the Habitats Directive, or, in the case of Norway, under which Appendix of the Bern Convention it is listed, in some regions (some) large carnivore species may be managed via hunting (Hansen et al., 2019; Pohja-Mykrä & Kurki, 2014; Trouwborst, 2014). Norway is the only country that uses land-sparing as its principal plan, aiming to separate large carnivores from livestock through specified management zones in which either a specific large carnivore species or livestock is prioritised. Some of the zones, however, overlap geographically (Hansen et al., 2019; Risvoll et al., 2016; Strand, 2018). Large carnivores are generally tolerated outside their prioritised zones, but it is supposed to be easier to obtain permission for culling in these areas compared to inside the carnivore zones (Risvoll et al., 2016; Risvoll & Kaarhus, 2020). While no fully-fledged land-sparing/zoning system formally exists in the other countries, in Sweden and Finland, there are what can be described as 'no go territories' for wolves in the reindeer husbandry area. In those areas wolves often cause so much damage that they are culled through protective hunting (Hansson-Forman et al., 2018; Sjölander-Lindqvist et al., 2020). Similarly, in Bavaria, it is easier to get permission to cull wolves in areas where livestock cannot be adequately safeguarded with protective measures (i.e. on most parts of the alpine pastures). In Castilla y León, core bear habitats can be declared areas of special protection in which several human activities are limited or banned. As both the Bern Convention and the Habitats Directive allow for some degree of zoning, especially when this relates to creating areas where large carnivores are prioritised (Trouwborst, 2018), the difference in approaches between the case countries does not seem to be a direct result of the difference in legislative frameworks for European Union and non-European Union countries.

Instrument mix

The overarching structure of the instrument mix defined by the international agreements that set forth regulations regarding the protection of large carnivores has been extended by the countries through additional policy instruments. All case countries largely use the same kind of instruments (see Table 2), but differences can be found regarding their application. Some of this differentiation is a result of the different annexes of the Habitats Directive or appendixes of the Bern Convention that apply to certain regions.

The main pillar of the instrument mix in all case countries is some form of ex-post compensation which is coupled in all case countries except the Netherlands to the prerequisite of having put in place reasonable preventive measures (see online appendix B). Other widely applied instruments include permanent or pilot subsidy programmes for preventive measures (e.g. Frank & Eklund, 2017), culling/lethal control of problem-causing animals (Pellikka & Hiedanpää, 2017; Sjölander-Lindqvist, 2015), and licence- or quota hunting (e.g. Cinque, 2015; Mykrä et al., 2017). The grounds on which culling/lethal control of problem-causing animals is allowed differ, with the Netherlands applying the strictest regulations and Bavaria and Saxony being the most lenient, and the only cases in our study that allow the continuous culling of members within a wolf pack until no more damage to livestock occurs in the case that damage cannot be attributed to a specific individual wolf. Hunting is in all case countries only allowed when the large carnivore population has reached favourable conservation status or if the purpose is a measure to prevent illegal hunting or increase the acceptance of the

Table 2. Comparative overview of the large carnivore instrument mixes applied in the case countries.

Instrument	Country/Region						
	SE	NOR	FI	CyL (ES)	NL	SX (DE)	BY (DE)
Ex post compensation payments	x	x	x	x	x	x	x
Conservation performance payment system	x		x				
Subsidy– *for preventive measures*	x	x	x	x	x	x	x
for livestock & agricultural productions insurance				x			
for additional expenses as a result of wolf presence						x	
Releasing livestock owners from their duty to pay for damage caused by escaped livestock when the escape was caused by wolf presence							x
Culling of problem-causing animals (*lethal control of a specified individual*)	x	x	x	x	x	x	x
Licence or quota hunting/harvesting to regulate the population (*Note that different countries use different definitions for licence hunting, quota hunting, & traditional hunting*)	x	x	x	x			
Maximum to acceptable depredation loss	x						
Promotion of ecotourism				x			
Adaptive hunting quota plans for ungulates					x	x	x
Habitat- & food availability improvement through reforestation				x			

presence of large carnivores. The differences in motivation for the application of the hunting instrument are rooted in cultural differences and hunting traditions (see online appendix B).

Less common instruments are conservation performance payments, i.e. payments based on conservation outcomes/the number of rejuvenations of the large carnivore species (only used in some parts of Sweden and Finland) (e.g. Suvantola, 2013; Zabel et al., 2014), the explicit promotion of ecotourism (in Northern Castilla y León), reforestation measures to increase food availability and habitat (in Castilla y León), adaptive hunting quotas for ungulates (in the Netherlands, Saxony, and Bavaria), and a set maximum of tolerable loss of reindeer to large carnivore predation (in Sweden).

Discussion

Lack of vertical and horizontal coherence of the policy processes

Our analysis shows that there is a lack of vertical coherence of the policy processes. All cases, except the Netherlands and Saxony, have in accordance with international commitments initiated decentralisation processes to involve stakeholders and local level actors in the policy implementation process. However, the inclusion of actors at the regional level with mainly upward accountability to implement national policy objectives, with little room for regional adaptation, has paved the way for power-struggles rather than effective collaboration between governance levels (Sandström et al., 2018). Generally, centralised policy processes are often clearer in terms of where power and responsibility lie and reduce the need for coordination of policy objectives and principal plans across levels (Lange et al., 2013) and would therefore have less difficulty in achieving and maintaining vertical coherence. Yet, when a decentralised process style is chosen, vertical coherence of the policy processes of a policy mix can be enhanced by addressing and clarifying power-related issues. We argue that this can be achieved by explicitly spelling out who is responsible for what and when, who has the right to make what kind of decisions in which situations, and who is accountable to whom.

Our analysis also shows that, due to a siloed policy design, there is a lack in horizontal coherence in the case countries and, consequently, potential synergies and complementarities between policy sectors remain largely unrealised. The selection of interests that are currently included is limited, focusing in particular on traditional stakeholders, leaving other potentially relevant interests on the side-line. Hence, large carnivore policy would benefit from improving horizontal coherence through collaboration and structured information exchange across policy boundaries to better identify trade-offs and conflicts between policy objectives which need to be reconciled. In Germany, where federal state authorities can develop management plans independent from one another without an overarching national framework, there is a risk of incoherence and inconsistency

between different federal states. Better collaboration and tuning between the regions would be a way to reduce this risk. The processes in the Netherlands, where the provinces work closely together, could be used as an example of how this could be achieved. Overarchingly, these suggestions for improvements might also be relevant for other environmental policy mixes that follow a similar policy design.

Consistency of the elements

Our analysis suggests that the elements of the policy mixes in all the case countries are characterised by weak consistency due to dual objectives that are difficult to achieve simultaneously without significant trade-offs, due to e.g. a contrast between European and local goals and a discrepancy between public and private interest, and primarily neutral interactions between the instruments.

All case countries, except the Netherlands, have dual objectives: attaining/maintaining a favourable conservation status of the large carnivore species while simultaneously maintaining (traditional) sustainable pasture-based livestock/reindeer husbandry. The Dutch case only has an objective for large carnivores, which might be a consequence of the only very recent return of the wolf to the Netherlands. However, this creates friction with the Habitats Directive and Bern Convention, which stress that measures to enforce the agreement should consider economic, cultural, and social requirements and characteristics (Trouwborst & Fleurke, 2019). The duality of the objective in the other case countries can therefore be seen as a consequence not only of the overarching ideal of coexistence, but also as a result of the European legislative framework and it is a common characteristic of environmental policies in general (Baker, 2012). In the case of Norway, in a comprehensive and independent audit of the government, looking at both objectives of the policy mix, the Norwegian Environment Agency and Ministry of Climate and Environment have recently been criticised for favouring the objective of viable carnivore populations, failing to consider the dual objectives of the large carnivore policy (Office of the Auditor General of Norway, 2019). Our analysis of the other case countries suggests that this tendency is also present there, primarily due to a lack of flexibility to adapt the policy mix to local social-economic contexts. Giving both aspects of the dual objective equal weight by being more sensitive to the local context requires a more adaptive form of management. However, this is difficult to achieve under the current international legal framework, which focuses on strict protection and not active management.

Our analysis further shows that the applied instruments do not conflict with one another but are also not explicitly set up to reinforce one another, except for the compensation payments that depended on the application of preventive measures. We therefore argue that the consistency of the instrument mixes in all case countries could be further improved by strengthening the connections between the various instruments, ideally addressing both objectives simultaneously. For example, in some countries it might be useful to couple subsidies for preventive measures to efforts to improve large carnivore habitat. Alternatively, it could be aimed at including more instruments that attempt to incorporate both aspects of the dual objectives. The conservation performance payment system in the reindeer herding areas of Sweden and Finland could be used as an example for other countries. This instrument is assumed to give incentives for local people to create conditions under which large carnivores reproduce while simultaneously compensating reindeer herders for economic losses (Zabel et al., 2014). However, due to the increasing numbers of large carnivores in the reindeer husbandry area in Sweden and the Sámi carrying most of the associated costs, the instrument is now about to be replaced with a system based on tolerance levels (Swedisch Environmental Protection Agency, 2013).

Comprehensiveness of the policy mixes

We found three aspects related to institutional failure that may undermine the comprehensiveness of the policy mixes. First, there is a mismatch in temporal and spatial scale between the policy instruments and the socio-ecological dynamics. The economic instruments can be helpful for the short term to relieve some of the economic pressure that large carnivores cause. However, they do not guarantee the long-term continuation of (traditional) pasture-based livestock/reindeer husbandry practices under continued large carnivore

presence as they are ineffective in preventing, for example in the case of reindeer husbandry, herd collapse (Åhman et al., 2014) and do not address the long-term structural change in the socio-ecological system where herders/farmers report e.g. anger, frustration, and anxiety caused by factors including distrust in authorities, lifestyle changes, concern for and emotional attachment to livestock, and the feeling to produce food for carnivores rather than for human consumption (Zahl-Thanem et al., 2020).

The Norwegian zoning system, which attempts to spatially separate large carnivores from livestock over the long-term tends to neglect the socio-ecological dynamics, with the effect that the majority of sheep killings are concentrated in the boundary areas right outside the carnivore zones (Hansen et al., 2019). For reindeer husbandry, the zoning system is not appropriate due to the extensive land use for this purpose in combination with specific geographical characteristics leaving reindeer husbandry in a vulnerable situation (Risvoll & Kaarhus, 2020). Hence, land-sparing under these conditions can hardly be a panacea (Hansen et al., 2019; Risvoll & Kaarhus, 2020; Strand et al., 2019).

Subsidies for preventive measures applied to sheep farming or other domesticated livestock (reindeer husbandry not included) might be more effective in the long-term due to the potential to help to prevent damage altogether. However, in the case countries studied here, most of the subsidy instruments are limited in terms of available resources or implemented on a project basis and the take-up for implementing preventive measures is often poor (Zahl-Thanem et al., 2020). Furthermore, a recent study by Eklund (2019) shows limited scientific evidence of the effectiveness of most measures. Therefore, more research to identify (new) effective measures is urgently needed.

Secondly, there is a mismatch between the applied instruments and the underlying dynamics of the human-carnivore conflict. The focus on economic instruments disregards the fact that human-carnivore conflicts often transcend material and income issues. Cultural factors and the potentially serious social and psychological impacts of carnivore presence to local communities and individual farmers (Hansen et al., 2019; Risvoll & Kaarhus, 2020; Salvatori et al., 2020; Zahl-Thanem et al., 2020) are not considered within the current instrument mixes.

Thirdly, we found a lack of clear objectives that can be objectively evaluated. With the exception of Norway and Sweden, that have attempted to quantify their objectives by specifying 'favourable conservation status' in terms of large carnivore reproduction and population numbers respectively, and, in the case of Sweden, also the maximum tolerated loss of reindeer to large carnivore predation, all the other case countries lack quantifiable objectives. Although we acknowledge that objectives in the form of specific numbers can be contested, lack of clear objectives can result in disagreement over the status of goal fulfilment and hinders debates on the necessity of (potential) policy instruments.

Similar institutional failures as in our case countries have been found in other environmental policy contexts (e.g. Dressel, 2020; Mancheva, 2020; Ward et al., 2018), which is why several lessons can be learned from our study on how to improve the comprehensiveness of environmental policy mixes in general. These include the need to (a) ensure that the temporal and spatial scale of the instruments matches the temporal and spatial scale of the socio-environmental issue that the policy mix attempts to address, (b) address all underlying dynamics of the human-environmental conflict through the inclusion of diverse types of instruments, and (c) formulate objectives in terms that can be objectively evaluated.

Conclusion

This study set out to compare large carnivore policy mixes across European countries with the aim of contributing to cross-national learning and to provide guidance for the design of policy mixes with multiple objectives. Our analysis shows that the large carnivore policy mixes in the case countries are a patchwork of processes, principal plans, and policy instruments that lack a clear uniform underlying guiding strategy that accounts for the multiple temporal and spatial scales and components of the socio-ecological system that the policy mixes aim to address.

Based on the identified multiple institutional failures, we argue that the studied countries display signs of systemic failure and would therefore benefit from structural reforms that review the policy mixes in their

entirety. These reforms could preferably build on an ecosystem-based approach as displayed in the Convention on Biological Diversity, whereby humans and their cultural diversity should be recognised as an integral component of the ecosystem (UN, 1992).

Overall, the analytical framework that we applied enabled us not only to create an overarching overview of the large carnivore policy mixes, but also to draw lessons that can be useful for the design of environmental policy mixes more broadly. Ultimately, creating coherent, consistent, and comprehensive environmental policy mixes that build on multiple objectives requires stepping away from sectorized policy development. What is needed instead is a holistic, system-based approach with strong collaborative structures across policy boundaries and regions, the inclusion of diverse stakeholders, and constant care and attention to address all objectives simultaneously rather than in isolation. This could be done through policy integration, mainstreaming biodiversity conservation into all other policy sectors, focussing on processes within socio-ecological systems as a whole, and designing evidence based policy instruments. Furthermore, it should be recognised that policy instruments are often targeted at heterogenous groups; one single policy instrument is unlikely to work equally for all (Pedersen et al., 2020). The conceptual framework underlying the Intergovernmental Platform on Biodiversity and Ecosystem Services could be used as an inspiration for these reforms, offering an approach to environmental management and policy that acknowledges multiple worldviews, values, and knowledge systems (Díaz et al., 2015). This would create room for a more integrated and adaptive approach that allows for the testing of various policy strategies and instruments and that includes more options to manage conflicts.

Acknowledgements

We would like to thank the three anonymous reviewers for their insightful and valuable comments and suggestions.

Disclosure Statement

No potential conflict of interest was reported by the author(s).

Funding

The research funds for agriculture and food industry (FFL/JA), co-funding from the regional counties of Hedmark, Trøndelag and Nordland. Research Council of Norway grant no 267982.

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