Reindeer ecology in a changing Arctic: Snow, vegetation, and traditional ecological knowledge

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Abstract

The cumulative effects of anthropogenic development on reindeer (*Rangifer tarandus*, L.), and how these impacts interact with a changing climate, remain largely unknown. To fill this knowledge gap, this thesis begins by examining how presence and winter accessibility of reindeer forage is affected by silviculture in boreal forests of northern Sweden, as stands progress from clear-cut to mature forests. Original surveys show that the abundance of various lichen species generally increases with stand age, highlighting the roles of competition, grazing pressure and disturbance in this process. Snow depth is consistently shallower in old stands, and the number of ice layers in the snow column increases throughout winter, affecting the ability of reindeer to dig to ground-lying lichens. Overall, there is upto 61 % lower availability of forage in clear-cut sites compared to old stands, showing that changes in forest structure have notable impacts on reindeer grazing.

Next the effects of multiple forms of land-use, individually and cumulatively, on reindeer are examined. A model is created which considers the impact of silviculture, roads, mines, hydropower stations, settlements and four climate scenarios, on reindeer populations over 50 years. All scenarios saw a loss of 54-100 % of reindeer, and only 25 % resulted in economically sustainable herd sizes for reindeer herders. Climate had the greatest impact on reindeer survival within the model. The results highlight that current and many projected future scenarios of land development create an unsustainable environment for reindeer and herders.

Alongside natural science methodologies, the role of history, politics and economics in the lives of Indigenous Sámi herders are explored, arguing that to gain a fuller understanding of reindeer ecology in a changing system, it is essential to consider both the biological and human context surrounding them. The processes of trying to carry out research using both scientific and traditional ecological knowledges are discussed, providing suggestions for others undertaking interdisciplinary work in this field.

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Statement of Copyright

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Preamble

The primary aim of this thesis, as with others, is to report on original research conducted by the author. Here, that includes delving into the complex world of ecology to understanding the relationship between semi-domesticated reindeer, their forage species, and their environment in northern Europe. Intense anthropogenic changes in the built environment are occurring in that region, yet the cumulative effects of these developments on reindeer is largely unknown, and how their impacts interact with a changing climate is understood even less, making it an important topic of research.

This thesis also has a secondary purpose, which is to try to carry out this research in an interdisciplinary manner, bringing together both biosciences and anthropology. The history, present, and undoubtably the future of reindeer in northern Europe is tightly interwoven with the Indigenous Sámi who have hunted and herded them for centuries. The Sámi, in turn, have been affected by the colonisation of their lands, which still has political, social, cultural, and spiritual impacts today. These factors, along with economics and law, strongly influence how Sámi reindeer herders and state legislation manage herds of semi-domesticated reindeer. To gain a fuller understanding of the ecology of reindeer then, it is essential to consider both their biological environment as well as the human context within which they exist.

The reader is encouraged to read and reflect on this thesis then, as a work of science, as a work of anthropology, and finally as an experiment itself in interdisciplinarity, attempting to bring together multiple ways of knowing, to create a more holistic understanding of a complex system. Due to its interdisciplinary nature, the thesis will depart from the usual format of the natural sciences in places, yet it is hoped that these departures will serve to enhance the research and the reader's understanding of the topics covered.

The scientific research and process of experimenting with interdisciplinarity have been fascinating for me as a researcher. I hope through reading this thesis, you too can have your thoughts provoked, your assumptions questioned, and your understanding and appreciation of reindeer immeasurably increased.

Chapter 1:

The past and present of land use and power relations in Sápmi

"Those who are eighty years old have experienced an unbelievable change during their lifetime in this society…"

– Rune Stokke, reindeer herder from Udtja sameby

(Helander-Renvall and Mustonen 2004, p. 269)

1.1 Introduction

Once seen by many as a desolate wasteland, in the twenty-first century the Arctic is receiving growing attention. This northernmost region of the earth is filled with resources, from oil to trees, metals to furs, and as the extent of sea ice in its waters decreases, the Barents Sea is being eyed up as a lucrative trading route between Europe and the Far East. However, it is often overlooked that this region has been a homeland, and its resources the lifeblood, for numerous Indigenous peoples for thousands of years.

In the European Arctic, these Indigenous peoples are the Sámi, and for the past few hundred years they have experienced a complex and often tense relationship with incoming groups who have been eager to exploit the resources of the north (Oksanen and Riseth 2004). This has played out in the form of persecution, colonisation and disenfranchisement in a number of ways, from culture-shaming and forced assimilation policies, to sterilisation based on race and adherents to traditional belief systems being burned as witches (Lundborg 1921; Willumsen 1997; Berglund 2005). However, far from being victims, today there is a proactive movement by the Sámi to not only promote their visibility and cultural practices, but also to protect the lands and animals which are important to them (Andersen and Atle 1985; Plaut 2012). As a Sámi woman in northern Sweden told me: this today is the generation of activists, of fighters who are proud of who they are.

Many of the most notable debates and battles today are between the Sámi and large-scale extractive industries in their homelands, such as the mining and forestry industries which, many Sámi claim, are destroying their ability to live a traditional way of life by altering the environment (Widmark 2006; Sandström 2015). This is a trend that can be traced back to the original Nordic colonisation of the north hundreds of years ago in the lands where the Sámi already resided. At the same time, the Sámi themselves have been accused by the general public and state of destroying the lands they say they wish to protect, through overgrazing by their reindeer herds (Torp 1999; Moen and Danell 2003). In Sweden both Sámi and other Swedes are citizens of the Swedish state, and there are those who are unhappy at the idea of special concessions in terms of land use rights being given to one ethnic group within the country over the others (SVT 2019).

Climate change is causing many changes in the Arctic, creating uncertainty in the longevity and accessibility of some resources. This coupled with a growing demand for these limited resources from an increasing world population with rising material expectations, means that it is becoming increasingly critical for land users to find effective ways to use and share the materials their local environments provide. Effective use and sharing of resources requires a genuine understanding of the current impacts that each land user has on one another and on the ecosystems within which they work. Perhaps even more challenging is the need to understand, acknowledge and positively act on the historic and current power relations between land users. This is a process which will no doubt highlight uncomfortable truths, such as how the Indigenous Sámi remain a colonised people in colonial states, the disproportionate 'sharing' of the resources in this region where capital is put ahead of cultural, the structurally biased legal frameworks which pander solely to 'western' modes of thinking, and concepts of land ownership which are used to grant the official land rights to different interest groups today. It will raise difficult questions of ethnicity, what it means to be one nation with diverse peoples and the notion that there may not be a universally pleasing answer to these questions that benefits all.

In this chapter I examine the power relations between stakeholders in the European north, looking at the impacts of commercial industry and Sámi reindeer herders on one another, and the environment around them. This will cover historical and current power relations, touching on how these relationships affect the ability of the Sámi and their reindeer to adapt to change, and concludes with a brief discussion on what may help to build constructive, equitable and fruitful relationships between these groups in the future.

Figure 1.1. A map of Sápmi (blue shading), the Sámi homeland in northern Europe. (Picture: Wikimedia Commons/CC BY-SA 3.0).

1.2 Early Land Use and the Development of Reindeer Herding

Sápmi is a region spanning northern Norway, Sweden, Finland and the Kola Peninsula of Russia (see Figure 1.1). It is characterised by long cold snowy winters, being partially within the Arctic Circle, and summer days with 24 h sunlight. Along the spine of Sápmi runs the Scandes mountain chain, whilst the lowlands are home to tundra, marshland and extensive coniferous forests. It is here that the Sámi are said to have been "since time immemorial" (Torp 1999).

The Roman historian Tacitus is credited with being the first to provide written evidence[1] of the Sámi in Sápmi, in 98 A.D. He writes that at the time these people used the land around them as nomadic hunter-gatherers, with both men and women taking part in hunting for game (Tacitus 1999). For subsistence they relied on a variety of sources, from marine and freshwater life such as herring (*Clupea harengus,* L.) walruses and salmon (*Salmo salar,* L.), animals with useful pelts such as beavers (*Castor fiber,* L.) and Arctic foxes (*Vulpes lagopus,* L.), and smaller game such as hares (*Lepus timidus,* L.) and ptarmigan (*Lagopus muta,* M.), along with edible plants like cloudberries (*Rubus chamaemorus,* L.). Large game was also hunted, such as elk (*Alces alces,* L.) and reindeer (*Rangifer tarandus,* L.). Rock art, thought to be from 4200 B.C., indicates that the Sámi used corrals and pitfall traps to hunt and capture reindeer, which are often identified as one of the Sámi's most important food sources (see Figure 1.2).

Exactly when this relationship started to shift from hunting to herding is unknown, but by 890 A.D. King Alfred of England writes that the Sámi were tending 600 tame reindeer (*Rangifer tarandus*, L.) for a Norwegian Chieftain named Ottar, showing that the knowledge and potential for reindeer pastoralism was present (Simonsen 1996). Archaeological evidence shows that around 1300 A.D. reindeer were being used as draught animals (Salmi *et al.* 2021). Sometime between the 1400s and 1600s A.D., if not earlier, they began to be cared for more intensively for milking purposes and by the end of the 17th Century reindeer pastoralism is considered to have been a widespread practice (Tengren 1952; Storli 1996). The reason for this shift is not fully known, but it is speculated that when the nation states

[1] Tacitus writes about the 'Fenni', an equivalent to the 'Finns' which at the time was a reference to the Sámi.

of Scandinavia started taxing the Sámi in the 15th Century and growing numbers of incomers into the area generated more trade, there was a greater need for the Sámi to hunt for pelts and meat for payment. A notable drop in game numbers around that time is suggested by some to have triggered the disappearance of wild reindeer herds in favour of owned and protected ones, as the hunter-gatherer way of life which relied on a wide array of resources over a large area was no longer sustainable (Tegengren 1952; Oksanen and Riseth 2004). Today there are no wild reindeer in Sweden, though in southern Norway and Finland some historically wild herds still remain, as well as feral formerly semi-domesticated herds.

The early reindeer pastoralism was a very intensive practice but the crowding created, especially in milking grounds, caused the spread of disease in the animals and was for the most part given up in favour of a more extensive form of herding (Lundmark 2007). This extensive style, which continues today, involves the movement of reindeer over a range of different grazing pastures which are largely unfenced, allowing the animals to roam free. They are periodically gathered in corrals for slaughter, separation of herds and marking the ears of new calves to denote ownership. The reindeer graze on ground lying vegetation, and lichen especially makes up an important winter food source for them.

It is likely that during Tacitus' time the Sámi had some contact with southern cultures, as has been explored in studies on their trading habits and linguistics. These interactions became more frequent from the 16th century onwards as the number of people moving north, identified as 'Swedes' and 'Norsemen', started to increase (Oksanen and Riseth 2004). With them came agriculture, which continued to expand, causing vegetation changes due to the grazing of cattle and sheep, in addition to the grazing already done by reindeer in the area. This arrival of new people may have been partially peaceful, with some suggestions that the Sámi and the incomers assisted one another with lodgings and skill sharing (Nordin 2002; Oksanen and Riseth 2004). On the other hand, there is also evidence of forced eviction of the Sámi with their possessions being destroyed and their reindeer killed (Fjellheim 1999; Thomasson 2002; Oksanen and Riseth 2004).

A growing sense of ownership by authorities in the south led to the prioritisation of farming in Sápmi, and the areas in which the reindeer could roam were restricted by legislation. In 1751 and 1826 the state borders were drawn between today's Fennoscandinavian countries, further restricting the reindeer's movements (Tyler *et al.* 2007; Reinert *et al.*

2009). In Finland the loss of traditional migration routes to summer pastures in the northern, and now Norwegian, coast caused a complete change in the way that reindeer herding is undertaken today. Many reindeer stay year-round in smaller areas and require far more supplementary feeding to survive in Finland compared to their wide roaming counterparts in Sweden and Norway.

It is worth noting here that not all Sámi shifted to reindeer herding as a source of livelihood. Many remained as fishermen by the coasts, sometimes termed as the Sea or Coastal Sámi. Oksanen and Riseth (2004) argue that the reindeer herders were in many ways the Sámi elite, who looked down on the farmers from the south and on their own fisher people at the coasts. Today less than 10% of Sámi herd reindeer, with many living in cities and working office jobs. Each Sámi village and family are unique in their cultural expression and practices, showing that 'Sámi' is a description for a community that is internally quite diverse (Josefsen 2007).

As rights granted to the Sámi outside of reindeer herding, such as those related to fishing and hunting, are normally attached to those Sámi that undertake reindeer herding, this is the subpopulation that will be focused on within the present study. It is important to note, however, that within these following discussions on rights and power relations, the voices of Sámi who do not herd are virtually non-existent, which within itself is telling of both hierarchies within the Sámi community and of what is deemed the 'correct' form of Sámi lifestyle by nation states to warrant even the discussion of their rights.

Figure 1.2. Sámi rock art in Alta, Norway depicting wild reindeer being chased into a corral to be hunted. Some artwork in the area dates back to 4200 B.C. (Photo: World Heritage Rock Art Centre/CC BY-SA 4.0).

1.3 Colonisation and Assimilation

Despite some positive trading relations, even as far back as Tacitus' writing, the Sámi were often viewed as a curious 'other' with their skis and furs. What may have started as curious observation led to the Sámi being looked down upon by others, who saw them as a primitive culture (Dooner, unknown; Lindmark 2013). This view was especially prevalent in the 17th and 18th centuries when concerted efforts were made to bring Christianity to the north (Joy 2014). The Sámi historically held beliefs based on polytheism, animism, spiritual connection with animate and inanimate parts of the natural world, and the use of shamans who could move between spiritual worlds. This stood in stark contrast to the forms of Christianity being taught during the reformation at the time. This contrast, along with the difference between the Indigenous nomadic lifestyle and the Christianised settled agricultural lifestyle, as well as linguistic differences, may have led to expressions of superiority among the incoming community in an attempt to maintain their own identity and prevent the settlers from being attracted by Sámi practices and culture.

The attitude of the newcomers towards the Sámi became more aggressive and paternalistic, shifting their position from new migrants in the area to colonisers (Lindmark 2013). Christian missionaries came to teach their faith, but this eventually turned into forced conversion, with Sámi drums[2] and sacred objects being confiscated and either burned or sent to museums, where some are still held today (Joy 2014). Many shamans were even killed in the witch hunting trials at the time, as they were known throughout Europe for their expertise in 'the art of magic' (Berglund 2005). Carl Linneaus in his travels around Sweden commented on how some Sámi shamans (known as noaidi) would be cut and bled till fainting, until they begged for their lives and gave up their drum (Willumsen 1997; Berglund 2005). There is little written record of these instances of torture, so it is hard to know the extent and severity of the persecution experienced by the noaidi and Sámi communities in general. The timescale of persecution is similarly hazy. Whilst these sorts of events are

[2] The Sámi drum is a ceremonial object used by shamans to go into a trance, allowing their spirit to transcend the body and travel to other places or even other spiritual worlds. The drum could also be used to learn about the future. A brass or horn vuorbi (pointer) was place on the drum face, which was beaten with a drum hammer. The symbols on the drum face onto which the vuorbi landed would indicate what would come to pass.

largely known to have taken place in the 17th and 18th centuries, a Sámi man from northern Finland, Lauri Ukkola, recollected how even as late as the end of the Second World War some priests confiscated drums from people's homes and burned them (Joy 2014).

Aggressive attitudes went beyond religious realms. A local inhabitant in northern Sweden told the author of how, during the development of early mining in the north, the Sámi were forced into slavery, with those who refused to work at mining sites being dragged under lake ice until they consented or died. They were stereotyped as 'too lazy' to actually work inside the mines, but were forced to use their reindeer herds to transport goods to and from the mining sites (Dooner, unknown). Harsh treatment was also expressed in state-wide policies encouraged by the Swedish State Institute for Racial Biology. This institute, established in 1922, was the first institute of its kind sanctioned by any country, and interestingly was supported by all political parties in Sweden at the time.

In 1934 Sweden passed laws where people could be forced to undergo sterilization for medical and eugenic reasons. This affected people across Swedish society, including many Sámi who were forced to undergo sterilizations due to their perceived 'genetic inferiority'. These ideas of genetic hierarchies were led by the Swedish State Institute for Racial Biology, first headed by ethnic Swede, physician and Professor Herman Lundborg in 1922. Lundborg promoted the idea that the Sámi were an inferior race and would therefore weaken the superior Swedish population if allowed to mix genetically (Lundborg 1921; Lundborg 1922; Lundborg and Linders 1926). The Sámi artist Katarina Pirak Sunna has devoted much of her time to exploring the hurt created by the racist attitude held towards the Sámi by the State Institute for Racial Biology, through their actions such as photographing and measuring the skull sizes of Sámi people, treating them as specimens and trying to prove lack of intelligence due to their ethnicity.

The paternalistic attitude towards the Sámi by the state was further expressed in what could be termed cultural genocide through 're-education'. Sámi children were forced to attend boarding schools where use of their traditional languages and the joik[3] were banned. The

[3] Joik or yoik is a form of singing traditionally done by the Sámi. Rather than singing *about* something or someone, the joik is supposed to evoke the thing or person. Traditionally joiks have few or no lyrics, and as they were said to resemble magic spells they were forbidden by churches during periods of forced assimilation. In some areas today it is still controversial to joik in a church.

children were also forced to learn the state's culture, language and practices in the hope that their Sámi roots would be forgotten. The extent and impacts of these harsh assimilation policies are mostly documented in Norway, though they did also occur in Sweden and to a smaller extent in Finland (Partida 2008). Similar policies based on the cultural isolation and 're-education' of Indigenous peoples to 'save' them from their perceived inferior ways have been seen across the world, from Native Americans, First Nations peoples and Alaskan Natives in the U.S and Canada, to the Aboriginal peoples in Australia, Maori in New Zealand and the San in Botswana to name but a few (Smith 2009).

The boarding schools threw many children into a cultural limbo. By being torn away from their Sámi culture, many struggled to adjust to their traditional lifestyle and language when visiting family during the holidays, yet at the same time though they were being prepared to be part of the dominant Nordic society, they were still viewed as inferior, so could not belong there either (Smith 2009). Recent discussions between the author and a young Swede highlighted the psychological impact of this. She told me of how her family had only just learned that her grandparents were Sámi. These grandparents had felt so ashamed and fearful of revealing their cultural and ethnic roots that they had told no-one, not even their own children and grandchildren of their backgrounds.

Thankfully the boarding schools are a thing of the past, though their effects are still felt by many. However, whilst today the Sámi have the freedom to express themselves in their traditional arts and language, their 'freedoms' do not stray very far outside of the artistic realms. Aspects such as land rights for the traditional practices of reindeer herding are still limited, determined strongly by the state and its values and ideas of landscape management. This raises questions about de-colonisation and what is actually involved, and more to the point whether it has actually been done. As Howitt and Suchet-Pearson highlight in their piece on Indigenous land use in Australia (2006):

> *"Indigenous self-determination is reconstituted in indigenous affairs discourses and practices not as a right to self-governance, but as 'community management'. In doing so, the very real historical and administrative processes of dispossession, theft and genocide which produced the 'communities' that are to be managed, as well as assertions of sovereignty and identity, and aspirations of being-in-place on one's own terms, are all rendered invisible. Indeed, they are reconstituted as barriers to achieving orderly management of the people and resources involved within the systems of 'good governance'*

prescribed by the states which perpetrated the dispossession, theft and genocide."

Whilst Indigenous communities across the world vary in their relationship with the dominant or colonising cultures in their midst, this pattern of paternalism appears to continue in Sápmi as a growing number of stakeholders seek to utilise and exploit ever shrinking natural resources in the area. Recent history shows that often the economic and developmental incentives are put before the land use needs and priorities of the Sámi and the Sámi reindeer herders.

Figure 1.3. The First Sámi Congress held in Trondheim on 6th February 1917. The assembly contained around 150 Sámi people from both Sweden and Norway. (Photo: Schrøderarkivet/Sverresborg/CC BY-SA 4.0)

1.4 Early Conflicts: Hydropower and the Alta Protests

In 1917 the First Sámi Congress took place in Trondheim, Norway, initiated by Elsa Laula Renberg and chaired by Daniel Mortensen, both Sámi politicians and herders (see Figure 1.3). The assembly covered topics from the schooling of Sámi children to initiating a Sámi newspaper and the issues with the legislation governing reindeer herding at the time. Whilst views on the matters diverged amongst the 150 or so attendants, the assembly represented an important milestone in Sámi collectivisation which formalised their feelings of not being suitably represented by the colonising governments and political parties. The Congress also led to a proposal being made for a new reindeer herding act (Josefsen 2007). However, it was not until much later that the power relations between the Indigenous inhabitants of Sápmi and the state began to shift in full view of the wider world. This shift came about in the 1970s, centring on a historic set of protests against the construction of a hydroelectric dam.

Sweden has *ca.* 1800 hydropower plants (Swedish Agency Marine and Water Management 2019), and Norway a further 1070 (Statistics Norway 2017a), which harness energy from the movement of water carefully released from dammed reservoirs. With the well-publicised impacts that burning fossil fuels is having on Earth’s climate, having a large and growing renewable energy sector has been seen as a positive development, yet despite the benefits of reducing fossil fuel usage, there are also negative consequences from hydropower plants for both the environment and the local people. The loss of land caused by the construction of a hydropower station and the flooding of its reservoir is what led to an uprising amongst the Sámi.

The Alta Controversy, as it is now known, was a set of mass demonstrations in the 1970s by the Sámi together with environmentalists against the building of a hydroelectric dam in Alta, Norway. The construction of this dam would have flooded a Sámi village, along with important reindeer grazing grounds and migration routes (Klein 1971). There were many potential effects on other wildlife too, with the dam disrupting salmon migration routes thus altering their life cycle and the flow of nutrients they bring to animals upstream as a food source. Even today attempts to create bypasses for salmon at dams are still not fully effective (Rivinoja *et al.* 2001; Thorstad *et al.* 2017). Aside from the flooding, the subsequent decrease in water flow downstream due to its regulation and storage in the reservoir would alter habitat of flora and fauna in the river below the dam (Klein 1971; Ligon *et al.* 1995). This meant a plethora of negative impacts for herders, wildlife and anyone reliant on the fish as a source of food.

A notable feature of the conflict therefore was that it increased the visibility of the Sámi through actions such as protesters undertaking a hunger strike in front of the Norwegian parliament wearing their traditional clothing. Throughout the colonisation of Sápmi little was taught to the dominant society about the Sámi and their culture, making them an almost invisible people in their homeland, so these very visible actions sent a clear message to the country that the Sámi still existed, and were going to stand up for themselves. Other activists set up blockades at the dam construction site, with the situation becoming so significant that at one point 10% of Norway's entire police force were present at the protests, the largest mobilisation of police in peacetime Norway (Andersen and Atle 1985).

In the end the dam was built, but the chaos and discussions sparked by the protests raised the issue of Sámi land rights with the general public. This eventually led to the creation of a Sámi Rights Committee, the Sámi Parliament and the Finnmark Act which gave 95 % of the lands in the Finnmark region of Northern Norway from the state into the ownership of the residents, highlighting the rights of the Sámi (Finnmark Act 2005).

It should be noted that there are some attempts at reconciliation for the impacts that dam construction has on the Sámi, with one herder in Sweden informing the author of how some herders in Norway obtain compensation from hydropower companies. This does not happen in Sweden however, and he said that many reject this compensation on moral grounds, stating they do not wish to accept money from an industry that is causing harm to the land.

Despite the eventual construction of the dam at Alta, the protests were seen as a success, creating more recognition for the land-rights of the Sámi, and this event set a precedent over the next few decades for increased activism.

1.5 Mining and Indigenous Visibility

Sámi activism has come to the fore again in another more recent land use dispute in Sweden between reindeer herders and mining companies. Sweden produces over 90 % of Europe's iron ore and currently has 12 active mines (Mining inspectorate of Sweden 2021; SGU 2019a). Construction of these mines and their access roads involves the removal of plant life and animal habitats, and once built the day to day activity at the mines in terms of dust, noise and human occupation continues to cause disturbance to animals in the area. Reindeer are known to avoid areas of between 11-14 km around sites of heavy human activity (Boulanger *et al.* 2012; Skarin and Åhman 2014), and up to 4 km around roads and power lines (Vistnes and Nelleman 2001), meaning each mine represents a significant loss of potential grazing pastures for the animals.

Much habitat and grazing lands are lost to the physical structure of the mine, but huge areas are also lost due to the storage of mining waste. In 2010 Sweden produced 117.6 million tonnes of mining waste, almost doubling waste production from the preceding decade (Swedish Environmental Protection Agency 2005; SGU 2019b). A large portion of this is 'tailings', fine particles of rock and minerals separated from the ore during its processing. These particles are combined with water and poured into large areas creating a pond, and when the debris settles excess water is often extracted and reused. To give an idea of the size of these 'ponds', one at a mine in Kiruna, Sweden is held in by a dam of 4km in length whilst another in Aitik covers an area of 13 km^2 (Mainali *et al.* 2015). As with hydroelectric dams, these mine tailing dams hold a risk to human life if they burst, with a recent disaster at Brazil's Corrego do Feijão mine's tailing pond killing hundreds of people downstream (Almeida *et al.* 2019).

Even without disasters, there are concerns of the chemical impacts of the waste on the surrounding environment, with bacterial breakdown of compounds in the tailings leading to the release of trace metals into surrounding soil and water systems. A Sámi activist opposing the building of mines in parts of Sápmi told the author of a grim irony: First these mines are created against the will of many locals, he explained, then their waste is poured into valleys and inlets to be stored, and finally when the mining companies leave, some bring sewage from southern cities like Stockholm and cover the mine tailings with it. This technique of using sewage to cover tailings is an attempt to smother bacteria to slow their breakdown of compounds within the tailings into harmful substances, yet it is only a short term band-aid solution, as within a decade 85% of the sewage will have broken down itself, unsealing the tailings below (Nason *et al.* 2013). In terms of relationships between land users, one can also see how bad feelings may arise when an Indigenous community is barred from utilising a space they have used for centuries only to watch a mine be built there, its waste poured into lands important for traditional livelihoods, and finally sewage from the capital city of the colonial culture being dumped on top.

This lack of respect and consideration for the Sámi appears to be a recurring feature in interactions between the mining industry and reindeer herders. A recent high-profile case in 2014 involved the application by Beowulf Mining plc to build two iron mines in Gállok (also called Kallak). These mines would split some Sámi reindeer herding areas in two, according to some making herding in that region unmanageable. A notable feature of this conflict was a statement by the CEO of Beowulf, Clive Sinclair Poulton, who in a press conference documented in Tuorda (2014) said:

> *"I show this slide to people primarily in the UK and Ireland (shows photograph of a landscape with forests and lakes) because one of the big, one of the major questions I get is 'What are the local people going to go ahead and say about this project?'. I show them this picture and I say 'What local people?'"*

He went on to say that indeed there are some locals, but not many, and their reactions to the project had been positive. However, his statement of 'what local people' caused an outcry amongst the Sámi, highlighting further their lack of visibility when land use decisions are being made. These comments were not only showing ignorance of Sámi presence in the area on the part of the mining company, but also displaying a worryingly colonial attitude through the use of colonial rhetoric to justify occupation.

William Clifford Holden outlined the 'Empty Land Theory' (now known as the Empty Land Myth) in relation to South Africa at the time of its colonisation, the theory arguing that the lands had been mostly empty prior to European settlement. As, according to the theory, the Europeans and Indigenous peoples arrived at the same time, they therefore had the same rights to claim the lands and thus whoever had the greatest force had a fair right to control the area, this unsurprisingly being the Europeans (Holden 1866). His theory essentially erased any idea of original inhabitants with rights. This was due largely to a lack of understanding by the Europeans of how land was used by the often somewhat nomadic Indigenous peoples, and this theory was further used throughout Africa to legitimize colonisation. Comments by industry that Sápmi is devoid of people simply revives the Empty Land Myth and archaic colonial rhetoric used to legitimize taking land from Indigenous peoples. This did not go unnoticed, and Clive Sinclair Poulton's comments together with efforts to begin construction of a mine in Gállok, once again led to protests by the Sámi, their supporters and environmental activists including blockades leading up to the mine site. Norrbotten County subsequently rejected the mine in 2014, a decision which was over-ruled by a governmental body called Bergsstaten. Now, many years later, a final verdict is still being decided upon by the Swedish government.

Part of the herders' argument against the mine was that they had land rights within the current Swedish legal system yet were these were being ignored. According to Sweden's Reindeer Husbandry Act, the Sámi have the right to use and maintain both themselves and their reindeer on lands that they have used traditionally, or that are culturally or spiritually significant to them (Rennäringslag 1971). Proving this right is somewhat difficult however, as courts state that the Sámi must have evidence that they have regularly used the land for the past 90 years to claim ownership (Bengtsson 2004).

Showing evidence of regular land use through time is difficult, especially for a livelihood that does not create obvious lasting changes within the landscape at each use. The Sámi also have not traditionally kept written records of their herding, so providing this evidence has relied on potentially incomplete local bailiffs' records. These records may not encompass the variable nature of reindeer land use, where some pastures are crucial during years of poor climate, but receive little or irregular use otherwise. This disparity between cultures of documentation and ontologies of land use have made providing written or physical evidence to claim land rights difficult (Borchert 2001). Perhaps aided by the ambiguities of what counts as a valid land rights claim, mines have continued to be constructed in herding areas without the herders' consent. Yet, whilst the outcome of the Gállok case is still pending, the fact that Norrbotten County supported the Sámi, along with scientific evidence indicating the potential harm of mining to reindeer herding, shows a positive shift in the level of regard given to the herding community and its needs.

1.6 Forestry and the Law

Silviculture is another industry which has difficulties trying to operate alongside reindeer herding, leading to more court cases over land rights. Sweden has 23.5 million ha of productive forested lands, 0.9 million ha of which is felled each year (SLU 2018). In stark contrast to the mass deforestation caused by logging in some parts of the world, the country is actually seeing an average increase in forest size due to replanting (Official Statistics of Sweden 2019). Whilst the number of trees may be increasing, the changes in forest structure caused by logging has had some negative consequences for the local flora and fauna.

During winter, which is the period of greatest food stress for reindeer, the snow sometimes becomes too hard or thick for them to dig through to reach ground lying vegetation below (Kohler and Aanes 2004). At these times tree lichens, also known as arboreal lichens, become an important food source for them as they are still accessible above the snow line. Arboreal lichens only grow on the bark of older trees (Horstkotte *et al.* 2011), yet trees in Sweden are commonly felled between the ages of 45-120 years, and as commercial silviculture has grown in scale, taking larger areas into the cycle of felling and replanting, the volume of old-growth forests has declined (Swedish Forestry Act 1979; Axelsson 2001; Korosuo *et al.* 2014). This means the loss of an emergency food source for reindeer in many areas.

The availability of forage for reindeer is also affected by the physical action of logging machinery on the landscape. This often heavy equipment churns up the ground when venturing into forests, and intentional soil disturbance called scarification is sometimes used to assist the growth of new tree seedlings during replanting. Both these processes disrupt ground lying vegetation, reducing the abundance of lichens, once again reducing the amount of forage available to reindeer (Roturier and Bergsten 2006; Korosuo *et al.* 2014). Along with forage presence, the accessibility of food is also being affected by commercial logging. The debris of branches and bark left behind in an area when it has been clear cut can create a barrier between reindeer and ground lying vegetation (see Figure 1.4). Richard Länta (in Tuorda 2018), a reindeer herder from near Jokkmokk, Sweden, has spoken of how his reindeer, with their keen sense of smell, can detect lichen through the snow in clear-cut areas during winter:

"After having dug through 2.5 feet of snow it (the reindeer) is faced with an equally thick layer of broken branches and shrubs… We, and many generations to come, will have passed before this area is grazable again… getting through all this, to the ground where the lichen grows is more or less impossible. In order to get to the lichen it has to move on to new pastures and try again and again. At some point or another, the reindeer give up."

In this situation, arboreal lichens above the snow and the debris would be a useful alternative food source for the reindeer, if the surrounding trees were old enough to contain them.

Figure 1.4. The debris left behind in forests in Northern Sweden after clearcutting has taken place. This creates a barrier to reindeer when they are trying to dig through snow to access lichens below in winter (Photo: the author).

Snow conditions also strongly influence the accessibility of food for reindeer. This is partly due to natural climatic conditions, but is also affected by forestry because of the effect tree canopy structure has on the way snow lies (Inga 2007; Rees *et al.* 2008; Riseth *et al.* 2011). For example, the canopies of old coniferous forests trap snowfall, thereby reducing the depth of snow on the ground. In warm periods this allows snow to melt completely in old forests, whereas in younger aged stands the deeper snow is more likely to form ice crusts which reindeer cannot dig through (Kumpula and Colpaert 2007; Roturier 2016). In late winter this pattern is sometimes reversed as snow trapped in the canopy of old forests may fall, creating deep piles on the ground (Roturier 2011). Old growth forests are generally viewed as preferred pastures during periods of acute food stress in winter, as their more varied canopy structures potentially allow patchier ice formation, leaving accessible ground in between patches (Kumpula and Colpaert 2007; Roturier and Roué 2009; Roturier 2011). Removal of old forests, with their multi-aged canopies, reduces the diversity of the landscape. The resulting homogenous landscape, made up of same-aged planted monocultures reduces the range of pastures available for reindeer to graze throughout winter (Kivinen *et al.* 2010).

Despite the negative effects of logging on reindeer herding, it is often forest owners who have been bringing the herders to court over land disputes. As mentioned, reindeer herders have a right to graze their reindeer on almost any land in Swedish Sápmi according to the Reindeer Husbandry Act (Rennäringslag 1971). Whilst the law is largely being upheld, a few private forest owners have been taking the Sámi to court in recent years with the hope of getting them legally excluded from the forests, stating that the reindeer cause damage by rubbing their antlers on young trees (Borchert 2001). This argument appears questionable as little has been said about wild elk who also graze on young trees, with all the blame being placed on reindeer.

These legal battles can be costly and, as mentioned earlier, the complexity of providing written evidence of landscape use by the Sámi confounds this discussion. Despite all this, some of the cases have been successful against the Sámi, such as one in 2011 in Nordmaling where two Sámi collectives (called Samebys) were sued for using land to graze reindeer without 'prior agreement' with foresters (Torp 2013). Any attempts to resolve the issues through meetings between the two parties, issues which in some cases have resulted in these kinds of legal action, were generally deemed ineffectual (Widmark 2006; Sandström 2015).

It should be noted here that some degree of forestry practice has been and still is common among both reindeer-herding and non-herding Sámi across Sápmi. The wood gained is often either used for materials for e.g., handicrafts and construction projects, used for fuel or sold for economic gain (Markkula *et al*. 2019; Forbes *et al.* 2020). The majority of this forest use is small-scale and so is not incompatible with reindeer herding.

There have been attempts within the scientific community to bridge gaps between the needs of reindeer grazing and commercial silviculture. Harvest schedules have been created for Swedish forests which include corridors for reindeer to graze and travel through, with their implementation deemed to be low cost (St John *et al.* 2016). The benefits to reindeer of partial cutting as opposed to clearcutting of forests have also been shown (Stevenson 1986; Korosuo 2014). These as well as preserving patches of old growth forest for reindeer, however, have been deemed as leading to opportunity losses for the forestry industry leading to a reluctance to utilise these methods by many (Bostedt *et al.* 2003).

Not all forestry groups work by the same principles in their interactions with Indigenous land users. One reindeer herder the author interviewed in Sweden spoke positively of logging groups that are certified by the Forestry Stewardship Council (FSC). This is because the criteria to gain certification by the FSC state that Indigenous peoples must be given the power to grant or withhold consent for operations on their traditional lands (Forestry Stewardship Council 1996). Sweden and Finland, unlike Norway, have not ratified the UN Indigenous and Tribal Peoples Convention (known as ILO 169) so the timber industry there is not under pressure to respect Indigenous land rights unless they wish to gain certification by groups such as the FSC.

The actions of the mining and silvicultural industries have been largely detrimental to the Sámi livelihood of reindeer herding and the environment in general. At the same time, it is important to note that the actions of the reindeer herding community are not always environmentally beneficial. One factor affecting all these groups, and thus the way that they interact with the environment and one another, is that both are influenced by the basic human need to earn an income.

1.7 The Power of Economy

Part of what drives industry in the sparsely populated north is the provision of jobs, which leads many to welcome the prospect of a new mine or timber works in their area. This is not only reserved for general Scandinavian society. Whilst the Sámi lifestyle may once have been based upon subsistence hunting, gathering and trade, the Sámi today are much like their southern counterparts, living under the umbrella of capitalism and the need to earn money. Many herders have part time jobs to supplement their incomes. Cars are driven, permanent houses built and holidays to places such as the Mediterranean taken, which all need to be paid for. Of the many Sámi who do not herd reindeer, some actually own and are commercially logging the forests which are causing issues for the reindeer. Whatever one's ethnic origins, there is today a need for some form of income, and so many of the arguments over land use are also occurring *within* Sámi communities where individuals' priorities vary.

The connection between many herders and their reindeer is deeply rooted and culturally significant, yet for herding to succeed today in any widespread way it needs to be profitable. To sell reindeer meat commercially within the EU, EU regulated slaughter houses must be used to kill the animals (Reinert 2006). This ensures that certain standards of animal welfare are upheld, yet also means extra costs to the herders who often have to transport their animals for long distances to these slaughter houses instead of doing the work on-site themselves (Reinert 2008).

Herders also need more capital to afford new technologies. Machinery such as snowmobiles and helicopters are increasingly being used to gather the animals, easing the physical burden of work that used to be done entirely on skis. This means less time needs to be spent out following the reindeer during winter, allowing more time for other jobs and family life, but it also means additional costs for the equipment and the fuel to sustain it.

Reindeer are like currency among the Sámi. Some feel that having more reindeer is like having insurance to help one ride out a bad year if a few animals die, and of course a larger herd means more opportunity to sell the reindeer for actual financial gain (Johannesen and Skonhoft 2010). Growing costs have led some to want larger herds which in turn has sparked a new debate against the Sámi about landscape destruction, this time centering on claims of over-grazing by unsustainably large reindeer herds.

1.8 The Overgrazing Debate

Growing reindeer populations have coincided with widespread reductions in the abundance their forage species in parts of Sápmi, with lichen cover in some areas being reduced from 30 % to 1 % since the 1970s (Johansen and Karlsen 2005). Seeing these transformative effects that reindeer grazing and trampling can have on a landscape has led some to comment that reindeer herding is 'destroying the Swedish mountains' and causing 'ecological disaster' (SOU 1995; Torp 1999; Moen and Danell 2003). These concerns around overgrazing have led the Norwegian government to set herd size limits on the herders, forcing many to slaughter their reindeer to reduce the population size, which has raised more questions about the power relations at play when land use decisions are being made (Johnsen *et al.* 2015).

The debate around overgrazing is a contentious issue. Reindeer are clearly altering their surrounding plant communities, often causing a shift from shrub dominated communities to grasslands. This can cause a decrease, and in some cases even elimination of, important reindeer forage species like lichen from an area. However, some suggest that this change is not a negative change and that instead what is being seen is simply a shift to an alternative stable state in the natural environment (van der Wal 2006; Pape and Löffler 2010). Whilst lichens *can* make up a major portion of reindeer's diet, up to 80 % in winter (Heggberget *et al.* 2002), globally the proportion of lichens in the diets of various *Rangifer* subspecies varies considerably, with some herds surviving on the grassland plant communities in their environment to which some grazing areas within Fennoscandia are shifting. Reindeer show a strong preference for eating certain plant species depending on digestibility and availability, even in environments where the range of plants they can eat is actually much wider meaning there is room for adaptation (White and Trudel 1980; Danell *et al.* 1994). There is also evidence that shifts in plant species from heavier grazing can lead to an increase in the diversity and productivity of the local plant community (Suominen and Olofsson 2000; van der Wal 2006).

Reindeer grazing may even be beneficial for the environment. Some studies report shrub and tree expansion into the Arctic due to the warming climate (Moen 2008; Myers-Smith *et al.* 2011). Tundra soils store greater amounts of carbon than boreal forests, so there is a release of carbon when trees and shrubs shift north into the tundra (Hartley *et al.* 2012; Väisänen *et al.* 2015). Reindeer however can slow, or even halt, this upward shift by consuming colonising saplings, meaning they may be helping to reduce carbon loss from soil. Halting shrub expansion may also allow an area's albedo[4] to remain high and delay the date of snowmelt (Cohen *et al.* 2013). However, these effects of herbivory can vary, based on local climate, topography and grazing intensity, and in some areas herbivory can encourage birch seedling establishment by removing lichens, reducing the barrier effect of

[4] The albedo effect is when solar energy reflects off light coloured surfaces such as snow and clouds, sending this energy back out into the atmosphere and reducing its warming effect on the earth.

lichen mats between seedlings and the soil, and so aiding the colonisation of the tundra (Tømmervik *et al.* 2009). Therefore, the full extent of the positive effects of grazing are still in question.

Overall, it could be argued that an issue does not necessarily stem from shifts in vegetation communities, but rather in the levels of productivity in an ecosystem and in the change in amount of carbon held there above and below ground. Nonetheless, there is of course a tipping point and concerns about overgrazing are not completely unfounded. There are cases where grazing has become so intense that reindeer health has deteriorated as the carrying capacity[5] of an area is reduced (Bråthen *et al.* 2007). This once again ties into economics, as unstable grazing conditions year-on-year can create uncertainty in the survival rate of reindeer, which can lead to the desire on the part of herders for larger herds as a safety net or buffer against financial ruin (Næss and Bårdsen 2010). A larger herd would mean that sufficient reindeer would remain surviving after a 'bad year' for the herd to continue, but would also create even more pressure on the land leading to a case of the 'tragedy of the commons'[6].

Between 1995 and 2016 reindeer numbers in Sweden have remained roughly stable around 250,000 animals with small yearly fluctuations (Jordbruks verket 2017). The reindeer population in Norway on the other hand did rise from 250,000 to 350,000 between 1998 and 2011, though this number had reduced to below 300,000 by 2017 (Statistics Norway 2017b). These trends are relatively consistent across herding areas in each country, suggesting that differences in herd management rules and strategies between states significantly influence reindeer numbers. The number of reindeer herders in Norway and Sweden have remained relatively stable, so this is not a case of fewer people with larger herds (Käykö and Horstkotte 2017).

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[5] A carrying capacity is the limit of how large a population can be stably sustained in a given environment. In other words, the amount of food present will determine how many animals can be fed, and the number of these animals which can survive without a population crash is the carrying capacity of that environment.

[6] The tragedy of the commons is a concept created by Garret Hardin where individuals using a shared resource act in ways that are beneficial to only themselves. When everyone does this, it causes the shared resource to be used up or destroyed to the detriment of all. Hardin's example given was of cow farmers on shared pastures placing too many animals on to the area to maximise profits, leading to overgrazing.

Views within the reindeer herding community on overgrazing are varied. Many are worried about landscape degradation and have been critical of others with large herds. Some have argued that the root of the issue doesn't stem from increasing reindeer numbers but from a decrease in grazing grounds due to industrial development, which they see as the real cause of greater grazing pressure (Johnsen *et al.* 2015). There are also feelings of upset amongst many who have acknowledged that there are issues with land use in some areas, but when solutions for these problems are being sought the herders are not sufficiently consulted by the state, with the government imposed slaughters being forced on them (Johnsen *et al.* 2015). They felt the state had over-simplified the situation to 'too many reindeer' and was exerting too much control over them (Torp 1999), whilst ignoring their experience and traditional ecological knowledge, which told the herders that the problems stemmed from a complex mixture of biological and political factors.

This view is supported in a paper by Benjaminsen *et al.* (2015) which discusses the idea that claiming excessive numbers of reindeer are present is an old concept which has been brought up repeatedly in an attempt to alleviate conflicts with growing agriculture in the area. This narrative was then shifted to 'overgrazing' causing environmental degradation from the 1970s onwards, when concern for the environment grew. Years with unusual weather conditions causing mortality were used to evidence die offs out of context, saying it was due to overly large populations. The paper also highlights that attempts to control the population by limiting perceived overabundance of the animals failed, indicating this was not the real issue. Instead Benjaminsen argues that the carrying capacity of an environment varies year to year with environmental conditions, and so the idea of having stable 'sustainable' reindeer populations with firm upper and lower limits is fundamentally flawed. The populations will naturally cycle and change with the environment as they have done in the past, a fluid situation which is compatible with the Sámi way of life in the past, though perhaps not compatible with the market's need for a stable supply of reindeer meat today.

Clearly continual expansion of herd sizes will lead to problems in the long-term, and ways need to be found to keep them dynamically sustainable. However, the case put forward by reindeer herders that they require space and land to continue their livelihood and tradition must be considered, and this case has had support from the scientific and legal communities. In terms of power relations, the point to take away is that when these discussions on overgrazing were at their height, the arguments against the Sámi were consistently the dominant narrative, even when scientific evidence concurred with the herders, indicating that there was not an equal level of visibility and opportunity for each side to put forward their case (Johnsen *et al.* 2015).

1.9 Visibility and Recognition of Indigenous Rights

As communication becomes easier with advancing technology, and the potential for media to be distributed across the planet grows, the tide may be turning in terms of power relations in the north. The Sámi and especially reindeer herders are getting greater recognition in mass media, from films about issues affecting the Sámi such as 'Sami Blood' directed and written by Amanda Kernell, to Sámi joik-style singing by the band KEiiNO gaining the tradition worldwide exposure in the Eurovision song contest in 2019. Perhaps from this increased exposure alongside the scientific, historical and sociological research that is being generated about and with the Sámi, empathy may be growing to the difficulties faced by reindeer herders today. According to a researcher who has worked with Sámi herders and with the timber industry over many years, forest owners are now more wary of trying to exclude herders from forest lands. There is a fear that, with all this newfound attention, the herders may start to win these legal battles which could set a precedent that may even lead to the forest owners being legally excluded from lands. This is not mere speculation, with evidence showing that the voice of Sámi rights in the face of industrial development is becoming stronger. A recent example of this has been seen in Finnish Sápmi.

Shrinking sea ice in the northern oceans has led to plans to develop a shipping route between Europe and the Far East via the Barents Sea. In order to get the transported goods from northern Europe to the rest of the continent, plans have been made to build a railway line called the Arctic Corridor. One potential route of this railway line goes from Kirkenes in northern Norway to Helsinki in southern Finland, then through a tunnel to Estonia and beyond (Arctic Corridor 2019). This railway line would pass through six different reindeer herding areas in northern Finland, disrupting the movement of the animals and splitting up their pastures. A report written on the feasibility of the project deemed it unviable (Ministry of Transport and Communications 2019), with one of the major issues raised being the impact that the transport corridor would have on reindeer herding. The report concluded that reindeer herders must be consulted to ensure that “no irrevocable damage would be caused to their culture or livelihoods”, and Åsa Larsson Blind, the president of the Sámi Council, stated that there was no way for the railway to be built without harming Sámi culture, reindeer herding and fishing. Whilst the project has only been put on hold and is not totally abandoned, it appears that the needs of reindeer herding have been a hindrance to its development.

2020 saw another success for the Sámi community in terms of land rights, when a court case between the Sámi village of Girjas and the Swedish state led to the Sámi of the Girjas community regaining the ability to exclusively control fishing and hunting rights in their area. This was not only a case of preventing more loss of Sámi rights and lands, but of some reclaiming of it from the state.

The assertion of Indigenous rights over industrial ones is gaining momentum worldwide, as has been seen with the Waorani in Ecuador in 2019, who won a lawsuit to remove tribal lands from an auction that would have opened it up to oil exploration. This precedent may give greater power to the Sámi as well as many other Indigenous groups in future if they wish to oppose industrial developments in the territories historically used by them.

1.10 Social Tensions and Differing Ontologies

Each of the major forms of land use discussed here have an impact on both the environment and on one another in Sápmi. Reindeer herding, for better or for worse, causes changes in vegetation. On the other hand, commercial silviculture, mines and hydroelectric dams remove habitat from wildlife and disrupt the traditional migration routes of reindeer as well as other animals like fish (Vistnes *et al.* 2010; Hermann *et al.* 2014; Thorstad *et al.* 2017). It may be unreasonable to expect absolutely no environmental impacts from any of these activities as structures and transport routes do require physical space and resources to create some essential products, but the sheer scale of industrial development occurring in northern Europe is important to note. In Norway the last century has seen a 70 % reduction in undisturbed reindeer habitat which of course means greater disturbance of animal life, plant life, water cycles and even local microclimates (Nelleman *et al.* 2003). A 71 % loss of lichen-rich heaths to development has been seen in Sweden, and if these are not replaced by alternative edible plant communities, this results in a dramatic loss of pastures for reindeer, showing how industry in its current form is restricting the success and viability of reindeer herding.

As discussed, the impacts of industry on people are not wholly negative. In rural communities where people often have to hold down multiple jobs to make ends meet, the opportunities for income from industry are welcomed by some whether in the form of work or compensation. These jobs can help stem the flow of people moving to the south of Sweden for better opportunities, helping to sustain northern communities. In disputes the reindeer herders are often not asking for a complete monopoly on land rights, but rather simply that their activities are not infringed upon, their landscape not degraded, and their voice respected in consultations on land use. Issue is taken not with the existence of industry, but with its extent and the way that its activities can often ignore the needs of other land users.

Whilst neither reindeer herding nor mining and forestry can grow exponentially, there must be a more equitable balance that can be reached. Finding this balance will not be easy, and will require compromise and genuine consultation. This will no doubt unearth some uncomfortable truths, as through all of these discussions runs a strong current of politics and tainted social relations. The Sámi have historically 'owned' or resided in Sápmi, yet were colonised and their land rights ignored. Now many generations later there is a difficult situation. To return the land exclusively to the Sámi would mean taking it away from people who are not responsible for the historical land theft themselves, despite being beneficiaries of past colonial acts, yet continuing to withhold the land rights from the Sámi perpetuates this theft created by colonisation. This issue has no clear solution, and is further complicated by questions of who the Sámi are, whether they are only those who actively herd reindeer or whether they include those of ethnic Sámi origins yet who are the ones undertaking other forms of land use such as silviculture today.

Adding to the difficulty of solving these problems is the unwillingness to cooperate and enter into discourse coming from industry and herders. Many Sámi herders have experienced attempts at discourse in the past which have proven pointless with their voices being marginalised, and due to the history of being oppressed there are negative feelings and mistrust towards the state. There is also mistrust towards the Sámi as many people do not fully understand the colonial history of the region and fear their own rights may be taken away from them in favour of a group who they feel are getting unfair positive discrimination for being Indigenous.

This was exemplified in a recent dispute in Kiruna, Sweden. Here herders applied to get a periodic ban on recreational snowmobiling in reindeer calving areas when mothers and new-borns are likely to be disturbed, an application which was sparked by the death of a pregnant reindeer which was chased to exhaustion and death by a recreational snowmobiler. As part of the ban the herders were still allowed to use snow mobiles for their work with the animals as they would do it in appropriate and necessary ways. Implementation of this ban led to an explosion of criticism from local individuals and politicians who were upset at the idea of having their recreational rights restricted. They claimed it was not a question of reindeer protection but of the Sámi getting rights which were not extended to the general populace (SVT 2019). This mistrust from both sides towards the motives of the 'other' creates a tension which spills over into claims of racism and further upset, even leading to 59 academics publishing an open letter condemning the fact that lawyers in some legal cases were using the "rhetoric of race biology" (Dagens Nyheter 2015). In a similar situation, when the news was released that the Sámi had won the aforementioned Girjas legal case which gave herders in the northern Swedish district of Girjas exclusive control over hunting and fishing rights in the area, there was an immediate backlash from the non-Sámi community in the area, which ranged from hate speech to death threats towards the Sámi and even purported cases of their reindeer being shot by enraged locals (DeGeorge 2020).

A potential explanation for some of the accusations and perhaps misunderstandings is that the Sámi and the recreational snowmobilers, hunters, fishers, and indeed the majority of Swedish society, come from completely different ontologies. For the snowmobilers, nature is something that sits out there and waits for them to come and get enjoyment out of it, as enshrined in Sweden's and Norway's Freedom to Roam acts which allow people to ski, cycle, walk and camp on almost any land. It is a relationship where the people have intrinsic rights to come and enjoy, but have little responsibility or connection otherwise to the wildlife which is ‘out there’. For the herders on the other hand, nature is a practical space which they are constantly part of. Sometimes they do indeed get enjoyment out of it, but they also rely on the natural environment for the survival of their livelihood so the way that they use and alter it bears a lot of responsibility if the ecosystem is to remain healthy for long term use.

This is perhaps better exemplified by the whole concept of wilderness. In a discussion with a reindeer herder the author mentioned the term ‘wilderness’. He laughed, saying that the forests and mountains around us are no ‘untouched wilderness’. He said he could point out to me all the ways that the landscape had been altered and manipulated over the centuries by the Sámi, from the intense trampling of reindeer in milking grounds of the past causing changes in patches of plant communities, to the low number of predators from selective hunting, to even the completely different forest structure seen today due to logging. The landscape and the humans on it are completely entwined. This is not necessarily good or bad but simply a statement of reality. A more extensive discussion of this ‘wilderness myth’ can be found in Fletcher *et al.* (2021). However, for many recreational landscape users the forests of Sweden are a great untouched wilderness and an escape from the urban centers, to a separate natural world. These different ontologies allow misunderstanding and mistrust to develop. If the land is inwardly seen in different ways, the way this expresses itself in outward actions toward the land and other land users can lead to allowances being given to one group over another that can be seen as unfair.

1.11 Lessons from Others

When trying to find ways to create successful dialogue between these different land users, it may be helpful to examine the successes and failures that have happened in other co-management attempts around the world. A failure to address differences of ontology has been seen as a significant contributor to misunderstandings and failure of multiple co-management programmes between Indigenous and non-Indigenous peoples (Blaser 2009; Ojha *et al.* 2016; Pauwelussen and Verschoor 2017). As Blaser (2009) puts it: “the modern world or ontology sustains itself through performances that tend to suppress and or contain the enactment of other possible worlds”. Examples of this can be seen in the case of the

Sámi also, such as the Sámi having the opportunity of land rights but only if they can prove their land usage according to the dominant culture's system of bureaucracy as explored before. This is also seen in the way that situations have been expressed, such as the vegetation shifts from reindeer grazing being quickly labelled as environmental destruction, and limiting of recreational snowmobiling in grazing grounds being labelled as unfair land rights grabs, whilst the less dominant narratives of the reindeer herders have largely been ignored (Johnsen *et al.* 2015).

These biases from the current monopoly of state and industry over the reindeer herders has led to marginalisation and the disputes we see today, and it is clearly not ideal. However other examples warn that the pendulum should not be allowed to swing too far in the opposite direction. The role of Indigenous peoples as environmental stewards has been widely discussed within anthropology, with some supporting the idea that Indigenous people should be looked to as the standard as across the planet they have been inherently sustainable in their use of natural resources, and this has only been ruined by the arrival of colonial people in their lands (Stoffle 2005). Others state this is a myth, and rather that many Indigenous groups have caused environmental degradation and localised extinctions of flora and fauna (Krech 1999). Others still state that the reason for past positive relationships between Indigenous people and the environment has not been from design but rather from situation such as Hames' (2007) idea of epiphenomenal conservation, which "is a consequence of a human population's inability to cause resource degradation or a simple observation about long-term equilibrium with resources. It may be a consequence of low population density, limited technology, or consumer demand". Evidence is used to back all these claims, and discussed at length in Hames' piece. Perhaps the moderate line discussed in Krech (1999) and Pauwelussen and Verschoor (2017) is best to heed, that Indigenous people do generally have a very good understanding of ecological systems, but the way that they see their role as actors and the actions they choose to take can vary even within a community. This can be seen with the apparent dichotomy of some Sámi fighting the timber industry to save resources for the reindeer, whilst other Sámi are the foresters causing this problem.

As explored earlier, the motivation for involvement in industries and activities that may be deemed harmful to the local ecosystem may stem from economic stressors such as poverty and lack of jobs, a theme common in many rural Arctic locations. Individuals and families increasingly need financial capital to survive in a capital driven world, which can be gained from extracting resources from their natural environment. Harm caused by Indigenous populations towards local ecosystems today, then, could in many cases be linked to the legacy of colonisation and sudden cultural shifts from more subsistence to more capital driven lifestyles. Like with any community there is heterogeneity within, and thus to leave the decisions of land use entirely up to the Indigenous peoples of an area simply because they are ’Indigenous’ may not be a reliable route to sustainable resource use.

Similarly, the role that science plays in these discussions must be approached mindfully. Science at its core is a method of enquiry which produces results which are testable by repeating the experiments undertaken. As such it is an objective and impartial tool which can overcome differences in ontology. Of course, as a tool it does run some risk of bias depending on who is using it, and there are indeed examples of scientists arrogantly and wrongfully imposing ‘solutions’ on Indigenous populations, ignoring ‘unscientific’ Indigenous knowledge and leading to negative consequences both socially and for the particular habitat they were trying to ‘save’ (Albert 2001). A greater encouragement of the education of Indigenous people in the scientific method so they can utilise or even identify more ontological biases within the way it is undertaken is important, and the science generated can help to find invaluable impartial insights and assist discussions on how land can best be utilised for all land users. There has already been success with this Sámi-science collaboration in Finland looking at salmon rewilding, which succeeded because, as one Sámi woman said, the researchers were willing to take the Sámi seriously (Quinn 2019).

1.12 Any Chance of Success?

Enrique Leff (in Escobar 1998) states that creating a better alternative plan or framework for sustainability to those unsuccessful ones trialled so far depends on:

> *“… the functional structures of ecosystems that sustain the production of biotic resources and environmental services; on the energetic efficiency of technological processes; on the symbolic processes and ideological formations underlying the cultural valorization of natural resources; and on the political processes that determine the appropriation of nature”*

Strong influence must be allowed to come from reindeer herders as well as industry and state within these conversations. For industries, with their rights controlled by the state, to strike equitable deals with the Sámi would likely mean giving up some opportunities for profit, and for Sweden as a nation it would mean formally and widely admitting to a history of colonialization within its own borders. It would also require a shift in mindset in which one accepts that one's own world view is not the only or even necessarily the right one. There is clear reluctance to do this. Proposals were put forward in the 1980s to give the Sámi greater control over decisions of land use in areas where they pasture their reindeer, but it was put down by the national parliament, who instead stated that the Sámi must be consulted on land use, but that any recommendations or concerns they raise do not actually need to be acted on, leaving any consultations a token gesture rather than something with the potential for impact (Korsmo 1993). This suggests that colonisation is not a historical but an ongoing situation today. Yet as support for the Sámi grows in legal questions of land rights and in the growing sympathy towards this group in general media, the ability of state to ignore the voices of their Indigenous peoples may be in decline.

Untangling the suspicions, ill-feelings and fears of power and economic loss from discussions on land use when looking to the future is by no means a small task. However, there are some examples where collaboration has been somewhat successful. One of these examples can be seen in a study by Riseth and Vatn (2009) which compared the condition of two reindeer herding communities in Norway, one in the south and one in the north. The southern one had managed to maintain sustainable reindeer numbers over a long period of time whilst the other northern one was struggling with overpopulation. There were two main reasons for these different trajectories. The first was that the number of reindeer herders and size of reindeer herding units were much smaller in the south allowing for easier organisation, whereas the need to successfully coordinate was more critical in the north with its wider areas of landscape and greater limitations on pasture use. The second, and more crucial to this discussion, was the reindeer herders' reaction to increased state involvement in the herding livelihood. In the south subsidies from the state were used to increase productivity through consultation with science, rather than simply increasing herd size, and opportunities were seized to work within new policies. These policies were thus promoted with the Sámi communities allowing them to develop their ability to consult through greater experience. In the north however subsidies were used to increase the number of reindeer, perhaps as insurance or as a status symbol, and policies of reducing herd size imposed by the state were ignored and seen as interference.

As seen from the south, successful policies can be developed which lead to sustainable land use. The different situation in the north simply highlights that a blanket set of policies applied to different environments, even when working with the same livelihood and ethnic group, may not be appropriate. It is imperative then to know the exact problem and context, in order to find the most suitable solution.

Perhaps the simplest route to achieving this, and increasing understanding of one another's ontologies, is simply through education. A more widespread understanding of Sámi culture and their historical relationships with the dominant culture is key to help dispel racial and cultural ignorance. Examples of this have been seen in Truth and Reconciliation Commissions in Canada and South Africa, which have helped to repair some feelings of ill-will and invisibility. This education would be best coming primarily from within the Sámi community, with support from others, to ensure that they can relate their history through the intimate knowledge of first-hand accounts rather than relaying perceptions of their culture developed from those outside of it.

Secondly, understanding the needs of reindeer herding in each area, as well as those of industry, is important so that both can be taken into consideration. This requires an increase in ecological literacy across the board within policy and decision making groups, to allow the limited resources in Sápmi to be equitably and sustainably used in the future, and will thus require collaboration between herders, state, industry and the scientific community in order to create a reliable and accurate narrative of the situation. Howitt and Suchet-Pearson (2006) raise the point that capacity building of *all parties* is important to preclude successful discourses, stating:

> *"Power relations shift as Eurocentric 'capacity builders' become accountable to the local in terms of recognizing their own deficiencies (and their own need to have their capacity built), the impacts of their work (often colonizing in effect) as well as the strength of their work in delivering new capacities in changing circumstances. If coming from Eurocentric discourses, capacity builders need to be aware of their own biases, especially in regard to taken-for-granted concepts such as conservation, management, wildlife, development, democracy, representation, gender and education."*

This highlights what is perhaps obvious in theory but absent in practice, namely that consultations and discussions about land use and management need to be a two-way process, where each party must try to understand the needs, strengths, struggles and ontologies of one another. As reindeer herders today already live life with one foot in their traditional ontology and one in the ontology of the dominant culture, it is perhaps even more imperative that the industries, environmentalists and state representatives from the dominant culture approach the situation in a posture of learning. In return the Sámi must find ways to relay their cultural knowledge to the scientific communities and those involved in discussions, whilst the scientific community must aim to increase the ecological literacy of those involved in discussions as well as being self-aware of their own potential biases when carrying out research. This education should ideally extend to the Nordic societies at large so that the young people today will be armed with the necessary ecological and cultural literacy to be effective collaborators in the future.

Previous discussions have been tainted with being one sided on the part of industry and state, and a common thread is the lack of visibility and understanding about the Sámi and reindeer herders, which is perhaps sometimes wilful ignorance. However, armed with accurate and relevant knowledge, more fruitful collaborations can be created from which appropriate future steps can be developed. These future steps may require compromise from all parties, and will likely require this more so as the effects of climate change create greater strain on landscapes and lives across the planet. Starting from a firm foundation of knowledge, mutual respect and equal opportunity to contribute to consultation, the forums in which these discussions take place will be far better equipped to deal with the conditions on our planet today, and the changes to our social and environmental worlds in the future.

1.13 Conclusion

This chapter has examined the complex social and political environment within which reindeer exist, including how they and the lands around them are governed and managed. This context has important implications for the ability of these animals to adapt and respond to change, affecting their survival in the long run. Next, to understand reindeer ecology in this ecosystem, we will examine what these changes are that they are experiencing. Chapter 2 will explore the physical changes to landscape from a competing land user, silviculture, alongside how this affects reindeer grazing. Chapter 3 will pair the impacts of silviculture within the impacts of climate, and chapter 4 will continue to embrace complexity by examining the cumulative impacts of multiple forms of land use, alongside climate scenarios, on reindeer survival. The human influenced potential for adaptation by reindeer to all these changes, as we started to explore in this chapter, will run as a thread throughout these later chapters.

After concluding our results of this study, the chapter 5 will go on to situate this information within the knowledge system within which it was generated, namely western academic science. The importance of traditional ecological knowedge as a way of understanding our environment will be discussed, alongside important reflections on the benefits, challenges and considerations natural scientists should be aware of when wishing to bring together these two ways of examining and understanding. A conceptual framework of the key themes of this thesis can be seen in Figure 1.5 below.

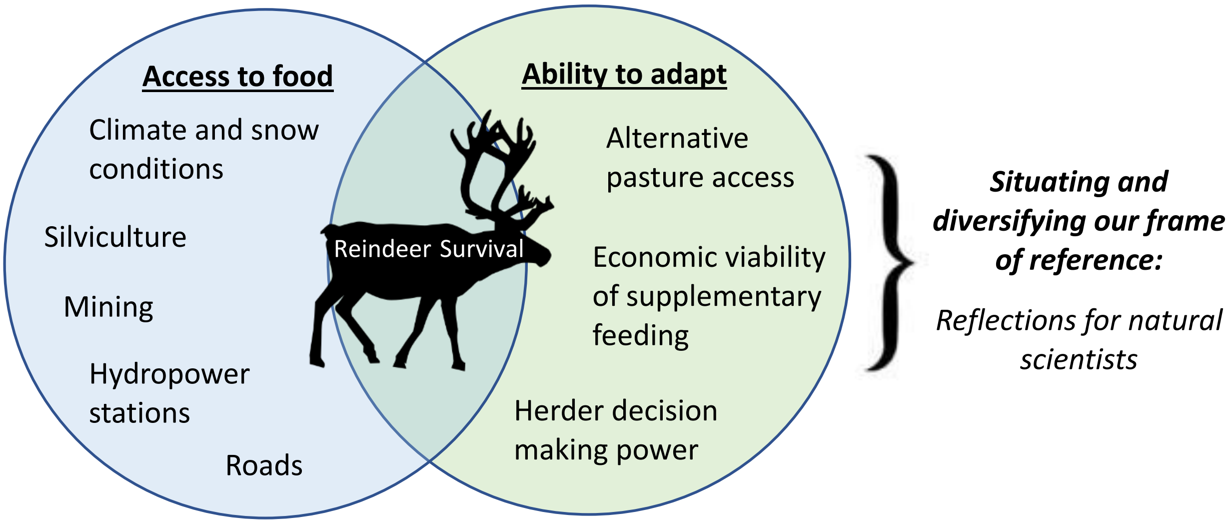

Figure 1.5. Conceptual model showing the key features of this thesis and how they interact with the core topic of we understand reindeer ecology in a changing system.

Chapter 2:

Silviculture, understorey community composition and the recovery of lichen in boreal forests of Northern Sweden

"Old forests are diminishing and the average age of trees has fallen… due to the new forest law that allows companies and private forest owners to clear-cut forests that are not fully grown. I have enjoyed being in the forest for many, many years since I started to follow my father when he went to see his herd, but now I have noticed changes. The trees are not as green as they used to be –the colours are different– and it feels to me like they are saying that they don't feel so well."

–Stefan Mikaelsson, former vice-president of Sámi Council

(Helander-Renvall and Mustonen 2004, p. 262)

2.1 Introduction

In recent years the number of reindeer has declined in multiple locations worldwide, and the wild population of the species is regarded as vulnerable by the International Union for Conservation of Nature or 'IUCN' (Gunn 2016; Uboni *et al.* 2016; Van Lanen *et al.* 2018). Many populations of semi-domesticated reindeer (*Rangifer tarandus*, L.) in Fennoscandia appear to have remained relatively stable in number, but they too have been experiencing increasing pressures. Pastures rich in lichen, an important forage for reindeer, have decreased by 70 % between 1953 and 2013 in Sweden (Sandström *et al.* 2016), with this loss accelerating over time (Horstkotte and Moen 2019). Additionally, herders in the region who normally leave the animals to find their own food during winter are having to purchase supplementary feed more regularly (Persson 2018; Axelsson-Lonkowski *et al.* 2020), indicating that the animals are increasingly unable to gain sufficient forage from their environment.

One source of great food stress for reindeer is the winter period itself. The environmental conditions during this season are not conducive to plant growth, meaning the abundance and diversity of forage species is low. Some lichens such as *Cladonia rangiferina* (L.) Weber ex F.H.Wigg. (1780), do remain available during this period and can become an essential food source, making up to 80 % of the reindeer diet (Heggberget *et al.* 2002). Reindeer are well adapted to their environment and are able, in winter, to detect lichen through up to 91 cm of soft snow (Helle 1984). However, moving and digging through this deep snow has a high energetic cost which may not always be proportional to the energy gain from the forage beneath, also contributing to the food stress of this season. Furthermore, if the snow hardens into ice, this layer may be too hard for the reindeer to break through, preventing them from accessing any food at all (Lie *et al.* 2008; Axelsson-Linkowski *et al.* 2020).

Altogether these winter conditions present significant barriers between reindeer and their ability to access sufficient food, yet whilst icing events are recognised as problematic by reindeer herders, some have stated poor winter conditions are not their main concern. Rather, they state that the root cause of many of today's pressures in herding in Fennoscandia is their reindeer's lack of ability to adapt to these poor conditions as they have been able to in the past (Rees *et al.* 2008).

2.1.1 Anthropogenic Impacts

Historically, reindeer coped with unfavourable grazing conditions by simply moving to new pastures where grazing was possible. However, increasing industrial development in the north has led to loss and degradation of grazing land, with one study in Norway identifying a 70 % loss of undisturbed reindeer habitat within the last century (Nellemann *et al.* 2003). These sources of disturbance include tourist sites (Helle and Särkelä 1993; Nellemann *et al.* 2000; Vistnes and Nellemann 2001; Anttonen *et al.* 2011; Helle *et al.* 2012; Panzacchi *et al.* 2013), towns (Anttonen *et al.* 2011; Polfus *et al.* 2011), mines (Wier *et al.* 2007; Boulanger *et al.* 2012; Hermann *et al.* 2014; Garry *et al.* 2018; Eftestøl *et al.* 2019), hydroelectric dams (Mahoney and Schaefer 2002; Nellemann *et al.* 2003; Hermann *et al.* 2014) and windfarms (Skarin and Åhman 2014; Skarin *et al.* 2018). Their impacts include both physical loss of pastures due to construction of infrastructure and the flooding of dam reservoirs, as well as noise and general disturbance discouraging reindeer from approaching nearby areas to graze. The construction of roads and railways has also been seen to fragment reindeer migration routes, and kill the animals in collisions (Wolfe *et al.* 2000; Dyer *et al.* 2001; Vistnes and Nellemann 2001; Lundqist 2007; Sorensen *et al.* 2008; Vistnes *et al.* 2010; Anttonen *et al.* 2011; Panzacchi *et al.* 2013).

This loss of pastures from infrastructure or reindeer avoidance has been widely noted in the literature. However, not all anthropogenic activity involves sustained human presence or complete removal of the vegetation community. Forestry, a key industry in many reindeer herding areas in northern Sweden, is one example of this.

2.1.2 Forestry

During the critical winter period in Sweden, reindeer tend to migrate to lowland boreal forests to graze until spring, making the state of these forests highly influential to reindeer survival (Käyhkö and Horstkotte 2017). As reindeer herding has been a traditional livelihood with deep cultural connections to the Indigenous Sámi, reindeer, and reindeer herders, have special customary usage rights of public and private land, allowing them to roam and graze in commercial forests in Sweden (Rennäringslag 1971). However, individuals and companies who own these forested grazing grounds also have the right to undertake commercial silviculture within these areas, which can alter the landscape in a way that leaves little to no forage remaining for the animals. Therefore, the reindeer still have the right to be present, but these forested areas may be degraded and functionally lost to them (Bostedt *et al.* 2003, Widmark *et al.* 2013). Indeed, Sweden has 23.5 million ha of productive forestry lands, producing 79.1 million m^3 of timber each year (SLU 2018), and in parts of the country commercial logging has caused up to a 50 % loss in reindeer winter pasture (Berg *et al.* 2008; Kivinen *et al.* 2012; Korosuo *et al.* 2014).

Degradation occurs in a number of ways: A large initial source of disturbance and degradation is the mechanical processes used in silviculture. Clear-cutting, where almost all trees are felled at a site, utilises heavy machinery such as harvesters and forwarders to fell, delimb and remove the logs. Equipment can weigh up to 20 metric tonnes, and is often driven over the same tracts of ground repeatedly. During replanting, scarification is standard practice. This is a process where soil is tilled, or turned to improve new seedling growth (Saursaunet *et al.* 2018). The soil disturbance created during clear-cutting and scarification are known to be detrimental to lichen growth and survival (Roturier and Bergsten 2006, Korosuo 2014; Vanha-Majamaa *et al*. 2017). In some cases, scarification has been seen to cause an up to 70 % reduction in lichen cover and biomass even 15 years after soil preparation has taken place (Roturier *et al.* 2011), although in other sites this decrease in lichen has been less dramatic (Roturier and Bergsten 2006). This in turn has led to reindeer showing reduced grazing in these areas (Eriksson and Raunistola 1990; Roturier and Bergsten 2006).

Of course, not all logging methods are detrimental to grazing. Thinning, where only a few trees, or parts of trees, are removed and the overall canopy remains intact, uses smaller machines as space and freedom of movement is limited by the remaining trees. This results in less disturbance, especially when handheld machinery is used, and can encourage lichen growth on the remaining trees (Stevenson 1986; Korosuo 2014). Harvesting trees during winter or using gentler forms of scarification compared to the conventional disc trenching can also significantly both reduce the loss of lichen and increase its rate of recovery from disturbance (Coxson and March 2001; Roturier *et al.* 2011). Additionally, the timing and location of tree harvesting have been scheduled through models in ways that minimise impacts on reindeer, creating corridors where they can travel through and graze as required, whilst allowing tree felling to continue (St John *et al.* 2016).

However, these alternative methods are yet to be widely adopted, argued by some to be uneconomical (Borstedt *et al.* 2003), whilst others within the forestry industry have expressed a reluctance to participate due to opportunity losses created by practices such as preserving certain forests for the benefit of reindeer (Bostedt *et al.* 2003). This disconnect between the needs of industry and reindeer herding have led to significant conflicts between these groups, even with mandatory consultations being in place since 1979 (Widmark *et al.* 2013). However, the potential is there to utilise practices that are beneficial to reindeer and incur little cost to the forestry industry (Bostedt *et al.* 2003; Sandström *et al.* 2006).

Aside from these initial impacts on the landscape, commercial logging has some more long-term effects upon reindeer grazing. A clear by-product of forestry is a reduction in old growth forests as areas are felled and replaced with young plantations (Kivinen *et al.* 2010; Horstkotte *et al.* 2011; Kivinen *et al.* 2012). Old growth forests are widely recognised as being optimum for grazing as they contain a higher abundance and diversity of lichens compared to younger stands (Lie *et al.* 2009; Horstkotte *et al.* 2011; Kivinen *et al.* 2012). The structure of old forests also gives reindeer greater opportunity to access forage through snow in winter. Older forests have multi-layered canopies from the diversity of tree ages and species present, unlike the single layered canopies often seen in younger commercial monocultures (Horstkotte and Roturier 2013; Korosuo *et al.* 2014). This canopy diversity allows snow to be trapped in branches at varying heights which is subsequently dropped at varying times over winter, leading to a patchy snowpack on the forest floor (Roturier 2011; Horstkotte and Roturier 2013). In a single layered stand, the homogenous snow pack means that the site will either be entirely accessible or inaccessible, whereas the variability within old-growth forests allows it to be used in a wide variety of snow conditions and thus allows the reindeer greater opportunities to adapt to adverse winter weather.

As seen above, much is already understood about the immediate effects of logging, as well as the benefits of forests once they have formed a mature stand. However, a large proportion of Sweden's forests are now at an intermediate stage where trees have grown back after clearcutting but have not yet attained the structure of old-growth forests. The processes of change and recovery of forage vegetation occurring in forests at intermediate stages of regrowth remain largely unstudied, especially in the important winter grazing areas, other than to establish broad characteristics of lichen and forest structure (Akujärvi *et al.* 2014; Sandström *et al.* 2016; Uboni *et al.* 2019). More detailed information in this area is therefore required.

2.1.3 Aim

The aim of this chapter was to examine how understorey plant communities in the boreal forests of northern Sweden recover from the processes of silviculture from the stage of being clear-cut towards becoming mature stands, and how the changes occurring during this process of recovery affect reindeer grazing. This was achieved by comparing the understorey vegetation community composition in boreal forests at various stages of regrowth, paying particular attention to lichens and other plant species that contribute to the diet of reindeer. The hypotheses are as follows:

1. Lichen biomass in forests increases as stands increase in maturity after clear-cutting.
2. The community composition of plants adjusts according to the processes of succession as forests progress from clear-cut to mature stands, beginning with rapid colonisation and eventually being restricted by competition.

The data collected contribute towards a greater understanding of the opportunities and challenges faced by reindeer and reindeer herders across a complex and multidimensional landscape, as well as providing valuable insights into boreal forest ecology and vegetation recovery from disturbance.

2.2 Materials and Methods

2.2.1 Study Area

Data collection was undertaken within an area of approx. 150 km^2 near Jokkmokk in Norrbotten, Sweden at 66°36'23" N and 19°49'23" E (see Figure 2.1). This work was done during June-July 2019. The research area forms part of the winter grazing grounds of three mountain Sámi siidas (herding groups) called Tuorpon, Sirges and Jåkkåkaskatjiellde, and one smaller forest Sámi siida called Slakka. The topography is undulating hills at *ca.* 250 m above sea level, and the Lilla Luleälven river runs through the area. Average temperatures range between -14.6 ^{o}C in January, and 14.5 ^{o}C in July (Norwegian Meteorological Institute 2019), though can vary widely. For example, in 2019, when there were temperatures of -35 ^{o}C in February and 28 ^{o}C in June. Mean annual precipitation is 475 mm (Climatemps 2017).

Figure 2.1. Map showing the approximate location of Jokkmokk (denoted by black star), alongside satellite imagery of the surrounding area. Scale of satellite imagery is 1:500000. (Map: Wikimedia Commons/CC BY-SA 3.0, Satellite imagery: Map Data ©2021 TerraMetrics)

The landscape is dominated by Scots pine (*Pinus sylverstris* L.) forests, interspersed with Norwegian spruce (*Picea abies* (L.) H. Karst.) and birch (*Betula* L.). Tree composition at sites measured was >95 % Scots pine. The understorey is characterised by shrub species such as *Empetrum nigrum* L., *Calluna vulgaris* (L.) Hull and *Vaccinium* spp. over a bed of *Pleurozium schreberi* (Brid.) Mitt and *Dicranum scoparium* Hedw. Lichens are prevalent in the area, mostly consisting of *Cladonia arbuscula* (Wallr.) Flot., and *C. rangiferina* (L.) Weber ex F.H.Wigg., although *C. pyxidata* (L.) Hoffm., *C. stellaris* (Opiz) Pouzar & Vezda, *Stereocaulon* spp. and *Nephroma arcticum* (L.) Torss. are also present. Arboreal lichen species *Hypogymnia physodes* (L.) Nyl. and *Bryoria fuscescens* (Gyeln.) Brodo & D.Hawksw. are present in some forest stands.

Commercial forestry is widespread, with data from locations within 70 km showing on average 19 % of the forests are >120 years old (Berg 2008; Horstkotte *et al.* 2011), with the remaining 81 % having experienced recent felling. Sweden's largest forestry company Sveaskog Förvaltnings AB operates within Norrbotten County, routinely using soil scarification to prepare clear-cut areas for replanting (Bureau Veritas Certification 2020). Visual evidence at sites showed that when clear-cutting occurs a small number of mature trees are left untouched, and tracks at clear-cut sites indicated the use of heavy machinery during tree felling and/or processing. Treatments used by multiple smaller forest owners may vary locally.

2.2.2 Vegetation Community Composition

Within the study, 16 forest stands were sampled using a stratified random approach, four stands being identified for each of the four maturity classes: 'clear-cut' (≥ 90 % of trees felled); 'young' (trees 0.5 m to 2 m in height); 'intermediate' (trees > 2 m in height, arboreal lichens absent); and 'old' (trees > 2 m in height, arboreal lichens present). The classification of old forests is based upon the finding that arboreal lichen only grows on trees over the age of *ca.* 63 years (Horstkotte *et al.* 2011). It should be noted that 'old' forests here have the monoculture and organised structure of commercial forests, as opposed to the tree diversity, abundance of deadwood and uneven tree placement found in virgin forests, indicating they have been artificially planted. Trees in old forest stands had a mean girth at breast height of 59.5 ± 2.9 cm, intermediate stands 19.6 ± 2.8 cm and young stands 7.3 ± 0.4 cm.

To minimise potential unwanted edge effects, sampled stands were > 20 m from a road or dwelling. Sites were aged according to dendrochronology procedures from Grissino-Mayer (2003), using a Mattson No. 4. Increment Borer (Sorbus International, Somerset, UK). Cores were extracted from four randomly chosen trees at breast height, or 50 cm if trees were below breast height. The annual growth rings were then counted by eye for each with an average of the four ages used. Old stands sampled had a mean age of 86 with a standard error (SE) of ± 13 yrs., intermediate stands 23 ± 4 yrs. and young stands 7 ± 1 yrs. Due to lack of trees in clear-cut forests, they were deemed to be zero years of age.

Species area curves were constructed for each forest maturity class, and 200 x 200 cm ($4\ m^2$) was deemed to be the most suitable quadrat size for subsequent understorey vegetation surveys (see Figure 2.2). At each site quadrats were randomly placed > 20 m apart, and the percentage cover of each plant species present was estimated by the lead author only, for consistency. Cover of bare ground and plant litter was also noted. Plants > 50 cm in height were not included so as to differentiate between understorey and overstorey.

Plant height was measured at five random points within each quadrat and a mean calculated. Measurements were taken from the soil surface, or if on moss beds from the point at which the moss started to turn brown (i.e., the point below which photosynthetic material was not present). Plants > 50 cm in height, regarded as overstorey, were not included. Lichen on windthrown branches within a quadrat were also ignored at this stage. This vegetation survey was repeated 10 times at each site.

The plant species present were classified as either 'edible' or 'non-edible' based upon whether reindeer are found to commonly eat them. The edible species were *Vaccinium vitis-idaea* L., *V. myrtillus* L., *Pleurozium schreberi* (Brid.) Mitt. , *Cladonia rangiferina* (L.) Weber ex F.H.Wigg., *C. arbuscula* (Wallr.) Flot., *C. stellaris* (Opiz) Pouzar & Vezda, *C. pyxidate* (L.) Hoffm., *Empetrum nigrum* L., *Deschampsia flexuosa* (L.) Trin., *Nephroma arcticum* (L.) Torss. and *Stereocaulun spp.*, and the non-edible were *Dicranum scoparium* Hedwig, *Calluna vulgaris* (L.) Hull, *Rhododendron tomentosum* Harmaja, *V. uliginosum* L., *Polytrichum* *juniperinum* Hedwig and *Diphasiastrum complanatum* (L.) Holub. (according to Nieminen and Heiskari 1989; Klein 1990; Danell *et al.* 1994; Inga 2007). It should be noted that there is a variability of preference in edible species, e.g., reindeer are not strongly inclined towards eating *V. myrtillus, Pleurozium schreberi* and *Empetrum nigrum,* but will do so when other more palatable species are unavailable (Danell *et al.* 1994).

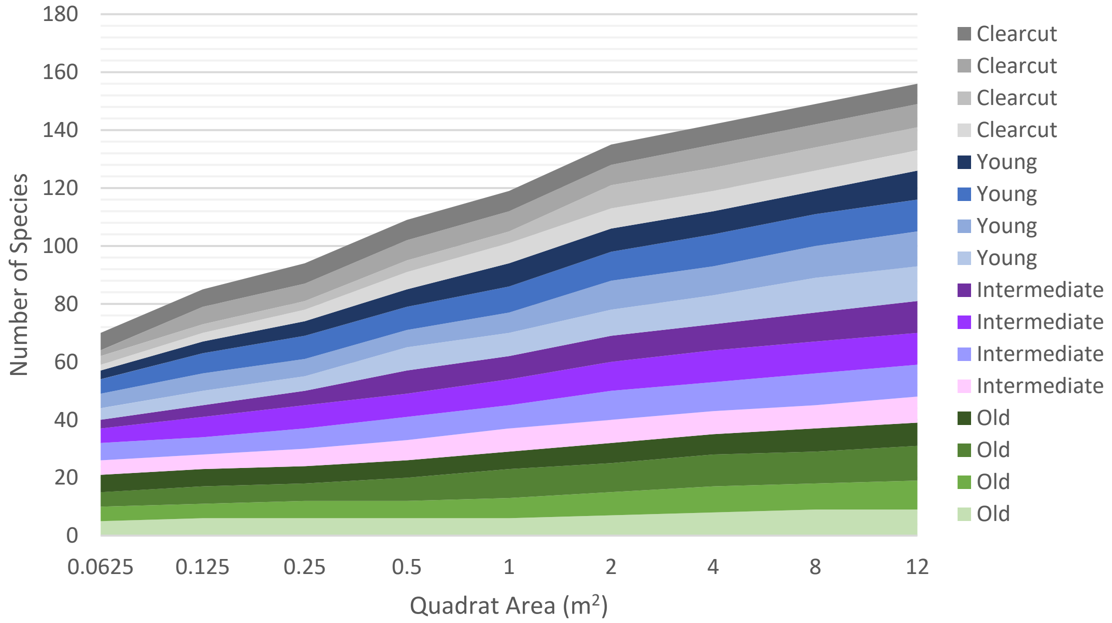

Figure 2.2. Species area curves for four replicates at each site maturity class. Species number within quadrats of increasing size are shown

2.2.3 Tree Density and Canopy Cover

Forest tree density and size were measured at all 16 sites using the point centred quarter method. At approximately 20 m along a transect, an object was thrown at random to determine a central point. The four quarters about this point were delineated based upon the cardinal points of the compass. The distance to the nearest tree from the central point within each quarter was measured in meters (*d*) and girth (*G*) of the tree at breast height (standardized at 1.4 m above ground level) measured in centimetres. Tree species was also noted. Five replicate point–centre plots were sampled per forest site.

The square root of the mean area (m^2) occupied by a tree ($\bar{d}$) was calculated using the formulae of Cottam *et al.* (1953), where q is the number of quarters, in this case being 20 per site:

$$\bar{d} = \frac{\sum d}{q}$$

Then the total tree density as trees per hectare (D_T):

$$D_T = \frac{10{,}000}{\bar{d}^2}$$

Tree basal area (*bA*) was calculated as:

$$bA = \frac{G^2}{4\pi}$$

Canopy cover was measured using a spherical densiometer (Spherical Crown Densiometer, Convex Model A, Forestry Suppliers, USA) according to methods in Werner (2019). Five replicate readings were taken per site, and measurements were made in all four of the 'young' sites plus three each of the 'old' and 'intermediate' sites as four could not be obtained due to access issues. Clear-cut areas, having no canopy, were not measured.

2.2.4 Ground, Arboreal and Branch Lichen Surveys

Within two of the 4 m^2 quadrats at each site described above, five smaller quadrants of 10 x 10 cm (area = 0.01 m^2) were placed in a stratified random manner on areas of higher lichen abundance. In these, lichen cover was noted, along with lichen height measured at three random points, with a mean taken. All lichen within the 0.01 m^2 quadrats was extracted, separated from surrounding soil and vegetation, and dried to a constant weight at 105 ^oC, after which it was immediately cooled and weighed (Satorius M-prove, Satorius AG, Goettingen, Germany) following Kumpula *et al.* (2000a). Mean lichen dry biomass across all quadrats within a site was then calculated as kg ha^{-1}. Quadrats used here (0.01 m^2) were smaller than those in Kumpula *et al.* (2000a) at 0.25 m^2 to allow for a greater number of replicates within the forest in the limited sampling time available.

Arboreal and branch lichen was only measured at 'old' sites as they were absent in other forest maturity classes (by definition). In each of the four 'old' forest sites, a tree was chosen at random, and its diameter measured at breast height (1.4 m). Lichen on the tree up to the height of 3 m collected, this being a combination of reindeer browsing height (<1.5 m according to Kumpula *et al.* 2011) and comments from local inhabitants that winter snow depth rarely exceeds 1.5 m. Arboreal lichen that had fallen onto the ground, termed 'branch lichen' as sometimes still attached to branches, was collected within a 4 m^2 quadrat adjacent to each of the five trees per site. Arboreal and branch lichen were dried and weighed using the same method as for ground lichen.

2.2.5 Statistical Analyses

All statistical analyses and generation of graphs were undertaken using the statistical software R, version 4.1.0 (R Core Team 2020). This included the packages FSA, gplots, ggplot2 and plotrix. Shapiro-Wilk normality tests were carried out to verify whether data was parametric. The impact of stand maturity class was measured for the following variables: lichen biomass, lichen height, lichen cover, understorey height, percentage cover of species edible and not edible to reindeer, percentage ground cover of plant litter, percentage cover of bare ground, and canopy cover. When data was normally distributed, variation in factors across sites was tested using a one-way ANOVA, followed by a Tukey's post-hoc test. For data that was not normally distributed, a Kruskal-Wallis test was carried out, followed by a Dunn's test. To identify any confounding effects, a scatterplot matrix of correlations for the above variables was constructed. A factorial ANOVA was then carried out to identify combined effects of other variables on lichen biomass. Data were nested by site to avoid pseudoreplication. A factorial ANOVA was constructed to identify relationships between lichen biomass and the following factors: canopy cover, understorey height, litter cover and bare ground cover. These factors were chosen as they were all the factors with parametric data.

2.3 Results

2.3.1 Forest Understorey Community Composition

The average number of plant species per forest site ranged between 7-10 species, differing little between forest maturity classes. Overall, sites were characterised by shrub species such as *Empetrum nigrum, Calluna vulgaris* and *Vaccinium spp*. over a bed of *Pleurozium schreberi* and some *Dicranum scoparium* (Figure 2.3). The grass *Deschampsia flexuosa* was found almost exclusively at 'young' sites. Clear-cut sites had more than double the mean plant litter cover (at 25.0 ± 6.6 %) compared to all older forest classes, which differed little from one another, although due to the spread of the data the difference in clear-cut sites is not significant ($F = 2.29$, $p > 0.05$; Table 2.1). Percentage cover of bare soil and rock was also greatest in clear-cut sites (13.9 ± 5.1 %) and almost non-existent in old sites (1.4 ± 1.0 %), though this pattern was not consistent with increasing forest maturity as intermediate forests had more bare ground than young forests, and this relationship was not significant statistically (*Chi sq.*= 5.19, $p > 0.05$; Table 2.1). Conversely, mean understorey height and percentage cover of species edible to reindeer were greatest in old forests, yet once again due to spread of data these relationships were not significant (Table 2.1; Figure 2.4).

Of the species classed as edible to reindeer, there was no significant difference in percentage cover between site maturity classes (*Chi sq.*= 4.52, $p > 0.05$; Table 2.1; Figure 2.4). However, edible species consistently dominated the vegetation community. The percentage cover of non-edible species decreased with forest maturity age, differing significantly between clear-cut and old sites at 68.7 ± 2.8 % and 44.8 ± 4.8 % respectively (Tukey's $p = 0.017$; Table 2.1; Figure 2.4). Canopy cover of trees increased with forest maturity, peaking in old sites at 70.8 ± 7.0 %. This was significantly higher than canopy cover at young sites (Dunn's Test 0.015; Table 2.1). Clear-cut sites were not included in this pairwise test due to complete lack of canopy.

Table 2.1. Mean and one standard error of factors in forests of differing stand maturity class. Only relationships with significant adjust-P values from post-hoc tests are shown in post-hoc test results. Test statistic is an F value for ANOVAs and a chi squared value for Kruskal-Wallis tests. DF denotes degrees of freedom.

	Clear-cut (C)	Young (Y)	Intermediate (I)	Old (O)	Replicates	Test	DF	Test statistic	P Value	Post Hoc Test	Post-Hoc results (p value)
Lichen biomass (kg ha^{-1})	416.1 ± 128.1	694.4 ± 201.9	870.4 ± 146.8	828.2 ± 81.4	n=16	ANOVA	3	1.97	0.173	-	-
Lichen height (cm)	1.6 ± 0.3	1.8 ± 0.3	1.8 ± 0.3	3.7 ± 0.4	n=16	ANOVA	3	8.0	0.003**	Tukey's	O-C (0.006) ** O-Y (0.009) ** O-I (0.011) *
Lichen cover (%)	20.7 ± 4.7	27.3 ± 5.5	29.3 ± 0.9	24.6 ± 3.9	n=16	ANOVA	3	0.83	0.503	-	-
Arboreal Lichen (kg ha^{-1})	0	0	0	41.5 ± 10.2	n=4	Kruskal -Wallis	3	14.62	0.002**	Dunn's	O-C (0.011) * O-Y (0.011) * O-I (0.011) *
Branch Lichen (kg ha^{-1})	0	0	0	9.6 ± 2.7	n=4	Kruskal -Wallis	3	14.62	0.002**	Dunn's	O-C (0.011) * O-Y (0.011) * O-I (0.011) *
Understorey height (cm)	10.0 ± 1.7	9.6 ± 1.3	9.2 ± 1.1	12 ± 0.9	n=16	ANOVA	3	0.94	0.452	-	-
Edible species cover (%)	92.0 ± 6.7	112.3 ± 25.7	121.1 ± 19.7	118.7 ±7.3	n=16	Kruskal -Wallis	3	4.52	0.2103	-	-
Non-edible species cover (%)	68.7 ± 2.8	55.1 ± 3.7	51.9 ± 6.7	44.8 ± 4.8	n=16	ANOVA	3	6.28	0.024*	Tukey's	O-C (0.017) *
Litter cover (%)	25.0 ± 6.6	9.8 ± 4.2	10.3 ± 4.7	9.9 ± 4.0	n=16	ANOVA	3	2.29	0.13	-	-
Bare ground cover (%)	13.9 ± 5.1	6.0 ± 2.1	8.2 ± 4.0	1.4 ± 1.0	n=16	Kruskal -Wallis	3	5.19	0.1583	-	-
Canopy Cover (%)	0 ± 0	4.6 ± 1.3	19.0 ± 7.4	70.8 ± 7.0	n=12	Kruskal -Wallis	3	12.47	0.018*	Dunn's	O-Y (0.015) *

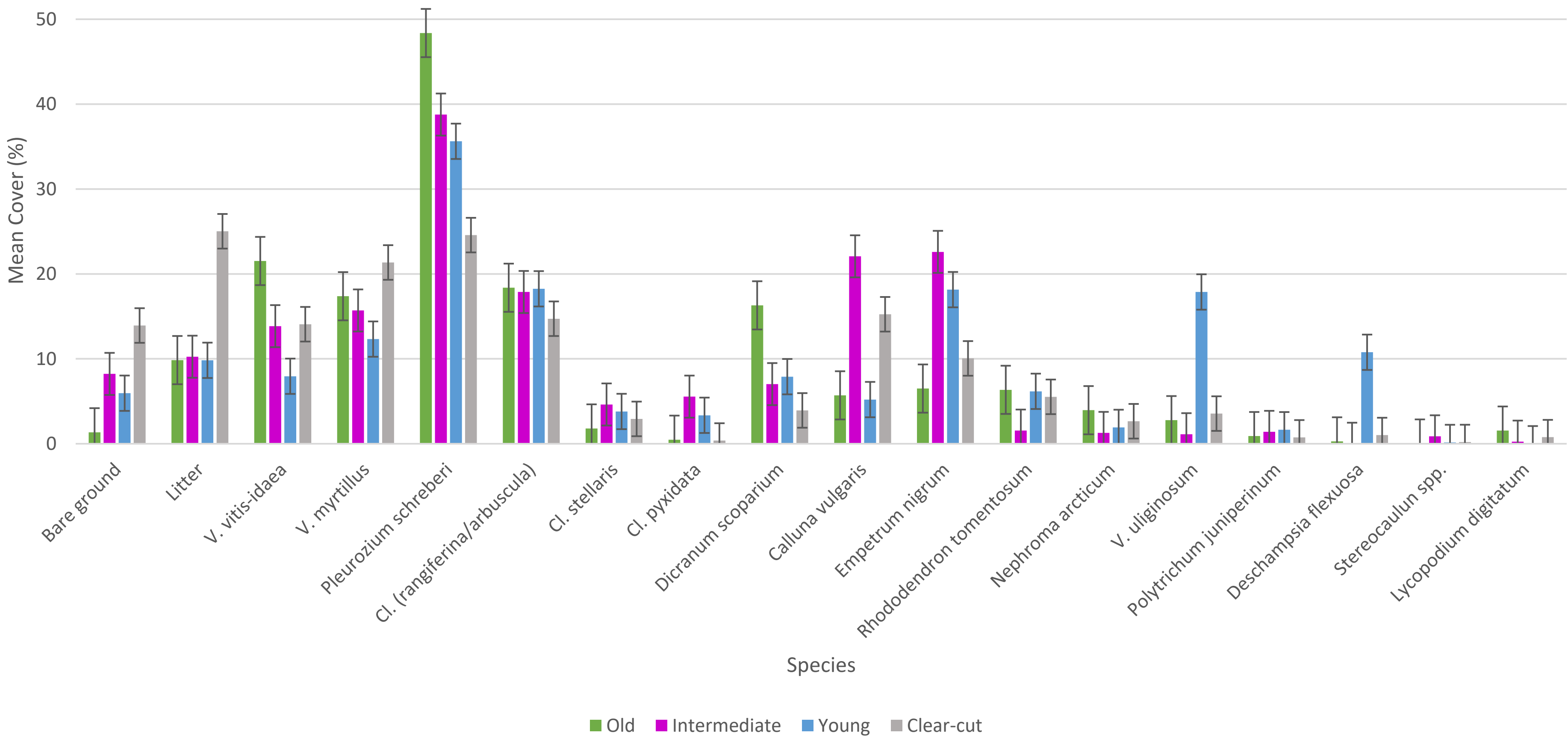

Figure 2.3. Mean percentage and one standard error of ground cover of plant species per quadrat at sites of the four forest maturity classes of old, intermediate, young and clear-cut (n=4). V. denotes Vaccinium, and Cl. Cladonia within the species name.

2.3.2 Lichen Abundance

Terrestrial lichen species present were *C. arbuscul*a, *C. rangiferina, C. pyxidata, C. stellaris, Stereocaulun* spp. and *Nephroma arcticum.* Lichen biomass here is represented as kg ha^{-1} to allow ease of comparison with other studies such as (Akujärvi *et al.* 2014). Mean lichen biomass was greater in intermediate and old sites (at 870.4 ± 146.8 kg ha^{-1} and 828.2 ± 81.4 kg ha^{-1}), respectively compared to clear-cut and young sites at 694.4 ± 201.9 kg ha^{-1} and 416.1 ± 128.1 kg ha^{-1}. However, these relationships were not significant (F= 1.97, p = 0.137; Table 2.1 ; Figure 2.4). Lichen cover also did not vary significantly according to site maturity (F= 0.83, p = 0.503; Table 2.1). Lichen height was significantly greater in old sites at 3.7 ± 0.4 cm when compared to all other site maturity classes (Tukey's, p < 0.05 for all, Table 2.1; Figure 2.4).

Arboreal lichens were found at all replicates of old sites, and none of the younger forests. The species present were *Hypogymnia physodes* (L.) Nyl. and *Bryoria fuscescens* (Gyeln.) Brodo & D.Hawksw. with a mean biomass of 41.5 ± 10.2 kg ha^{-1}, representing 5 % of the total lichen mass present. Lichen mass on wind-thrown branches was 9.6 ± 2.7 kg ha^{-1} representing 1.2 % of the mean site total (Table 2.1).

A factorial ANOVA was undertaken examining how lichen biomass was affected by all parametric variables (site maturity class, understorey height, lichen height, lichen cover, non-edible species cover and litter cover, see Table 2.2). This test showed that lichen biomass is significantly affected by forest stand maturity class (F= 7.74, p < 0.05), understorey vegetation height (F= 17.27, p < 0.01) and lichen height (F= 16.25, p < 0.01). Comparison of effects of each factor on one another can be seen in the scatterplot matrix in Figure 2.5.

Table 2.2. Table summarising the results from a factorial ANOVA where lichen biomass was tested against site maturity class, understorey height, lichen height, lichen cover, non-edible species cover and litter cover. D.F denotes degrees of freedom whilst Pr (>F) denotes the p value of the factorial ANOVA.

	D.F	Sum sq	Mean Sq	F Value	Pr (>F)
Stand Maturity class	3	80680	26893	7.741	0.013 *
Understorey height	1	59995	59995	17.269	0.004 **
Lichen height	1	56456	56456	16.250	0.005 **
Lichen Cover	1	16715	16715	4.811	0.064
Non-edible species cover	1	5756	5756	1.657	0.239
Litter Cover	1	700	700	0.201	0.667
Residuals	10	47490	4749		

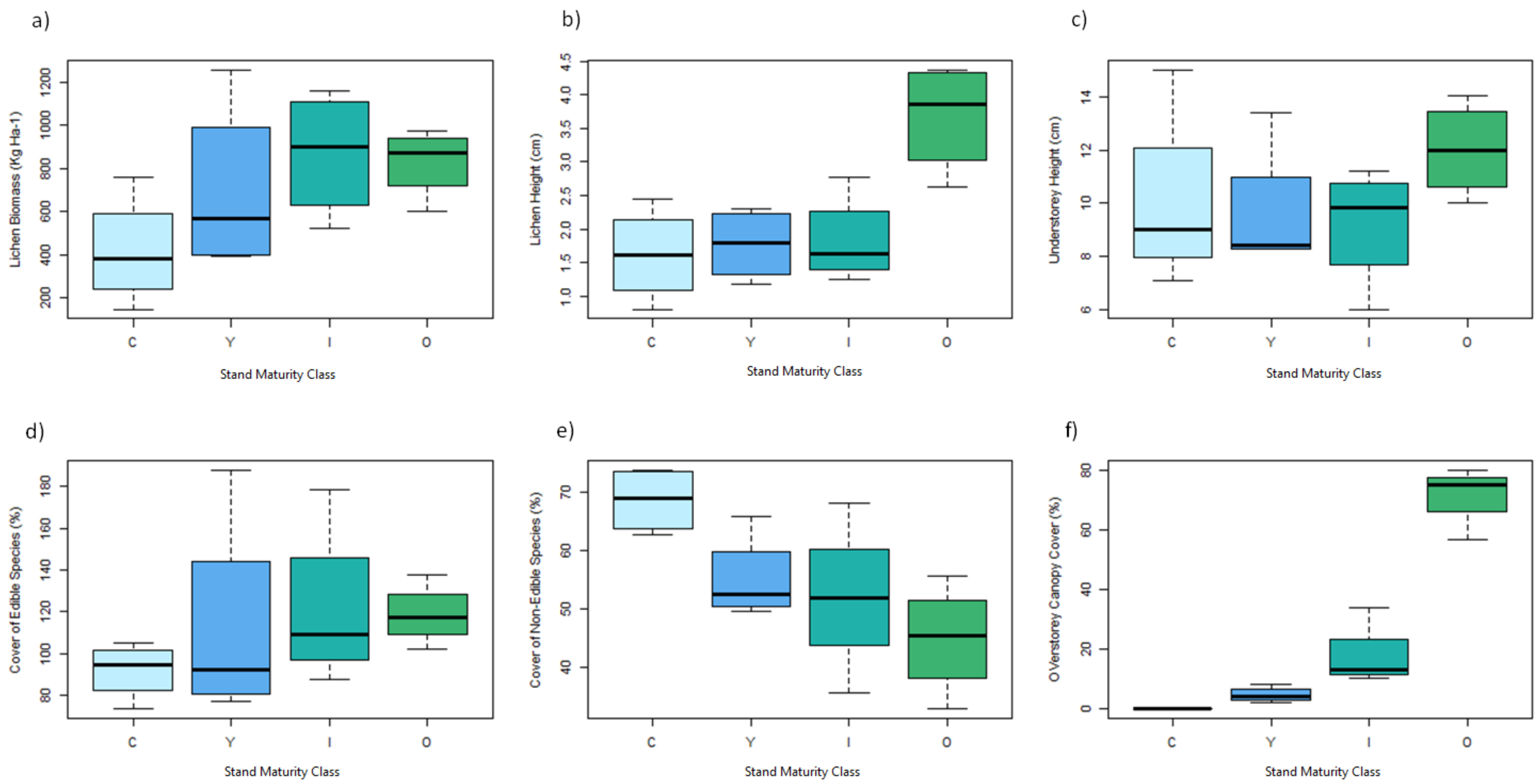

Figure 2.4. Boxplots showing the median, minimum, maximum and 1st and 3rd quartile of data at each of four forest stand maturity classes, with C denoting 'Clear-cut', Y denoting 'Young', I 'Intermediate' and O 'Old' aged stands. Lichen biomass (a), lichen height (b), understorey vegetation height (c), cover of species edible to reindeer (d), cover of species not edible to reindeer (f) and overstorey canopy cover (f) are included.

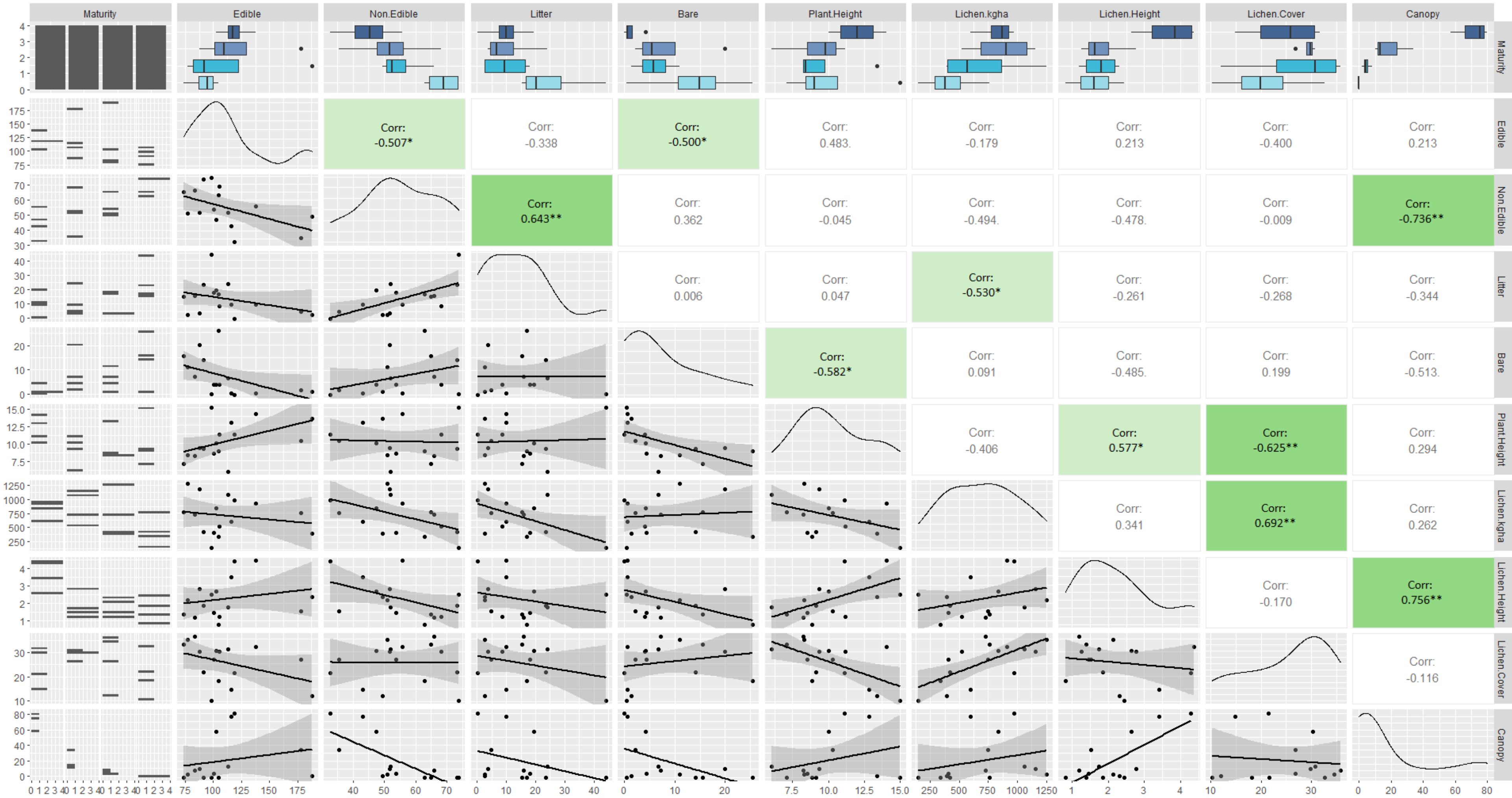

*Figure 2.5. Matrix of correlations for forest stand and understorey vegetation characteristics measured in a boreal forest in northern Sweden during summer 2019. Maturity indicates stand maturity class of 'clear-cut (1), 'young' (2), 'intermediate' (3) and 'old' (4). Edible and non-edible indicate percentage cover of species relative to their palatability to reindeer. Litter and bare indicate ground cover and canopy is measured in percentage cover. * p<0.05; ** p<0.01*

2.4 Discussion

This study shows that the understorey vegetation community in boreal forests are part of a complex system. Here it was seen that stand maturity significantly affects lichen height and the cover of species not edible to reindeer, as well as canopy cover which consistently increased with forest maturity class. Whilst lichen biomass was not significantly affected by forest stand maturity alone, it was significantly affected by a combination of stand maturity, understorey vegetation height and lichen height, indicating some synergstic effects between these factors. These results broadly support the first hypothesis of this study, with stand maturity being one of three interacting factors which cause an increase lichen abundance, although the strength of this relationship would be better studied with greater numbers of replicates of each stand maturity class.

2.4.1 Initial Impacts of Silviculture on Lichens

Stand maturity and height of lichens and other understorey species are are all impacted by clear-cutting and the subsequent process of recovery by the vegetation community. More broadly, clear-cutting is known to be detrimental to lichen growth and survival, due to soil disturbance from machinery during processes such as scarification (Eriksson and Raunistola 1990; Roturier and Bergsten 2006; Korosuo 2014; Vanha-Majamaa *et al.* 2017). Here sites which had most recently experienced felling of trees and preparation of soil had the lowest lichen abundance, and indeed abundance of all plant species when compared to older forests further in the recovery process.

Immediately following clear-cutting, stands were characterised by a higher prevalence of bare ground and disturbed soil. These kinds of stressful conditions give lichens an initial competitive advantage over bryophytes and higher plants, allowing them to recolonise areas from where they were removed during clear-cutting, as well as allowing the lichens to experience rapid growth. Indeed, the greatest increase in lichen abundance occurred between clear-cut and young stands. Additionally, the colonising species *D. flexuosa, E. nigrum* and *V. uliginosum* were seen to take advantage of the disturbed environment (Kershaw and Kershaw 1987; Linder *et al.* 2017), increasing between clear-cut and young forests, though they were present in lower abundances or even absent in old forests perhaps due to being out-competed at this later stage. These data support the second hypothesis, stating that succession was occurring within the understorey community as the forest recovered from clear-cutting. The role of competition within this will be examined in the following section of this chapter.

Past studies suggest that lichens favour areas with reduced canopy cover due to greater access to sunlight (Pharo and Vitt 2000; Uboni *et al.* 2019). However, tree harvests carried out in winter, causing less disturbance to the soil, have been seen to lead to far more abundant lichen communities than tree harvests in summer, where greater soil disturbance occurs (Coxson and Marsh 2001). This again indicates that soil disturbance has a notable effect on lichen growth, greater than any advantages from greater access in open canopies.

Conversely, alternative forestry methods which involve less soil disturbance, such as winter harvest or alternative scarification methods, can significantly reduce both the loss of lichen and increase its rate of recovery from disturbance (Coxson and March 2001; Roturier *et al.* 2011). Other alternative methods such as thinning and carefully scheduling tree harvests to allow for travel corridors for the reindeer have also been discussed in the introduction of this chapter (Stevenson 1986, Korosuo 2014; St John *et al.* 2016). Whilst these alternative methods are yet to be widely adopted, they do show that silviculture can occur in a manner which is less detrimental to reindeer grazing.

2.4.2 The Role of Competition

Following initial disturbance from silvicultural processes, the effects of competition on the recovery of species like lichens is important to consider. It should once again be noted that forests in this study which are classed as ‘old’ have the monoculture and organised structure of commercial forests, as opposed to the tree diversity, abundance of deadwood and uneven tree placement found in unplanted old forests. This indicates that these ‘old’ stands have been artificially planted and so are not the same as old-growth forests, but rather forests which are at a later stage of recovery from silviculture.

In old stands bare ground was almost non-existent and understorey vegetation taller than in younger stands, suggesting plants were experiencing vertical competition for access to light and space. Mean lichen cover even declined slightly between intermediate and old stands, perhaps due to being effectively 'squeezed' inward and upward by competing moss species such as *Dicranum scoparium* and *Pleurozium schreberi* (Sulyma and Coxson 2001). Other less adaptable colonisers such as the grass *D. flexuosa* were eventually outcompeted, whilst lichens maintained a foothold in the mature forests.

Competition has the potential to interact with the processes of silviculture in its effect on understorey vegetation. Whilst increased sunlight through more open canopies is associated with greater lichen abundance (Pharo and Vitt 2000), it has also been seen that thinning mature pine forests, thus opening the canopy, maintains the diversity of ground vegetation but does not actually promote the growth of species important to reindeer grazing (Vitt *et al.* 2019). This could be because despite having access to greater sunlight, the lichens were experiencing too much competition from other vegetation to be able to expand. This is supported by research in the study area which show that regular historic fires would open the canopy in old forests and crucially also suppress some ground-lying vegetation, reducing the competition experienced by lichens and thus encouraging their growth once they have been re-established after the fire event (Östlund *et al.* 1997; Berg *et al.* 2008; Cogos *et al.* 2019). Therefore, the results suggest that the disturbance effects of forestry on lichen and their rate of recovery are additionally dependent on the level of competition experienced, with taller understorey vegetation height being a proxy for that competition here.

2.4.3 Effects of Reindeer Grazing

Alongside stand maturity and understorey height, the results showed that lichen height is an additional influencing factor on lichen abundance. Lichen height can show the success with which lichen has grown, but it can also show the level of top-down pressure lichen has received through grazing by reindeer. For example, Akujärvi *et al.* (2014) studied lichen biomass relative to reindeer grazing in different aged, Scots pine-dominated forests in Finland. Despite parallels between that study and the present one, mean lichen biomass in grazed sites was 270 kg ha^{-1}, less than a third of that found in intermediate or old stands here. Reindeer herding in Finland occurs in smaller areas and supplementary feeding occurs more regularly, allowing the reindeer population to be maintained above the natural carrying capacity of the environment and in greater densities compared to Norway and Sweden where herding is more extensive spatially (Helle and Jaakkola 2008). Lichen height is largely affected by the grazing action of reindeer and thus quite naturally the biomass of lichen present is affected by this grazing action.

Conversely, aspects of silviculture affect the ability of reindeer to graze. Here it was seen that clear-cut stands have a high proportion of plant litter on the ground, in some sites covering up to 44 % of the ground over understorey vegetation. This litter is largely made up of a brash of the discarded branches from felled and de-limbed trees, and its presence has been seen to be detrimental to lichen growth, smothering them and their access to sunlight (Bråkenhielm and Persson 1980). Perhaps more problematic, these branches can be nearly impossible for reindeer to dig through, especially in wintertime when they are also covered in snow (Eriksson 1976). Litter presence concerns herders, as it means that even if forage is present and snow conditions are conducive to digging, reindeer still will be unable to access food and may waste precious energy following the smell of forage, only to be stopped by branches near the bottom of the snow column (Tuorda 2018).

2.4.4 Data Gaps and Opportunities

The study undertaken here focused upon forests dominated by Scots pine which make up the largest proportion of productive forest stands at 35 %, followed by spruce, mixed stands and deciduous stands in Sweden (Official Statistics of Sweden 2019). Mixed stands, in comparison to the large monocultures often cultivated in commercial logging areas, have been seen to promote plant productivity and understorey richness in Sweden (Jonsson *et al.* 2019), yet the author's personal discussions with reindeer herders indicate that they hold far less lichen than pine stands. Further research into how differing ages of mixed and deciduous dominated stands affect reindeer forage species would allow a better understanding of the effects of logging on reindeer grazing over a wider range of ecological habitats.

Another area of research which is essential to consider is that of the barrier effects of snow. As mentioned in the introduction, altering the structure of the tree canopy through silviculture also has an effect on the structure of snow. This in turn impacts the ability of reindeer to actually access the forage that is present within boreal forests during winter. Arboreal lichens are often considered by reindeer herders to be an emergency food source when snow conditions are poor, preventing reindeer from reaching terrestrial lichens. However, whilst arboreal lichens were present in old stands here, the biomass which was accessible to reindeer was low. This reflects results from Korosuo *et al.* (2014) who stated that the current biomass of arboreal lichen present in forests is so low that it is unlikely to make a significant difference to reindeer survival in poor snow conditions. Understanding the effects of snow on reindeer's ability to graze then, is essential to allow a more holistic understanding of the ecology of winter grazing, especially as the biomass of arboreal lichens which provide a safety net for survival in poor snow conditions are currently insufficient.

2.4.5 Conclusion

This chapter has shown that ecological relationships in boreal forests are complex and multifaceted. Lichen biomass was affected by a combination of stand maturity, understorey height and lichen height. These factors are perhaps representations of wider ecological phenomena, with stand maturity and age reflecting recovery from the disturbance of clear-cutting, understorey height being a proxy for the level of competition occurring between species, and lichen height giving some indication of the level of grazing pressure being experienced in the area. Species that are not edible to reindeer appear to thrive in clear-cut areas, declining with stand maturity, and clear-cut sites also show a greater mean cover of plant litter which could have impacts on the accessibility of forage below. Old forests do contain arboreal lichens, but only in small amounts in this study area, which may not be sufficient to offset terrestrial forage lost to poor snow conditions.

Whereas this chapter has examined how silviculture affects reindeer forage abundance, the next chapter will examine how it affects the accessibility of this forage though it's impacts on forest structure and subsequent snow conditions. These will be combined to show realised forage availability through time, both within the winter season and as forest stands recover from clear-cutting to become mature stands.

Chapter 3:

Effects of forest structure on snow conditions and reindeer access to forage in Subarctic Sweden

"Snow is wetter than before and arrives later. Temperatures in the autumn have varied from cold to warm making the snow-covered pastures more unreliable for reindeer herding. But of course it depends quite a lot on the first snow. If the ground is warm when the snow comes, the resulting ice on the ground makes it difficult for the reindeer to feed themselves."

–Lars Anders Baer, reindeer herder from Luokta-Mavas sameby

(Helander-Renvall and Mustonen 2004, p. 263)

3.1 Introduction

The relationships between organisms in an ecosystem are dynamic. For example, the availability of food to an animal may vary throughout the year, depending upon how the cycling of the seasons affects the growth and fruiting of relevant plant species. Similarly, temporary landscape features may affect the ability of an organism to access resources, such as the reliance of polar bears upon sea ice to allow them to hunt (Ferguson *et al.* 2000), or the role that river ice plays in allowing the movement of many Arctic terrestrial species through a landscape (Leblond *et al.* 2016). When assessing the quality of a habitat in relation to a species then, one must not only consider the presence of their resources in space but also how the presence and accessibility of the resources change over time. Knowledge of this realised availability is especially critical in environments with notably harsh and annually varying climates. This is especially true in the case of reindeer when they are trying to reach ground-lying food during winter. In their Arctic and Subarctic habitats, snow and ice formation has the potential to create impenetrable barriers between the reindeer and terrestrial vegetation, despite the forage being physically present (Rennert *et al.* 2009; Hansen *et al.* 2011).

Reindeer are increasingly struggling to gain sufficient forage in boreal forests during winter. In Fennoscandia this can partially be seen in the increased purchasing of supplementary feed by reindeer herders, when historically the reindeer would be self-sufficient in their grazing (Tyler *et al.* 2007; Persson 2018). As explored in chapter 2 of this thesis, commercial silviculture causes a notable loss of forage for reindeer, both through the mechanical processes of tree felling and the removal of old trees (Eriksson and Raunistola 1990; Lie *et al.* 2009; Horstkotte *et al.* 2011; Kivinen *et al.* 2012). This research on the impacts of silviculture has thus far only focused on the loss of forage presence. Changes in forage accessibility in these contexts are yet to be fully explored. As boreal forests often tend to be the winter grazing grounds of mountain reindeer populations, it is the conditions associated with winter which are likely to have the greatest impact on forage accessibility, affecting the survival of reindeer.

3.1.1 The Effects of Winter

Winter is a time of particular food stress, as the forage which remains during this period is relatively nutrient-poor. To extract as much as possible from this poor diet, reindeer undergo physiological changes within their alimentary tract which allows them to absorb nutrients more efficiently (Staaland and White 1991). One of the few forage types that does remain available throughout winter is lichens, often of the genus *Cladonia* P. Browne, which become a staple of the diet, constituting up to 80 % of reindeer forage intake (Heggberget *et al.* 2002). However, even with food being present, nutrients being efficiently extracted and with grazing areas that the animals are willing to enter, this does not mean that forage is accessible. Snow can act as a significant barrier between reindeer and ground-lying vegetation.

Deep soft snow drifts require a great deal of effort from reindeer to move and to dig through (Brown and Theberge 1990; Lie *et al.* 2008), a level of energy expenditure which may not be matched by the energy gained from the forage beneath. When snow reaches over 91 cm in depth, the reindeer are barred from even being able to detect lichens below its surface (Helle 1984). During these conditions, reindeer may respond by moving to areas where the snow is shallower, such as on hillsides (Pruitt 1959). However, these areas can also be problematic, as whilst snow in exposed locations may be shallower, it is also more prone to being compressed by wind into dense ice layers (Heggberget *et al.* 2002), and so the energy expenditure required in pursuit of food may once again not be in favour of the reindeer. Coping with these winter conditions is difficult, yet this is an environment in which the reindeer have survived for many thousands of years. Their migratory behaviour, morphological adaptations and their ability to dig through snow (known as cratering) mean that they can normally find a way to survive, even with notable weight loss as they expend fat reserves, until the spring melt arrives (Tyler 1986; Stein *et al.* 2010).

3.1.2 Extreme Weather Events

Extreme weather events have been a growing cause for concern in recent years. Increasingly frequent, warm periods in winter have been causing temperatures to rise above 0˚C, allowing precipitation to fall as rain instead of snow. This freezes when temperatures drop again or when the rain comes into contact with the snow surface, forming a sheet of ice which may subsequently be buried by further snowfall, in what is called a "rain-on-snow" event. Alternatively, surface layers of snow may melt in the warmth and later refreeze to ice, termed a "freeze-thaw" event (Putkonen and Roe 2003; Hansen *et al.* 2011, Axelsson-Linkowski *et al.* 2020). These ice sheets may simply be more energetically costly for reindeer to break through to reach forage below, but some can also be so hard as to be impenetrable to the animals. Herders say some of these ice layers form within the snow column each year, such as in areas exposed to wind, but extreme weather events may lead to ice formation over a much wider area. Icing has led to cases of mass starvations of animals as they cannot reach ground-lying forage (Reimers 1982; Lie *et al.* 2008; Rennert *et al.* 2009; Hansen *et al.* 2011). Some recent examples saw the starvation of 61,000 reindeer on the Yamal Peninsula in the winter of 2013-14, with a further 80,000 stariving in that area in 2020-21 due to icing (Department of Economics 2015; Staalesen 2021).

Despite their clear negative impacts, extreme weather events have been seen to stabilise some wild reindeer populations. This is through ensuring reindeer numbers do not exceed the carrying capacity of the environment, that the community is made up of more resilient and fit individuals, and that the future sensitivity of the population to these types of events is lower due to reduced competition for food (Hansen *et al.* 2019). However, within wild populations there is leeway for population crashes to occur, whereas the semi-domesticated herds in northern Fennoscandia are part of a market system, where certain numbers need to be slaughtered each year to provide a stable income for herders (Pekkarinen 2018).

3.1.3 The Need for Spatial and Temporal Data

From the information presented above, icing events are, therefore, a concern for reindeer herders, both for their economic implications, and because many herders have a deep cultural and emotional connection to their animals and so do not want to see many dying from starvation. These icing events also show how, in addition to yearly snow conditions, rarer, yet more extreme, climatic conditions can be very influential to the ability of reindeer to utilise resources within a landscape. Despite this clearly important role of snow in reindeer grazing, there are research gaps in this field of study. Some have examined the effects of varying tree canopy on snow depth with relation to its effects on grazing (Horstkotte and Roturier 2013), and there is a growing body of work involving traditional ecological knowledge and interviews with herders, providing detailed information on both the impacts of snow on reindeer and the causes of varying snow conditions in their pastures (Reinert *et al.* 2009; Roturier and Roué 2009; Roturier 2011; Eira *et al.* 2018). Whilst valuable in their own right, much of these data are qualitative rather than quantitative in nature, and there remains much scope for more of these phenomena and relationships to be quantified.

The changes in snow throughout the winter season account for much of the temporal changes seen in reindeer grazing pastures, but there is also a need to consider how these changes vary spatially. Many mountain reindeer herding districts in Sweden have their reindeer in the mountains in summer and in the lowland boreal forests in winter (Käyhkö and Horstkotte 2017). These lowland forests are felled on rotation, meaning that they are a patchwork of different ages and maturity classes. As we have seen in the previous chapter, this impacts the presence of plant species edible by reindeer. Snow conditions in these forests patches of differing maturity should therefore be considered.

3.1.4 Aim

Chapter 2 of this thesis brought together data to examine the spatial presence of forage in forests of different maturity. The aim of the present chapter is to examine the spatial and temporal impacts of snow on the ability of reindeer to *access* this forage throughout winter. This was carried out by studying snow conditions in four different forest maturity classes in November, January and March over the 2019-20 winter period, measuring snow depth, hardness and density. These data were used to infer the ability of reindeer to graze within these different forests. The hypotheses are as follows:

1. If forests have a larger, more mature canopy, ground-lying snow conditions are more heterogenous.
2. As winter progresses, the snow column in forest stands has increased presence of impenetrable ice layers.

3. Mature forest stands have greater realised availability of forage for reindeer compared to recently clear-cut stands.

The expected heterogeneity of snow conditions in old forests is derived from the work of Horstkotte and Roturier (2013), who studied the impacts of tree canopy complexity, being single or multi-layered, on snow conditions below. The knowledge gained in this study will provide valuable insights into how human activity through forestry is affecting reindeer survival during the critical winter period, both quantitatively and in greater detail than studies that focus primarily on the presence or absence of forage in an area.

3.2 Methods

3.2.1 Study Area

The study was undertaken in the winter of 2019-20 in the landscape surrounding Jokkmokk in Norrbotten, Sweden at 66°20’N. A more detailed description of this area can be found in the methods of chapter 2 of this thesis. Commercial silviculture occurs in the study area, resulting in the forest comprising patches of different-aged trees. As explained in chapter 2, these forest stands have been categorised into four maturity classes of ‘old’ (86 ± 13 years SE), ‘intermediate’ (23 ± 4 years), ‘young’ (7 ± 1 years) and clear-cut (0 years). Old stands did have the structure of commercial forests rather than the diversity and uneven distribution of trees found in ‘virgin’ forests, indicating that, despite their advanced maturity, they are not old-growth forests. Within each maturity class, four sites were identified, yielding a total of 16 sites within which data were collected. These were the same sites used in chapter 2 to allow direct matching of vegetation and snow data.

3.2.2 Snow Pits

Snow pits were dug and snow measurements taken during the period of 17-20th November 2019 and repeated in 17-19th January 2020 and 6-9th March 2020, yielding data for early, middle and later winter periods. At each of the 16 sites, six snow pits were dug. These were > 20 m from the site edge to reduce any potential edge effects, and at distances of > 10 m from one another. At young, intermediate and old sites, snow pits were placed using stratified random sampling, with three pits placed within 0.5 m of a tree and the other three placed in open areas between the trees as far from their base as possible. This was to account for the effects that the tree canopy exerts on snow characteristics (Fassnacht *et al.* 2006). At clear-cut sites the six snow pits were randomly distributed within the site due to the absence of trees, maintaining the 20 m site edge buffer and the 10 m buffer with other snow pits (Figure 3.1).

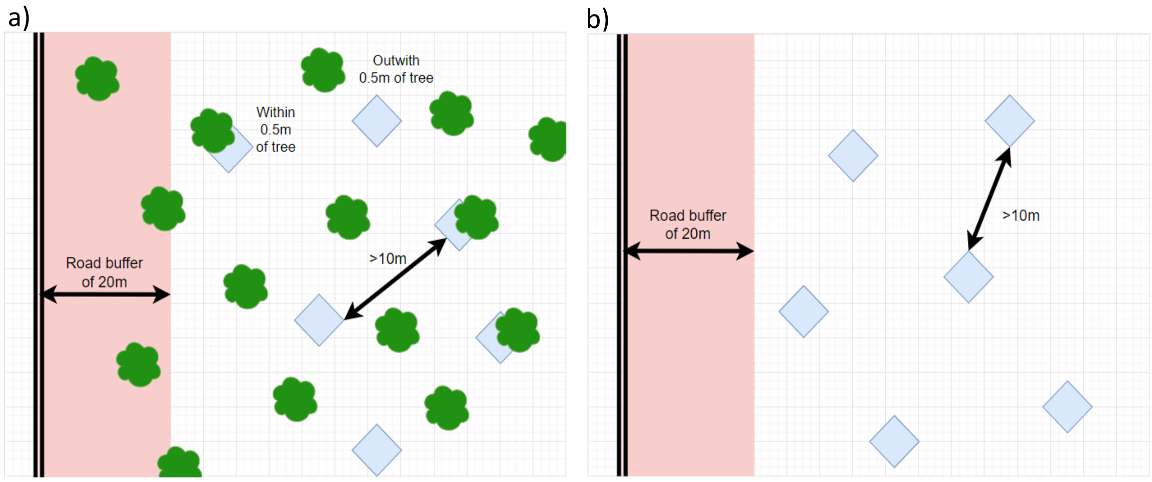

Figure 3.1. a) Experimental setup for forested sites. Blue squares indicate quadrats which are placed greater than 20 m from the road, indicated by the double parallel lines, and greater than 10 m from one another. Three quadrats are within 0.5 m of a tree, indicated by the green curved shapes, and three are place greater than 0.5 m from a tree in an otherwise random placement. b) The experimental design is repeated in clear-cut areas, although due to the absence of trees all six quadrants are simply randomly placed within the site regardless of vegetation.

Snow pits were dug to ground level and the distance from the snow surface to the ground was measured. A finger was lightly run down the snow profile to identify changes in snow hardness. The thicknesses of each distinct layer were measured with a ruler, and their vertical order in relation to one another noted. Hardness of each layer was determined using the hand hardness test as described in Höller and Fromm (2010). Measurements were undertaken by the same person (the author) to ensure that despite the subjective nature of this method, data were collected in a consistent way and the results were directly comparable.

Snow density was measured by extracting a known volume of snow (250 cm^3) using a density cutter (Snowmetrics RIP 2 Cutter, Snowmetrics, Fort Collins, USA, Figure 3.2), and weighing this on an accurate balance (Satorius M-prove, Satorius AG, Goettingen, Germany) in grams. Measurements were undertaken as many times as snow depth allowed, e.g. the density cutter was 10 cm in height so could be used four times in snow of 48 cm depth, but only once in snow less than 20 cm in depth. The presence of ice crusts on the snow surface and impenetrable ice layers on the ground were noted. Evidence of reindeer by the presence of greater than two aggregations of reindeer droppings and/or the presence of at least two craters dug by the animals was also noted.

Figure 3.2. Snowmetrics snow cutter used in this study. This equipment can contain a volume of 250 cm^3 of snow, which together with measuring the snow's weight can be used to obtain snow density.

3.2.3 Reindeer Digging Thresholds

There is no clear published limit of snow hardness through which reindeer can dig. One study observed that reindeer can crater through snow with a density of at least 2000 g cm^{-2} (Fancy and White 1985; Figure 3.3), although at this snow density, the cratering was very energetically demanding. On the other hand, Skogland (1978) created an equation on reindeer cratering rates which showed that they ceased digging at a snow ram hardness of 1985 g cm^{-2} (Figure 3.3). Ram hardness is a measurement of the force exerted on a cone tipped metal rod, known as a ramsonde penetrometer, as required to break through a layer of snow. These two studies indicate that snow hardness of *ca.* 2000 g cm^{-2} is on the cusp of the reindeer's ability to crater.

Hand hardness is a common alternative method used to measure snow conditions, where snow is categorised according to whether it can easily be penetrated by a fist, four fingers, one finger, a pencil or a knife, or then is otherwise categorised as 'ice'. Whilst somewhat subjective in nature, hand hardness values do have approximate equivalent values in the units of kPa as shown in Höller and Fromm (2010), which can be converted to the more commonly used g cm^{-2} by multiplying the number by 10.197 (see Table 3.1). Considering snow hardness of approximately 2000 g cm^{-2} is on the cusp of reindeer's ability to crater, this indicates that reindeer grazing is severely impacted (either highly energetically costly, or impossible), in snow of the hand hardness categories 'pencil', 'knife' and 'ice' – henceforth collectively termed PKI layers.

Table 3.1. Hand hardness values and their equivalent values in kPa and g cm^{-2} as derived from data in Höller and Fromm (2010).

Hand Hardness Value	Mean Equivalent force (kPa)	Mean Equivalent Force (g cm^{-2})
Fist	0.5	5
Four Fingers	5.5	56
One Finger	55	561
Pencil	550	5608
Knife	1000+	10197

Whilst the ability of reindeer to break through ice layers depends upon multiple factors, Sámi reindeer herders identify the section of snow closest to the grounf as båddne, and they say that ice layers occurring here are specifically difficult for reindeer to break through. Therefore, whilst PKI layers in the snow column are here regarded as very energetically costly for reindeer to break through, and potentially sometimes impossible to break, båddne PKI layers will be regarded as always impenetrable to cratering.

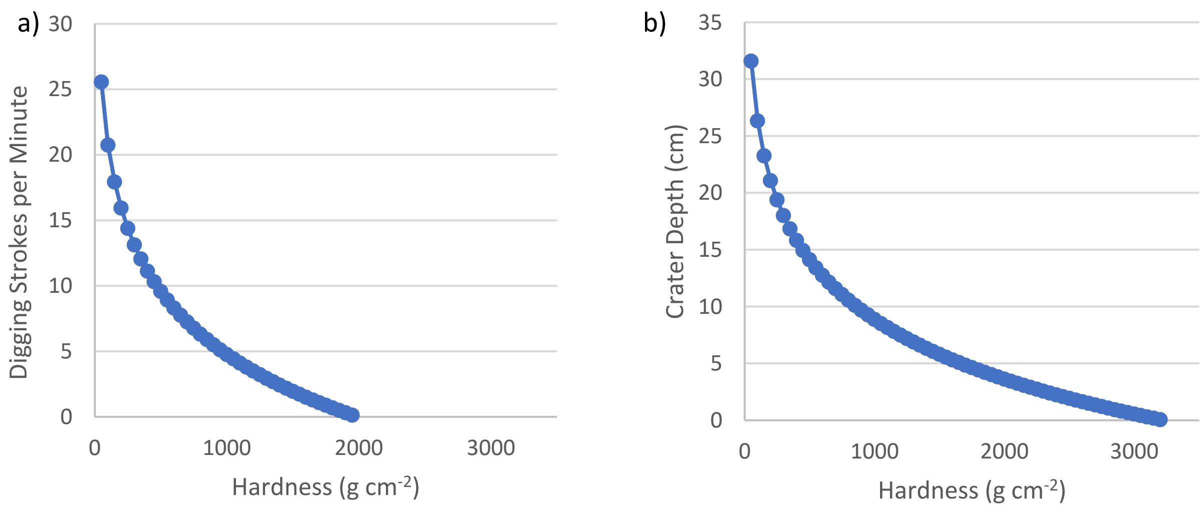

Figure 3.3. Graph a) showing the relationship between reindeer digging strokes per minutes and snow hardness in g cm^{-2} from an equation from (Skogland 1978). Graph b) illustrates the depth of reindeer craters relative to snow hardness at the site from an equation from Collins and Smith (1991).

3.2.4 Representativeness of Single Season Measurements

The data shown in this chapter cover the span of only one winter, making them unavoidably limited, as extra data collection could not be undertaken due to international movement restrictions from the Covid-19 pandemic. To assess how representative these data on the conditions during winter 2019-20 were, further pertinent analyses of related relevant meteorological data were undertaken.

Data from the Swedish Meteorological and Hydrological Institute was accessed from their weather station at Jokkmokk airport (66°29'27.6'' N and 20°8'31.2'' E), showing daily temperatures for winters starting in 2006 until the winter of 2019-20 (SMHI 2021a). Data from their Murjek D station, at 66°29'2.0'' N and 20°52'44.8'' E, showed daily snow depth for the same period (SMHI 2021b). Jokkmokk airport sits approx. 15 km southeast of Jokkmokk town, and all field sites are between these two locations, whilst Murjek D weather station is located 50 km east of Jokkmokk and 32 km east of Jokkmokk airport.

Winter was defined as 1st October to 20th April, corresponding to the period when reindeer are allowed to be present in their winter grazing grounds (Rennäringslag 1971). For each winter the maximum, minimum and mean temperatures were noted, as well as the number of days > 0 °C. The maximum and mean snow depth was also noted, along with the dates and number of days of consistent snow cover (> 3 days in a row of > 0 cm snow).

3.2.5 Statistical Analyses

All statistical analyses and generation of graphs were undertaken using the statistical software R, version 4.1.0 (R Core Team 2020), using the packages bitops, FSA, gplots and ggplot2. Shapiro-Wilk normality tests were carried out to verify normality of data. Comparison of the strength of effects at different sites was made using a one-way ANOVA followed by a Tukey’s Test for data that were normally distributed, or a Kruskal-Wallis test followed by a Dunn’s Test if data were not normally distributed. For statistical analyses of snow conditions near to and far from trees, post-hoc tests were not included as only two factors were involved (i.e. ‘near to’ and ‘far from’ trees). Data were nested by site, to avoid pseudoreplication.

3.3 Results

3.3.1 Snow Depth and Layers

Snow depth varied significantly between months ($p < 0.001$, Dunn's test, Table 3.2), being shallowest in November at 25.6 ± 0.92 cm (standard error), followed by January at 48.7 ± 1.68 cm and deepest in March at 80.3 ± 1.12 cm. In November and March, old sites had significantly lower snow depth compared to all other sites, at up to 19 % shallower (Tukey's Test, $p < 0.05$ for all, Figure 3.4, Table 3.2). The other three forest maturity classes showed little difference in snow depth between each other (Figure 3.4). The deepest snow in November and January was found in young-aged forests at 28.2 cm and 52.6 cm respectively. In March clear-cut areas had the deepest snow at 86.2 cm.

The number of distinct layers within the snowpack also varied throughout winter. November had an average of 1.9 layers which did not vary significantly between forest maturity classes. In January the number of layers was significantly higher in clear-cut forests compared to all other maturity classes (mean 6.6 layers, Tukey's Test $p < 0.01$ for all, Table 3.2). In March, mean number of snow layers varied from 6.6-8.5 but there was no significant difference between sites ($F= 0.24$, $p > 0.05$, Table 3.2).

Table 3.2. Analysis of effects of forest stand maturity on differing snow characteristics by month of data collection. Analysis of Variance (ANOVA) and Tukey's post-hoc test are used for parametric data, and Kruskal-Wallis (K-W) and Dunn's post-hoc test are used for non-parametric data according to a Shapiro-Wilk test. P values for significant post hoc test relationships only are shown, comparing the months November (Nov), January (Jan) and March (Mar), or forest stands of different maturity, including old (O), intermediate (I), young(Y) and clear-cut stands (C).

Variable	Month	Statistical Test (post hoc test)	DF	Test statistic	P Value	Posthoc stand Comparisons	Posthoc P value
Snow Depth	Nov	ANOVA (Tukey's)	3	7.04	0.006**	O-C O-I Y-O	0.024* 0.024* 0.006**
	Jan	ANOVA (Tukey's)	3	4.44	0.026 *	Y-O	0.029*
	Mar	ANOVA (Tukey's)	3	11.96	0.001**	O-C O-I Y-O	0.001** 0.027* 0.004 **
	Between Months	K-W (Dunn's)	2	99.24	<0.001***	Nov-Jan Jan-Mar Nov-Mar	<0.001*** <0.001*** <0.001***
Number of Snow Layers	Nov	K-W	3	0.62	0.618	-	-
	Jan	K-W (Dunn's)	3	16.35	<0.001***	I-C O-C Y-C	<0.001*** <0.001*** 0.010**
	Mar	K-W	3	1.66	0.238	-	-
	Between Months	K-W (Dunn's)	2	79.52	<0.001***	Nov-Jan Nov-Mar	<0.001*** <0.001***
Number of PKI layers	Nov	K-W	3	0.44	0.933	-	-
	Jan	K-W	3	8.91	0.031*	-	-
	Mar	K-W	3	1.54	0.263	-	-
	Between Months	K-W (Dunn's)	2	79.07	<0.001***	Nov-Jan Nov-Mar	<0.001*** <0.001***
PKI layer thickness	Nov	ANOVA	3	0.46	0.928	-	-
	Jan	ANOVA	3	1.81	0.198	-	-
	Mar	ANOVA	3	0.87	0.488	-	-
	Between Months	K-W (Dunn's)	2	83.89	<0.001***	Nov-Jan Nov-Mar Jan-Mar	<0.001*** <0.001*** 0.004**
Snow Density	Nov	ANOVA	3	1.50	0.265	-	-
	Jan	ANOVA	3	1.17	0.398	-	-
	Mar	ANOVA	3	0.18	0.91	-	-
	Between Months	K-W (Dunn's)	2	66.80	<0.001***	Nov-Jan Mar-Nov	<0.001*** <0.001***

3.3.2 Ice Crusts, Density and Hardness of Snow

The number of snow layers deemed 'very difficult to impossible' for reindeer to dig through (PKI layers) varied significantly according to forest maturity throughout winter (Kruskal-Wallis $p < 0.001$), with significantly fewer PKI layers in November (0.2 ± 0.06 SE) in comparison to January at 2.6 ± 0.12 and March at 2.4 ± 0.13 (Dunn's test, all $p < 0.001$, Table 3.2). The overall thicknesses of PKI snow layers also differed significantly between forest maturity classes, with the mean thickness increasing from 0.8 ± 0.3 cm in November, to 11.2 ± 0.8 cm in January and 15.8 ± 1.4 cm in March (Dunn's Test, all $p < 0.01$, Table 3.2, Figure 3.4). In November, 21 % of old sites, 13 % of intermediate and young sites, and 17 % of clear-cut sites haf PKI layers, although these were all in hand hardness category of 'pencil'. January and March saw some PKI layers in all the forest sites sampled.

The position of the PKI layer within the snow column also changed throughout the season and between forest maturity classes. None of the sites had båddne PKI layers in November. In January they were present in none of the intermediate sites, 8.3 % of old sites, 25 % of young sites and 33.3 % of clear-cut sites. In March, båddne ice was present in 4.2 % of old sites and 8.3 % of clear-cut sites, a decrease in both, though most notably in clear-cut sites. Intermediate sites had båddne PKI layers at 16.7 % of sites, and young sites 33.3 % (Table 3.2, Figure 3.4).

Snow density did not vary significantly between forest maturity groups in November, January or March (all $p > 0.05$, Table 3.2). However, density did increase through time, being significantly lower in November compared to January and March ($p < 0.001$, Dunn's Test, Table 3.2). Density in January and March differed little from one another.

In January, 50 % of all clear-cut, intermediate and old forest sites showed signs of cratering, as did 75 % of young sites. In March 50 % of old sites also showed signs of cratering, which rose to 75 % in young and intermediate sites, but no clear-cut sites showed recent cratering activity. There were no clear signs of cratering in November at any sites.

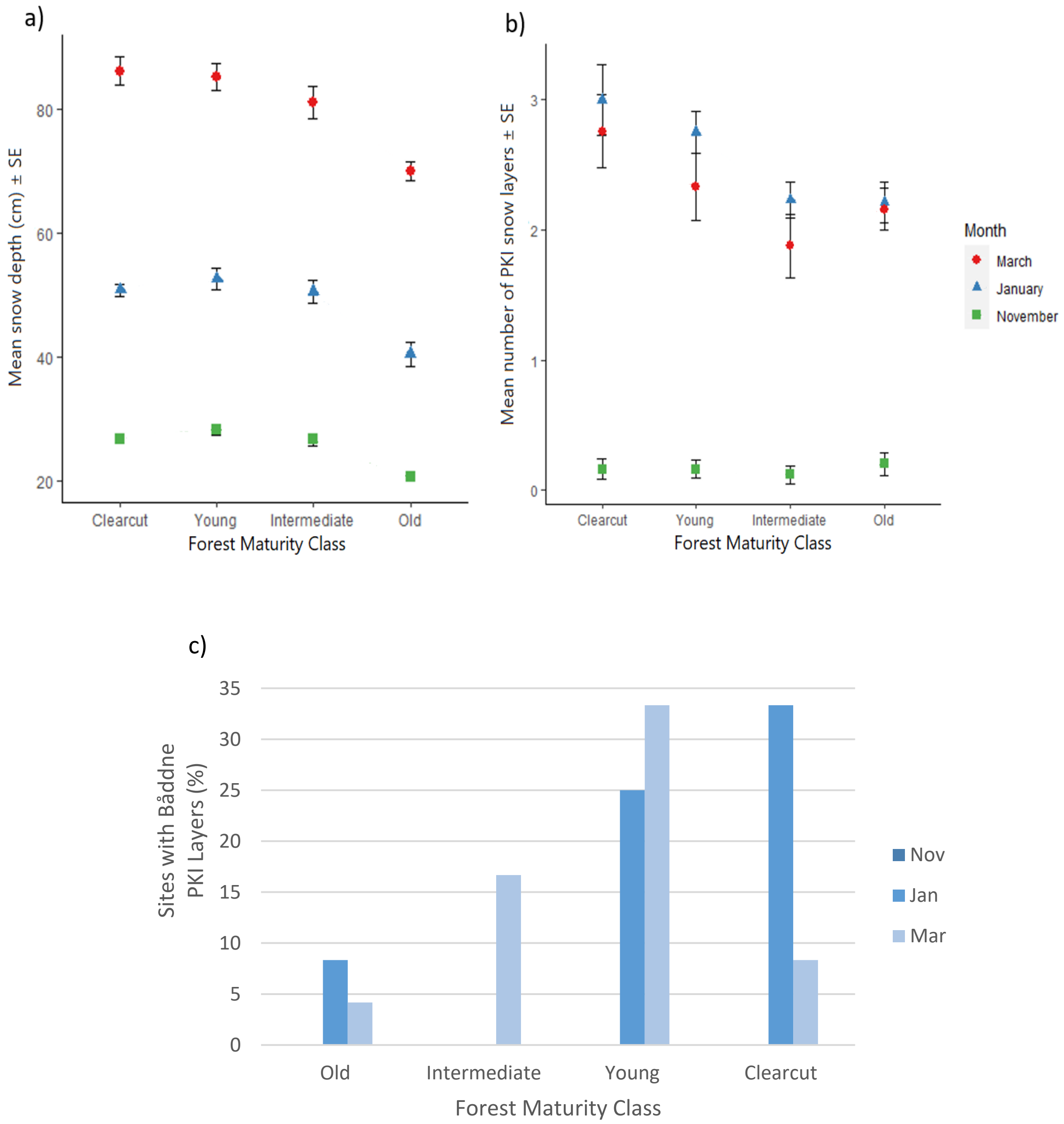

Figure 3.4. Snow characteristics according to forest maturity class measured throughout the winter of 2019-20 in boreal forests of Northern Sweden. a) Shows mean snow depth and b) mean number of PKI snow layers (impenetrable to reindeer) in the snow column, with bars showing standard errors where large enough to be shown (n=16 per month). Graph c) Percentagen of sites within each age group that had ground lying 'båddne' PKI layers (impenetrable for reindeer to dig through) per month measured. No sites had båddne PKI layers in November (n=24 per ages class per month).

3.3.3 Snow Conditions in Proximity to Trees

Mean snow depth was significantly shallower in close proximity to trees in November (F = 7.1, p = 0.01), January (*Chi sq*. = 13.19, p < 0.001) and March (*Chi sq.* = 4.98, p= 0.02; Table 3.3) compared to data from pits further from the tree bases. The overall number of layers within the snow differed significantly in January (*Chi sq*. = 6.38, p= 0.012; Table 3.3) but not in other months. The number of PKI layers in the snow column did not vary significantly near or far from trees throughout winter. The thickness of PKI layers did not differ in November or March, but in January these layers were significantly thicker in snow pits near to trees at 13.1 ± 1.5 cm compared to far from trees at 8.7 ± 1.0 cm (*Chi sq*.= 7.04, p = 0.008, Table 3.3).

Table 3.3. Effect of tree proximity to snow pit, measured as 'near to' or 'far from' trees, on variables relating to snow conditions. Data are separated according to the month in which measurements were taken.

Variable	Month	DF	Statistical Test	Test Statistic	P value	Mean 'near' tree compared to 'far from'
Snow Depth	Nov	1	ANOVA	7.10	0.010 *	Lower
	Jan	1	Kruskal-Wallis	13.19	<0.001 ***	Lower
	Mar	1	Kruskal-Wallis	4.98	0.026*	Lower
Number of Layers	Nov	1	Kruskal-Wallis	1.3816	0.2398	Lower
	Jan	1	Kruskal-Wallis	6.3751	0.012*	Lower
	Mar	1	Kruskal-Wallis	1.9475	0.1629	Higher
Number of PKI layers	Nov	1	Kruskal-Wallis	2.6453	0.104	Lower
	Jan	1	Kruskal-Wallis	1.2151	0.270	Lower
	Mar	1	Kruskal-Wallis	1.1862	0.276	Higher
PKI Thickness	Nov	1	Kruskal-Wallis	2.80	0.094	Lower
	Jan	1	Kruskal-Wallis	7.04	0.008 **	Higher
	Mar	1	Kruskal-Wallis	1.81	0.179	Higher

3.3.4 Forage Availability

Båddne PKI layers, occurring closest to the ground within the snow column, affected whether reindeer could access terrestrial forage. All lichen was available for grazing during November. In January 29.2 % of clear-cut sites and in March, 7.3 % of clear-cut sites were inaccessible to reindeer due to the presence of båddne PKI layers in the snow. Young stands had a 25 % and 33.3 % loss of lichen availability in January and March respectively, whilst intermediate sites only had båddne PKI layers in March at 16.7 % of sites, remaining accessible earlier in the winter season. Old stands saw an 8.3 % and 4.2 % loss of access to lichen in January and March respectively. These proportional losses affected the total abundance of lichen available for reindeer to graze throughout winter.

Multiplying data from chapter 2 (Table 2.1) of this thesis on lichen biomass with data from this chapter on percentage accessibility of sites due to snow conditions (Figure 3.4 c), the realised abundance of lichen available to reindeer can be calculated. Within old forest stands, lichen availability ranged between 828 and 759 kg ha^{-1} on average, varying little throughout winter (Figure 3.5). Similar available biomasses of lichen were seen in intermediate forests of *ca*. 870 kg ha^{-1}, although this decreased in March to 725 kg ha^{-1} (Figure 3.5). Young forests had considerably lower abundance of available lichen decreasing between November and March from 694 kg ha^{-1} to 463 kg ha^{-1} (Figure 3.5). Finally, clear-cut stands consistently showed the lowest availability of lichen. This decreased between November and January from 416 kg ha^{-1} to 295 kg ha^{-1}, subsequently increasing in March to 386 kg ha^{-1} (Figure 3.5).

Overall, the abundance of lichen available for grazing at clear-cut sites was notably lower when compared to old and intermediate forests. There is 50% less lichen available for grazing in November, 61% in January and 47% in March at the recently logged sites compared to intermediate and old growth forests.

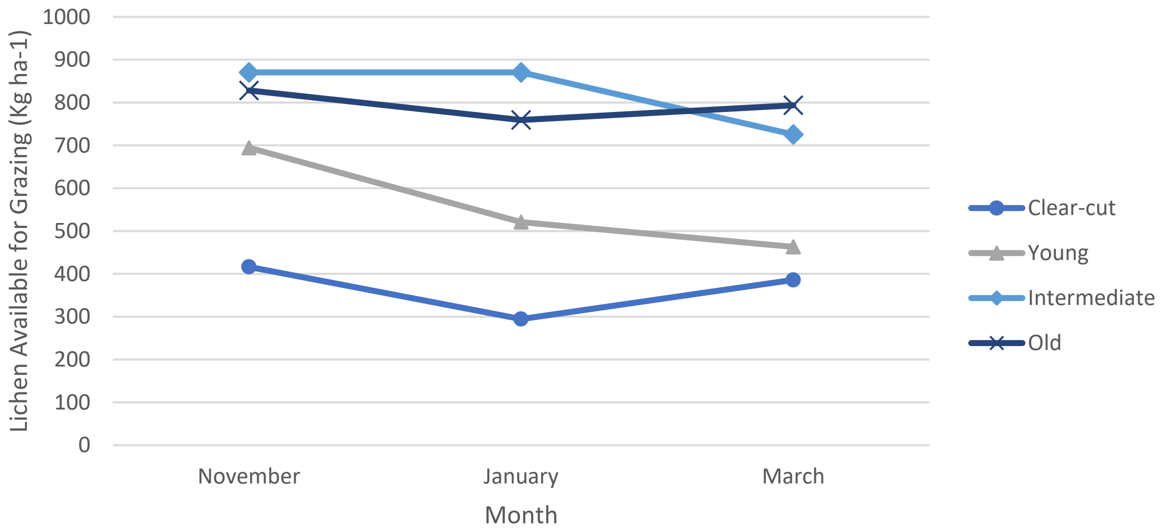

Figure 3.5. An estimate of the abundance of lichen present and accessible for reindeer to graze in clear-cut, young, intermediate, and old forest maturity classes shown throughout winter. Data calculated using mean lichen biomass multiplied by the proportion of sites without ground lying 'båddne' PKI layers, so deemed accessible to grazing (n=12).

3.3.5 Environmental Conditions

Weather data from Murjek D weather station in the study region showed that the days of consistent snow cover during the winter starting in 2019, when research was undertaken, did not differ significantly compared to winters starting in 2006 until 2018 for which data were available (Dunn's Test $p > 0.05$). Mean depth of snow during the winter only differed compared to four of the previous 13 years (Dunn's test $p < 0.05$). Three of these years (2008, 2010 and 2018) showed significantly different snow depths compared to at least 7 of the 13 other winters (see Figure 3.6). This suggests that they were anomalies in terms of yearly climate condition, and therefore our winter of 2019 was consistent with the broader 'normal' conditions of the region.

Temperature data from Jokkmokk airport weather station showed that the mean temperature in the winter starting in 2019, this being -5.7 °C, was not significantly different in comparison to past years (Dunn's test $p > 0.05$), though it should be mentioned that the maximum temperature was 2.7 °C cooler at 5.7 °C and the minimum 4.8 °C warmer at -20.7 °C compared to other winters (see Figure 3.7). Finally, during the winter of 2019-20 the number of days with temperatures > 0°C was 57. This did not differ significantly from previous years (Dunn's test $p > 0.05$).

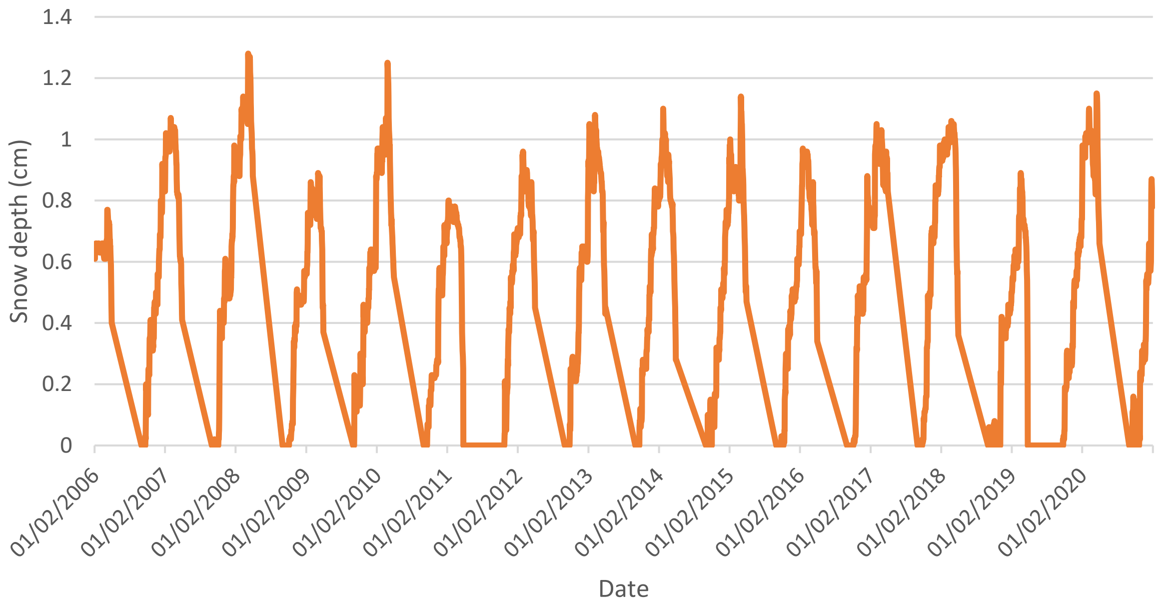

Figure 3.6. Daily snow depth (cm) measured at Murjek D weather station between the years 2006-2019 derived from SMHI (2021b).

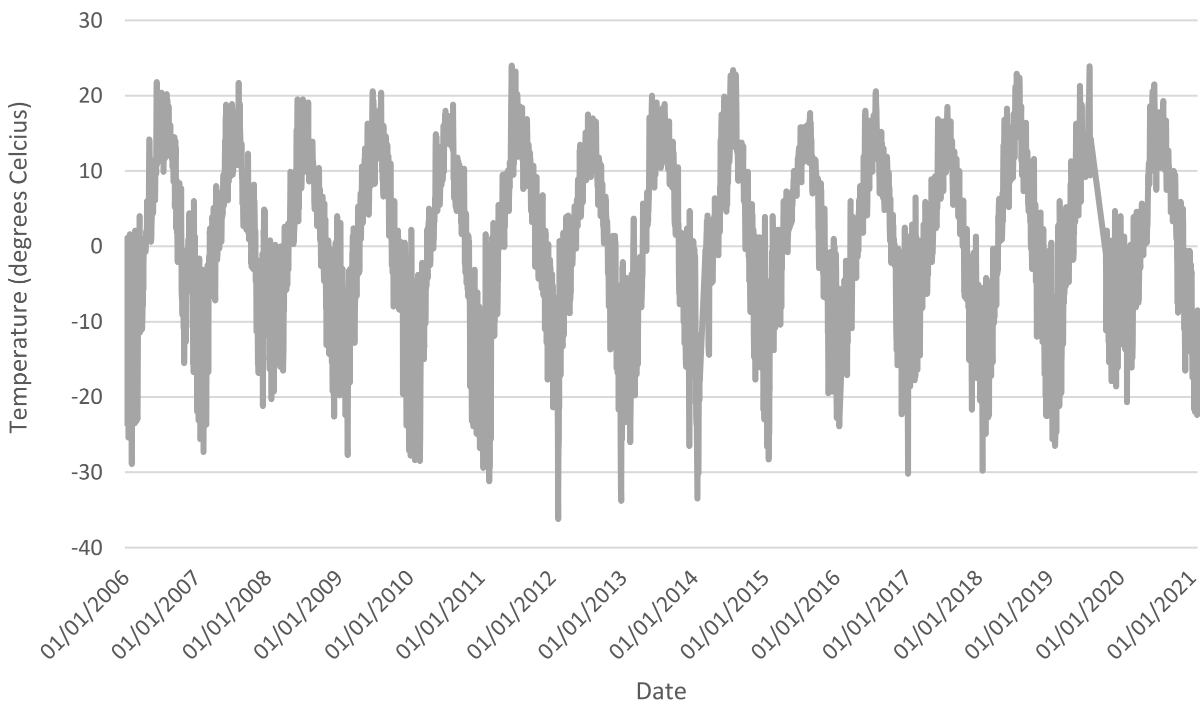

Figure 3.7. Daily temperature (degrees Celsius) measured at Jokkmokk airport weather station derived from SMHI (2021a).

3.4 Discussion

The greatest variance in snow conditions within the forests appeared to be through time. Snow depth increased significantly from November to March, having had sufficient time to accumulate. Snow depth was also significantly shallower in old sites and deepest in clear-cut and young forests, a pattern consistent with previous studies (Horstekotte and Roturier 2013). Reindeer herders say this occurs because small trees in young pine forests encourage snow to accumulate, whereas in older stands snowfall is trapped in the canopy above the ground (Roturier 2011). Here snow in January was significantly shallower near trees, supporting the idea that depth is affected by snow being held in tree canopies.

Older stands have multi-layered canopies, meaning later in the winter snow can drop to the ground from these canopies in a patchy manner (Roturier 2011). This creates heterogenous snow conditions on the ground, with both soft patches and patches that are impenetrable to reindeer. The wide variety of snow conditions increases the opportunity for reindeer to encounter suitable grazing conditions during adverse winter weather (Horstkotte and Roturier 2013, Korosuo *et al.* 2014).

Here the number of PKI layers did not differ significantly according to stand maturity, but did increase in number over winter as more snow accumulated and there was greater time for icing events to occur. Towards the end of the season some of the snow held in the canopy starts to drop onto the ground lying snowpack, often with force. This can compress patches into harder layers, thus leading to a greater variation in snow characteristics and a greater number of hard PKI layers (Roturier 2011). This is mirrored in greater snow density being seen in January and March compared to November. Depth and density measurements provide important information on the snow conditions in the area, although this does not encompass the whole complexity of snow structure and characteristics. The fine grain structure of the snow can further vary, from large heavy grains that are loosely attached, to large powdery light snow with few very hard thin sheets of ice.

3.4.1 PKI Ice Layers

Snow is already known to be a critical factor affecting reindeer grazing in winter (Roturier and Roué 2009; Reinert *et al.* 2009; Roturier 2011). Reindeer vary where they graze throughout this period as conditions within pastures change (Roturier 2011). All sites had the hard PKI layers in January and March, yet still showed some signs of cratering despite suggestions from Skogland (1978) that they should be inaccessible. Together with the data from Fancy and White (1985), this indicates that PKI layers are on the cusp of reindeer's cratering threshold, being highly energetically demanding yet not always impossible to break through. This suggests that reindeer are willing to go to sub-optimal grazing areas, perhaps due to lack of better alternatives in their area. At clear-cut sites, where the number of PKI layers was highest in March compared to other forest maturity, no reindeer cratering was seen, suggesting that the conditions made grazing here impossible.

Herders have specifically highlighted the dangers of the bottom-most layers of the snow, termed *båddne,* being iced as it is especially hard for reindeer to break through. No *båddne* PKI layers were present in November. A third of clear-cut sites and almost a quarter of young sites had *båddne* PKI layers in January, whilst in March this feature occurred at a third of young sites, 17 % of intermediate sites, and less than 10 % of clear-cut sites. As reindeer were seen to crater at sites with PKI layers present, only *båddne* PKI layers were deemed completely impossible to dig through, and therefore used in the later forage availability calculations. Of course, as mentioned, estimates on the proportion of sites which are inaccessible are likely to be conservative, as some areas with PKI layers other than *båddne* would also be impenetrable.

3.4.2 Loss of Available Forage

As seen in chapter 2, research has explored some of the effects of forestry on lichen abundance (Akujärvi *et al.* 2014, Korosuo 2014, Sandström *et al.* 2016, Uboni *et al.* 2019), as well as how forest structure alters snow conditions (Roturier and Roué 2009, Roturier 2011). However, realised forage availability depends on both forage abundance and its accessibility though snow. When combining these two datasets, from chapter 2 and chapter 3 of this thesis, a clear pattern emerges. The realised availability of lichen in old forests remains relatively consistent throughout winter, supporting the first hypothesis. Realised forage availability in intermediate forests also remains relatively stable throughout the season although this drop slightly in March. Young stands have significantly lower realised availability of lichen which decreases throughout the season, supporting the second hypothesis explored in this chapter. This makes them better suited to early winter grazing as seen in Axelsson-Linkowski *et al.* (2020). Finally, clear-cut sites consistently show the lowest availability of forage for reindeer, partly due to a reduced abundance of lichen, and partly due to greater occurrence of *båddne* PKI layers (Collins and Smith 1991; Horstkotte and Roturier 2013).

Altogether clear-cut sites have up to 61 % less biomass of available forage than old stands. Older sites then provide far more food for reindeer over a longer period of time, whilst clear-cut sites are mainly useful as pastures in early winter, becoming inaccessible later in the season. These results support the third hypothesis explored in this chapter.

Widespread loss of realised forage can create difficulties for reindeer. Either they are forced to remain with insufficient food, or must migrate elsewhere in search of food (Stein *et al.* 2010). However, if poor snow conditions are present on a broader scale, such as due to icing events, reindeer may be restricted by the reindeer herding area boundaries, making their movement not only a biological issue of survival but also a political one.

3.4.3 Opportunities and Limitations

Thus far I have explored how variation in snow conditions affects access to forage, but these differences in snow also impact the forage species themselves. When sites become locked by ice and reindeer cannot graze, the lichen is protected and allowed to grow the following year in greater abundance. The interaction between where reindeer can, and choose to, crater and the grazing experienced by ground-lying vegetation, could have wider implications for how heterogeneous lichen patches are within the landscape, with lichen-free areas indicating high grazing, and lichen rich patches indicating areas that remain largely locked by ice during winter. This hypothesis could be an interesting topic of future study. On the other hand, thick snow on warm or wet soil is thought to encourage the growth of mould on forage, making it inedible in subsequent years (Kumpula *et al.* 2000; Riseth *et al.* 2011), so the negative impacts of thick snow cover should also be investigated.

Aside from opportunities highlighted, there are some limitations to the work undertaken in this chapter. Data only spans the duration of one winter season, partially due to the limited funding period for this PhD and partially due to restrictions in movement due to the COVID-19 pandemic. Inter-annual climatic variations can impact the prevalence and extent of icing, with warmer temperatures allowing more freeze-thaw and rain-on-snow events to occur. Caution should be taken then when suggesting that data from one winter is broadly representative of general conditions in the area. However, climatic data collected from SMHI showed that the number of days with temperatures > 0 °C and mean winter temperatures were not significantly different from previous years. Mean snow depth only differed compared to four out of the 13 previous years, three of which appeared to be climatic anomalies themselves. This suggests snow depth was also broadly consistent with the norm. Weather stations providing this climatic data were 32-50 km from our field sites, but the broad trends seen within the data suggest that the area was not experiencing unusual climatic conditions during the winter of 2019-20 when our measurements were taken. Therefore, whilst future studies could benefit from sampling during multiple winters, the results here can nevertheless be regarded as broadly representative of an average winter within this environment.

As with the data in chapter 2, this chapter is also limited somewhat by the volume of data, this being taken from 16 sites further categorised into four maturity classes with four replicates each. Once again, this low number of replicates allowed more detailed research into snow conditions to be undertaken within the time available but did lower the statistical power of tests employed. Future studies would benefit from a greater number of sites being sampled.

3.4.4 Conclusion

This chapter shows that snow conditions vary in boreal forests throughout winter, affecting the ability of reindeer to reach ground-lying forage. When combined with data on vegetation presence from chapter 2, we see that intermediate and old forests have notably more realised forage compared to younger stands. Indeed, clear-cut sites have up to 61 % lower biomass of available forage compared to old sites, estimates which are likely to be conservative. These results highlight the important role old forests play in allowing reindeer to survive through the critical winter period, and by extension the notable impact the forestry can have on this reindeer ecology.

Silviculture is just one of multiple forms of land use occurring simultaneously with reindeer grazing. Understanding the cumulative effects of other industries and infrastructure on grazing then would provide a more accurate picture of the pressures reindeer in northern Sweden are facing today. This would be valuable information not only ecologically, but also for reindeer herders who are currently bearing the brunt of this grazing pressure, having to increasingly purchase supplementary feed for their reindeer, which is both an expensive option and one which is broadly detrimental to the reindeer's health despite allowing their survival in the short term. The cumulative effects of multiple forms of land use on reindeer is the topic of the next chapter of this thesis.

Chapter 4:

Modelling the ecology of reindeer survival in a multi-use landscape

"In my part of Sapmi, on Swedish side, the industrial forestry, as well as building of hydro-plants and dams, has caused a rapid change of the environment in a very short limit of time."

– Stefan Mikaelsson, former vice-president of the Saami Council

(Helander-Renvall and Mustonen 2004, p. 357)

4.1 Introduction

Throughout the preceding chapters, I have explored some of the factors influencing the ecology of reindeer in northern Sweden, notably the effects of commercial silviculture in their winter grazing grounds. Silviculture affects wide areas of landscape, and thus the impact of this industry on its own merits study. However, in practice silviculture is just one of the many industries that are simultaneously competing for land-use with reindeer herds in these regions. For instance, where there are commercial forests, there are roads, both for industrial traffic and everyday use by citizens. Also, within the landscape can be other large-scale industries. Approximately 45 % of Sweden's energy supply comes from hydropower, from the *ca.* 1800 stations scattered around the country (Energimyndigheten 2020; Swedish Agency Marine and Water Management 2019). The initial construction of hydropower dams involves a lot of human activity which can disturb reindeer, causing avoidance behaviour that can continue even after construction has been completed (Mahoney and Schaefer 2002). Additionally, construction of hydropower stations involves the creation of reservoirs, which can flood wide areas of reindeer pastures (Hermann *et al.* 2014). Construction of hydropower stations, through the process of flooding pastures and Sámi villages, and further filling the landscape with associated railways and property, has also been called out as a form of soft colonialism (Össbo and Lantoo 2011).

Another industry which has created much contention and competition for land with reindeer herders is mining. Sweden is responsible for producing up to 93 % of all of Europe's iron ore (Svemin 2021). Iron, along with other metals, comes from 12 currently active mines within Sweden, though there are applications and planning permissions for more, which are in various stages of processing (Mining Inspectorate of Sweden 2021). These mines also cause avoidance behaviour in reindeer due to activity at mining sites (e.g. Eftestøl *et al.* 2019), as well as causing physical loss of grazing from both the mine itself and from tailings ponds which are used to process mining waste, these ponds reaching up to 13 km^2 in size (Mainali *et al.* 2015).

Mining, hydropower and forestry, along with the roads and settlements associated with them, all exist within the Swedish landscape together, meaning their effects on reindeer are cumulative rather than isolated. Additionally, changes in climate are increasing the occurrence of extreme icing events, further barring reindeer from forage and exacerbating the pressures which they are already facing (Tyler 2010). Conflicts between reindeer herders and other land users have become commonplace (Borchert 2001; Mustonen and Jones 2015; Johnsen *et al.* 2017; Persson *et al.* 2017). Anecdotally herders in the area have expressed concern that their livelihood will soon become unsustainable due to these cumulative pressures, and the future of their reindeer remains unknown. In order to safeguard these animals and the livelihood of reindeer herding then, it is essential to understand the ecology of the reindeer's ecosystem and not simply their relationship with one aspect of their environment.

Models can be very valuable tools, assisting us to predict future scenarios, and allowing us to compare the strengths and impacts of different factors within an environment upon the population in question, helping to triage issues in the ecosystem.

4.1.1 Current models

Using models to explore reindeer ecology is not a new concept. Uboni *et al.* (2019) used modelling approaches to investigate the types of forest and broad landscape characteristics which tend to be lichen-rich, showing that forests dominated by Scots pine on gentle south- and west-facing slopes are optimum for abundant lichen. Other studies have modelled the impact of a changing winter climate upon reindeer populations (Kohler and Aanes 2004; Hansen *et al.* 2019). These studies found that, whilst increasing thickness of ice formed in the snow column can drastically decrease the reindeer population size, over time this can actually have a stabilizing effect on the overall population dynamics, reducing the number of explosions and crashes, at least in wild herds in Svalbard. At the same time, predicted decreases in snow cover duration has, in models, been seen to lead to a decrease in plant biodiversity and the extinction of certain plant species eaten by reindeer across the Arctic (Niittynen *et al.* 2018).

Human impacts on the animals have also been studied, including wind farms (Skarin and Alam 2017), forestry (Horstkotte *et al.* 2011), roads and railways (Lundqvist 2007; Panzacchi *et al.* 2013), mines (Anttonen *et al.* 2011; Eftestol *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). Each of these structures have been found to have varying zones of influence, stretching from hundreds of meters to several kilometres, where reindeer are less inclined to be present and graze. This avoidance behaviour creates a functional loss of grazing despite the forage being present. Man-made structures, especially ones linear in nature, such as roads and railways, have additionally been seen to reduce the 'reachability' of forage due to habitat fragmentation (Lundqvist 2007).

As the environment affects the reindeer, so do they affect their environment. Studies and models have tried to delve into the impact of reindeer density on their forage abundance which have contributed to wider controversial discussions on overgrazing (Kumpula *et al.* 2000a; Moen and Danell 2003; van der Wal 2006; Pekkarinen 2018) and the subsequent impact of forage wastage due to grazing and trampling (Pekkarinen *et al.* 2017). Broadly these studies have noted a shift within the vegetation community towards graminoids when heavy grazing occurs.

Certain of the studies discussed so far have received interest from the forestry industry and those working in connection to it. This has led to new, more applied models being developed to try to solve some land use conflicts between reindeer herding and forestry (Widmark *et al.* 2013; St John *et al.* 2016; Johnsen *et al.* 2017). It has also led to research that suggests how herders can optimize their slaughter rates and their supplementary feeding to maintain healthy herds (Pekkarinen *et al.* 2015; Tahvonen *et al.* 2015). Steps have also been taken by the Metsähallitus forestry group in Finland to use the Akwè: Kon method (Secretariat of the Convention on Biological Diversity 2004) in their 2022-2027 Natural Resource Plan, which involves consulting with herders before creating forestry plans to minimise negative impacts (Metsähallitus, *pers. comm.*).

Whilst the field of reindeer ecology may at first sight appear to be saturated with models, these models tend to focus on singular or very few interactions occurring within the ecosystem. The models hold an inherent value, but are limited in their scope as reindeer exist in a complex environment where multiple relationships are influencing them at the same time. Some models have taken steps to embrace this complexity, examining the cumulative effects of human activity and infrastructure upon reindeer (Vors *et al.* 2007; Sorensen *et al.* 2008; Polfus *et al.* 2011), yet these have mainly focused on caribou in Canada which live in a different ecosystem to reindeer in Europe, and this research has focused solely upon human disturbance. Wider relationships, such as that between reindeer and their climate, forage presence, slaughter rates and human disturbance have yet to be brought together in one model.

4.1.2 Aim

The aim of this chapter is to continue to expand the scope of reindeer ecology models and to attempt to embrace greater complexity. No doubt this introduces extra uncertainties with the increasing complexity of inputs, yet it will also help identify the multiple relationships occurring in the ecosystem.

The research objects for the chapter are as follows:

1. Compile original data on forage availability in forests of different maturity classes from chapters 1 and 2, with data from grey literature on the impacts of roads, mines, hydropower stations, settlements and climate on reindeer access to forage.
2. Using a deterministic model, analyse the impacts of these multiple forms of land use on reindeer forage presence within an originally modelled hypothetical landscape.
3. Use this model, adding further data on reindeer birth rates, death rates, annual slaughter numbers, nutritional requirements and four climate scenarios affecting access to forage, to develop a deterministic model of reindeer survival over 50 years.
4. Analyse and compare the impacts of each factor on final reindeer population after 50 years.

4.2 Methods

4.2.1 Data incorporated into the model

The basis of this model is a simulated landscape 500 km^2 in extent, composed of a grid of 1000 squares of 0.5 km^2 extent each. 13 squares are designated as lakes and two as population centres, whilst the remaining 985 comprise a forested landscape (Figure 4.1). The model is treated as uniformly flat in topography to avoid overcomplicating with factors such as slope, aspect etc. Vegetation data incorporated from chapter 2 of this study were also taken from sites with minimal slope.

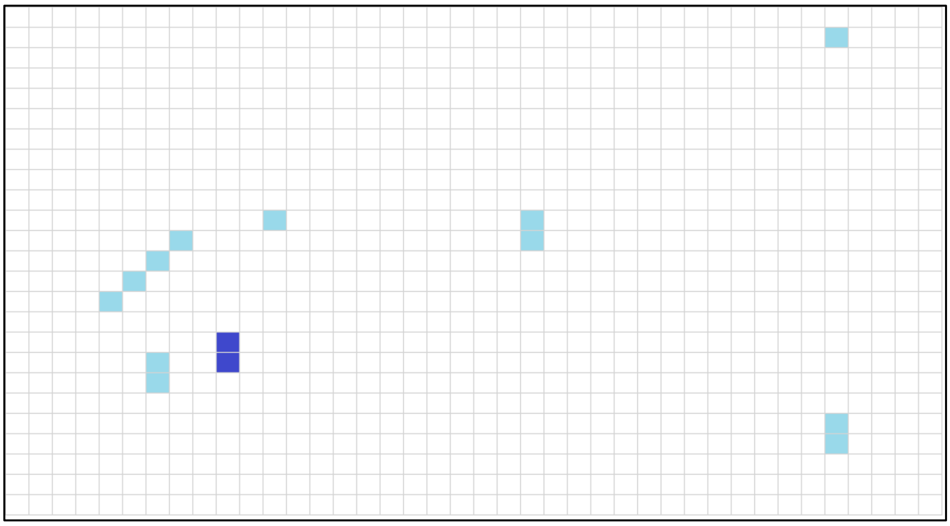

Figure 4.1. Visual representation of the basis of the model used within the study. Each square represents 0.5km^2 of land. Light blue squares indicate lakes whilst dark blue squares indicate a population centre.

Births and Death Rates

Heard (1990) reported the birth rate of reindeer in the Canadian archipelago led to an increase in herd size of approximately 26 % per year. The proportion of the herd dying from disease is unknown. These data were collected in an environment where death by predators or humans, either from shooting or collisions, was absent. The situation differs in northern Fennoscandia however, with one study in Norway identifying a natural death rate of 22 % within a reindeer population, with 14 % of these dying from predation and the rest from causes such as road accidents and disease (Nybakk *et al.* 2002). Combining a growth rate of 26 % and death rate of 22 % leads to an overall growth in reindeer population of 4 % per annum. Therefore, the reindeer population within the model was considered to grow by 4 % each year.

Grazing requirements

Within the model, reindeer survival is based upon whether sufficient lichen is consumed. A review of research on reindeer forage requirements showed that mean dry matter intake was 2.6 ± 0.8 kg per day for an adult (Table 4.1). According to Swedish legislation reindeer can only be in winter grazing areas between 1st Oct and 30th April, a maximum of 212 days, so only grazing data between these dates were included in the model (Rennäringslag, 1971). Therefore 2.6 kg of forage for 212 days gives an average dry matter intake requirement of 553 kg/reindeer for the total winter season.

A review of the literature revealed that the mean proportion of lichen within the winter diet of reindeer or caribou is *ca.* 54 % (Table 4.1). From this we may assume that in best possible grazing conditions, lichen will make up 54 % of the diet and hence 46 % of the diet will be composed of other food sources. Therefore, on average each reindeer requires *ca.* 299 kg lichen over the course of winter, calculated as 54 % of the 553 kg of general forage required. If reindeer avoid an area with lichen or lose lichen to the construction of infrastructure, they will also lose the opportunity to graze on the other edible species present.

Lichen regrowth

According to Scotter (1963) the growth rate of lichens in the Northwest Territories of Canada, specifically *Cladonia* species such as *C. rangiferina* (L.) Weber which are eaten by reindeer, was 3.4-4.1 mm year^{-1}. In Gustavsson (1989) lichen growth rate of relevant species in northern Sweden is 3-4 mm year^{-1}. Between these two works the mean lichen growth rate is 3.6 mm year^{-1}. According to data collected in chapter 2 of this thesis, the mean lichen height across all sites was 220 mm. With a growth rate of 3.6 mm this would mean a 1.6 % increase in size per annum. If this is applied to biomass, we can also assume a 1.6 % increase in lichen biomass per annum.

Table 4.1. Studies examining the amount of forage intake of reindeer per day and the period of time within which the data was collected, along with the proportion of lichen within a reindeer or caribou's diet

Period	Dry matter intake kg/day	Mean lichen in diet (%)	Source
Jan-April	3.7-6.9	-	Hanson *et al.* 1975
Jan-March	4.9	-	Holleman *et al.* 1979
Sept-May	1.8	-	Staaland and Hove 2000
March	0.8-1.4	-	Storeheier *et al.* 2003b
March	0.5-0.9	-	Storeheier *et al.* 2003a
Oct	-	50	Heggberget *et al.* 2002
Nov	-	50	
Dec	-	55	
Jan	-	55	
Feb	-	55	
Mar	-	53	
Apr	-	48	
Oct-May	-	62	Boertje 1990
Nov-Dec	-	54	Kojola *et al.* 1995
Jan-Mar	-	53	
Mean ± S.E.	**2.6 ± 0.8**	**54 ±1.2 %**	

Forest impacts

To calculate the biomass of lichen within the model's simulated landscape, data from chapter 2 of this thesis were used. These data, showing lichen biomass in four different forest age groups, were collected in quadrats of 4 m^2 and the data extrapolated to the 0.5 km^2 landscape squares used within the model (Table 4.2).

The average distribution of forest maturities in locations near field sites used in chapter 2, was 'clear-cut' 10 %, 'young' 35 %, 'medium' 35 % and 'old' 20 % (Berg *et al.* 2008, measured in Akkajaur/Abraur and Eggelats). This is a ratio of 1:3.5:3.5 between clear-cut, young and medium forests. Medium forests may for the purposes of this study be regarded as equal to intermediate forests. From this ratio the age distribution of each of the 985 0.5 km^2 landscape squares in the model was calculated (Table 4.3). This was undertaken for five scenarios, where none (0 %), 20 %, 40 %, 60 % and 80 % of the forest experienced logging at some point but had yet to return to the status of mature forest. The number of old sites was deemed to be the leftover percentage that had not been logged. Rounding to whole integers was done, so when site numbers did not add up the difference was added or subtracted from the clear-cut site category for consistency. This range was chosen as 80 % corresponds to the current level of commercial logging in Sweden (Berg *et al.* 2008), whilst the lower percentages allow us to simulate scenarios where less active forestry is occurring due, for example, to land preservation/conservation efforts.

Lichen biomass per forest maturity square (Table 4.2) was multiplied by the number of eachsquare present, producing the biomass of lichen in the entire simulated environment for each of the forestry scenarios (Table 4.3).

Table 4.2. Data from chapter 2 of this thesis showing the biomass of lichen found per m^{-2} based on measurements within a 4 m^2 quadrat, and extrapolating this to fill 0.5 km^2 landscape squares. Shown are both the total and the data broken up by whether it is ground-lying lichen or arboreal lichen.

Forest maturity	Ground lichen in site (kg 0.5 km^{-2})	Arboreal lichen in site (kg 0.5 km^{-2})	Total (kg 0.5 km^{-2})
Clear-cut	20269	0	20269
Young	35419	0	35419
Intermediate	44636	0	44636
Old	41687	2075	43762

Table 4.3. The number of 0.5 km^2 landscape squares per forest maturity class based on the level of forestry practised and the ratios of forest classes in relation to one another. The number of sites in each maturity class was multiplied by the lichen volumes seen in Table 4.2, giving total lichen biomass.

Level of forestry (%)	Clear-cut sites	Young sites	Intermediate sites	Old sites	Ratio (C:Y:I:O)	Total lichen biomass (tonnes)
None	0	0	0	985	0:0:0:1	43106
20	25	86	86	788	1:2:2:19.5	41876
40	50	172	172	591	1:2:2:7.5	40646
60	75	258	258	394	1:2:2:3.3	39417
80	100	344	344	197	1:2:2:1.2	38187

4.2.2 Infrastructure Data

Roads and population centres

Various studies have identified zones of influence (ZOI) around structures within which reindeer show avoidance behaviour (Table 4.6). Panzacchi *et al*. (2013) measured reindeer road avoidance at distances of 1 km, 5 km, and 10 km. The data on 5 km and 10 km avoidance were discounted as in this particular study the avoidance appears to be a residual from the avoidance occurring within the first 1 km (46 %).

The mean ZOI of road avoidance by reindeer, according to past literature, was 1 ± 0.25 km (Table 4.6). Avoidance is likely to be highest closest to the source of disturbance, reducing with distance, so percentage reindeer avoidance was only calculated from studies which examined a 1 km ZOI for roads. This gave a mean avoidance of 43 ± 3 % within 1 km of a road.

Three increasing magnitudes of extent of roads within the model environment were imposed: Level 1 (15 km of road), level 2 (37 km) and level 3 (49.5 km, see Figures 4.2-4.5). When a mine was present road levels 1 and 2 increased by 9.5 km and level 3 by 1.5 km due to access roads. When a hydropower station was present road levels 1 and 2 had an additional 5 km of road. The lengths of these roads were determined according to how they fit on a map of this simulated landscape (see Figures 4.2-4.5).

The model includes a population centre of 1 km^2, made of two adjacent landscape squares. Previous studies show a mean ZOI of 6.5 km (to the nearest 0.5 km), with a mean reduction in reindeer presence of 63 ± 8 % around population centres (Table 4.6).

Mining

A mine was included within the model environment. Little public data were available on the surface ground cover of the 12 currently active mines in Sweden, so this was calculated using satellite imagery hosted on Google Maps (CNES 2020; Table 4.4). Two mines did not have suitable imagery to determine a size, so only 10 were used, giving a mean ground cover of 13.1 ± 7 km^2. Rounded to the nearest 0.5 km^2, the mine displayed in the model is 13 km^2 in size, occupying 26 landscape squares.

Studies have shown a variety of ZOIs and percentage avoidance of mines by reindeer (Table 4.6). As Polfus *et al.* (2011) found a significant difference in caribou avoidance between summer and winter only data from winterwas included. The mean zone of influence was 1.5 ± 0.7 km to the nearest 0.5 km. This corresponds well to findings in Eftestol *et al.* (2019) where use was reduced by 35 %, so these were the parameters included in the model.

Table 4.4. List of all mines active in Sweden in 2019, showing their county, the mineral mined, and the approximate ground surface area of the mine as measured from satellite imagery on Google Maps.

Mine	County	Mineral	Approximate ground surface area (km^2)
Aitik	Norrbotten	Gold, copper, silver	62.3
Kiirunavaara	Norrbotten	Iron	45.0
Malmberget	Norrbotten	Iron	6.7
Leveäniemi	Norrbotten	Iron	No recent image
Kaunisvaara	Norrbotten	Iron	6.1
Kristineberg	Västerbotten	Lead, gold, copper, silver, zinc	3.0
Kankbergsgruvan	Västerbotten	Lead, gold, copper, silver, zinc	0.1
Renström	Västerbotten	Lead, gold, copper, silver, zinc	0.1
Björkdalsgruvan	Västerbotten	Gold, copper, silver	5.6
Garpenberg	Dalarna	Lead, gold, copper, silver, zinc	2.2
Lovisagruvan	Örebro	Lead, silver, zinc	Difficulty locating
Zinkgruvan	Örebro	Lead, copper, silver, zinc	0.1
		Mean	13.1 km^2

Hydropower stations

A hydropower station was also included in the model environment. Data on the sizes of hydropower dams and the land flooded by their reservoirs could not be found for Sweden directly. However, data could be obtained on the energy production of the hydropower stations in Sweden, and it is known that the average net land occupation of Norwegian storage hydropower plants is 0.027 km^2 yr^{-1} GWh^{-1} (Dorber *et al.* 2018). Much of the data used in parameterisation of this model was derived from the area of Sweden surrounding the Luleälven river in Norrbotten, which has multiple hydropower plants along it, so data on energy production by these specific power plants were used to indirectly calculate the area of land lost to each reservoir (Finnish Barents Group Oy, 1998; Table 4.5). The mean land area occupied by the hydropower station and reservoir was 27.5 km^2 (to the nearest 0.5 km^2), equivalent to 55 landscape squares. As aforementioned, access roads of 5 km were added to landscapes with hydropower stations in road extent scenarios “1” and “2” (See Figure 4.4).

Table 4.5. The kWh energy production of hydropower stations along the Luleälven in Sweden, showing the net land occupation calculated using equations from Dorber et al. (2018).

Reservoir	Energy production (GWh)	Net land occupation (km^2)	Location
Ritsem	460	12.4	Mountains
Vietas	116	31.3	Mountains
Seitevare	850	23.0	Mountains
Porjus	1290	34.8	Winter graze
Ligga	790	21.3	Winter graze
Akkats	585	15.8	Winter graze
Harsprånget	3481	94.0	Winter graze
Messaure	1901	51.3	Winter graze
Letsi	1770	47.8	Winter graze
Porsi	1150	31.1	Winter graze
Laxede	835	22.5	Winter graze
Randi	235	6.3	Winter graze
Parki	100	2.7	Winter graze
Vittjärv	230	6.2	Winter graze
Boden	490	13.2	Winter graze
	Mean	**28.6**	

Studies shown in Table 4.6 described the difference in reindeer presence relative to distance from a hydropower station, labelled as the percentage of the population ‘near’ (so within the ZOI) or ‘far away’ from the structure. Assuming an even distribution of 50 % of reindeer within and without the ZOI before the construction of the hydropower station, the percentage loss after construction was calculated as the percentage of reindeer within the ZOI in relation to 50 % of the entire population. The mean ZOI was 3.5 ± 0.5 km with a 60 ± 2 % loss in reindeer presence, which is supported by the results of a review study by Driedger (2014). It is assumed this is a ZOI around the physical dam itself and not the entire reservoir.

Multiple ZOIs

Panzacchi *et al.* (2013) identified a cumulative effect of roads and population centres on reindeer avoidance. Using the data from Table 4.1 and equation 6 from that study (shown here in Equation 4.1), the cumulative impact of roads and populations centres within a 1 km buffer, regarded as the mean ZOI of roads here, was calculated.

Equation 4.1. Equation describing the decrease in use of area near roads and population centres by reindeer as seen in Panzacchi *et al.* (2013).

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

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

From this calculation, decrease in use by reindeer is 100 %, meaning that where roads and population centres have overlapping zones of influence, no reindeer are present. This same theory is applied in the present model in cases where the ZOIs of hydropower stations, mines, roads and population centres overlap with one or more other ZOIs.

Table 4.6. Studies showing the zones of influence (ZOIs) of reindeer avoidance for differing parameters included in the model. Shown also are the ZOI distances and percentage avoidance of reindeer used within the model, made up of the mean of relevant results (read in text for more details on parameter selection).

Factor	ZOI (km)	Avoidance (%)	Source	ZOI used in model (km)	Avoidance used in model (%)
Road	0.25	-	Dyer *et al.* 2001	**1**	**43**
	1.0	33-47	Lundqvist 2007		
	0.25	54.2	Sorensen *et al.* 2008		
	1.5	-	Anttonen *et al.* 2011		
	2.0	-	Polfus *et al.* 2011		
	1.0	-	Polfus *et al.* 2011		
	1.0	46	Panzacchi *et al.* 2013		
Population Centre	2.5	-	Anttonen *et al.* 2011	**6.5**	**63**
	8	52 and 55	Helle and Särkelä 1993		
	4	-	Helle *et al.* 2012		
	10	Mean 60	Nellemann *et al.* 2000		
	5	Mean 85	Nellemann *et al.* 2001		
	9	-	Polfus *et al.* 2011		
Mine	1.5	35 %	Eftestol *et al.* 2019	**1.5**	**35**
	4.0	33.3 %	Weir *et al.* 2007		
	0.0 - 1.5	-	Anttonen *et al.* 2011		
	0.25	-	Polfus *et al.* 2011		
Hydropower station	3	58 %	Mahoney and Schaefer 2002	**3.5**	**60**
	4	62 %	Nellemann *et al.* 2003		
Multiple ZOIs	-	-	-	**-**	**100**

Lichen loss from Avoidance

Data on ZOIs and avoidance behaviour is summarised in Table 4.6 above. Using these data, number of squares within the ZOI of each infrastructure was calculated (Table 4.7). These squares are later used to calculate the amount of lichen lost to potential reindeer grazing due to their avoidance of these areas. A visual representation of these ZOIs can be seen in Figures 4.2-4.5.

From data collected in chapter 2 it can be estimated that a 0.5 km^2 landscape square (in a subarctic landscape *ca.* 66 °N) in a clear-cut forest contains 20,269 kg of lichen, in a young forest contains 35,419 kg, in an intermediate forest 44,636 kg and in an old forest 43,761 kg of lichen. As stated earlier, the ratio of forest landscape square maturity varies according to the proportion of the forest that is logged (see Table 4.3). The maturity of forest squares within ZOIs were considered to adhere to these ratios, and are shown in Appendix A. Forest maturity ratios from Table 4.3 were multiplied by the biomass of lichen known to be contained within one square of each of these forest maturity for the ZOIs (Table 4.2), and then further multiplied by the proportion of reindeer avoiding the area according to the forms of infrastructure present (see Table 4.6). This gave a figure for the lichen lost to potential grazing in each scenario due to avoidance of infrastructure and varying forest composition by reindeer (Appendix A).

Table 4.7. The number of 0.5 km^2 landscape squares within the zone of influence (ZOI) of each type of infrastructure scenario. The naming system is as follows: R1, R2 and R3 indicate the number of through roads from one to three respectively, N indicates a mine is absent whilst Y indicates it is present, and _H indicates the presence of a hydropower station.

Scenario	Road (ZOI squares)	Population centre (ZOI squares)	Mine (ZOI squares)	Hydropower (ZOI squares)	Multiple ZOIs
R1N	46	210	0	0	71
R2N	158	150	0	0	131
R3N	176	88	0	0	193
R1Y	68	176	60	0	117
R2Y	145	150	55	0	148
R3Y	199	114	55	0	184
R1N_H	46	139	0	12	107
R2N_H	158	79	0	12	167
R3N_H	174	79	0	12	167
R1Y_H	68	109	60	12	149
R2Y_H	145	79	55	12	184
R3Y_H	197	79	55	12	184

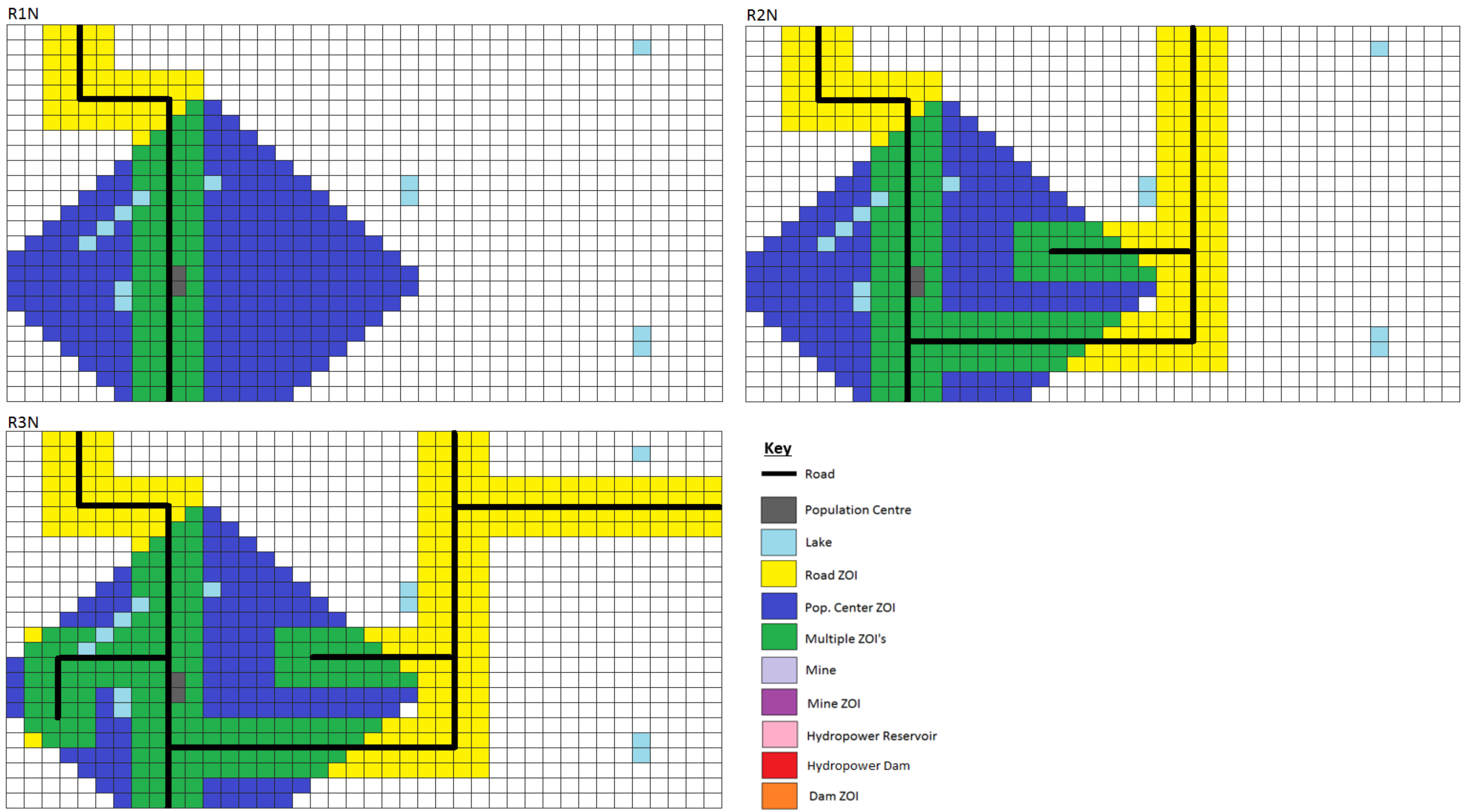

Figure 4.2. Landscape map of the model used in this study, with each square representing an area of 0.5 km^2. Coloured squares indicate infrastructure and landscape features as well as their zones of influence as described in the key. A population centre and roads are present but a mine and hydropower dam and reservoir are absent. R1N has only one through road, R2N has two through roads with one dead end, and R3N has three through roads and two dead end roads. Data used to make these maps are found earlier in the methods section.

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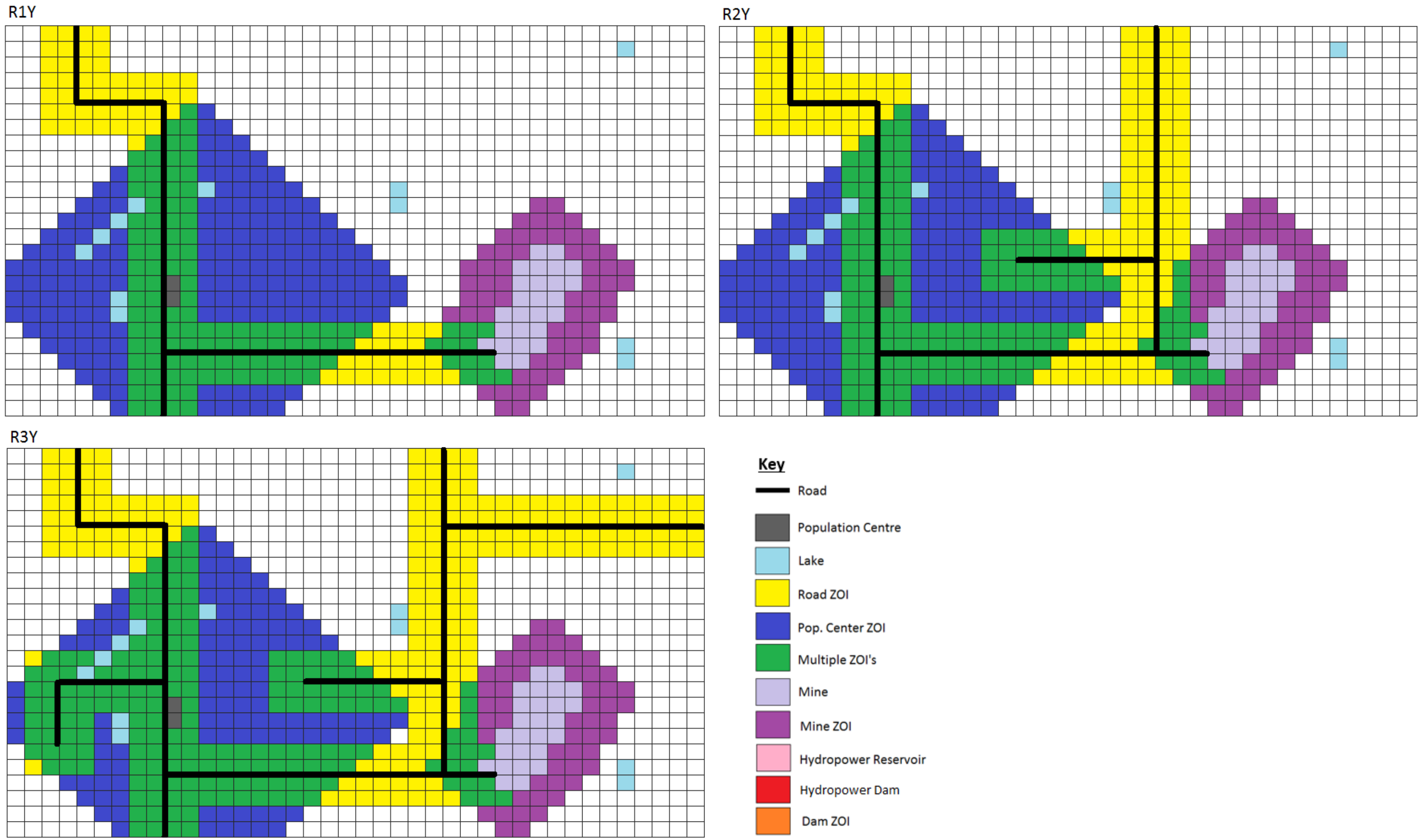

Figure 4.3. Landscape map of the model used in this study, with each square representing an area of 0.5 km^2. Coloured squares indicate infrastructure and landscape features as well as their zones of influence as described in the key. As well as roads and population centre as in Figure 4.2, a mine and its ZOI is also included. R1Y indicates one through road with mine present, R2Y is two through roads with mine present, and R3Y three through roads with mine present.

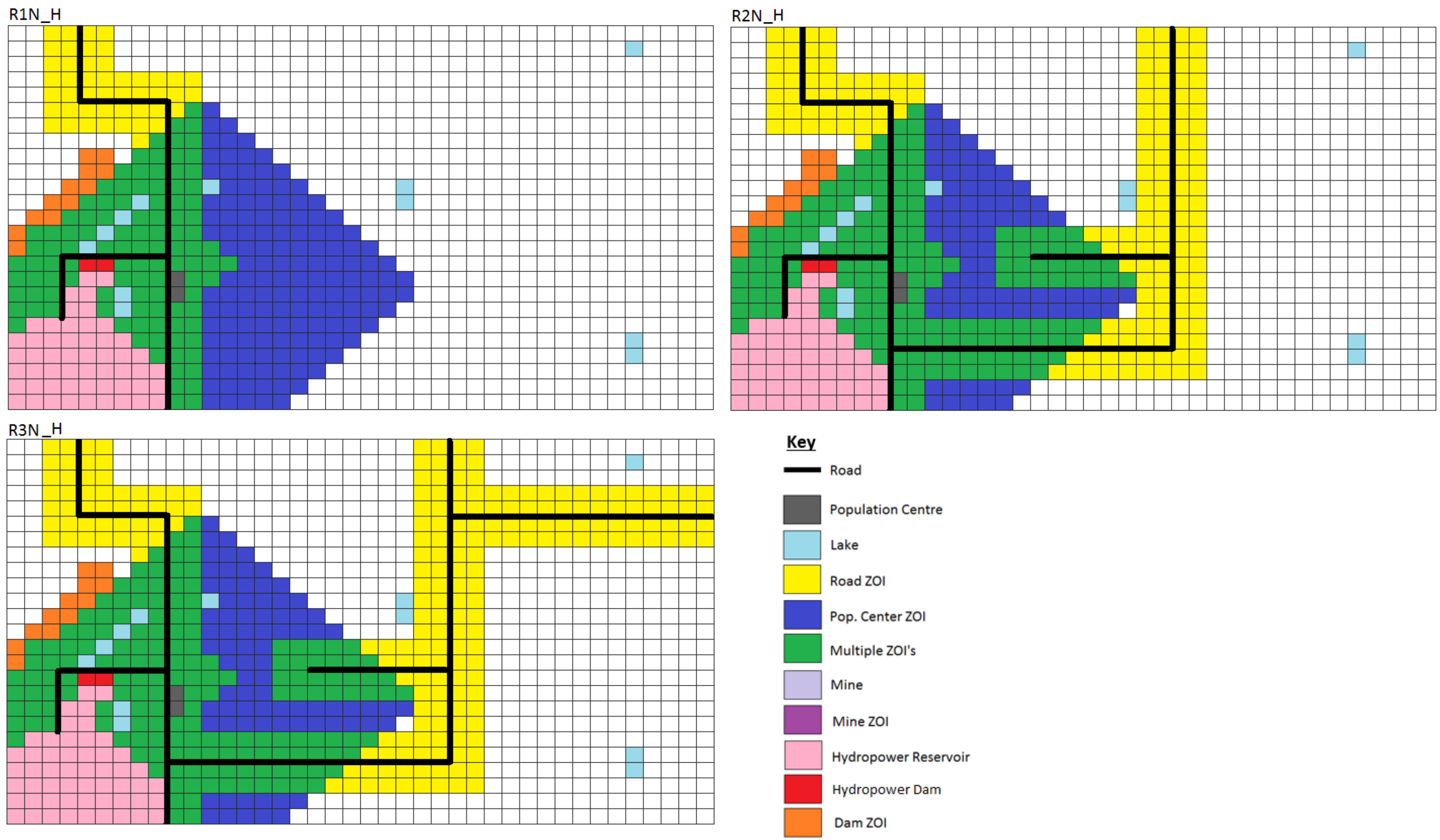

Figure 4.4. Landscape map of the model used in this study, with each square representing an area of 0.5 km^2. Coloured squares indicate infrastructure and landscape features as well as their zones of influence as described in the key. As in Figure 4.2, roads and a population centre are present, as well as a hydropower dam and reservoir. R1N_H indicated one through road with no mine but a hydropower station present, R2N_H the same but with two through roads, and R3N_H with three through roads.

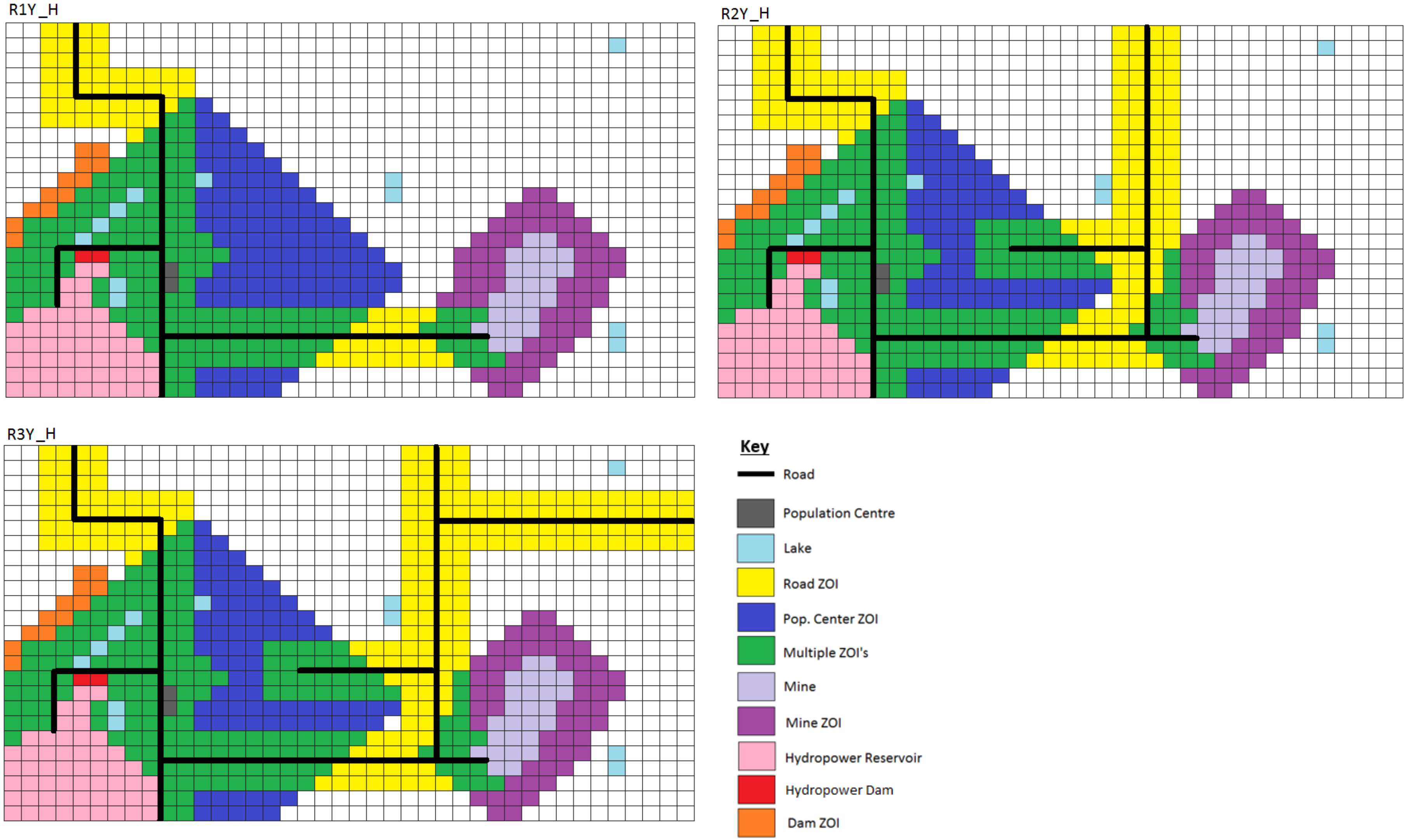

Figure 4.5. Landscape map of the model used in this study, with each square representing an area of 0.5 km^2. Coloured squares indicate infrastructure and landscape features as well as their zones of influence as described in the key. In addition to the population centre and roads, both a mine and hydropower station are present. R1Y_H indicates a map with one through road and a mine and hydropower station present, R2Y_H the same but with two through roads, and R3Y_H with three through roads.

Physical Lichen Loss

In addition to loss of potential grazing from avoidance behaviour by reindeer, the presence of infrastructure in an area creates a physical loss of lichen due to e.g. laying concrete and tarmac to create the structures. Within the model it was assumed that all roads had two lanes, being a total of 8 m in width. Road width was multiplied by the road length, as seen in Figures 4.2-4.5, giving the area covered by road. The ground being covered was always assumed to be old forest, for consistency, which is made up of 87514 kg km^{-2} of lichen. From this the biomass of lichen lost due to road infrastructure within each model scenario was calculated (Table 4.8).

Table 4.8. The biomass of lichen lost in the model landscape due to the physical road structures constructed. Road length is multiplied by 0.008, indicating the 8 m width that all the roads are assumed to have.

Scenario	Length (km)	Road area (km^2)	Lichen lost (kg, area × old forest lichen)
R1N	15.0	0.120	10503
R2N	37.0	0.296	25907
R3N	49.5	0.396	34659
R1Y	24.5	0.192	16804
R2Y	38.5	0.308	26957
R3Y	51.0	0.408	35709
R1N_H	20.0	0.160	14004
R2N_H	42.0	0.336	29407
R3N_H	49.5	0.396	34659
R1Y_H	29.5	0.232	20305
R2Y_H	43.5	0.348	30458
R3Y_H	51.0	0.408	35709

The physical structure of the mine covered 26 landscape squares. The maturity of the forest within these squares, and thus the biomass of lichen present without the mine was assumed to correlate to the ratios in Table 4.3. The total biomass of lichen lost due to the construction of the mine was calculated by multiplying the number of sites affected within each maturity class of landscape square with the biomass of lichen found within each forest maturity class (shown in Table 4.9). This was repeated for the lichen lost due to the presence of a hydropower dam and reservoir, with this structure occupying 55 landscape squares (Table 4.9).

Table 4.9. The biomass of lichen lost due to the construction of a mine or hydropower station and reservoir within the model landscape. Lichen biomasses are calculated based on the level of forestry occurring within the landscape and the subsequent ratio of forest maturity classes within each landscape square affected

			Mine		Hydropower Station	
Level of Forestry (%)	**Forest maturity**	**Lichen per square**	**Sites lost**	**Total Lichen lost (Tonnes)**	**Sites lost**	**Total Lichen lost (Tonnes)**
20	C	20269	1	1099	3	2307
	Y	35419	2		4	
	I	44636	2		4	
	O	43761	21		44	
40	C	20269	2	1061	4	2246
	Y	35419	4		9	
	I	44636	4		9	
	O	43761	16		33	
60	C	20269	4	999	7	2145
	Y	35419	6		13	
	I	44636	6		13	
	O	43761	10		22	
80	C	20269	5	961	8	2085
	Y	35419	8		18	
	I	44636	8		18	
	O	43761	5		11	

4.2.3 Snow Impacts

The impact of snow upon grazing was included within the model, drawing on data from chapter 3 of this thesis in which the lowest proportion of cratering occurred in March, the period of greatest food stress for reindeer and thus a potential bottleneck to their survival. According to data in chapter 3, considered to be collected in a broadly ‘normal’ year climatically, a mean of 15 % of pastures experiences ground-lying or ‘*båddne’* ice layers. Anecdotal evidence from herders states that there is always some loss of forage availability during winter from snow, but that there is a notable difference between ‘normal’ and ‘bad’ years. Therefore, a year considered ‘bad’ climatically would have more than 15 % of the ground experiencing icing that completely bars reindeer access to ground-lying lichens. The severity of these ‘bad’ climatic years are likely to vary, but for the purposes of this model, a ‘bad’ year was standardised to 35 % less lichen availability that in a ‘normal’ year, broadly reflecting anecdotal comments from herders. However, this is an approximation and is a figure that would benefit from greater research on the extent of icing during poor weather. A more severe extent of icing (50 %) was also incorporated into the model as a differing climate scenario, but this will be discussed later in the methods.

Using this assumption, that a 'bad' year would have 35 % lower presence of food/accessibility to grazing, we can calculate the results seen in Table 4.10 on accessibility of forage through snow.

Table 4.10. Using data of reindeer cratering evidence in March 2020 from chapter 3 of this thesis, the percentage of sites that were deemed possible for reindeer to graze in within each forest age group is noted. 'Normal' data are from the study in Chapter 3, whilst 'bad' data are the same but with an additional 35 % of the area deemed impossible to graze due to a predicted increase in icing.

Forest maturity	Proportion of area deemed grazable (%)	
	'Normal' March	'Bad' March
Clear-cut	0	0
Young	67	44
Intermediate	67	44
Old	50	33

The percentage of the landscape that remained available to graze during snow cover, relative to the level of forestry occurring, was calculated. This was done by multiplying the ratio of landscape square maturity classes from Table 4.3 by the grazability of each forest maturity as seen in Table 4.10. These numbers were added together for each level of forestry. This process was repeated assuming 100 % grazability within all sites, and the real grazability was then calculated as a proportion of 100 % grazability, giving an overall percentage of sites available to graze within each level of forestry (Table 4.11). The factor to convert between the mean 'normal' landscape possible to graze (51.8) and the mean 'bad' landscape possible to graze (39.6) was 76 %, used later within this model's calculations.

Table 4.11. The mean total area deemed possible to graze within each scenario of forestry extent and within a year considered 'Bad' in climate and 'Normal' in climate. The ratio of forest squares as seen in

Table 4.3 were multiplied by the percentage possible to graze (data in Table 4.10) and the sum of these were divided by the sum of the ratios assuming 100 % possibility of reindeer grazing, producing the mean total proportion of the landscape that is possible to graze within each scenario.

Level of Forestry (%)	**Climatic conditions**	**Ratio of forest maturity squares to one another (Ratio × % grazable)**				**Mean proportion of landscape possible to graze (%)**
		Clear-cut	Young	Intermediate	Old	
20	'Normal'	0	134	134	975	50.7
40	'Normal'	0	134	134	375	51.4
60	'Normal'	0	134	134	165	52.2
80	'Normal'	0	134	134	60	52.9
20	'Bad'	20	174	174	1365	42.2
40	'Bad'	20	174	174	525	40.5
60	'Bad'	20	174	174	231	38.7
80	'Bad'	20	174	174	85	36.9

4.2.4 Dataframe Used in the Model

To calculate the total biomass of lichen present in each scenario for reindeer to potentially graze, the loss of lichen from avoidance behaviour and from the physical presence of infrastructure was combined. This was subtracted from the biomass of lichen present in the absence of infrastructure (Appendix B). A dataframe was then produced to be used in the construction of the model, using the 'lichen remaining' data in Appendix B. These data were multiplied by the percentage grazable figures in a 'Normal' year from Table 4.11 to include the effect of snow cover on the availability of lichen. The contents of the dataframe are shown in Table 4.12. Lichen abundance in a scenario with no forestry or infrastructure, yet with the grazing barrier of snow in a normal year was 22,415,120 kg. This was later used to calculate the percentage loss of lichen due to infrastructure when compared to this 'original' (disturbance-free) scenario.

Table 4.12. Table of data used within the dataframe which comprises the basis of the ecological model in this study. Column titles depict the scenario in terms of infrastructure whilst row headings show the percentage of forestry occurring within the landscape. Numbers within the dataframe show the abundance of available lichen in the simulated landscape in kilograms, this being lichen presence data from Appendix B multiplied by percentage grazable data in a 'normal' year from Table 4.. Column titles 'nothing', nothing2' etc... were necessary gaps for the formatting of the dataframe so that it performed its function correctly in the model.

	lichen_r1N	lichen_r2N	lichen_r1Y	lichen_r3N	lichen_r2Y	lichen_start	nothing	nothing2	lichen_r3Y
l_20	16307850	14845588	14595673	10053723	13639293	15298167	0	0	12881934
l_40	16087854	14643692	14402122	10020741	13452273	15509384	0	0	12734207
l_60	15890392	14466807	14266003	10018095	13316714	15750775	0	0	12618143
l_80	15623734	14279772	14041286	9989709	13139623	15961993	0	0	12435424

…cont

lichen_r1NH	lichen_r2NH	lichen_r1YH	lichen_r3NH	lichen_r2YH	nothing3	nothing4	nothing5	lichen_r3YH
15216754	13742581	13525603	13599142	12548198	0	0	0	12066183
15022928	13565361	13366482	13418050	12394253	0	0	0	11919030
14835586	13439372	13247682	13296718	12289278	0	0	0	11838054
14587648	13243687	13052020	13106163	12124442	0	0	0	11675662

4.2.5 Model Construction

Using the data outlined above, a model was created which runs for 50 years, producing data on reindeer survival and lichen abundance throughout this period. A period of 50 years was chosen as this would be an approximate working lifespan of a reindeer herder. The model was constructed using the software R version 4.1.0 (R Core Team 2020).

Inputs

According to Norway's reindeer husbandry act, herds of less than 250 reindeer per siida (herding group) are considered uneconomical (Lov om reindrift 2007); so, within the present model, a reindeer herd below the size of 250 animals is considered unsustainable. The options of initial reindeer population size in the area available within this model are 500, 2500, 5000, 7500 and 10,000 individuals.

The number of reindeer slaughtered each year can be varied, options including 25, 50, 100, 250 and 500 reindeer being killed per annum. 500 is the maximum as it is equal to the minimum number of reindeer present in the landscape. The proportion of the landscape experiencing logging, as with the data shown earlier, can be varied between 20 %, 40 %, 60 % and 80 %. There is an option to have a mine and/or a hydropower station with reservoir present or absent within the model, and the extent of road infrastructure can be varied between levels "1", "2", and "3", corresponding to one through-road, two through-roads and three through-roads respectively, with some additional side roads present (shown in Figures 4.2-4.5).

In addition to infrastructure impacts, the effect of snow on reindeer grazing in four climate scenarios was included. Climate scenario 1 (C1) is a business as usual scenario, including periodic 'bad years' with greater icing which reduce the ability to graze at all sites to a value of 76 % of what it would be in a 'normal' year. These 'bad' years were inserted every five years in the model, which runs for a total of 50 years. A five-year interval was chosen as a study in the island of Svalbard noted four poor years causing extensive icing within a 20-year period (Hansen *et al.* 2019). Though this is quite a different environment from Subarctic northern Fennoscandia, being located much further north with different dominant flora and climate, as well as a lack of reindeer herding by humans occurring in the area, we can use it as a best-available, initial, approximation that a ‘bad’ year occurs at five year intervals, which in this model were chosen to be years 3, 8, 13, 18, 23, 28, 33, 38, 43 and 48, respectively.

Another climate scenario (C2) saw the frequency of these ‘bad years’ increase over time, with one event in the first decade, two in the second decade, three in the third, four in the fourth and finally five ‘bad years’ during years 41-50 in the model. When these occurred within their decade was randomised. The subsequent climate scenario (C3) saw an increased severity of icing during the ‘bad years’, these still occurring every five years as in C1 but where only 50 % of grazing was accessible. In the final scenario (C4) frequency of ‘bad years’ increases as in C2 and the icing is more severe as in C3.

Model Calculations

The initial biomass of lichen available is based upon the scenario selected. In the first year this is multiplied by the lichen growth rate of 16 % per annum, and then the biomass of lichen which has been grazed away by reindeer is subtracted. This biomass of lichen already grazed away is calculated by subtracting the yearly slaughter from the initial reindeer population and multiplying this number by 298.62 (kg) for the lichen requirement of this population. The results show the biomass of lichen remaining available for the following year.

If the reindeer population is above the carrying capacity of the environment, i.e. if the lichen requirements of the population are greater than the lichen available in the model landscape, the reindeer population in the following year will be limited to this carrying capacity, with the number of reindeer exceeding this being assumed to starve. If the reindeer population is below the carrying capacity, the following year’s population is the current year’s population added to its growth rate of 4 % per annum. Depending upon which climate scenario is selected, there may be a change in the amount of lichen that is functionally available for reindeer to graze, although it is not lost from the environment, but simply unusable for that year (e.g. if an icing event occurs).

The birth of reindeer when the population is below the carrying capacity is subjected to a two year time lag as reindeer mate in spring, gestate for seven and a half months and take a further four months to wean their calves, meaning that only in their second year after surviving a winter will females have calved and their calves reached the age where they will contribute to grazing. This assumes within the normal birth/death rate parameters that the same proportion of pregnant females survive each year. Lichen abundance in the model was capped at 22,415,120 kg which is the calculated 'best possible' lichen scenario for the environment as it is assumed lichen cannot increase in abundance indefinitely.

A function of this model was created within the coding software R, and then a loop applied which ran through each different combination of the following: initial reindeer population, number of reindeer slaughtered each year, presence or absence of a mine and/or hydropower station, the extent of road development in the landscape, the proportion of forest being logged and climate scenario. Each of these model scenarios produced a final number of reindeer present and the abundance of lichen remaining in the landscape after 50 years. The R script used to create the model is shown in Appendix C. A schematic of the model can be seen in Figure 4.6 and the equations used within the model are shown in Equation 4.2 to 4.5.

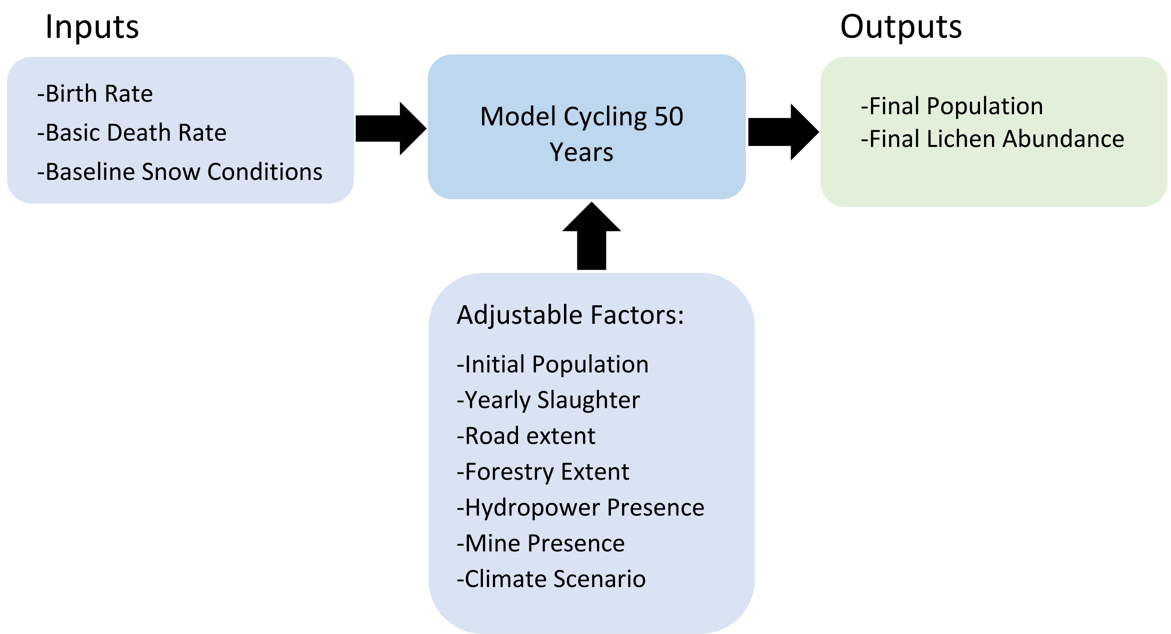

Figure 4.6. Schematic of the landscape model

Statistical Analysis

Initial changes in lichen abundance due to infrastructure, before the running of the model over 50 years, was tested by comparing the lichen abundance available (as seen in Table 4.12) compared to a null scenario with assumed 100 % cover of old forest and no human infrastructure present. This null scenario was still subjected to the effects of snow in a 'normal' year (mean 52 % accessible), creating 22,415 tonnes of lichen within the model landscape. The impacts of forestry, roads, hydropower stations and mines on the percentage loss of lichen from the null were calculated using a Kruskal-Wallis Test followed by a Dunn's test using the statistical software R, version 4.1.0 (R Core Team 2020). The Kruskal-Wallis test was used due to the data being non-parametric according to a Shapiro-Wilk test.

The model was cycled through all the potential scenarios, producing a dataframe of the reindeer population and abundance of available lichen after the passing of 50 years in the simulated landscape. Linear regression models were carried out on model data to compare the effects of variables within each factor on final reindeer population and lichen volume after 50 years. The factors included were extent of forestry, road extent, presence of a mine, presence of a hydropower station, climatic scenario, number of reindeer slaughtered yearly and the initial reindeer population.

Model runs a cycle of 50 years (T50). At each time step (T_i):

Equation 4.2. Lichen (N) at T_{i+1} is the lichen added to its yearly growth (N_GRO), with the lichen grazed away by reindeer subtracted. R_i indicates reindeer population, SL number slaughtered, and F lichen grazed by an individual reindeer.

$$\boldsymbol{Ni+1 = (Ni \times N\_GRO) - ((Ri - SL) \times F)}$$

Equation 4.3. The carrying capacity (CC) of reindeer in the environment is the lichen abundance divided by lichen requirement of an individual reindeer

$$\boldsymbol{CC = Ni/F}$$

Equation 4.4. If reindeer population (R_i), minus the animals slaughtered is greater than the carrying capacity, the following year's population will be limited to the carrying capacity of the environment.

$$If \quad \boldsymbol{Ri - SL > CC}$$

$$then\ \boldsymbol{Ri + 1 = CC}$$

Equation 4.5. If the reindeer population is below carrying capacity, the following year's population will be the current population multiplied by the population growth rate (R_GRO).

$$Otherwise\ \boldsymbol{Ri + 1 = Ri * R\_GRO}$$

4.3 Results

4.3.1 Initial lichen presence

When infrastructure is simply placed on the model environment, before running the model over 50 years, the maximum percentage loss of lichen compared to a null landscape is 55.4 % (Figure 4.7). Greatest loss of lichen occurs in scenario 16 which had the largest extent of forestry (80 %) and roads (level 3) but an absence of a mine or hydropower station. The minimum lichen loss is 5.2 %, this coinciding with the scenario with minimal road extent and forestry, and an absence of a mine or hydropower station. 48 out of the initial 52 scenarios have a lichen loss of over 25 %.

Forestry did not have a significant effect on the initial loss of lichen (Dunn's test, $p > 0.05$). However, road extent, mine presence and hydropower station presence all had significant effects (Dunn's Test, $p < 0.05$; see Table 4.13). According to the Z-statistics, presence or absence of hydropower stations had the lowest impact on lichen loss ($z = -2.00$), showing greater loss when the hydropower station was present. The same pattern was seen with mine presence ($z = -3.27$). Increased road extent had larger effect on lichen loss compared to low levels or absence of roads (see Table 4.13).

Table 4.13. The effect of each factor on the loss of lichen from a null scenario. Shown are the p-values and Z-statistics of all factors that were significant according to a Dunn's Test. Degrees of freedom are from a preceding Kruskal-Wallis Test. Hydropower station and mine factors are "yes" and "no" for presence, whilst road factors are 0 (no roads) followed by levels 1, 2 and 3 of road extent which are further clarified in the methods.

	Factors differing significantly	Degrees of Freedom	Dunn's Test Z statistic	Dunn's Test Adjusted P-Value
Hydropower	N-Y	1	-2.00	0.045 *
Mine	N-Y	1	-3.27	< 0.001 ***
Road (level of road extent)	0-2	3	-2.98	0.017 *
	0-3		-4.57	< 0.001 ***
	1-3		-4.61	< 0.001 ***
Forestry	-	3	-	> 0.05

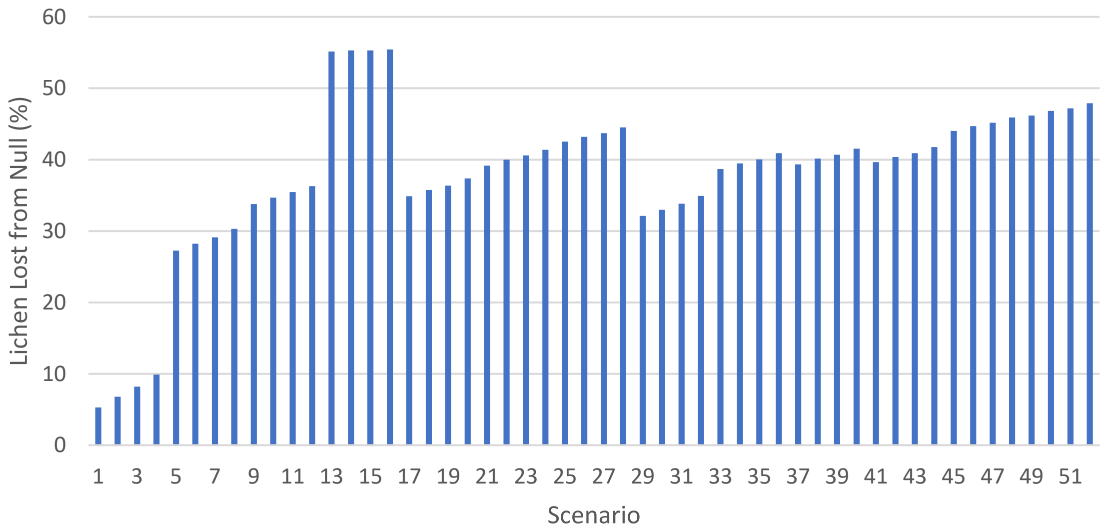

Figure 4.7. Loss of lichen in each model scenario before the running of the model. Loss is shown as a percentage of that present in a null scenario with no infrastructure. The abundance of lichen in the null scenario is 22,415,120 kg.

4.3.2 Final reindeer population

After running the model, the final reindeer population was significantly affected by the climate scenarios, road extent, the number of reindeer slaughtered yearly and the initial population size (all $p < 0.05$; Table 4.14). A linear regression model showed that increases in both initial population size had a positive effect on the final reindeer population (coef = 0.1 and R^2 0.127), as did the number of reindeer slaughtered (coef = -5×10^{-9} and R^2 0.127; Table 4.14).

A linear regression model carried out on road extent also showed that road levels 1 and 3 significantly affected final reindeer population, with road level 1 leading to the highest mean final reindeer population at 489, followed by level 2 at 434 and level 3 at 369. However, once again the explanatory R-squared value was small ($R^2 = 0.002$; Table 4.14).

Climate scenario on the other hand accounted for 44 % of the variation in final reindeer population when a linear regression model was applied ($p < 0.001$ for all scenarios, Table 4.14). C1 had the greatest mean final reindeer number at 1426 animals, followed by C2 (increased frequency of icing events) at 254, C3 (increased severity in impact) at 44 and C4 (increased frequency and severity) at 1 (Table 4.14; Figure 4.8).

Forestry and the presence of a hydropower station or mine did not have a significant effect (Table 4.14). Across all scenarios there was a loss of reindeer ranging between 54 % and 100 %. Only 25 % of scenarios resulted in an economically sustainable reindeer population greater than 250 in size.

Table 4.14. Results from a linear regression model showing the effects of a variety of factors on reindeer population after 50 years. Road extent and climate scenarios (C1, C2, C3 and C4) are further described in the methods. For most factors the coefficient represents the slope of the line within the linear regression. For the four climate scenarios, the coefficient for C1 is the mean final reindeer population for scenario C1 whilst the coefficients of C2, C3 and C4 are the difference between their mean and the mean of C1. This is the same for road levels with the coefficient of road levels 2 and 3 being the mean compared to the coefficient of road level 1. When appropriate third order polynomial regressions were used within the analysis as indicated within the table.

Factor	Coefficient	D.F	P-value	Test	Adjusted R^2	F-statistic
Hydropower presence	-8.4	2878	> 0.05	ANOVA	-	-
Mine presence	-28.8	2878	> 0.05	ANOVA	-	-
Forestry	-0.5	2878	> 0.05	ANOVA	-	-
Number Slaughtered	0.1	2878	0.013 *	1st Order Polynomial	0.002	6.2
Initial Population size	-5×10^{-9}	2876	< 0.001 ***	3rd Order Polynomial	0.127	140.9
Road level 1	489.4	2877	< 0.001 ***	ANOVA	0.002	4.5
Road level 2	-55.0	2877	> 0.05	ANOVA		
Road level 3	-119.7	2877	0.003 **	ANOVA		
C1 (business as usual)	1426.3	2876	< 0.001 ***	ANOVA	0.444	767.2
C2 (frequency)	-1172.3	2876	< 0.001 ***	ANOVA		
C3 (severity)	-1382.8	2876	< 0.001 ***	ANOVA		
C4 (both)	-1425.7	2876	< 0.001 ***	ANOVA		

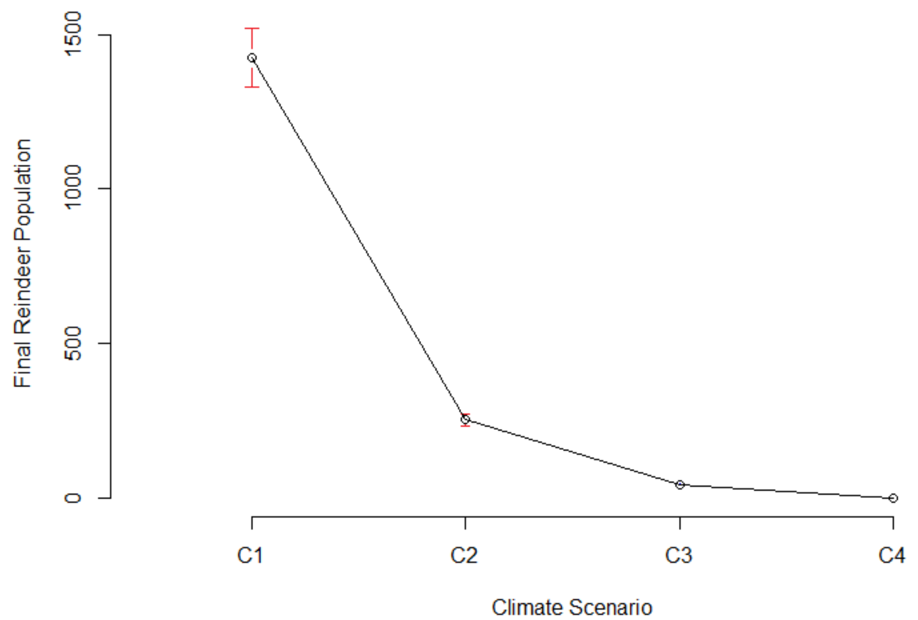

Figure 4.8. The mean final reindeer population in relation to each climate scenario. Red bars indicate the standard errors. Bars are not present on C3 and C4 due to small standard errors. C1 is a business as usual climatic scenario, C2 is a climate scenario with increased frequency of icing events, C3 has events of an increased severity in impact and C4 is a combination of increased frequency and severity.

4.3.3 Lichen loss after running the model

After running the model, the greatest loss of lichen compared to the null scenario was 99.8 %. The smallest was a loss of no lichen, with it having regrown throughout the model to achieve the best possible abundance. Increased yearly reindeer slaughter caused a decrease in lichen loss, with a linear regression model showing an adjusted R-squared value of 0.08 (coef. = -2971, $p < 0.001$, Table 4.15). Increases in the initial population size on the other hand also caused an increase in lichen loss, with a linear regression model showing a coefficient of 454 ($p < 0.001$; $R^2 = 0.08$; Table 4.15). Road levels 1 and 3 had a significant impact on final lichen loss, with a mean loss of 947,808 kg and 1,449,597 kg respectively ($R^2 = 0.01$; Table 4.15). Finally, all climate scenarios had a significant impact on lichen loss with the greatest loss being seen due to C1 at 2,834,727 kg of lichen lost (Table 4.15). Forestry and the presence of hydropower stations or mines did not have a significant impact on final lichen loss according to linear regression models ($p < 0.05$; Table 4.15).

Table 4.15. The effect of each factor on lichen loss after 50 years from linear regression models. Road extent and climate scenarios are further described in the methods. For most factors the coefficient represents the slope of the line within the linear regression. For the four climate scenarios, the coefficient for C1 is the mean final reindeer population for scenario C1 whilst the coefficients of C2, C3 and C4 are the difference between their mean and the mean of C1. This is the same for road levels with the coefficient of road levels 2 and 3 being the mean compared to the coefficient of road level 1. When appropriate second order polynomial regressions were used within the analysis as indicated within the table.

Factor	Coefficient	D.F	P-value	Test	Adjusted R^2	F-statistic
Hydropower presence	-54752	2878	> 0.05	ANOVA	-	-
Mine presence	-42909	2878	> 0.05	ANOVA	-	-
Forestry	2233	2878	> 0.05	ANOVA		
Number Slaughtered	-2971	2878	< 0.001 ***	2nd Order polynomial	0.084	264.7
Initial Population size	454	2878	< 0.001 ***	1st Order polynomial	0.084	264.7
Road level 1	947808	2877	< 0.001 ***	ANOVA	0.001	2.271
Road level 2	143808	2877	> 0.05	ANOVA		
Road level 3	501789	2877	0.037 *	ANOVA		
C1 (business as usual)	2834827	2876	< 0.001 ***	ANOVA	0.043	44.01
C2 (frequency)	-2778198	2876	< 0.001 ***	ANOVA		
C3 (severity)	-2579240	2876	< 0.001 ***	ANOVA		
C4 (both)	-1329843	2876	< 0.001 ***	ANOVA		

4.4 Discussion

4.4.1 Effects of infrastructure and forestry on initial lichen loss

The loss of lichen in the simulated landscape prior to running the model followed expectation, with more infrastructure generally leading to greater loss of lichen. The only exceptions were scenarios 13-16, which had the following settings: no mines or roads present, level 3 of road extent, and forestry scenarios including 20 %, 40 %, 60 % and 80 % of the landscape experiencing commercial silviculture. Scenarios 13-16 had the highest loss of lichen although the reason is unknown. All scenarios had a lower abundance of lichen present in comparison to the null landscape where infrastructure and forestry are absent.

Forestry did not have a statistically significant effect on initial lichen loss. This follows on from data in chapter 2 of this thesis which highlights that whilst lichen abundance varies broadly in relation to forest maturity, the heterogeneity of these patterns means the differences are not statistically significant. This may seem at odds with past studies arguing the importance of old forest and concerns about the negative impacts of perceived as excessive commercial logging on reindeer grazing (Berg *et al.* 2008; Kivinen *et al.* 2010; Moen and Keskitalo 2010). However, as explored in chapter 3, the majority of the impacts of forestry on grazing come from the effects of forest structure on snow conditions, altering how much forage reindeer can reach. As initial lichen loss due to presence of infrastructure does not take into account snow conditions, the effects of forestry cannot be seen.

All forms of infrastructure measured here, these being road extent and the presences of a hydropower station and/or a mine, did have significant effects on lichen lost from the landscape before running the model, with the greatest impacts coming from road levels 2 and 3 followed by the presence of mines.

4.4.2 Factor impacts on model outcomes

After running the model for 50 years, the final reindeer population remaining in the landscape was impacted by a variety of factors. The initial population size affected how quickly reindeer reached the carrying capacity of their environment and thus if/when the population subsequently crashed. It would also impact how long after the crash the population would have had time to recover once more. This factor accounted for 12.7 % of the variation within the model. Yearly slaughter on the other hand provides a regular control on the reindeer population number, and thus a higher yearly slaughter rate means it will take a longer period for the carrying capacity of the environment to be surpassed, creating a positive influence on population dynamics, much like those seen in Hansen *et al.* (2019). Road extent, having the greatest initial impact on lichen loss from the landscape, would have altered where the carrying capacity lies more strongly than the other forms of infrastructure, thus explaining its significant effect on final reindeer population. However, the R-squared value showed that together road extent and yearly slaughter only accounted for 0.4 % of the variation within the model.

As for lichen loss from the landscape after running the model, once again number of reindeer slaughtered, initial population size and road extent all had a significant effect. Increases in the number of reindeer slaughtered decreased lichen loss in the form of exponential decay, as with fewer reindeer present less grazing could occur. Conversely an increased initial population size would lead to greater grazing, and a wider extent of roads would mean more physical loss of lichen from infrastructure. Altogether these accounted for 16.9 % of variation within the model.

4.4.3 Climate

Another factor significantly affecting both final reindeer population and final lichen loss after running the model was climatic conditions. The business as usual scenario (C1) resulted in the largest final reindeer population. As climatic conditions became more extreme, with icing events increasing in frequency, severity, or both, the final reindeer population was reduced. Increased severity caused a greater decrease in final reindeer population compared to increased frequency of icing events. The impacts of climate scenario on lichen loss mirrored these scenarios. The greatest lichen loss was seen in C1, as this was also when there was the greatest presence of reindeer leading to higher grazing. Loss of lichen decreased subsequently as climate scenarios allowed for lower final reindeer populations.

As previously mentioned, some studies have shown that extreme weather events can be beneficial to reindeer populations, stabilising the number of animals (Hansen *et al.* 2019). Here however, scenarios with more extreme climate conditions than currently experienced have a detrimental effect on reindeer over time, suggesting that any stabilising effects of climatically induced die offs may only work within very specific circumstances. The population studied in Hanssen et al. (2019) is also a wild population. Even if high mortality rates may have some stabilising effects, the deaths of many semi-domesticated reindeer in Sweden would cause additional problems to reindeer herders if they are losing a significant portion of their herd, affecting their income and ability to continue herding.

All scenarios here showed a 54-100 % loss of the reindeer population by the end of 50 years, and only 25 % of scenarios resulted in economically sustainable reindeer populations of > 250 animals as delineated by Lov om reindrift (2007). This suggests the reindeer population overall is at risk of experiencing a concerning decline. According to the R-squared value, climate here accounted for 44.4 % of the variation in final reindeer population, with this being based on only four hypothetical climatic scenarios. Research by Kohler and Aanes (2004) found that ground ice thickness in winter accounted for 80 % of the variation in the local reindeer population growth rate in Svalbard. Their study, and the results of this model highlight the importance of gaining a greater understanding of the effects of extreme icing events on reindeer as well as developing ways to adapt to these changes.

Icing events only accounted for 4 % of the variation in lichen loss, meaning of all the factors mentioned, they only account for 21 % of lichen loss statistically. This could be because the response of lichen abundance to factors such as reindeer population change experiences a lag before populations hit carrying capacities, meaning linear and polynomial regression models will not pick up these complex relationships.

4.4.4 Model justification and data omissions

A deterministic model was used in this study, as due to the inherent complex range of data sources and the heavy use of mean figures in data, as stochastic and agent based modelling approaches were deemed to add to much additional uncertainty to the accuracy of the model outcomes. Models based more heavily on ground-truthed case studies, e.g.

measuring lichen abundance in forests adhering to differing scenarios, could not be undertaken due to time and funding constraints. Construction of the model in Rstudios was done as the author had limited coding knowledge outside this coding software, and by using the ShinyApp function, the author was able to create the interactive interface for the model to more easily select and visualise each scenario manually during model construction to note any flaws or unusual patterns in the data quickly.
As with all ecological models, whilst built to serve a certain function, this model cannot encapsulate the entire complexity of the landscape and so some data omissions occurred which are worth mentioning and discussing. When building the model, past literature on reindeer ZOIs from infrastructure were gathered for use in the model parameters, but some studies were excluded due to being incompatible with this research. For example, some studies were undertaken during the calving season when females with calves are likely to show greater avoidance behaviour than in winter, making it a less accurate proxy for infrastructure avoidance (Wolfe *et al*. 2000). Details on studies omitted can be found in Appendix D.

The impacts of wind farms were also omitted. All the studies identified on the topic in Fennoscandia were deemed insufficient to provide data for this study, as the data on reindeer avoidance was either collected during the calving season (Skarin and Alam 2017; Skarin *et al.* 2018), in very small enclosed areas (Flydal *et al.* 2004) or in very specific parts of the landscape such as peninsulas where reindeer movement was highly influenced by topography (Colman *et al.* 2012; Colman *et al.* 2013) making inferences about the impact of the wind farms more difficult to make. For this reason, the impacts of wind farms were omitted from this model, but with greater data in future it would be a valuable factor to add.

On the other hand, some data was included which may otherwise have been omitted. To maximise the utilisation of scarce data, results from studies in Norway, Finland, and Canada were included in the model parameters as well as those from studies 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 reindeer live in a different environment with greater numbers of predators and very little active reindeer herding taking place, differing much 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.

Another piece of information which was not so much omitted as simplified related to the proportion of lichens within the reindeer's diet. Chapter 2 of this thesis found that for each percentage increase of lichen cover, edible species cover decreases by 2.9 %, a pattern also seen in Kojola *et al.* (1995). However, here we assumed the proportion of edible forage species would vary in tandem with changes in lichen within the model environment. At the same time, many 'edible' species such as *Empetrum* spp., *Vaccinium myrtillus* and *Pleurozium schreberi*, are strongly not favoured by reindeer, likely due to their high content of acid detergent fibre which reduces their digestibility compared to lichens (Danell *et al.* 1994). This means that other edible species may be present in lower abundance with increasing lichen population, and may remain largely uneaten by reindeer aside from in emergency situations. Overall, these relationships were deemed too complex to include in the model and so it was assumed that the proportion of lichen within the diet remains constant throughout winter. The proportion of lichen within the diet is already a mean number, and so any increases or decreases in proportion were assumed to average out to this mean.

4.4.5 Impacts on migration

The effects of habitat fragmentation were not considered in the model but deserve attention nonetheless. Whilst variations in traffic levels between seasons does affect reindeer avoidance of roads (Dyer *et al.* 2001), there is evidence that without a physical barrier, reindeer will often continue to cross even though they will not linger to graze within close proximity (Reimers *et al.* 2007). The desire to continue following an established migration route remains strong despite increases in traffic or decreases in reindeer population (Reimers and Colman 2006; Reimers *et al.* 2007), meaning without physical barriers roads do not necessarily cause habitat fragmentation.

The desire of reindeer to stay in close proximity to traditional migration routes is notable especially in relation to hydropower reservoirs (Dahle *et al.* 2007). Some of the major impacts of constructing hydropower stations include loss of shoreline habitats to flooding, artificial shoreline constructions, and dangerous ice conditions because of shifting currents and water levels (Mahoney & Schaefer, 2002). River valleys are often an integral component of reindeer migration routes so their change, and the instability of the ice covering the reservoirs in winter which the reindeer must now cross, can become a barrier to traditional migration routes (Sandström 2015).

Evidence shows the natural reaction of reindeer to poor climatic conditions for their grazing is to move elsewhere to find forage (Stein *et al.* 2010). However, in northern Sweden reindeer herding areas are tightly regulated, and so the animals are limited in their movements. Further cluttering of the landscape with a greater abundance of infrastructure, and subsequently with ZOIs covering larger areas, may not allow natural stopping and resting points for reindeer, and thus may become an issue to their ability to move or find other areas to graze.

4.4.6 Supplementary Feed and Slaughter Rates

Two other factors which undoubtedly have an effect on reindeer winter ecology, but which were not included in the model, were supplementary feeding and variable slaughter rates. In the model it was assumed that slaughter rates would remain constant due to the reindeer herder selling the meat requiring a stable income year to year. In reality this stability does not always occur. Some years more reindeer will be slaughtered, perhaps from economic need or due to the herd being larger than required or desired. Alternatively, fewer reindeer may be slaughtered despite this creating an unprofitable situation for the herder, for reasons such as to maintain herd size. Indeed, many reindeer herders and herding families have secondary jobs to supplement their income when herding is unprofitable. Variable slaughter rates through time were not included in the model as they are based on very individual decision making by herders, a level of complexity which was deemed too difficult to include at this stage.

The effects of supplementary feeding was also excluded. Use of commercial feed to help reindeer survive through difficult grazing periods is becoming increasingly common (Kojola *et al*. 1995; Majala and Nieminen 2001; Oksanen 2001). It is a very costly practice, and has been seen to lead to poor health outcomes for the animals in terms of disease from poor hygiene and overcrowding as well as through causing digestive issues (Majala and Nieminen 2001; Oksanen 2001). However, it does allow a larger proportion of the herd to survive during certain years with unfavourable grazing conditions. The decisions surrounding supplementary feeding are once again highly individual to each reindeer herder, so this was deemed too complex a factor to include within the model.

The decision making of herders nevertheless is important, as it has allowed reindeer herding to survive through much change over the past few centuries. Principles of flexibility, diversity and mobility which are embedded in traditional reindeer herding are seen as essential for the health of this livelihood (Turi 2008). There has been much criticism of the increasing state regulation of reindeer herds in Norway, based on a more static agricultural and production based model, rather than allowing as much room for the flexibility and adaptability of traditional herding (Turi 2008; Tyler *et al.* 2007; Löf 2013). Even basing reindeer herd management around carrying capacities, viewing an environment somewhat statically rather than cyclically, has been criticised. It has been said to undermine the resilience of the reindeer's ecosystem compared to when traditional herding methods and philosophies are used to manage herds overall (O'Brien *et al.* 2009; Tyler *et al.* 2007).

4.4.7 Link to interactive model

An interactive online version of this model has been created to help users visualise what is happening in the simulated environment. The parameters are the same, apart from icing event occurrences are not randomised but are set. In C1 and C3 icing events occur every five years from year three to year 48 as in this chapter. For C2 and C4, when climate events increase in frequency through time, the events occur on years 3, 8, 13, 18, 23, 25, 28, 33, 35, 38, 40, 41, 43, 45, 47 and 48. These static occurrences of icing events, rather than the more randomised occurrences used in the chapter, were done for ease of interactive model construction. The interactive model is best viewed on a laptop or PC device, or a tablet with a large screen, and can be found here or at:
https://reindeerecology.shinyapps.io/ThesisReindeerModel/

4.4.8 Conclusion

This model shows that, aside from direct factors such as initial reindeer population and yearly slaughter number, climate has the greatest influence on the final reindeer number after a period of 50 years. Presence of a mine and hydropower station did significantly reduce lichen volume, and thus the food available to reindeer, but this did not directly appear to affect their population. Similarly, forestry appeared to not significantly affect reindeer population in the long term. The low impact of infrastructure and forestry could be related to their temporal roles. The barriers to grazing caused by younger forests are exacerbated by icing events, so their effect may appear in this category instead, whilst the beneficial impacts of old forests would only occur in cases where little food and lichen is found elsewhere. These benefits are very valuable, yet the circumstances where these benefits are seen can be infrequent, so may not have been picked up by the model.

Infrastructure like mines and hydropower stations reduce the amount of lichen from the null notably, but once present do not appear to further suppress the potential reindeer population. This may be as the population will adjust to the new lower carrying capacity caused by reduced access to food within the first few years. Therefore, whilst infrastructure alters the carrying capacity of the environment, initial reindeer population size and factors causing reindeer mortality (e.g. slaughter rates and climatic events causing die offs) alter the time it takes to reach the carrying capacity and thus have a stronger effect on the final population through time. However, it should be remembered that the carrying capacity is not a static figure between or even within years.

This model gives a broad idea of strengths of relationships between factors affecting reindeer grazing in winter but doesn’t encapsulate the entire complexity of the environment. However, what is clear is that infrastructure can cause notable losses of potential grazing, up to almost 47 %, whilst general snow conditions as well as extreme climatic events have some of the greatest negative impacts on herds in the long term. Altogether these lead to a 54-100 % reduction in the reindeer population over the span of 50 years, suggesting that current forms of reindeer herding together with current levels of competition in land use with other industries are unsustainable, and something needs to change to maintain the future of this species and the herding way of life.

Chapter 5:

Learning to collaborate - Bringing together scientific and traditional ecological knowledge

"The imbalanced and biased western knowledge regime has led to overall problems. There must be a balance of knowledge in relation to the ecological and environmental factors. Due to their constant stay in nature, the Sámi and other Indigenous groups are able to make continuous observations that can be utilized in the overall monitoring of global change. One should encourage local observations of changes in relation to the environment and incorporate this and other TEK information into scientific databases… If we lose touch with TEK, we might also lose essential knowledge about our survival potential in the rapidly changing world."

– Dr Elina Helander, Sámi and researcher from Utsjoki

(Helander-Renvall and Mustonen 2004, p. 305)

5.1 Introduction

Throughout this thesis I have explored a number of scientific topics and reached some conclusions. Chapter 2 highlighted the complex relationship between forest maturity and understorey vegetation relevant to reindeer. This included themes of disturbance and recovery, competition, and physical barriers created by tree felling processes. Chapter 3 then went on to describe how the greatest effects of silviculture on reindeer grazing appear to come from how it impacts snow conditions rather than the physical presence of forage. Then chapter 4 focused on the cumulative effects of anthropogenic factors, modelled as reindeer survival over the span of 50 years in a multiple-use landscape. This model noted the impacts of various forms of infrastructure on lichen loss, and highlighted climatic changes as having the greatest negative impacts on reindeer survival over time.

Alongside the scientific conclusions, a number of anthropological conclusions were also made. Chapter 1 explored the complex history and power relations between Sámi reindeer herders and wider society in Fennoscandia, which continue to shape the lives of herders and their reindeer today. Chapters 2 and 3 noted some of the ongoing disputes between commercial forestry groups and herders, and highlighted the role economy plays in the decisions that are made about reindeer e.g. whether supplementary feed can be afforded. Chapter 4 then noted that despite our best efforts to scientifically quantify the ecology of reindeer in northern Sweden, factors such as the decision making of herders is also likely to be highly influential. This decision making is dynamic, responsive and unique to each individual or siida, and so cannot easily or statistically be modelled. Here then we have a thesis which has explored the science of an ecosystem, but acknowledges that economic, historical, political, social, and ultimately human factors play an inseparable role within this ecosystem too.

At this point it is valuable to discuss the process of using multidisciplinary approaches to research. The original plans for this thesis involved extensive collaboration with Sámi reindeer herders. However, much of this did not come to fruition due to a variety of circumstances, both within and out-with the power of the researcher to influence. In this chapter some of the opportunities and challenges surrounding the collaboration between Indigenous/local communities and researchers will be discussed, explored through the lens and experiences of the author, a non-Indigenous western researcher. Finally, some suggestions will be shared for others hoping to do collaborative research, to help them avoid some of the pitfalls whilst finding meaningful ways to cooperate in this important new form of integrative research partnership.

5.2 Brief overview of research relationships

Indigenous peoples have long formed a variety of sovereign nations across the planet. Depending on their role in the community, each individual has held detailed knowledge about their physical, ecological, social and spiritual environment, which have contributed to the broader knowledge culture and informed the practices of their nation. Understanding the natural world has been particularly vital for the survival of those practicing subsistence hunting, fishing, gathering and herding.

Despite this wealth of local experience, common modes of research historically and still today are colonial in nature, with data collection being undertaken in an area under colonial administration with little regard for the rights and interests of the local Indigenous nations (Clarke, 2007; Minasny *et al.* 2020). This kind of practice has often lead to information being misrepresented, and research being undertaken which was unethical and harmful to local communities, such as in the examples outlined later in this chapter (e.g. Laid and Noejovich 2002; Smith, 1999; Mosby 2013).

Today, this colonial trend in research practices can be seen in 'helicopter research', where scientists extract data from another country or area, once again with minimal input or collaboration with local researchers or individuals. The prior knowledge of these scientists on the area is often solely based upon the published work of other academic researchers who have also only spent a limited period of time there, and the data are subsequently processed and published elsewhere. The result is often that locals may never see the results of the study or experience any benefits from the work being done (Minasny *et al.* 2020; Santos 2008). Carlson (2017) argues that helicopter research perpetuates a very disconnected and detached theorizing, as outlined in Martineau and Ritskes (2014), and that new research must strive to go further to be anti-colonial in method. Indeed, Carlson argues that much work has been done in this direction already by some 'white settler' researchers, with examples ranging from applied natural science research projects by academics like

Sandström (2015), to a growing body of literature by Carlson herself, and academics like Absolon (2011) and Hart (2009) which critique and explore current Indigenous research methodologies. In addition to the practices of white settler researchers gaining scrutiny, more scientists studying the natural world are starting to recognise the value of knowledge beyond western science. This knowledge is often termed traditional ecological knowledge (TEK), and is associated with information and understandings held by Indigenous communities, although the detailed definition of TEK varies widely (see Whyte 2013). If incorporated into data interpretations, TEK can make research richer and more detailed, and can highlight gaps in scientific knowledge when trends or processes unknown to 'western' science are brought to light (Snively and Corsiglia 2001). It is also sometimes seen as an act of social justice to include Indigenous voices within studies, especially about the lands and ecosystems which they have traditionally inhabited (McKinley 2007; Ball and Janyst 2008).

Within the scientific literature, many articles recommend having greater collaboration or co-creation of projects with Indigenous and local people, and some scientists are already carrying this out in practice (e.g. Huntington 2000; Albert 2001; Reinert *et al.* 2009; Riseth *et al.* 2011; Pape and Löffler, 2012; Sandström *et al.* 2012; Riseth *et al.* 2020). Anecdotally, researchers long-established in their field shared at the Arctic Science Summit Week 2021 conference how they see that the new generation of scientists is much more aware of the ethics surrounding research, in terms of how they interact with local and Indigenous people, and are striving for greater collaboration and co-creation.

Despite many good intentions, the recent rush to include Indigenous perspectives in scientific work has led to some problematic methodologies being used, which have resulted in harm to the individuals and communities sharing their knowledge. TEK has been seen one-sidedly as a valuable asset for science, a colonial stance in its own right (Brunger and Wall 2016; Morton Ninomiya and Pollock 2017). It was at this early point of personal understanding that I entered my PhD research, during which I would experience first-hand some of the value and benefits of collaborating with Indigenous people on research, as well as myself perpetuating some of the problematic practices that come with trying to do this complex task of collaboration without the necessary wisdom and knowledge.

It is important to note that, there is an existing and growing body of work on decolonising methodologies, and science/TEK interactions by Indigenous scholars themselves such as Linda Tuhwai Smith (1999, Ngāti Awa and Ngāti Porou iwi), Zoe Todd (2016, Métis), Jessica Hernandez (2022, Maya Ch'orti' & Binnizá) and Robin Wall Kimmerer (2015, Potowatomi). Indeed even broad terminologies such as 'Native American', Māori and 'Indigenous' have been highlighted as complex and at times problematic in their homogenising assumptions (Smith 1999; Yellow Bird 1999). This critical discussion is invaluable, and will be drawn on throughout this chapter, but unfortunately it was a discussion I was not exposed to during my natural sciences training and so when entering my PhD was one I was little aware of.

5.3 My research

5.3.1 Step 1- Research fatigue

From the outset of my PhD study, I was determined that my research would not simply consist of answering an interesting question. Rather, I wanted to do work that could be of some use to those within the reindeer herding community, either striving to answer questions they had, or more than likely scientifically 'answering' questions they already had the answers for themselves. This scientific 'answering' would perhaps allow this information to be quantified and published by an academic, putting it in a form that may be more heeded by policymakers, or those in involved in land-use court cases. I was aware this would perpetuate the power dynamics of scientific work being viewed as the only credible source of knowledge. This is an attitude that I felt should shift, yet also on which would likely take a while to do so, especially in official forums. In the short term then, perhaps my research could be an interim step, at least allowing ongoing legal cases to draw on science done with locals and not only research done in their absence.

To begin familiarising myself with the field, I read the academic literature, as well as books like Hugh Beach's *A Year in Lapland* (2001) and Robert Paine's *Herds of the tundra: a portrait of Sámi reindeer pastoralism* (1994), accounts by outsiders of spending time within Sámi reindeer herding communities and learning about herding through the cycle of a year. I also read more general non-academic literature on Sámi culture and history. Due to language barriers, I could not source much original material written by reindeer herders, as this was mostly in Norwegian and Swedish. Why I did not attempt to venture into the

Finnish literature using my clumsy yet roughly fluent grasp of the language was somewhat arbitrary, as I had simply at some point decided my focus would be on the herding systems in Norway and Sweden, which differ somewhat from Finnish herding. My understanding of reindeer herding then was still purely theoretical, and if I was to develop some supposed expertise of reindeer ecology by the end of my PhD, some practical experience was necessary.

To try to gain this practical experience of herding, I began by contacting some reindeer herding individuals or institutions to see if I could volunteer for a few months, providing free labour in return for receiving valuable experience and knowledge. I advertised my background of growing up in a rural northern community and working with livestock (although not reindeer), hoping to somewhat legitimise myself in their eyes as someone who would perhaps be of some help and not a total burden. My emails and phone calls received no replies, save one. A prominent herder from northern Sweden kindly agreed to call with me, but seemed non-plussed by my enquiring if he knew of any herders who would want to take an eager researcher along with them to the hills and tundra. With courtesy, and an undertone of frustration and weariness, he informed me that this was not something that he could arrange. At this point I started to become aware of research fatigue.

Research fatigue occurs when often marginalised, minority or Indigenous groups, are repeatedly approached and incorporated into research projects, being surveyed, questioned and asked to partake in workshops to share part of their knowledge or understanding of an area or topic (e.g. Chilisa and Tshenko 2014). During this process, participants are often not treated as an equal by the researcher and do not experience any perceivable positive change due to participation in the project, i.e. the research benefits the researcher but not the local community. This often leads the local and/or Indigenous participants to feel weary and disinclined towards further involvement (Clark 2008). It is an issue all too common in studies connected with the Sámi, who have a joke stating “There are five people in my family- me, my husband, the two children, and the anthropologist”, showing a humorous weariness of being the constant subject of study. Also, some Sámi say, the joke has been repeated so many times it’s no longer funny.

Helga West, a Sámi Theologian, shared a reply she wrote to a researcher from France, asking to interview her on Sámi culture. Her reply, written after wrangling with the wish to be helpful yet also feeling the strain of constantly being interviewed, encapsulates research fatigue well:

> *"Thank you for your kind message. Unfortunately I see very little relevance in your study to me personally and my community. Please understand my refusal and don't take it personally. I'm just tired. I'm tired of strangers who constantly approach my people for the sake of science."*
>
> *Kindly, Helga West (West, 2020)*

Many Sámi are saying they need to substitute the anthropologist in the joke with a natural scientist these days, as whether from genuine interest in collaboration, or as a token gesture, fields like environmental science and ecology are increasingly valuing the inclusion of Indigenous perspectives within research. This was a trend I was perpetuating. Upon reflection the lack of response by most herders I had contacted, and the one herder's polite refusal to put me into contact with any others, made total sense. After all, I would be a bit apprehensive if a stranger emailed me asking if they could move in and study my daily actions. The herders did not know me, trust me, have any obligation or duty towards me, or any investment in the research I wanted to do. More than a bit embarrassed at my thoughtlessness, I went back to the drawing board.

Aware now of my problematic approach, I contacted some relevant official institutions. In emails I explained my research and desire to carry it out ethically, alongside my uncertainty of the best way to do this, asking if they had any advice or protocols. This time I received two replies, both expressing happiness that I was considering these topics and redirecting me to another more relevant body. Perhaps they were too busy, or did not see it as their job to informally advise PhD students on research practice, but this other body did not reply. With time ticking on my PhD, I decided I had to start moving forward with the science by myself.

5.3.2 Step 2- Relationships

After a few more months of thinking and reading, I had identified some gaps in the academic literature, and set out to do some preliminary study, staying at the home of some family friends living near Jokkmokk in northern Sweden. For ethnic Swedes, Jokkmokk is a small village in the wilds of Sweden. However, for the Sámi it is an important cultural centre.

The Northern Sámi name for Jokkmokk, *Dálvvadis,* derives from the Sámi *dálvi* meaning winter, and it has been a gathering point for the Sámi for many hundreds of years, still being a centre for Sámi culture and arts today. Part of what has drawn Sámi to the area is its relevance to reindeer, and the movement of herders has naturally brought many non-herding Sámi to the area also.

Each reindeer herding group (siida) has a designated area where they can graze their reindeer in winter and summer. Many of the winter pastures overlap in the areas surrounding Jokkmokk. Three mountain siidas, have their winter grazing pastures in the area, these being Sirges, Tuorpon and Jåhkågaskatjiellde. Corresponding with the cultural importance of the area, Sirges is one of the largest siidas in Sweden, with over 100 herders and a maximum limit of 15,500 reindeer. Tuorpon has 55 herders with a maximum of 9000 reindeer, and Jåhkågaskatjiellde has around 45 herders with a limit of 4,500 reindeer in their pastures (Sámediggi 2021). Two forest siidas, Udtja and Slakka, also have year-round pastures near Jokkmokk. They are much smaller with only ten and three herders leading the groups respectively, although family and friends may be involved in the herding activities even when not considering themselves herders (Sámediggi 2021).

Although neither reindeer herders nor Sámi, my family friends had spent many decades in the area, one being born there, and later as a couple raising their family in the area. They both knew the landscape, culture and history well, being fully embedded in community life and being curious individuals themselves who had a clear love of the place. Many evenings were spent in discussion about all aspects of Jokkmokk and the Norrbotten region's past, present and potential future in relation to reindeer, community life and the changing environment. These discussions were both enjoyable and genuinely interesting from a personal point-of-view, but also gave some valuable background understanding to my later work.

Whilst I did begin to do some botanical research in the area myself, the hope to work with the local reindeer herders had not been forgotten. I decided that a workshop might be the way forward, asking only a few hours of the herders' time. I thought that this could create a space where reindeer herders could highlight research they wanted to be done, or issues in how science represents their understanding, or perhaps some comparison of scientific and experiential knowledge to see where the two merge and converge. I would not however, approach everyone and anyone. After all they would not know me and it would, I felt, be a little invasive. Instead, I tried to contact representatives and spokes-people. This included the contact persons for local siidas, the Jokkmokk Sámi Parliament branch, and Dálvvadis ekonomisk förening, a local association linking the local herding communities with others to foster collaborations for the benefit of reindeer herding within the Jokkmokk municipality.

The staff at Dálvvadis ekonomisk förening were clearly endeared when I stood in their office, attempting to explain my proposal of collaboration in very broken Swedish, but they kindly said they were unable to help. A Sámi Parliament representative I ran into was, once again, polite but seemed uncomfortable and apprehensive at being approached by a researcher. Only one of the siida representatives replied. He raised my invite to the workshop at a herder's meeting, afterwards sharing with me that unfortunately both he and the other members were too busy. On further reflection I again realised I was approaching strangers and asking them to give their time and energy to someone they did not know, and to a project they had no investment in. As I was later told by a local herder, many representatives within the Sámi and herding communities are repeatedly approached by newspapers, blogs, vlogs, magazines, individuals, researchers etc. She explained that she herself could sometimes receive up to 5 emails a day asking her to share her knowledge and culture. Ignoring these emails was not personal, but simply necessary for people who have jobs to do, families to raise, and a desire for some peace and quiet in their free time. Indeed reindeer herding itself is at times incredibly time consuming. Not wanting to continue to intrude, I stopped trying to contact the local reindeer herders.

During a long field season spent in the area, I happened to befriend a herder. I wanted this connection to be genuine, without ulterior motives, and so didn't try to leverage the friendship to incorporate her into research. Throughout my visits to Jokkmokk we had many informal chats, and she kindly introduced me to two other local Sámi. One was a very experienced herder, who kindly invited me to his home where we talked for hours over strong coffee about reindeer, covering topics from lasso equipment to tourism, changes in climate to how the youth today seem to know more about lions in Africa than the wildlife in their own back garden. This disjoint between the Sámi and the local, non-herder understanding of reindeer was further made clear in a discussion in a shop. A local who spent much time outdoors in nature said he knew nothing about reindeer whatsoever, other than that they get their ears marked in summer to show who owns them. Otherwise, he said, he is not Sámi so he doesn't know.

The second person I was introduced to was a local Sámi man who, whilst not a herder, was an activist for Sámi land rights, a topic which has overlapping interests with herding. Once again welcomed by kind hospitality, we sat in his kitchen eating some freshly caught Arctic char discussing the impacts of mining and forestry on the landscape, and the complex politics involved in land use arguments. Throughout these discussions it was clear thatthese were people who loved their lands, and loved the life that roamed on these lands, reindeer included. They were also people who had been forced to watch these lands and lives be destroyed, and were still experiencing the racism and disregard that came along with that process, something that was deeply painful and frustrating for them.

These informal conversations made up the greatest portion of my personal learning during this PhD study, teaching me more than any book or article ever could. They also felt like the most natural part of my connecting with the locals. As with the family friends, we were simply having conversations based on common interests, and as the conversations were not formal or destined to be published, they did not feel invasive. I tried to use the information I learned to guide my work, and these individuals seemed quite enthusiastic that someone from abraod was taking an interest in some of the issues facing the Sámi. However, it was clear they were not interested in being personally involved. They were currently fighting their own battles and whilst happy to cheer others on in their work, they could only spread their energy and focus so thinly.

Distrust

Relationships are a fundamental part of research, though often overlooked. Each interaction, between researchers, supervisor and supervisee, participant and academic, research and technical staff, has a unique relationship. These relationships have differing levels of formality, and differing power dynamics, affected by all kinds of variables including gender, language and race (Riley *et al.* 2003; Chen 2011; Muhammad *et al.* 2015). When ecological research has included Indigenous perspectives, there has been a notable lack of attention given to power relations between different parties, or of critical reflection given to the collaborative practices within many these works (Whyte 2013; Ford et al. 2016; Mosurska and Ford 2020; Singleton *et al.* 2021). As it stands, the researcher often holds the power- they come with funding, and they will be largely responsible for how data is interpreted, used and presented in the end product. This imbalance of power has led to cases of completely unethical research, such as by the Swedish State Institute for Racial Biology, which undertook incredibly problematic racially focused studies on many Sámi individuals, especially in residential schools, without their permission. Similar examplescan be seen in state sponsored programmes in Canada in the 1940s and 1950s (Mosby 2013).

In relation to ecological knowledge, data and specimens, researchers again have abused their position of power by patenting and claiming rights to materials shown to them or shared with them by Indigenous groups, such as Loren Miller's patenting of a traditional ceremonial drink called *ayahuasca* in 1994, despite it being long used by Amazonian peoples (Laid and Noejovich 2002). These kinds of patenting issues have occurred multiple times in multiple contexts (Battiste and Youngblood, 2000), showing the ease with which knowledge can be appropriated and claimed by others outside of a local/Indigenous community, and understandably making some groups wary of sharing ecological information. Indeed, Zoe Todd (2016) writes at length of how concepts long espoused in Indigenous communities can get rebranded, 'discovered', or surpassed by very similar concepts created within western academic institutions, ensuring credit for these ideas remains solely with western researchers and not the source community.

Free prior informed consent (FPIC) is one of many practices and pieces of legislation put into place to protect the individual or group sharing data with researchers. It aims to fully inform the participant how data will be used, may outline the level of recognition or anonymity they will receive, and often gives them the opportunity to withdraw from the study along with their data until a certain time period has elapsed. A more extensive discussion on FPIC can be found in Laid and Noejovich (2002). Even with FPIC in place however, in practice those who provide data may not always see how it is interpreted or utilised later down the line (e.g. Dalton 2002), and consent may take many different forms depending on the circumstances in which it is given, to whom, and how (e.g. Sankar 2004).

With multiple cases of historical misuse of data or knowledge such as those outlined above, distrust is understandably common within local and Indigenous communities towards researchers. Indeed, Linda Tuhiwai Smith has written:

> *"Research is probably one of the dirtiest words in the Indigenous world's vocabulary. When mentioned in many Indigenous contexts, it stirs up silence, it conjures up bad memories, it raises a smile that is knowing and distrustful."* (Smith, 1999)

The Sámi are not exempt from these feelings of distrust. Some feel that researchers arrive with their own agenda and so are less inclined to consider local needs. For this reason, during the Finnish Truth and Reconciliation Commission hearings some Sámi participants stated they did not wish any researchers to be commissioners throughout the Truth and Reconciliation process (Juuso 2018). For others, the character of the person rather than their status, as e.g. a researcher, mattered most. As one stated:

> *"[We want] the sort of person who has knowledge, ability and a good heart. A really good heart and understanding. As some professor said, anyone can acquire knowledge and skills. But they should also have wisdom. I believe that wisdom sees and understands that, when someone else tells something, why he tells it like that. Not that he tells it like that, but can see why, what he is really saying and why he feels that way. I would choose the wisest people of all for that."* (Juuso 2018)

This statement shows the importance of building relationships with communities where study will occur. Strong relationships on the one hand better inform the researcher of local conditions and realities. On the other hand, for the local and Indigenous people who are the gatekeepers for the knowledge and information they hold about their environement, becoming acquainted with the character of the researcher can allow them to determine if they feel the researcher has the capacity to take in, process and present information given to them in an accurate, constructive and fair way, helping them to decide what they feel they can share with the researcher. Building up this kind of understanding of someone's character is not a quick process, and in many ways is quite instinctive, building off body language, expressions, immediate reactions to topics and time spent informally together. These instinctive impressions cannot be gained from email, and so whilst an easy way to reach out to someone new, it is often not the arena of building a strong relationship. My most meaningful connections throughout this PhD came from people with whom I had met in person, or spoken to on the phone, both in terms of Sámi and reindeer herders, but also academics and policymakers. For humans to connect, having a shared space to siply be together is invaluable. It's the coffee breaks between the talks at conferences, the pub after work, the chance meetings on the streets. These are where we can often go beyond sharing information, to sharing connection. Time then, and sincere personal relationships are essential between researchers and local communities if they are to work together effectively.

5.3.3 Step 3- Timelines

Throughout subsequent visits to field sites and informal meetings with herders, I tried to let the information they shared with me guide the project. For example, when I mentioned the idea of creating a map of the local area, showing which pastures are more suitable for reindeer grazing and why, some herders were apprehensive about the idea. A map like that, they said, might be misused by mining companies to target less grazed areas as potential development sites, when in fact these lesser used areas may still be important for the reindeer. Not wanting to do work that could be potentially harmful, I opted instead to create a map of a hypothetical location based on broad, northern Swedish landscape parameters.

Part way through my second year of fieldwork the COVID-19 pandemic hit, and lockdown regulations restricted movement. As a result, I was unable to spend more time in Norrbotten meeting new herders or interacting with those I had already met, preventing the development of relationships which may have led to collaboration. On reflection, I felt that I had not been successful in planning the project from the beginning with the herders, or carrying out the work together. Although I had entered the field of research with good intentions and a genuine desire to collaborate, I had also entered with a lack of knowledge and perspective. By the time I had started to gain some of this knowledge and perspective, there was insufficient time to carry out the research and partnerships in a way that I would have felt was 'good enough'. Good intentions were not enough, and I started to realise that as a newcomer to the field and to Jokkmokk, even the span of two or three PhDs may not have been enough. To truly get to know people, build trust and create collaborative research together would have required far more commitment, embedding long term into the community and gaining fluency in the Swedish language. I knew that I did not intend to commit a significant period of my future life to the area, and so any further attempts to build relationships were at serious risk of perpetuating patterns of helicopter research if I did not return after the completion of my PhD.

As a last-ditch attempt at including some kind of partnership in the project, I thought perhaps it could be interesting to give herders a platform to critically assess this science I had done without them, comparing it to their own TEK of the system. This could highlight the discrepancies that can arise when collaboration is not successfully undertaken early in a project, and having this kind of critical assessment of science by herders in a published format would, I hoped, allow the information to be referred to and considered by other academics. Perhaps as me being someone who they did not know well, they would feel less compelled to be polite, and for better or worse give a very honest assessment.

A survey was created where herders were walked through the reindeer ecosystem model created in chapter 4 of this thesis. They were then asked to reflect on their own experiences of change within their environment and how this has affected the reindeer. Finally, they were asked to assess the accuracy of the model compared to their own knowledge. This survey was translated into Swedish, involved FPIC, and allowed anonymity as well as the option to be given credit for involvement in the project. Although not part of any official protocol, I decided to include payment for taking part, to avoid expecting participants to give up their free time without compensation.

I began sending the survey to herders I had come to know, inviting them to take part, but got no initial replies. In the meantime, I attended multiple seminars discussing how scientists and local communities do, and should, interact when carrying out research. These seminars, often attended or organised by Indigenous individuals and Indigenous scholars, reminded me once again of topics like research fatigue, disinterest when individuals do not have a personal stake in the project, the importance of including local/Indigenous communities early in the research, and why co-creation and collaboration can be far more valuable than the model of researcher-informant that I was using. It no longer felt appropriate to start another barrage of emails to get survey respondents to fit my wants for the project, so I stopped. For this project, I had simply run out of time.

Time is an inconvenient yet important consideration. The modes of passing on information common in many Indigenous communities involve sharing practical skills as well as stories, feelings, understandings, and showing experiences first hand. Ecological knowledge can be shared by observing the cycle of the seasons together, noticing how the natural world reacts. Sometimes information can only be shared when an individual is deemed ready to receive that knowledge. These modes of teaching and sharing not only require relationships, which can take time to develop, but can also take months, years or lifetimes to share and convey (Armitage *et al.* 2011).

The time required to do justice to learning TEK is completely at odds with the timelines of western scientific practice (Castleden *et al.* 2012). PhDs or post-doctoral research projects may only last three or four years, if not less, and academic papers must be published regularly with novel data in each. Work must be published before potential 'competitors' within the field publish something similar. In addition to all this, an academic researcher juggles with teaching, winning grants, giving conference talks, reviewing other's research and supervising students. It is a field inherently pushed for time where information must be processed in high volumes and immediately.

The relentless push of time within the world of an academic researcher then often proves incompatible with the slow pace of sharing knowledge found in many Indigenous communities in this and other studies e.g. Russell-Mundine (2012), or at times the beaurocracy of academia is too slow for communities who need data quickly e.g. on how an industry is rapidly harming their surroundings, so that it can be counteracted. The tension this pull of time creates can be difficult to navigate. Indigenous participants at the Arctic Science Summit Week 2021 conference seminars discussed how they often have to carry the emotional baggage of western researchers who are frustrated at the pace of progress or the difficulties with building trust with local/Indigenous participants, or who are frustrated when modes of communication between these groups differ, such as the preference in some Indigenous communities to discuss matters informally and over time rather than having rigid question-and-answer sessions.

It can be a frustrating experience and a steep learning curve for the researcher to navigate between effective research and respectful and ethical interactions. However, it can also put extra fatigue on the local and Indigenous participants who have to repeatedly play the role of a therapist for each new researcher going through these struggles. Forcing local communities to meet academic timelines then can be ineffective, yet for the academic to continue to receive funding and in some cases employment, this push is necessary, creating a problematic paradox.

5.3.4 Step 4- Attempts to reconcile

Realising I had run out of time to carry out the ideal kind of collaborative research, I tried to make the work I had done as ethical as possible. I wanted to give credit to those who had influenced my project, and so contacted them with the results of early research which I was trying to publish, giving them a summary of the results and inviting any comments before I submitted the manuscript to publishers. I also asked if I could acknowledge them within the research. Some responded with excitement and interest in the results, and suggested practical uses of the research paper once published, although they were not interested in commenting extensively.

I had hoped to return to Jokkmokk and give a talk as part of a regular talk series hosted by locals and visitors there, once again to return to the community with the results so that locals could see how data about their area had been used, but travel restrictions due to COVID-19 lockdown rules prevented this. In writing my chapters I tried to find research and literature carried out by or with Sámi reindeer herders to ensure their perspectives and interpretations were given precedence. Finally, I decided to write this chapter to highlight for others some of the concepts, challenges and opportunities I came across in my attempts to work with an Indigenous group on research, so that readers can perhaps start their work more informed and better equipped for a successful collaboration.

Had I known at the beginning of my project what I know now about research partnerships with Indigenous and local communities, perhaps a more appropriate timeline and expectations of the project could have been created, and fewer people would have had to be needlessly emailed, approached and questioned. However, these are lessons and conversations not often covered within scientific forums. For those about to embark on a journey of collaborative research then who are perhaps less experienced in this arena as I was, below are a few suggestions to aid that process.

5.4 Suggestions for methods in future research

Much of this chapter has contained examples of local/Indigenous-researcher dynamics from across the entire globe. It is essential to remember that each community, its knowledge, needs, fears, wants, ways of communicating and ways of working are unique (Aikenhead and Ogawa 2007). As McKinley has stated, ‘‘Indigeneity is a heterogeneous, complex concept that is contextually bound’’ (2007, p. 202). The circumstances and individuals, or as in the Strathern’s work on ‘dividuals’ the differences in perception on what it means to be part of a community (1988), are unique for each potential collaboration. Therefore, there is no strict formula on how successful collaboration can occur. However, there are some broad guiding principles which can contribute towards creating more successful western science-TEK collaborations.

5.4.1 Research planning

Many concerns over research methodologies in relation to Indigenous and local people base on the idea of doing research *on* rather than *with* these communities (Ball and Janyst 2008). Doing research *on* them involves extracting data for use within science, and is focused on the needs and wants of the researcher. Doing research *with* local and Indigenous communities requires their involvement from the outset to create a more even power balance, so that they can help formulate research questions that are of interest and relevance to them too. Some kind of prior relationships are likely required here between the researcher and the local/Indigenous collaborator. One cannot always rely on the good relationship between a community and a colleague or supervisor, but may need to build this relationship themselves. It would be of benefit for researchers to take time to reflect on what qualities and associations they as individuals will bring into the research partnership in terms of their personal background, experience, race, gender etc., with this awareness allowing any imbalances in the make-up of the research team to be addressed (Riddell *et al.* 2017).

There are also practical benefits to initially planning research projects together *with* the relevant local and Indigenous communities. For example, local/Indigenous timelines and those of academic or commercial institutions may differ vastly. Discussing expectations of timescales at initial stages can allow more appropriate types of funding to be considered, minimising possible tensions that may otherwise arise later.

As with issues of time, there can be many other issues arising in research collaborations, including politics, fears, prejudices, tensions between world views, personality differences, misunderstandings etc. To minimise these, it is worth as a researcher to educate oneself on the historical and current politics and research politics surrounding the community in question.

To avoid unethical research practices being carried out, many Indigenous organizations have created guidelines and principles for researchers. This includes Canada's Tri-Council's Statement on Ethical Conduct for Research Involving Humans (CIHR *et al.*, 2010; CIHR *et al.* 2014) and the United Nations Declaration on the Rights of Indigenous People (United Nations 2007), and the Inuit Tapiriit Kanatami's National Inuit Strategy on Research (ITK 2018), alongside ethical engagement guidelines currently being developed by the Sámi Council and Inuit Circumpolar Council (Muotka 2020; ICC 2021). These guidelines highlight themes like FPIC, full participation, privacy and confidentiality, intellectual property rights, respect towards Indigenous ontologies, plans for dissemination of results and acknowledgment of colonial impacts. They have their greatest value when written by the Indigenous groups themselves as they are more likely to reflect the expectations and stances of this community on collaboration. Ethical guidelines created by universities, research institutions and governing bodies must be approached critically and mindfully with the aim being to carry out ethical research rather than to strictly serve ethical guidelines, which may end up leading to harm for the community involved (Brunger and Wall 2016; Morton Ninomiya and Pollock 2017).

Throughout all the processes mentioned above, stress and frustrations can arise. The emotional burden of these frustrations should not, however, be put onto the Indigenous collaborators. Whilst discussing issues with colleagues and sharing stresses with friends is a natural part of life, Indigenous collaborators should not be led to become fatigued by having to constantly take care of their non-Indigenous academic partners. Additionally, even if a researcher is mindfully working to foster ethical collaborations, it must also be acknowledged that, as in all matters of consent, no means no. Sometimes even when best practice is being followed, Indigenous individuals and communities may not wish to be involved in certain research and this must be respected.

5.4.2 The role of collaborators

The role that individuals play within a research project must be carefully considered and explicitly communicated. Indigenous and local people have often been included as research subjects within projects. This is problematic, as it is often done for the benefit of the researchers rather than the community being studied, and is generally criticised for not being a respectful form of knowledge exchange (Ball and Janyst 2008). It also allows for interpretation of data by non-Indigenous/non-local researchers, and we naturally interpret information through our own lens of understanding, which may subtly or significantly alter it from its original meaning. Chilisa and Tsheko (2014) highlight this stating:

> *"the dominant language that includes gesture, tone, expression, theoretical frameworks, methods of data collection, and data analysis communicates dominant cultures and is most likely to misrepresent and render silent the experiences of the majority researched and relegated to the position of Other."*

Another role Indigenous individuals often play within a project is that of a research assistant or informant, providing data through workshops/interviews, or taking part in the physical carrying out of fieldwork. How assistants and informants are credited for their input varies widely. Some remain unmentioned in academic papers, as with the criticised case of US scientists 'discovering' a new species of giant wasp in Indonesia with no reference to locals who had likely been well aware of its existence (Kimsey and Ohl 2012; Minasny *et al.* 2020). Others are included in acknowledgements. However, there is no standard practice on how local and Indigenous contributors are credited.

Then comes the question of co-authorship. This can be a contentious and sensitive issue even within the scientific community. Opinions vary on the level of involvement in a project that would result in co-authorship. Some believe it should only be extended to those actually writing the academic article, and therefore the interpreting results, whilst others believe “a significant intellectual property input” is the main criteria, opening the opportunity for a wider group to be considered (Castleden *et al.* 2010). In some cases, this has been extended to include entire communities even if they were not directly involved in the research project, as it was felt that the entire community is responsible for generating and maintaining the knowledge that was shared in the research project (Castleden *et al.* 2010).

Co-authorships have multiple benefits. Indigenous co-authors are given more clear credit for their involvement in the research, perhaps giving TEK greater credibility within academia (Castleden *et al.* 2010). These Indigenous co-authors are also more likely to be given the opportunity to access and contribute to manuscripts before submission, meaning they can improve accuracy of interpretations as well as develop a greater sense of empowerment by being involved in this process. This blending of perspectives, sometimes termed a mixed methods approach, allows data and information to be viewed through multiple lenses (Chilisa and Tsheko 2014). Learning how to allow this blending to occur, including perspectives such as history, spirituality and ideology, can perhaps initially feel unnatural to those within the natural sciences, but the potential for more meaningful, respectful and informative work is great.

As there are benefits, so there are also some difficulties associated with co-authorship. If multiple community members, even entire communities, are listed as co-authors it creates the assumption that all agree with what is being stated in the scientific article. In reality this may not be the case, and reaching a consensus amongst all members of a community can be near impossible (Titz et al. 2018; Mosurska and Ford, 2020). It may also prevent certain data being portrayed accurately, for example if there are opportunities for gain or loss for individuals in how information is presented that feeds into local politics. In some cases, members of the community may decide they no-longer want to share data, a cause for worry for collaborating academics who may be losing years of work (Castleden *et al.* 2012). Finally, as mentioned earlier, community based research strategies that include multiple

Indigenous individuals as co-authors often work on a much slower timescale, which prevents the high publishing rates expected for many natural scientists to remain employed (Castleden *et al.* 2010).

When beginning a research project with hopes of collaboration then, it is essential to reflect carefully on the role each person will play, the power balances that this division of credit and labour will create, and how it will affect the eventual output of results.

5.4.3 Indigenous involvement

Finally, for many projects it can be very valuable to promote and enable the Indigenous collaborators to be at the forefront. This can either be as assistants, co-creators or co-authors, or even project leaders, but should not just be done as a token gesture. As one Inuit participant in the Arctic Science Summit Week 2021 conference said, *"Inuit can do research, we're capable, just give us the tools"*. Indigenous individuals are in many cases leading the way in scientific-Indigenous collaborations and commentaries, such as Inupĩaq Dr. Victoria Buschman, Gwich'in Evon Peter, Mi'kmaq Dr. Marie Battiste and Dr. Stanislav Ksenofontov who is Indigenous to Yakutia (e.g. Battiste 2000; Buschman 2019; Ksenofontov *et al.* 2018), amongst many others thus far cited in this thesis. This allows someone with a foot in 'both worlds', the Indigenous knowledge system and the world of western academia, to use this dual understanding to reconcile some of the differences between the two, and translate each world for the other group. That inclusion and acknowledgment of capacity is not only empowering. The support of higher order institutions in these sorts of processes is also crucial in allowing a richer, more diverse science to be created which generates greater learning due to the range of voices involved than would be created if a more homogenous group of academics are at the forefront of the research project (Armitage *et al.* 2011). Epistemologically, Harraway (1988) highlights how all knowledge can be seen as situated within the conditions of the knowledge producer, whether scientific, Indigenous or otherwise. All knowledge then, including 'subjugated' knowledge (Harraway 1988), should be approached critically. However, with a backdrop of colonial silencing and a general lack of heterogeneity amongst natural science researchers, there is importance in new understanding generated from Traditional Ecological Knowledge being critically assessed by those linked to, or rooted in, that knowledge form. This should be done in a way that is cognisant of, and tries to reduce, the barriers placed in front of these Indigenous scholars trying to fulfil this position, including delegitimization and institutional racism (e.g. see Todd 1999).

There are increasing numbers of natural science research projects which are Indigenous-led. Examples include the Skolt Sámi-led Näätämö salmon conservation co-management project (Brattland and Mustonen, 2018), efforts in parts of the Resource Extraction and Sustainable Arctic Communities (REXSAC) project, and hopes for the International Western and Indigenous Science Hub for Fish (IWISH) group which is currently being established, along with bowhead whale research being overseen by the Inupĩaq formed Alaska Eskimo Whaling Commission (Albert 2001). These projects can often be slow, misunderstandings can arise between collaborators from differing backgrounds and ontologies (e.g. Brattland and Mustonen 2018; Holmberg 2018), and tensions can arise due to a mismatch between academic requirements and the natural timelines of Indigenous communities (Armitage *et al.* 2011; Castleden *et al.* 2012). However, these projects also show positive progress towards equalising power balances and tend to produce quite applied outcomes, leading to positive social and ecological results (Armitage *et al.* 2011; Ball and Janyst 2008).

5.5 Conclusions

Much has been learned about interdisciplinarity in the last few years, especially the bringing together of traditional and scientific knowledge, yet still more work needs to be done. Keeping in mind Kyle Whtye’s discussion on the diversity of TEK may be (Whyte 2013), knowledges held by Indigenous communities and individuals about their surroundings have an inherent value in their own right, not just in relation to science. Sometimes this means TEK cannot be simplified into generalisable principles which are often sought after in scientific research. However, this is what makes the field of ecology an ideal training ground for learning how to successfully undertake inter- or multidisciplinary collaboration. Ecology is by its very nature a field that must embrace complexity and uncertainty, making it more open to this task.

The purpose of this thesis was two-fold. Firstly, as with any PhD thesis, the aim was to conduct original research into a topic. This has been done, through data collection, analysis, creation of a model and discussions of results. The second aim of the thesis was to be an experiment within itself, exploring the potential for research to be carried out in an interdisciplinary manner. Here the scientific methods have highlighted and quantified the impacts of forestry, climate change, snow conditions and cumulative land-use competition with infrastructure on reindeer grazing, whilst the methods of anthropology have highlighted the importance of power dynamics, individual herder decision making and the dynamic nature of the ecosystem when trying to understand reindeer ecology. The conclusions of this thesis have seen that both have contributed to a greater understanding of reindeer ecology in northern Sweden.

We are currently entering an exciting new chapter within scientific research, where by creating a more diverse input of voices into the scientific process we have the opportunity to create even greater understanding and roadmaps to guide action regarding our natural world and our human world, and indeed all the things that tightly connect the two. There is much learning required from all sides: the overcoming of prejudices, development of open-mindedness and cultivation of empathy, intangible qualities often not spoken of as part of the scientific process. However, the benefits that can be gleaned from these collaborations by all parties are vast, and that will most definitely make the time, energy and effort worth it.

Appendix

Appendix A. The amount of lichen lost in different model scenarios due to reindeer avoidance in zones of influence (ZOIs) of infrastructure. R1, R2 and R3 indicate one to three through roads respectively, N and Y indicate absence and presence of a mine, and _H within the scenario name indicates the presence of a hydropower station. Squares refer to the 0.5 km² landscape squares which make up the model. In the forest age column C denotes 'clear-cut', Y is 'young', I is 'intermediate' and O 'old' forest maturity, using lichen presence data for these maturity groups from chapter 2 of this thesis.

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R1N	20%	C	20269	1	9	0	0	2	8715.67	114925.2	0	0	40538	164178.9	9434480
		Y	35419	4	17	0	0	6	60920.68	379337.5	0	0	212514	652772.2	
		I	44636	4	17	0	0	6	76773.92	478051.6	0	0	267816	822641.5	
		O	43762	37	169	0	0	57	696253.4	4604200	0	0	2494434	7794887	
R1N	40%	C	20269	4	17	0	0	6	34862.68	204311.5	0	0	121614	360788.2	9079625
		Y	35419	7	34	0	0	11	106611.2	758675	0	0	389609	1254895	
		I	44636	7	34	0	0	11	134354.4	956103.1	0	0	490996	1581454	
		O	43762	28	127	0	0	43	526894.5	3473828	0	0	1881766	5882489	
R1N	60%	C	20269	6	26	0	0	9	52294.02	319236.8	0	0	182421	553951.8	8718083
		Y	35419	11	51	0	0	17	167531.9	1138012	0	0	602123	1907667	
		I	44636	11	51	0	0	17	211128.3	1434155	0	0	758812	2404095	
		O	43762	18	84	0	0	28	338717.9	2288315	0	0	1225336	3852369	
R1N	80%	C	20269	7	34	0	0	11	61009.69	421392.5	0	0	222959	705361.2	8404936
		Y	35419	15	68	0	0	23	228452.6	1517350	0	0	814637	2560440	
		I	44636	15	68	0	0	23	287902.2	1912206	0	0	1026628	3226736	
		O	43762	9	41	0	0	14	169358.9	1130372	0	0	612668	1912399	

Scenario	Forest felled (%)	Age	Lichen (kg) per age square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R2N	20%	C	20269	6	6	0	0	5	52294.02	89386.29	0	0	101345	243025.3	12379464
		Y	35419	13	12	0	0	11	197992.2	267767.6	0	0	389609	855368.8	
		I	44636	13	12	0	0	11	249515.2	337448.2	0	0	490996	1077959	
		O	43762	126	121	0	0	104	2371025	3280837	0	0	4551248	10203110	
R2N	40%	C	20269	13	12	0	0	10	113303.7	153233.6	0	0	202690	469227.3	11947586
		Y	35419	25	24	0	0	21	380754.3	535535.3	0	0	743799	1660089	
		I	44636	25	24	0	0	21	479837	674896.3	0	0	937356	2092089	
		O	43762	95	91	0	0	79	1787678	2481305	0	0	3457198	7726181	
R2N	60%	C	20269	19	18	0	0	15	165597.7	229850.5	0	0	304035	699483.2	11500330
		Y	35419	38	37	0	0	32	578746.5	803302.9	0	0	1133408	2515457	
		I	44636	38	37	0	0	32	729352.2	1012344	0	0	1428352	3170048	
		O	43762	63	60	0	0	52	1185513	1654204	0	0	2275624	5115341	
R2N	80%	C	20269	25	25	0	0	22	217891.8	319236.8	0	0	445918	983046.6	10998756
		Y	35419	51	49	0	0	42	776738.7	1071071	0	0	1487598	3335408	
		I	44636	51	49	0	0	42	978867.5	1349793	0	0	1874712	4203373	
		O	43762	31	29	0	0	25	583347.5	799531.7	0	0	1094050	2476929	
R3N	20%	C	20269	8	4	0	0	7	69725.36	51077.88	0	0	141883	262686.2	13682264
		Y	35419	14	7	0	0	16	213222.4	156197.8	0	0	566704	936124.2	
		I	44636	14	7	0	0	16	268708.7	196844.8	0	0	714176	1179730	
		O	43762	140	72	0	0	154	2634472	1929904	0	0	6739348	11303724	
R3N	40%	C	20269	14	7	0	0	15	122019.4	89386.29	0	0	304035	515440.7	13199370
		Y	35419	28	14	0	0	31	426444.8	312395.6	0	0	1097989	1836829	
		I	44636	28	14	0	0	31	537417.4	393689.5	0	0	1383716	2314823	
		O	43762	106	54	0	0	116	1994672	1461213	0	0	5076392	8532277	

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R3N	60%	C	20269	22	11	0	0	22	191744.7	140464.2	0	0	445918	778126.9	12697495
		Y	35419	42	22	0	0	47	639667.1	468593.4	0	0	1664693	2772954	
		I	44636	42	22	0	0	47	806126.2	590534.3	0	0	2097892	3494553	
		O	43762	70	36	0	0	77	1317236	964952.1	0	0	3369674	5651862	
R3N	80%	C	20269	28	15	0	0	32	244038.8	191542.1	0	0	648608	1084189	12149602
		Y	35419	57	29	0	0	62	868119.7	624791.2	0	0	2195978	3688889	
		I	44636	57	29	0	0	62	1094028	787379	0	0	2767432	4648839	
		O	43762	34	17	0	0	37	639800.4	468691	0	0	1619194	2727685	
R1Y	20%	C	20269	2	7	2	0	4	17431.34	102155.8	14188.3	0	81076	214851.4	11749154
		Y	35419	6	15	5	0	10	91381.02	312395.6	61983.25	0	354190	819949.9	
		I	44636	6	15	5	0	10	115160.9	393689.5	78113	0	446360	1033323	
		O	43762	54	142	48	0	93	1016154	3859808	735201.6	0	4069866	9681030	
R1Y	40%	C	20269	5	14	4	0	9	43578.35	178772.6	28376.6	0	182421	433148.6	11333908
		Y	35419	11	28	10	0	19	167531.9	624791.2	123966.5	0	672961	1589251	
		I	44636	11	28	10	0	19	211128.3	787379	156226	0	848084	2002817	
		O	43762	41	107	36	0	70	771524.1	2922426	551401.2	0	3063340	7308691	
R1Y	60%	C	20269	9	21	8	0	14	78441.03	280928.3	56753.2	0	283766	699888.5	10865128
		Y	35419	16	43	14	0	28	243682.7	937186.7	173553.1	0	991732	2346155	
		I	44636	16	43	14	0	28	307095.7	1181069	218716.4	0	1249808	2956689	
		O	43762	27	71	24	0	47	508076.8	1929904	367600.8	0	2056814	4862396	
R1Y	80%	C	20269	11	29	10	0	18	95872.37	357545.2	70941.5	0	364842	889201.1	10468090
		Y	35419	22	57	19	0	38	335063.7	1271896	235536.4	0	1345922	3188418	
		I	44636	22	57	19	0	38	422256.6	1602879	296829.4	0	1696168	4018133	
		O	43762	13	34	12	0	23	244629.6	937382	183800.4	0	1006526	2372338	

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R2Y	20%	C	20269	6	6	3	0	6	52294.02	89386.29	21282.45	0	121614	284576.8	13658332
		Y	35419	12	12	4	0	12	182762	267767.6	49586.6	0	425028	925144.2	
		I	44636	12	12	4	0	12	230321.8	337448.2	62490.4	0	535632	1165892	
		O	43762	115	121	44	0	118	2164031	3280837	673934.8	0	5163916	11282719	
R2Y	40%	C	20269	12	12	4	0	11	104588	153233.6	28376.6	0	222959	509157.2	13203536
		Y	35419	23	24	9	0	24	350293.9	535535.3	111569.9	0	850056	1847455	
		I	44636	23	24	9	0	24	441450	674896.3	140603.4	0	1071264	2328214	
		O	43762	87	91	33	0	89	1637136	2481305	505451.1	0	3894818	8518710	
R2Y	60%	C	20269	17	18	7	0	17	148166.4	229850.5	49659.05	0	344573	772249	12703507
		Y	35419	35	37	13	0	36	533056	803302.9	161156.5	0	1275084	2772599	
		I	44636	35	37	13	0	36	671771.8	1012344	203093.8	0	1606896	3494106	
		O	43762	58	60	22	0	59	1091424	1654204	336967.4	0	2581958	5664553	
R2Y	80%	C	20269	23	25	8	0	23	200460.4	319236.8	56753.2	0	466187	1042637	12192407
		Y	35419	47	49	18	0	48	715818	1071071	223139.7	0	1700112	3710141	
		I	44636	47	49	18	0	48	902093.6	1349793	281206.8	0	2142528	4675621	
		O	43762	28	29	11	0	29	526894.5	799531.7	168483.7	0	1269098	2764008	
R3Y	20%	C	20269	9	5	3	0	8	78441.03	63847.35	21282.45	0	162152	325722.8	15188567
		Y	35419	16	9	4	0	15	243682.7	200825.7	49586.6	0	531285	1025380	
		I	44636	16	9	4	0	15	307095.7	253086.1	62490.4	0	669540	1292212	
		O	43762	158	92	44	0	146	2973190	2508875	673934.8	0	6389252	12545252	
R3Y	40%	C	20269	16	9	4	0	16	139450.7	127694.7	28376.6	0	324304	619826	14636312
		Y	35419	32	19	9	0	29	487365.4	401651.5	111569.9	0	1027151	2027738	
		I	44636	32	19	9	0	29	614191.4	506172.2	140603.4	0	1294444	2555411	
		O	43762	119	70	33	0	110	2239302	1874764	505451.1	0	4813820	9433337	

Scenario	Forest felled (%)	Age	Lichen (kg) per age square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R3Y	60%	C	20269	24	14	7	0	23	209176.1	191542.1	49659.05	0	466187	916564.3	14076147
		Y	35419	48	28	13	0	44	731048.2	602477.2	161156.5	0	1558436	3053118	
		I	44636	48	28	13	0	44	921287	759258.4	203093.8	0	1963984	3847623	
		O	43762	79	46	22	0	73	1486595	1240653	336967.4	0	3194626	6258841	
R3Y	80%	C	20269	32	19	8	0	30	278901.4	229850.5	56753.2	0	608070	1173575	13554708
		Y	35419	64	37	18	0	59	974730.9	825616.9	223139.7	0	2089721	4113209	
		I	44636	64	37	18	0	59	1228383	1040465	281206.8	0	2633524	5183579	
		O	43762	39	22	11	0	36	733888.7	606541.3	168483.7	0	1575432	3084346	
R1N_H	20%	C	20269	1	6	0	0	4	8715.67	76616.82	0	0	81076	166408.5	9366285
		Y	35419	4	12	0	1	9	60920.68	245453.7	0	21251.4	318771	646396.8	
		I	44636	4	12	0	1	9	76773.92	309327.5	0	26781.6	401724	814607	
		O	43762	37	112	0	10	85	696253.4	3060277	0	262572	3719770	7738872	
R1N_H	40%	C	20269	4	11	0	1	9	34862.68	153233.6	0	12161.4	182421	382678.7	8989985
		Y	35419	7	23	0	2	17	106611.2	490907.3	0	42502.8	602123	1242144	
		I	44636	7	23	0	2	17	134354.4	618655	0	53563.2	758812	1565385	
		O	43762	28	85	0	7	64	526894.5	2288315	0	183800.4	2800768	5799778	
R1N_H	60%	C	20269	6	17	0	1	12	52294.02	229850.5	0	12161.4	243228	537533.9	8674189
		Y	35419	11	34	0	3	26	167531.9	736361	0	63754.2	920894	1888541	
		I	44636	11	34	0	3	26	211128.3	927982.4	0	80344.8	1160536	2379992	
		O	43762	18	56	0	5	43	338717.9	1516353	0	131286	1881766	3868123	
R1N_H	80%	C	20269	7	23	0	2	16	61009.69	280928.3	0	24322.8	324304	690564.8	8355803
		Y	35419	15	45	0	4	35	228452.6	1004129	0	85005.6	1239665	2557252	
		I	44636	15	45	0	4	35	287902.2	1265431	0	107126.4	1562260	3222720	
		O	43762	9	27	0	2	21	169358.9	744391.6	0	52514.4	919002	1885267	

Scenario	Forest felled (%)	Age	Lichen (kg) persquare (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R2N_H	20%	C	20269	6	3	0	0	6	52294.02	51077.88	0	0	121614	224985.9	12334761
		Y	35419	13	7	0	1	14	197992.2	133883.8	0	21251.4	495866	848993.4	
		I	44636	13	7	0	1	14	249515.2	168724.1	0	26781.6	624904	1069925	
		O	43762	126	64	0	10	133	2371025	1736914	0	262572	5820346	10190857	
R2N_H	40%	C	20269	13	6	0	1	13	113303.7	76616.82	0	12161.4	263497	465578.9	11882843
		Y	35419	25	13	0	2	27	380754.3	290081.6	0	42502.8	956313	1669652	
		I	44636	25	13	0	2	27	479837	365568.8	0	53563.2	1205172	2104141	
		O	43762	95	49	0	7	100	1787678	1295793	0	183800.4	4376200	7643471	
R2N_H	60%	C	20269	19	10	0	1	21	165597.7	127694.7	0	12161.4	425649	731102.8	11403522
		Y	35419	38	20	0	3	40	578746.5	423965.4	0	63754.2	1416760	2483226	
		I	44636	38	20	0	3	40	729352.2	534292.9	0	80344.8	1785440	3129430	
		O	43762	63	32	0	5	66	1185513	854671.9	0	131286	2888292	5059763	
R2N_H	80%	C	20269	25	13	0	2	27	217891.8	178772.6	0	24322.8	547263	968250.2	10949622
		Y	35419	51	26	0	4	54	776738.7	557849.3	0	85005.6	1912626	3332220	
		I	44636	51	26	0	4	54	978867.5	703017	0	107126.4	2410344	4199355	
		O	43762	31	16	0	2	32	583347.5	413550.9	0	52514.4	1400384	2449797	
R3N_H	20%	C	20269	8	3	0	0	6	69725.36	51077.88	0	0	121614	242417.2	12612428
		Y	35419	14	7	0	1	14	213222.4	133883.8	0	21251.4	495866	864223.6	
		I	44636	14	7	0	1	14	268708.7	168724.1	0	26781.6	624904	1089118	
		O	43762	138	64	0	10	133	2596837	1736914	0	262572	5820346	10416669	
R3N_H	40%	C	20269	14	6	0	1	13	122019.4	76616.82	0	12161.4	263497	474294.6	12164189
		Y	35419	28	13	0	2	27	426444.8	290081.6	0	42502.8	956313	1715342	
		I	44636	28	13	0	2	27	537417.4	365568.8	0	53563.2	1205172	2161721	
		O	43762	104	49	0	7	100	1957037	1295793	0	183800.4	4376200	7812830	

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R3N_H	60%	C	20269	21	10	0	1	21	183029.1	127694.7	0	12161.4	425649	748534.2	11671554
		Y	35419	42	20	0	3	40	639667.1	423965.4	0	63754.2	1416760	2544147	
		I	44636	42	20	0	3	40	806126.2	534292.9	0	80344.8	1785440	3206204	
		O	43762	69	32	0	5	66	1298419	854671.9	0	131286	2888292	5172669	
R3N_H	80%	C	20269	28	13	0	2	27	244038.8	178772.6	0	24322.8	547263	994397.2	11204340
		Y	35419	56	26	0	4	54	852889.5	557849.3	0	85005.6	1912626	3408370	
		I	44636	56	26	0	4	54	1074835	703017	0	107126.4	2410344	4295322	
		O	43762	34	16	0	2	32	639800.4	413550.9	0	52514.4	1400384	2506250	
R1Y_H	20%	C	20269	2	5	2	0	6	17431.34	51077.88	14188.3	0	121614	204311.5	11633755
		Y	35419	6	9	5	1	12	91381.02	200825.7	61983.25	21251.4	425028	800469.4	
		I	44636	6	9	5	1	12	115160.9	253086.1	78113	26781.6	535632	1008774	
		O	43762	54	88	48	10	119	1016154	2398595	735201.6	262572	5207678	9620201	
R1Y_H	40%	C	20269	5	9	4	1	12	43578.35	127694.7	28376.6	12161.4	243228	455039.1	11182265
		Y	35419	11	18	10	2	24	167531.9	379337.5	123966.5	42502.8	850056	1563395	
		I	44636	11	18	10	2	24	211128.3	478051.6	156226	53563.2	1071264	1970233	
		O	43762	41	67	36	7	89	771524.1	1792054	551401.2	183800.4	3894818	7193598	
R1Y_H	60%	C	20269	9	13	8	1	18	78441.03	178772.6	56753.2	12161.4	364842	690970.2	10745833
		Y	35419	16	27	14	3	36	243682.7	580163.2	173553.1	63754.2	1275084	2336237	
		I	44636	16	27	14	3	36	307095.7	731137.7	218716.4	80344.8	1606896	2944191	
		O	43762	27	44	24	5	59	508076.8	1185513	367600.8	131286	2581958	4774435	
R1Y_H	80%	C	20269	11	18	10	2	24	95872.37	229850.5	70941.5	24322.8	486456	907443.2	10326128
		Y	35419	22	36	19	4	48	335063.7	780989	235536.4	85005.6	1700112	3136707	
		I	44636	22	36	19	4	48	422256.6	984223.8	296829.4	107126.4	2142528	3952964	
		O	43762	13	21	12	2	29	244629.6	578971.3	183800.4	52514.4	1269098	2329014	

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R2Y_H	20%	C	20269	6	3	3	0	8	52294.02	51077.88	21282.45	0	162152	286806.4	13590137
		Y	35419	12	7	4	1	15	182762	133883.8	49586.6	21251.4	531285	918768.8	
		I	44636	12	7	4	1	15	230321.8	168724.1	62490.4	26781.6	669540	1157858	
		O	43762	115	64	44	10	146	2164031	1736914	673934.8	262572	6389252	11226704	
R2Y_H	40%	C	20269	12	6	4	1	16	104588	76616.82	28376.6	12161.4	324304	546046.8	13099276
		Y	35419	23	13	9	2	29	350293.9	290081.6	111569.9	42502.8	1027151	1821599	
		I	44636	23	13	9	2	29	441450	365568.8	140603.4	53563.2	1294444	2295629	
		O	43762	87	49	33	7	110	1637136	1295793	505451.1	183800.4	4813820	8436001	
R2Y_H	60%	C	20269	17	10	7	1	23	148166.4	127694.7	49659.05	12161.4	466187	803868.6	12606699
		Y	35419	35	20	13	3	44	533056	423965.4	161156.5	63754.2	1558436	2740368	
		I	44636	35	20	13	3	44	671771.8	534292.9	203093.8	80344.8	1963984	3453487	
		O	43762	58	32	22	5	73	1091424	854671.9	336967.4	131286	3194626	5608975	
R2Y_H	80%	C	20269	23	13	8	2	30	200460.4	178772.6	56753.2	24322.8	608070	1068379	12103756
		Y	35419	47	26	18	4	59	715818	557849.3	223139.7	85005.6	2089721	3671534	
		I	44636	47	26	18	4	59	902093.6	703017	281206.8	107126.4	2633524	4626968	
		O	43762	28	16	11	2	36	526894.5	413550.9	168483.7	52514.4	1575432	2736876	
R3Y_H	20%	C	20269	8	3	3	0	8	69725.36	51077.88	21282.45	0	162152	304237.7	14535605
		Y	35419	16	7	4	1	15	243682.7	133883.8	49586.6	21251.4	531285	979689.5	
		I	44636	16	7	4	1	15	307095.7	168724.1	62490.4	26781.6	669540	1234632	
		O	43762	157	64	44	10	146	2954373	1736914	673934.8	262572	6389252	12017046	
R3Y_H	40%	C	20269	15	6	4	1	16	130735.1	76616.82	28376.6	12161.4	324304	572193.9	14018584
		Y	35419	32	13	9	2	29	487365.4	290081.6	111569.9	42502.8	1027151	1958671	
		I	44636	32	13	9	2	29	614191.4	365568.8	140603.4	53563.2	1294444	2468371	
		O	43762	118	49	33	7	110	2220484	1295793	505451.1	183800.4	4813820	9019349	

Scenario	Forest felled (%)	Age	Lichen (kg) per square (L)	Road ZOI (Rsq)	Town ZOI (Tsq)	Mine ZOI (Msq)	Hydro ZOI (Hsq)	Multiple ZOI's (RTMHsq)	Road lichen loss (kg) (Rsq x L x 0.43)	Town lichen loss (kg) (Tsq x L x 0.63)	Mine lichen loss (kg) (Msq x L x 0.35)	Hydro lichen loss (kg) (Hsq x L x 0.6)	Multiple ZOI lichen loss (kg) (RTMHsq x L x 1.00)	Total lichen loss (kg/age)	Total lichen loss (kg)
R3Y_H	60%	C	20269	25	10	7	1	23	217891.8	127694.7	49659.05	12161.4	466187	873594	13465861
		Y	35419	47	20	13	3	44	715818	423965.4	161156.5	63754.2	1558436	2923130	
		I	44636	47	20	13	3	44	902093.6	534292.9	203093.8	80344.8	1963984	3683809	
		O	43762	78	32	22	5	73	1467777	854671.9	336967.4	131286	3194626	5985328	
R3Y_H	80%	C	20269	31	13	8	2	30	270185.8	178772.6	56753.2	24322.8	608070	1138104	12946860
		Y	35419	64	26	18	4	59	974730.9	557849.3	223139.7	85005.6	2089721	3930447	
		I	44636	64	26	18	4	59	1228383	703017	281206.8	107126.4	2633524	4953257	
		O	43762	38	16	11	2	36	715071.1	413550.9	168483.7	52514.4	1575432	2925052	

Appendix B. Data on the loss of lichen for grazing from avoidance behaviour and physical structures within the model. These were combined to give total lichen loss, which was then subtracted from the biomass of lichen present in a scenario with no infrastructure. This provided the biomass of lichen remaining in the landscape in each scenario

Scenario	Level of Forestry (%)	Loss from avoidance (kg)	Physical loss road (kg)	Physical loss mine (kg)	Physical loss hydro (kg)	Total lichen loss (kg, LL)	Lichen without structures (kg, LO)	**Lichen remaining (kg, LO-LL)**
R1N	20	9700028	10503	0	0	9710531	41875915	32165384
R1N	40	9336430	10503	0	0	9346933	40646260	31299327
R1N	60	8964736	10503	0	0	8975239	39416603	30441364
R1N	80	8642142	10503	0	0	8652645	38187114	29534469
R2N	20	12568770	25907	0	0	12594677	41875915	29281238
R2N	40	12130680	25907	0	0	12156587	40646260	28489673
R2N	60	11676506	25907	0	0	11702413	39416603	27714190
R2N	80	11167309	25907	0	0	11193216	38187114	26993898
R3N	20	13793409	34659	0	0	13828068	41875915	29281238
R3N	40	13306831	34659	0	0	13341490	40646260	28489673
R3N	60	12800569	34659	0	0	12835228	39416603	27714190
R3N	80	12248288	34659	0	0	12282947	38187114	26993898
R1Y	20	11971442	16804	1099360	0	13087606	41875915	28788309
R1Y	40	11548830	16804	1060934	0	12626568	40646260	28019692
R1Y	60	11071275	16804	999016	0	12087095	39416603	27329508
R1Y	80	10666647	16804	960590	0	11644041	38187114	26543073
R2Y	20	13847639	26957	1099360	0	14973956	41875915	26901959
R2Y	40	13386631	26957	1060934	0	14474522	40646260	26171738
R2Y	60	12879684	26957	999016	0	13905657	39416603	25510946
R2Y	80	12360960	26957	960590	0	13348507	38187114	24838607
R3Y	20	15332693	35709	1099360	0	16467762	41875915	25408153
R3Y	40	14774896	35709	1060934	0	15871539	40646260	24774721
R3Y	60	14209191	35709	999016	0	15243916	39416603	24172687
R3Y	80	13683397	35709	960590	0	14679696	38187114	23507418
R1N_H	20	9542078	14004	0	2306511	11862593	41875915	30013322
R1N_H	40	9159086	14004	0	2245684	11418774	40646260	29227486
R1N_H	60	8836597	14004	0	2145340	10995941	39416603	28420662
R1N_H	80	8512702	14004	0	2084513	10611219	38187114	27575895
R2N_H	20	12434314	29407	0	2306511	14770232	41875915	27105683
R2N_H	40	11979417	29407	0	2245684	14254508	40646260	26391752
R2N_H	60	11495932	29407	0	2145340	13670679	39416603	25745924
R2N_H	80	11037868	29407	0	2084513	13151788	38187114	25035326
R3N_H	20	12711980	34659	0	2306511	15053150	41875915	26822765
R3N_H	40	12260762	34659	0	2245684	14541105	40646260	26105155
R3N_H	60	11763964	34659	0	2145340	13943963	39416603	25472640
R3N_H	80	11292587	34659	0	2084513	13411759	38187114	24775355
R1Y_H	20	11772021	20305	1099360	2306511	15198197	41875915	26677718
R1Y_H	40	11314509	20305	1060934	2245684	14641432	40646260	26004828
R1Y_H	60	10873241	20305	999016	2145340	14037902	39416603	25378701

R1Y_H	80	10448701	20305	960590	2084513	13514109	38187114	24673005
R2Y_H	20	13689689	30458	1099360	2306511	17126018	41875915	24749897
R2Y_H	40	13195851	30458	1060934	2245684	16532927	40646260	24113333
R2Y_H	60	12699110	30458	999016	2145340	15873924	39416603	23542679
R2Y_H	80	12192003	30458	960590	2084513	15267564	38187114	22919550
R3Y_H	20	14635157	35709	1099360	2306511	18076737	41875915	23799178
R3Y_H	40	14115158	35709	1060934	2245684	17457485	40646260	23188775
R3Y_H	60	13558273	35709	999016	2145340	16738338	39416603	22678265
R3Y_H	80	13035107	35709	960590	2084513	16115919	38187114	22071195

```
#Dataframe
lichenDF<-read.csv("lichen_dataframe.csv")


###Model Function###
#Final Population
model<-function(renpop,ice,cull,mine,hydro, road, volume){
  N0<- lichenDF[volume, ((road*mine)+hydro)]
  N1<-((N0*1.016)-((renpop-cull)*298.62))
  if ((renpop-cull)>(N1/298.62)) {R1<-((N1/298.62))} else {R1<-(renpop)}
  N2<-((N1*1.016)-((R1-cull)*298.62))
  if (((R1+(renpop*0.04))-cull)>(N2/298.62)) {R2<-((N2/298.62))} else {R2<-
((R1+(renpop*0.04)))}
  n<-N2
  r_prev<-R1
  r_current<-R2
  for (i in 1:48){ n <- ((n*1.016)-((r_current-cull)*298.62))
  r_temp<-r_current
  if (((r_current+(r_prev*0.04))-cull)>(n/298.62)) {r_current<-(((n/298.62)
*ifelse(i  % % 5 == 1, ice, 1)))} else {r_current<-((r_current+(r_prev*0.04
))*ifelse(i  % % 5 == 1, ice, 1))}
  r_prev<-r_temp}
  return(r_current)
}


##Loop of inputs
data<- data.frame(initialPop=numeric(0),icingpresent=character(0),slaughter
ed=numeric(0),minepresent=numeric(0),hydropresent=numeric(0),roadextent=num
eric(0),forestry=numeric(0), finalPop=numeric(0))


for (renpop in c(500,2500,5000,7500,10000)){
  for (ice in c(0.77,1)){
    for (cull in c(25,500,1000)){
      for (mine in c(2,1)) {
        for (hydro in c(9,0)){
          for (road in c(2,3,5)){
            for (volume in c(1,2,3,4)) {
              result <- model(renpop,ice,cull,mine,hydro,road,volume)
```

```
              ice_ <- if(ice == 0.77) "Yes" else "No"
              mine_ <- if(mine == 2) "Yes" else "No"
              hydro_ <- if(hydro == 9) "Yes" else "No"
              road_ <- if(road ==2) "1" else (if (road ==3) "2" else "3")
              volume_ <-if(volume ==1) "20" else (if (volume ==2) "40" else 
(if (volume ==3) "60" else "80"))
              result_ <- if(result <0) 0 else result
              data[nrow(data) + 1,] = c(renpop,ice_,cull,mine_,hydro_,road_
,volume_, result_)
            }
          }
        }
      }
    }
  }
}
```

Appendix D. Studies omitted from calculations on reindeer avoidance behaviour, and reasons for being omitted

Topic	Reason Omitted	Study
Road Avoidance	Undertaken during calving season	Dau and Cameron 1986; Vistnes and Nellemann 2001
Road Avoidance	Measured lichen abundance rather than reindeer behaviour (not comparable)	Dahle *et al.* 2007
Population Centre Avoidance	Undertaken during calving season	Vistnes and Nellemann 2001
Mine Avoidance	Undertaken in summer	Boulanger *et al.* 2012

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