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Modelling impacts of infrastructure and climatic factors on reindeer forage availability in winter

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ABSTRACT

The cumulative impacts of climate change and human activities on species are often studied in isolation, limiting understanding of their combined effects. The present research addresses this limitation by proposing a novel conceptual model to assess the cumulative impacts of various anthropogenic developments and environmental conditions on ungulates. The conceptual model is applied to semi-domesticated reindeer, specifically in the context of winter grazing in northern Fennoscandia, as this species is facing an increasing range and intensity of stressors detrimental to its health and survival.

The conceptual framework for the model is described, measuring forage loss due to physical, behavioural and climatic factors. Using data from previous studies, this framework is applied to assess loss of reindeer forage in winter pastures due to construction of roads, mines, hydropower stations and population centres. Results of this case study show that excluding behavioural impacts would lead to an 86 % underestimation of forage loss, while ignoring access limitations caused by snow conditions would result in an 11 % underestimation. Additionally, synergistic effects from multiple infrastructures impact 22 % of the area.

Although the model does not yet account for factors like habitat connectivity or inter-annual weather variability, it provides a multi-faceted framework for evaluating cumulative impacts, offering a more holistic approach than existing models. Its adaptability also allows for application to other regions, species, or land-use scenarios. These findings emphasise the necessity of considering cumulative impacts within environmental impact assessments used to inform sustainable land-use and conservation strategies.

1. Introduction

Human activities are increasingly having profound impacts upon individual species, and the functioning of entire ecosystems (e.g. Dirzo et al., 2014). Studying these impacts can be challenging, as they are often comprised of multiple, interacting factors (Piggott et al., 2015), an inherent complexity that sometimes conflicts with tendencies to prefer simplicity within scientific research (Montgomery et al., 2019). However, when ecological theory is translated into the practical application of land management or public policy, simplification can pose risks, as generalised trends may not reflect local specificities, potentially leading to inappropriate or ineffective management decisions (see e.g. Yates et al., 2018; Drees et al., 2021).

To support more effective land management approaches, research tools that consider some of the more complex relationships in ecological systems are desirable. This includes tools that allow us to understand the synergistic impacts of multiple stressors upon a system; identified as a key priority for ecologists by academics and organisations such as the Organisation for Economic Co-operation and Development (Larsen et al., 2017; Hodgson and Halpern, 2018; OECD, 2021). Responding to this need, there is a growing field of research upon multiple stressors, aiming to improve predictive power through increasing ecological complexity, as well as temporal scale and degree of realism, within studies (Orr et al., 2020). Models are one of the basic tools of this predictive work. The aim of this paper is to contribute to the discourse of multiple stressor assessment, by proposing a new and more complex conceptual model for assessing the cumulative impacts of human activities upon a species. To provide some grounding to the theory, the modelling approach reported here is developed in relation to a case study of reindeer ecology. Whilst going forward we will focus upon the details, justification and discussion of the model in relation to this case study, we suggest that the principles and process described are suitable

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for application to other species in other settings, if appropriate input data is available.

1.1. Modelling reindeer ecology

Semi-domesticated reindeer (*Rangifer tarandus* L.), are classed as vulnerable on the IUCN red list, due to a declining global population (Gunn, 2016), although this trend, and the factors contributing to it, vary by region. A regularly cited factor is the impact of poor climatic conditions, particularly the development of ice layers within the lowest level of the snow column in winter. These ice layers, often formed during warm spells when snow melts or precipitation falls as rain and subsequently freezes, can completely prevent reindeer digging to reach ground-lying forage, leading to their starvation (Hansen et al., 2011; Axelsson-Linkowski et al., 2020). Yet, whilst climate can significantly affect reindeer access to forage, especially during the critical winter season, other factors also play a notable role. One such is presence of forage. Lichen comprises a significant portion of a reindeer's diet during winter, yet over the last 60 years lichen-rich forests in Sweden have declined significantly (c. 71 % Sandström et al., 2016). This low winter forage abundance, alongside predation, restrictive land-use policy and competing human activities, have all created significant stresses for reindeer, and reindeer herders (Magga et al., 2011; Pape and Löffler, 2012; Pogodaev and Oskal, 2015). Many have long felt that cumulative stresses from these multiple, incremental changes in the landscape are making herding untenable, and that its loss would significantly harm local livelihoods, cultures, and environmental processes connected to the reindeer (Turi, 1910; Furberg et al., 2011; Forbes et al., 2020). Additionally, competition for access to land has reached such an extent that Sámi reindeer herders and other land users are now regularly involved in legal conflicts with one another, adding to financial and emotional pressures on both sides (Borchert, 2001; Johnsen et al., 2017; Persson et al., 2017).

To better inform management strategies, policies and environmental assessments such as those influencing discourses around reindeer herding, it can be valuable to have some kind of quantification of the system. Predictive models have long been used to assess the cumulative impacts of changing land use, habitat quality and disturbance on species and ecosystems (Hodgson and Halpern, 2018). Those specifically relating to reindeer include that of Uboni et al. (2019), investigating the relationship between lichen-rich pastures and broad landscape characteristics such as slope. Impacts of a changing winter climate have been investigated in the models of Kohler and Aanes (2004) and Hansen et al. (2019), who note that reindeer starvation from icing can have long-term stabilizing effects on cyclical explosions and crashes in population size, at least in wild herds in Svalbard. Other models have attempted to elucidate the impact of reindeer density on abundance of lichen, contributing to discussions on such factors as overgrazing (Kumpula et al., 2000; Moen and Danell, 2003; van der Wal, 2006; Pekkarinen, 2018) and the subsequent impact of forage wastage from trampling (Pekkarinen et al., 2017).

The impacts of human activities and infrastructures upon reindeer have also been studied. These include consideration of wind farms (Skarin and Alam, 2017), forestry (Horstkotte et al., 2011; Kater and Baxter, 2022), roads and railways (Lundqvist, 2007; Panzacchi et al., 2013), mines (Anttonen et al., 2011; Eftestøl et al., 2019), hydropower dams (Mahoney and Schaefer, 2002; Nellemann et al., 2003), settlements (Nellemann et al., 2000; Nellemann et al., 2001), and tourism (Helle et al., 2012; Niebuhr et al., 2023). Many of these studies have identified zones of influence (ZOIs), from hundreds of meters to several kilometres in extent, where reindeer exhibit avoidance behaviour, creating a functional loss of grazing despite the forage being physically present. Large human-built structures, especially linear ones such as road and rail, have also been seen to reduce the 'reachability' of forage due to habitat fragmentation; for example, where a lichen-rich area is surrounded by linear structures the reindeer are less willing to cross (Lundqvist, 2007). Stoessel et al. (2022), have mapped the extent to which these forms of disturbance, together with pressure from predation, overlap with one another to create cumulative impacts. They found that, in northern Fennoscandia, 85 % of the region is affected by at least one form of anthropogenic disturbance, whilst 60 % is affected by multiple concurrent ones. Niebuhr et al. (2023) have also highlighted the cumulative impacts of multiple of the same kind of infrastructure, in their case study tourist cabins, depending on their level of clustering, suggesting that multiple small and scattered features may have a greater impact than one large tourist resort. Finally, Fohringer et al. (2021) undertook a spatial analysis in the Laevas reindeer herding community's grazing grounds in Northern Sweden, providing data on disturbance from multiple ZOIs. The authors categorised the pastures as "functionally unavailable" and "undisturbed", finding that 34 % of grazing grounds are functionally unavailable due to multiple forms of disruptive land use.

The models considered so far either outline trends and processes relating to reindeer ecology, or assess the impacts of a limited range of infrastructures. Of the models that have taken steps to embrace more complexity by assessing the cumulative effects of human activity (e.g., Vors et al., 2007; Sorensen et al., 2008; Polfus et al., 2011), these have mainly focused upon caribou in Canada, whose ecosystem characteristics and usage differ to herded reindeer in Europe, and the research has focused solely upon human disturbance. In the case of Fohringer et al. (2021), the results whilst valuable only provide a categorical assessment of disturbance. There remains a need to more reliably assess the cumulative impacts of current or proposed infrastructure in Fennoscandia (OECD, 2021).

1.2. Study aims and objectives

The aim of this study was to improve assessments of cumulative impacts by proposing a method of conceptual modelling, and by displaying through the model results how inclusion of multiple, sometimes synergistic factors affect the outputs. This conceptual model is then applied to a case study that considers the effects of multiple forms of anthropogenic development, in conjunction with snow conditions, on reindeer grazing in winter pastures in northern Fennoscandia. The types of infrastructure considered include the major ones of roads, mines, hydropower stations, settlements and forestry. Whilst the parameters and input data reported are relevant to our case study, we hope that the conceptual model can be generalisable and applied to other locations and potentially other species, according to the expertise of practitioners using it.

2. Methods

2.1. Outline of methodology

The conceptual model proposed here involves the layering of multiple, sometimes synergistic factors onto a basemap to calculate functional loss of forage to potential grazing due to these factors. As each layer can be added successively, considering forage presence and its accessibility due to physical and behavioural factors, the output allows for an assessment of these factors cumulatively. The model is constructed as follows:

1. Determine forage availability in the absence of the infrastructure being considered through combining:
   a. the abundance of forage in the absence of the infrastructure.
   b. the accessibility of this forage in the absence of infrastructure. This may include considerations of the barrier effects of snow or geomorphology, for example.
2. Add the infrastructure to the landscape:
   a. Determine the forage lost to the physical structures.

b. Determine the forage lost to avoidance behaviours exhibited by the species within a zone of influence around the infrastructure.
c. Consider any synergistic effects of multiple forms of infrastructure being present.
d. Combine these to calculate the total forage lost to potential grazing due to the infrastructure.

A conceptual model for this process is shown in Fig. 1.

2.2. Methodology applied to case study

This model is applied to a hypothetical case study of forested winter grazing grounds of reindeer, based on data relevant to Fennoscandia, particularly Norbotten in Sweden. As silviculture is widespread across the region, an 'undisturbed' environment in the absence of infrastructure is still assumed to have some variation in forest age.

Forage abundance in the absence of infrastructure is determined using original data on forage biomass per km^2 in reindeer winter grazing grounds far from notable infrastructure, taking into consideration the variation of forage abundance in forest patches of different ages. The major factor affecting access to forage during winter is the barrier effects of snow, particularly the formation of ice layers, which prevent reindeer from digging to ground-lying forage. Therefore, forage accessibility in the absence of infrastructure is taken to be the percentage of ground area that can be reached by reindeer through digging, again including consideration of how this varies across forest ages. Forage abundance is multiplied by accessibility to give overall availability as kg km^{-2}.

The model was created using QGIS version 3.34.6. A basemap of an area of Norbotten proximal to winter grazing grounds was used. Reindeer grazing grounds vary in size according to the area allocated to each reindeer herding village (siida). For example, Sirges siida has an area of 13,485 km^2 with approximately 25 % of this being winter grazing, Tuorpon siida has an area of 13,180 km^2 with approximately 50 % of this being winter grazing, and Jåhkågaska tjiellde siida has an area of 9922 km^2 with approximately 33 % of this being winter grazing (Sámediggi., 2025), all three being based in Norbotten. Grazing grounds of other siidas can be smaller, there is overlap between some of these grazing grounds, and not all grazing grounds can constantly be accessed. The area considered within the hypothetical map here is 1000km^2 (31.6 km × 31.6 km) in extent.

Infrastructure included in the map were roads, a population centre, a mine, and a hydropower station and associated reservoir. They were assumed to be circular for ease of construction, aside from roads which were linear. The size of these structures in the hypothetical landscape here were determined according to data representative of infrastructure already present in Fennoscandia, reported in the results. The area taken up by the physical structure was multiplied by the forage availability per km^2 as calculated earlier, to determine total forage loss from its presence.

ZOIs, where reindeer exhibit avoidance behaviour, were calculated from academic literature, noting the distance of impact from each form of infrastructure and the percentage of reindeer avoidance within that zone. ZOIs were created as buffers around the infrastructure in the QGIS map, using the clip tool to remove the area physically covered by the feature, and to remove water features from the basemap where forage would not be present. Overlapping buffers were created as a separate feature to buffers including the ZOI of only one feature. These calculated areas were multiplied by forage availability per km^2, and were then further multiplied by percentage avoidance gave total forage loss from avoidance. The synergistic effects of multiple overlapping ZOIs on percentage avoidance by reindeer was considered.

Forage loss due to accessibility, physical structures and avoidance in zones of influence were added together to give overall forage loss. This figure could be compared to an idealised undisturbed environment, or by comparing figures to a landscape with different forms and extents of infrastructural development. Actual input data used in the case study model is outlined in the results.

3. Results

3.1. Forage availability

Lichen is used as a proxy for reindeer forage, making up a mean proportion of 54 ± 1.2 % of their diet (using data from Boertje, 1990; Kojola et al., 1995; Heggberget et al., 2002). Whilst reindeer do eat other forage, lichen is regularly used in research as a marker for their habitat quality (see e.g. Dahle et al., 2007; Kumpula et al., 2000; Sandström et al., 2016; Uboni et al., 2019), and data from Kater and Baxter (2022) show that the proportion of lichen and other edible species within winter grazing grounds remain approximately proportional to one another.

Abundance and accessibility of lichen in the absence of infrastructure was determined using data from Kater and Baxter (2022). That particular study quantified lichen biomass per km^2 in forests of four different age classes in Norbotten, Sweden, these being clear-cut (zero years of age), young stands (7 ± 1 years), intermediate stands (23 ± 4 years), and old stands (86 ± 13 years). These age classes correspond to those in larger inventories of surrounding forests, with classes of 0–2 years, 3–10 years, 11–60 years, and 61+ years (see Table 3.4 in Skogstyrelsen, 2014). Using these inventories, we calculated the proportion and ratio of each forest age class in a representative landscape, as shown in Table 1.

Assuming a constant distribution of forest age classes, we calculated the dry mass of available lichen within 1 km^2 of a representative model landscape. This availability is calculated by multiplying the biomass of lichen present per km^2 with the percentage accessibility of this lichen to reindeer grazing during winter due to snow conditions (Table 1, converted from kg ha^{-1} to kg km^{-2}; Kater and Baxter, 2022). The original data on accessibility of lichen through snow used data collected in November, January and March to account for changes throughout the season. Here we have averaged across this period to produce the figures of percentage lichen accessibility in Table 1.

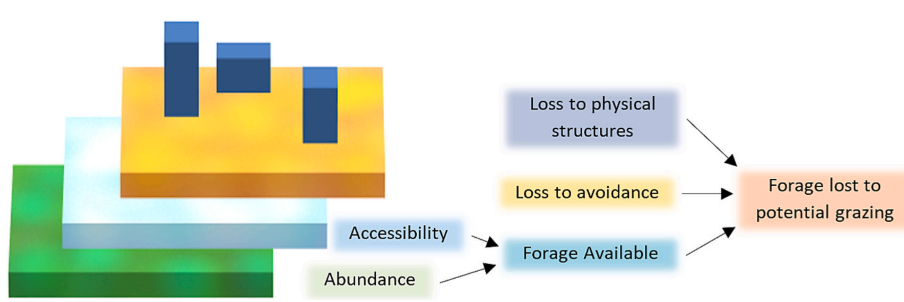

Fig. 1. Conceptual model showing the construction of the ecological model. ZOI is the zone of influence within which reindeer show avoidance behaviour.

Table 1
Biomass of lichen available per km^2 of a forest in which the age distribution of trees is representative for Norbotten county, Sweden. This is calculated by multiplying biomass of lichen present per km^2 with the percentage accessibility of this lichen due to snow conditions, producing lichen availability in each forest age class during winter. Data are taken from Kater and Baxter (2022). This availability is further multiplied by the proportion of each age class within a forest representative of Norbotten, to produce the representative lichen availability.

Age Class	Lichen present ($kg\ km^{-2}$)	Lichen accessible (%)	Lichen available ($kg\ km^{-2}$)	Proportion of age class in total forest (%)	Lichen availability in representative forest (kg)
Clearcut	41,610	86.1	35,826	2	717
Young	69,440	80.6	55,969	6	3358
Intermediate	87,040	94.4	82,166	44	36,153
Old	82,820	95.8	79,342	48	38,084
Total lichen (kg/km^2)					**78,312**

3.2. Loss of forage to physical infrastructure

The area covered by a physical piece of infrastructure is taken to lead to 100 % forage loss in that area. Data for average infrastructure sizes in the region according to published research are outlined in Table 2. The mean surface ground cover of mines was 13.1 ± 6.9 km^2. Due to difficulty in ascertaining hydropower dam and associated reservoir extents for northern Sweden, an estimate from available data for Norwegian storage hydropower plants was derived. Dorber et al. (2018) calculated that Norwegian storage hydropower plants have an average net land occupation of 0.027 $km^2\ yr^{-1}\ GWh^{-1}$, and Finnish Barents Group Oy (1998) provide data on the kWh energy production of hydropower stations along the Luleälven river in Norbotten, Sweden. Mean estimated land loss was 27.6 ± 6 km^2 to the nearest 0.1 km^2.

The population density of Norbotton County is 2.6 persons km^{-2} (Statista, 2022). In a theoretical landscape of 1000 km^2, this would total a population of 2600. The most analogous urban centre in the region is Jokkmokk with a population of 2786 censused in 2010, so assuming a similar urban density of 776 people km^{-2} (Statistics Sweden, 2011), the population in a model of 1000 km^2 would occupy 3.6 km^2 to the nearest 0.1 km^2. Roads are also included as infrastructure in this model but given the relatively small size of most roads within the region, lichen loss due to their physical structure is not included.

3.3. Avoidance and ZOIs

The literature has reported areas surrounding infrastructure where reindeer show reduced presence and grazing due to avoidance. The ZOIs corresponding to each piece of infrastructure are summarised in Table 3, including roads, population centres, mines, hydropower stations and multiple overlapping ZOIs. Available data on roads show a mean ZOI of 1 ± 0.24 km, with reindeer avoidance of 43 ± 3 % within a 1 km ZOI. Literature on population centres reveal a mean ZOI of 6.4 ± 1.2 km, with a mean reduction in reindeer presence of 64 ± 11 % (using avoidance in a 5 and 8 km ZOI). Mines show a mean ZOI of 1.8 ± 0.8 km and 35 ± 0 % avoidance using avoidance data in a 1.5 km ZOI. The mean ZOI around hydropower stations, but not the accompanying reservoir, is 3.5 ± 0.5 km^2 with an avoidance of 60 ± 2 %.

Panzacchi et al. (2013) identify a cumulative effect of roads and population centres upon reindeer avoidance. Data from Table 1 and the following equation, both from Panzacchi et al. (2013), were used to estimate this potential cumulative impact within a 1 km buffer zone:

$$\textit{Decrease in use} = -\left(1 - e^{((\beta_D \times n)+(\beta_E \times n))}\right) \times 100 \tag{1}$$

Where, for a 1 km buffer:

β_D (tourist cabin, direct effects) = −17.66.

β_E (road, direct effects) = −0.61.

n = 1 for tourist cabins and 1 km roads.

The approach of Panzacchi and colleagues showed that where roads and population centres have overlapping ZOIs, no reindeer are present. We apply the same theory here within our model for any overlapping ZOIs.

3.4. Model outputs

A visualization of the model can be seen in Fig. 2. A total of 14,675 metric tons of lichen were lost to potential grazing due to the infrastructure added to the model landscape. Of this, 2091 tons were lost due to the physical structures (mine, population centre, novel flooding from the hydropower reservoir), with the remaining 12,583 tons due to avoidance behaviour. Consideration of only the physical impacts of infrastructure on forage, omitting the behaviour impacts, would therefore have caused impacts to be underestimated by 86 %. Areas of

Table 2
List of all mines active in Sweden in 2019 according to Mining Inspectorate of Sweden (2021), showing their county, the mineral mined, and the approximate ground surface area of the mine as measured from satellite imagery on Google Maps. Also shown the kWh energy production of hydropower stations along the Luleälven in Sweden from Finnish Barents Group Oy (1998), showing the net land occupation calculated using equations from Dorber et al. (2018).

Mine	Approximate ground surface area of mine (km^2)	Hydropower Station	Hydropower Energy production (GWh)		Net land occupation of dam and reservoir (km^2)
Aitik	62.3	Ritsem	460		12.4
Kiirunavaara	45.0	Vietas	116		31.3
Malmberget	6.7	Seitevare	850		23.0
Leveäniemi	No recent image	Porjus	1290		34.8
Kaunisvaara	6.1	Ligga	790		21.3
Kristineberg	3.0	Akkats	585		15.8
Kankbergsgruvan	0.1	Harsprånget	3481		94.0
Renström	0.1	Messaure	1901		51.3
Björkdalsgruvan	5.6	Letsi	1770		47.8
Garpenberg	2.2	Porsi	1150		31.1
Lovisagruvan	Difficulty locating	Laxede	835		22.5
Zinkgruvan	0.1	Randi	235		6.3
		Parki	100		2.7
		Vittjärv	230		6.2
		Boden	490		13.2
Mean	**13.1 km^2**			**Mean**	**27.6 km^2**

Table 3
Derivations for the Zones of Influence around infrastructure, where reindeer display avoidance behaviour. Shown also are the percentage avoidance of reindeer within each study when those data are present.

Factor	ZOI (km)	Avoidance (%)	Source	Study location	ZOI used (km)	Avoidance used (%)
Road	0.25	-	Dyer et al. 2001	Alberta, Canada	1	43
	1.0	Mean 40	Lundqvist 2007	Jämtlands, Sweden		
	0.25	54.2	Sorensen et al. 2008	Alberta, Canada		
	1.5	-	Anttonen et al. 2011	North Finland		
	2.0	-	Polfus et al. 2011	B.C., Canada		
	1.0	-	Polfus et al. 2011	B.C., Canada		
	1.0	46	Panzacchi et al. 2013	South Norway		
Population Centre	2.5	-	Anttonen et al. 2011	N Finland	6.4	64
	8	52 and 55 %	Helle and Särkelä 1993	N Finland		
	4	-	Helle et al. 2012	N Finland		
	10	Mean 60 %	Nellemann et al. 2000	S Norway		
	5	Mean 85 %	Nellemann et al. 2001	S Norway		
	9	-	Polfus et al. 2011	B.C. Ca		
Mine	1.5	35 %	Eftestol et al. 2019	North Norway	1.8	35
	4.0	33.3 %	Weir et al. 2007	Newfoundland, Canada		
	1.5	-	Anttonen et al. 2011	North Finland		
	0.25	-	Polfus et al. 2011	B.C., Canada		
Hydropower station	3	58 %	Mahoney and Schaefer 2002	Newfoundland, Canada	3.5	60
	4	62 %	Nellemann et al. 2003	South Norway		
Multiple ZOIs	-	Derived from equation as 100 %	Panzacchi et al. 2013	South Norway	-	100

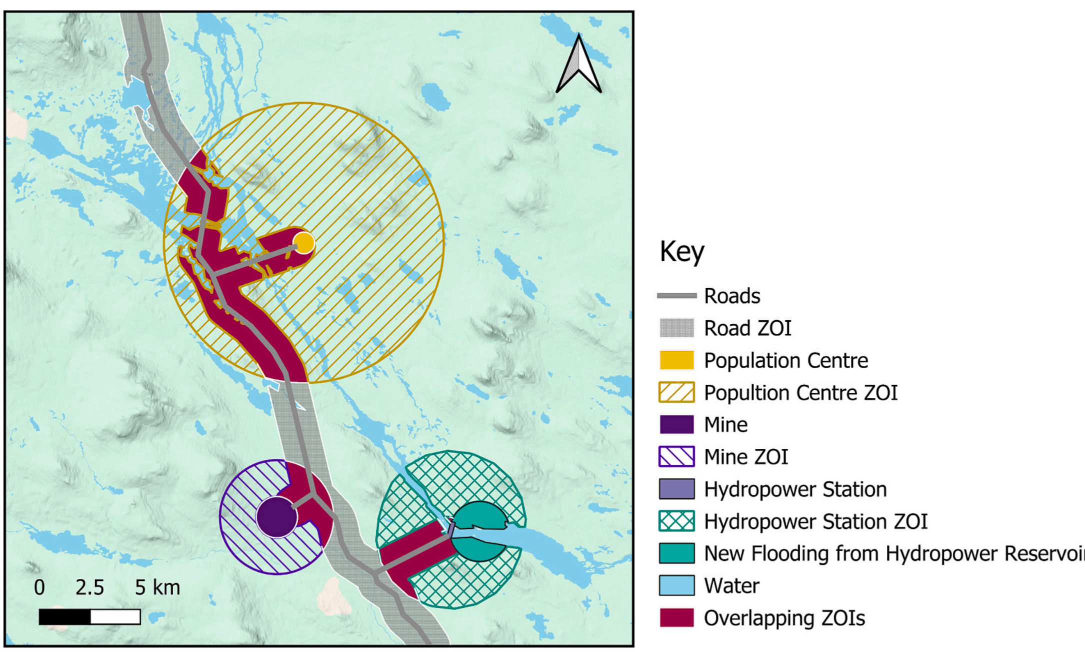

Fig. 2. Visualization of a model, showing the area impacted by physical infrastructure, and the surrounding area affected by their associated zones of influence (ZOIs) within which reindeer exhibit avoidance behaviour.

overlapping ZOIs, within which avoidance behaviour is greater due to synergistic effects, made up 26 % of the total ZOI areas, and 22 % of the total impact combining avoidance and physical structures (see Fig. 3). The biomass of lichen lost due to the various features added to the model are outlined in Table 4.

4. Discussion

The results of this study do two things. First, they show how the conceptual model we put forward can be applied in practice, using the reindeer grazing case-study. Second, they highlight the importance of considering a range of factors, including physical, behavioural and synergistic, in models assessing environmental impacts of development. If behavioural data would have been absent from this model, for example, this would have led to an underestimation of lichen loss of 86 %, a significant omission. Further, the levels of reindeer avoidance within each ZOI varied between 35 and 64 % according to the type of infrastructure, yet 26 % of this area was composed of multiple overlapping ZOIs, the synergistic effect of which is 100 % avoidance by reindeer. Omission of this factor would also lead to a significant underestimation within the model. It should be noted that in this model the instances of infrastructure are placed relatively distant to one another, but if more proximal in other configurations the proportion of overlapping ZOIs, and therefore the cumulative impacts of the structures, would be higher. Finally, the impacts of snow as a barrier to grazing was also included as an influencing factor. On average 89 % of pasture areas were accessible to grazing, with this accessibility varying with forest age. Omission of this factor would lead to biomass of forage loss being underestimated by approximately 11 %.

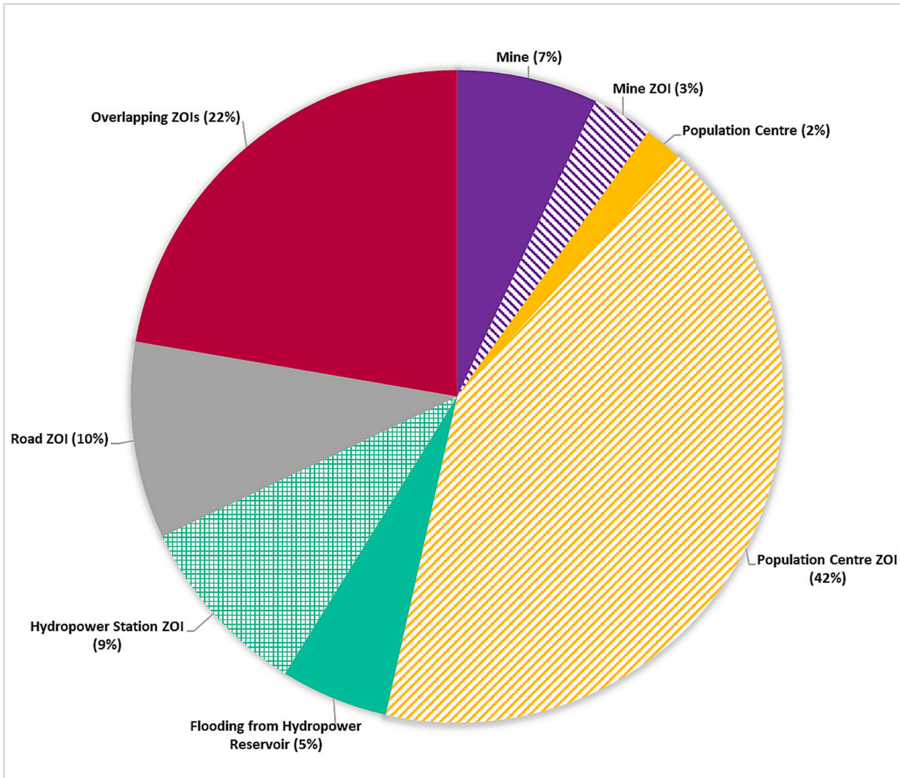

Fig. 3. Proportional impact of each factor included within the model on overall loss of lichen to reindeer grazing. ZOI indicates zone of influence within which reindeer exhibit avoidance behaviour, whilst named structures refer to the total loss of lichen under the respective physical structure.

Table 4
For each layer of the model, below is displayed the area affected by the respective feature and the percentage of forage loss within this area (either due to avoidance behaviour of physical destruction of forage due to infrastructure). The area, multiplied by percentage forage loss, is then multiplied by lichen availability as kg/km^2, giving a figure of the total lichen lost to grazing due to the feature (displayed in metric tons).

Layer	Area affected (km^2)	Forage loss in affected area (%)	Lichen kg km^{-2}	Lichen Loss (metric tons)
Physical Mine	13.1	100	**78,312**	1026
Mine ZOI	15.9	35	**78,312**	436
Physical Population Centre	3.6	100	**78,312**	282
Population Centre ZOI	121.9	64	**78,312**	6110
New Flooding from Reservoir	10.0	100	**78,312**	783
Hydro station ZOI	28.4	60	**78,312**	1334
Road ZOI	42.5	43	**78,312**	1431
Overlapping ZOIs	41.8	100	**78,312**	3273
			Total	14,675

4.1. Data considered within the model

The factors considered within this model included snow conditions, and the physical and behavioural impacts of mines, population centres, roads and hydropower stations. Windfarms, which have received significant attention in relation to their impacts on reindeer recently (e.g. Cambou et al., 2021), were not included. This is because currently available data on reindeer avoidance were either collected during the calving season, when female reindeer are more avoidant compared to other times of year (Skarin and Alam, 2017; Skarin et al., 2018); in very small, enclosed, areas (Flydal et al., 2004); in summer pastures (Eftestøl et al., 2023); or in very specific parts of the landscape such as peninsulas where reindeer movement is highly influenced by topography (Colman et al., 2012; Colman et al., 2013). However, with greater availability of representative data in future, this would be a valuable factor to add. Research by Dau and Cameron (1986) and Vistnes and Nellemann (2001) on roads and population centres were also excluded due to being undertaken in the calving season, whilst research by Boulanger et al. (2012) on avoidance of mines was omitted due to being collected in summer grazing pastures.

Other data were included in the model that may otherwise have been omitted, such as results from studies in Norway, Finland, and Canada alongside studies undertaken in Sweden. Whilst reindeer herding practices are relatively similar in Norway and Sweden, reindeer are far more sedentary in Finland, which may alter their behaviour. In Canada, caribou, the close relative of reindeer, live in an environment with greater numbers of predators and no active reindeer herding taking place, so differing greatly from Sweden. However, semi-domesticated reindeer still receive little handling compared to other fully domesticated livestock. Due to this, and the scarcity of data, studies from these other countries were included in determining the parameters within this model. The data on avoidance behaviours across these locations, as reported in Table 3, are reassuringly similar enough to suggest the validity of this decision.

4.2. Limits of models in encompassing ecological complexity

Whilst our model provides a more detailed assessment of the impacts of infrastructure on reindeer ecology, the results should be interpreted as conservative estimates of true impact. For example, our model focuses on how infrastructure affects the ability of reindeer to graze, but does not include impacts on reindeer movement. Driven by a strong instinct to follow established migration routes, reindeer often continue to cross roads so long as physical barriers such as fences are absent (Reimers and Colman, 2006; Reimers et al., 2007), yet they tend not to linger near these structures (Dyer et al., 2001), with this avoidance behaviour removing natural stopping and resting pastures along their migration route. Changes to topography from infrastructural developments can also affect reindeer movement. For example, frozen river valleys are often integral components of traditional winter migration routes. However, the ice covering reservoirs behind hydropower dams is more unstable due to shifting currents and water levels, making it dangerous or impossible for reindeer to traverse (Dahle et al., 2007; Mahoney and Schaefer, 2002). Inclusion of these factors within a generalised model is difficult, as they are likely to be highly locally variable. Nevertheless, it valuable to be aware of these additional potential impacts of industrial development.

More broadly, year to year the weather, number of reindeer, herd composition, number of herd-owners, their ages and their level of experience all vary, meaning herd management must also vary. As stated by herder Mikkel Nils Sara in Paine (1994, pp.171) "unforeseen change is a law which necessarily sets its stamp on reindeer pastoralism and its pastoralists". As the herd varies, so do the features of their environment and the ways in which they interact with one another. Herders have created strategies to adapt to, and manage through, the periodically unfavourable weather in the region (Reinert et al., 2009). This including having herds of diverse individuals with diverse functional niches, encapsulated in the concept of a *čáppa eallu* (phenotypically diverse "beautiful herd", in the North Sámi language), which has a greater ability to withstand environmental disturbance and unfavourable climatic conditions (Reinert et al., 2009; Magga et al., 2011; Borgenvik, 2014). Another strategy is making use of diverse pastures, so if forage in one becomes inaccessible, others can be used (see e.g. Paine, 1994; Turi, 2008; Reinert et al., 2009; Magga et al., 2011; Eira et al., 2018). This requires maintaining access to areas of varying topography, wind exposure, plant community composition, and forest age, to name but a few. Determining what kind of pasture is needed and when involves complex and tacit knowledge, often built up through generations of herding experience.

Modelling necessarily simplifies a complex range of interactions, but there is a risk if this simplification is not tempered with consideration of how these data relate to real-life systems and local variation (see e.g., Jernsletten and Beach, 2006; Vuojala-Magga et al., 2011; Marin et al., 2020). State-imposed herd management strategies based upon more static agricultural and production-based models, excluding the experiential knowledge of reindeer herders, have been widely criticised as being inappropriate to the needs and functioning of the reindeer herding system (e.g. Tyler et al., 2007; Turi, 2008; Löf, 2013; Eira et al., 2018). As such, whilst the model presented here does start to introduce relevant complexity and accuracy to conceptual modelling approaches, it still only captures elements of the multifaceted relationships occurring in the ecosystem. Models like this would be best applied in the spheres of policy and land management as one of many tools, with management plans ideally being created in collaboration with reindeer herders. This would provide more accurate assessments of the system, drawing on both the quantified generalisable data within the scientific literature, and on the more locally nuanced experiential knowledge of herding practitioners.

4.3. Conclusion

The conceptual model presented in this study offers a more nuanced framework for evaluating the cumulative impacts of existing or proposed infrastructure on reindeer winter grazing pastures, as applied to the context of northern Sweden. By incorporating multiple, often synergistic impacts, the model highlights the critical importance of considering both physical and behavioural effects. Notably, our findings demonstrate that excluding behavioural impacts of infrastructure underestimates total impact by 86 %, while neglecting the accessibility of forage through snow underestimates impact by 11 %. This framework has practical applications, such as assessing the effects of planned infrastructure projects on reindeer forage availability or evaluating various land-use scenarios to identify those with minimal adverse effects.

The model can be refined and expanded as additional relevant data become available, including data on infrastructure such as wind farms, and being applied to summer grazing grounds and calving areas also. The model framework could also be applied to other geographic locations, and perhaps even other species, if adequate and locally appropriate data for these contexts are available.

The estimates of the model likely remain conservative, as it does not account for factors such as habitat connectivity disturbances, migration route disruptions, or inter-annual variability in weather conditions. Nevertheless, by integrating multiple forms of human activity and environmental conditions, we suggest that our model provides a more accurate and holistic assessment of the impacts of environmental change, in our case, upon reindeer ecology.

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Declaration of competing interest

The authors declare no conflict of interest.

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Data availability

All data used is reported within the core text

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