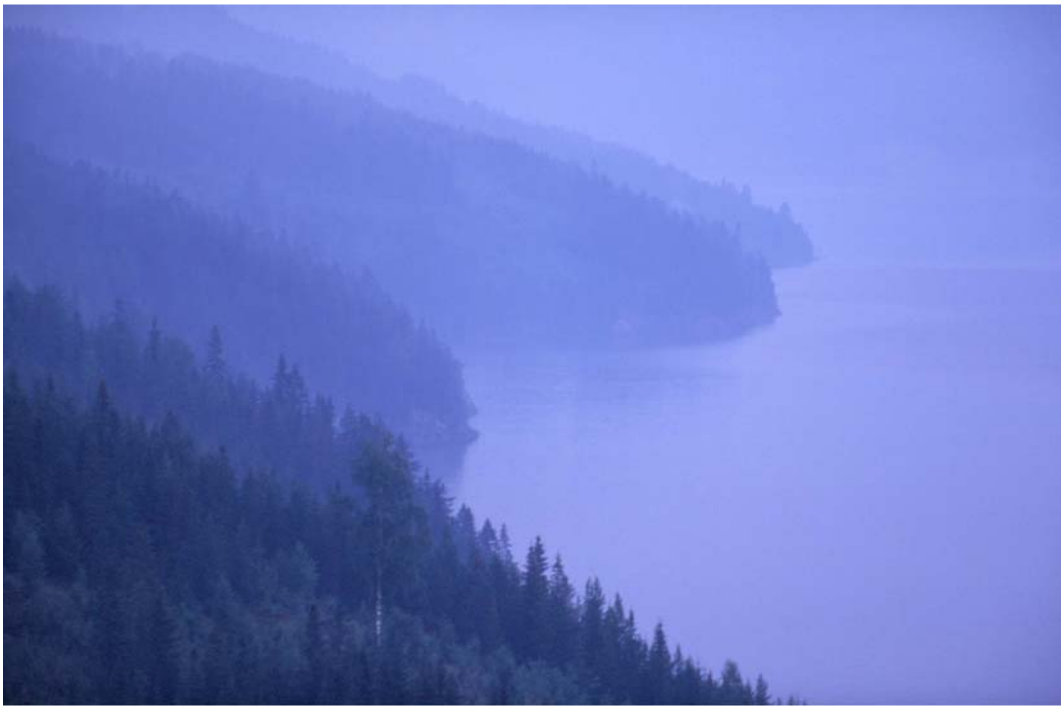
NINA Report 62

Spatial aspects of managing natural resources and conserving biodiversity

Integrating the global and the local

John D. C. Linnell



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Abstract

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The issue of scale has recently attracted much importance in ecology. It is also of crucial importance in the context of managing nature. The question that constantly appears is “at what level should we manage our natural resources?” From the point of view of the social sciences, there is a trend to move resource management to the local level, to satisfy objectives such as the desire for social equity, social justice and local empowerment. However, from the point of view of ecology, the recent development of ecosystem thinking calls for management to be moved up to larger spatial scales such that holism can be maintained. This apparent contradiction of the “local ecosystem” is actually imbedded in the Convention on Biological Diversity’s Malawi principles. Finding ways to include both the local and the global is clearly a key need for improving natural resource management.

The first part of the report presents some concrete analyses where the importance of scale issues for Norwegian natural resource management is highlighted. The examples include (1) an analysis of the various spatial scales at which ecological processes operate, (2) using data on movement of radio-collared lynx to improve national level monitoring, (3) using data on wolf dispersal and Baltic sea ice conditions to explore the potential for wolves having re-colonised Scandinavia from Finland on their own, (4) drawing up principles for the spatial aspects of planning large carnivore recovery in Norway that combine both ecological, administrative and social aspects, and (5) a set of analyses to see to what extent scale varies within and between species, and the identification of ways of predicting scale. The main message from this section is that all scales need to be considered. At least when it comes to large carnivores, where each individual uses several municipalities, “local” management must be considered on scale of 10s or 100s of square kilometres.

The second part of the report is a review of the scientific literature dealing with the issues of scale and resource management from the viewpoints of many different disciplines. We firstly look at the experience with community based conservation, various decentralization or devolution projects, and co-management systems especially those that have been conducted in tropical countries and the arctic. The experiences from these attempts to pass greater resource management responsibility to the local level are mixed. Many of those from tropical countries are negative, with the co-management tradition from the arctic offering the most optimistic outcomes. Problems with lack of capacity, elite capture, and corruption often led to a decline in the sustainability of resource use, and often a decrease in equity as well. Secondly, we examined the literature dealing with democracy and environmental justice, which provided many insights into the issue of scale, NIMBYism (Not In My Back Yard) and decision making within the context of public goods and local costs. Again, the experience at the local level is poor, especially when the costs and benefits of resource conservation are felt at different scales. Thirdly we look at some of the characteristics of different resources in light of how suitable they might be for more local – or less local – level management, and develop some guiding principles.

Finally, we attempted to gather these various inputs together and come up with a conceptual framework for future thinking. It is apparent that resource management cannot be conducted at a single scale – be it local or national or global or someplace in between. There is a need to consider all scales – with different decisions being appropriate for different scales. Management should be viewed as a nested hierarchy, where upper levels set general frameworks of guidelines and principles, and the lower levels make increasingly detailed, and locally adapted, decisions, within the framework set by the upper levels. This approach should in principle allow “freedom within limits”, and follows the principle of subsidarity. The challenge for the future is to ensure the effective coordination of management at multiple scales and to ensure democratic representation of the lower (local) levels in the upper level decision making processes.

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Sammendrag

Linnell, J. D. C. 2005. På hvilke nivå skal vi forvalte naturen? Integrering av det lokale og globale - NINA Report 62. 38 pp.

Betydningen av *skala* har etter hvert fått stor oppmerksomhet innen økologien. Skala har også avgjørende betydning i naturforvaltningssammenheng. Spørsmålet som stadig dukker opp er: "På hvilket nivå skal vi forvalte naturressursene våre?". Fra et samfunnsvitenskapelig ståsted er tilrådningen ofte å legge forvaltningen til lokalt nivå for å tilfredsstille kravet om rettferdighet og lokal bestemmelse over egne omgivelser. På den annen side vil det fra en økologisk synsvinkel – og særlig innenfor nyere økosystemtenkning – være mer hensiktsmessig å legge forvaltningen til nivåer som dekker områder over en langt større skala enn bare det lokale, slik at helheten i økosystemet ivaretas. Det motsetningsfylte i det en kan kalle "lokale økosystem" gjenspeiles like fullt i de såkalte *Malawi-prinsippene* i *Konvensjonen om biologisk mangfold*. Naturforvaltningen har et åpenbart forbedringsbehov i det å finne måter å inkludere både det lokale og globale på.

Første del av rapporten presenterer konkrete analyser som viser betydningen av skala for norsk naturforvaltning. Eksemplene inkluderer (1) identifisering av ulike romlige skalaer for økologiske prosesser, (2) bruk av bevegelsesdata fra radiomerkede gauper til å forbedre nasjonal bestandsovervåkning, (3) bruk av data på spredningsavstander hos ulv og på isforholdene i Østersjøen for å undersøke ulvens potensiale for naturlig rekolonisering i Skandinavia, (4) utvikling av arealprinsipper for bevaringen av store rovdyr, der en kombinerer både økologiske, administrative og sosiale forhold, og (5) en rekke analyser av hvordan skalaen for arealbruk varierer innenfor en og samme art og mellom ulike arter, og identifisering av måter å forutsi dette på. Hovedbudskapet i denne delen er at alle skalaer må tas med i betraktningen. "Lokal" forvaltning er nødt til å betrakte områder på skalaer fra noen titalls til hundrevis av kvadratkilometer, i hvert fall når det kommer til arter slik som store rovdyr, der et enkelt individ kan bruke områder som dekker flere kommuner, fylker og nasjoner.

Andre del av rapporten gjennomgår litteratur fra en rekke fagdisipliner der en ser på skala og nivå for ressursforvaltning. Her går vi først gjennom erfaringene fra ulike modeller for lokalsamfunnsbasert naturvern, prosjekter med desentralisering og økt selvstyre, samt systemer der det er lagt opp til medvirkende forvaltning. Mesteparten av erfaringen er hentet fra land i tropiske strøk og i Arktis. De ulike forsøkene på å overføre mer ansvar til lokalt nivå har gitt blandede resultater. Mens mye av erfaringen fra de ulike forsøkene i tropiske land synes å være negativ, ser vi de mest positive erfaringene fra Nord-Amerika og Arktis og de systemene en der har for medvirkende forvaltning. Mangel på kapasitet på lokalt nivå, tilstedeværelsen av korrupsjon, samt dominans fra lokale og/eller globale maktpersoner/-aktører har i mange tilfeller ført til både nedgang i bærekraftig ressursutnytting og redusert lokal rettferdighet. Videre i denne delen gjennomgås litteratur om demokrati og miljømessig rettferdighet der en tar for seg en rekke skalarelaterte problemstillinger, den såkalte "ikke i min bakgård" - holdningen, og beslutningstakning i en verden der hensynet til fellesskapsgoder ofte kan gi lokale kostnader og ulemper. Igjen synes erfaringene på lokalt nivå å være negative, særlig når fordelene og ulempene – for eksempel ved den valgte naturvernpolitikken – oppleves forskjellig på ulike nivåer. Videre igjen ser vi på egenskaper ved ulike naturressurser i lys av hvor egnede de er for større eller mindre grad av lokal forvaltning, og vi foreslår i denne sammenheng noen veiledende prinsipper.

Til slutt forsøker vi ut fra dette kunnskaps- og erfaringsgrunnlaget å gi et begrepsmessig rammeverk for fremtidig tenkning. Det er åpenbart at naturforvaltning ikke kun kan foregå på en enkelt skala – enten den er lokal, nasjonal, global eller et sted midt i mellom. Det er behov for å betrakte alle skalaer, der ulike beslutninger passer på ulike nivå. Naturforvaltning må sees på som et sammenvevd hierarki der de øvre nivåene definerer overordnede mål og generelle rammer, og de understående nivåene har ansvar for utforming av mer detaljerte og lokalt tilpassede forvaltningsgrep innenfor disse rammene. En slik tilnærming gir "frihet under ansvar". Dette er også i tråd med det såkalte *nærhetsprinsippet* i EU. Fremtidens utfordring er å sikre effektiv koordinering av forvaltning over mange ulike skalaer, og å sikre demokratisk representasjon fra de lavere (lokale) nivåene i den overordnede beslutningstakningen.

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Contents

Foreword

The organisational structures of Norwegian nature management are in a period of change, as more authority is passed to a local level. Simultaneously, Norwegian nature management has to conform to international treaties, and ecological research is focusing on large scale processes and moving in the direction of ecosystem thinking. How can we balance these conflicting trends towards both the global and the local? This review does not provide the definitive answer, neither does it present a robust, statistical analysis of the alternatives. Instead it seeks to clarify the issues and views potential solutions through the lenses of different disciplines. It concludes with some tentative suggestions for the way forward.

This review has been developed within the project entitled "Biological and administrative perspectives on defining the spatial scale for management of interacting resources" funded by the Research Council of Norway (NFR), the Directorate for Nature Management (DN) and the Norwegian Institute for Nature Research (NINA). It has been a fascinating opportunity to read widely, follow my curiosity, think, ponder, and discuss controversial issues. The challenge was to try and condense 5 years of random and often unconnected thoughts. I hope that the product helps to inform, and that it will contribute to a more reflected debate.

I am grateful to all of my colleagues and students who have contributed in various ways to the background papers, and to the many discussions. These include Erlend Birkeland Nilsen, Unni Støbet Lande, Ivar Herfindal, Einar Asbjørnsen, John Odden, Reidar Andersen, Erling Solberg, Ketil Skogen, Hans Chr. Pedersen, Børre Dervo and Håkon Hustad, who has translated the document from English to Norwegian.

John D. C. Linnell, November 2005.

1 Introduction

1.1 The contradiction of “local ecosystems”

The last 30 years have seen an increasing awareness of the threats facing global biodiversity and the rampant overexploitation of valuable natural resources. In attempts to halt biodiversity decline and increase the sustainability of natural resource exploitation there have been a number of paradigm shifts in the way that scientists, conservationists, and the international community recommend that natural resources be managed. These paradigm shifts are the result of independent and parallel activities that have been going on for decades. The most serious attempt to integrate them into an international agreement is the Convention on Biological Diversity's (CBD) and its Malawi principles (Prins 1999; Swanson 1999). Some of the most important paradigm shifts that lie behind this document include (Vogt et al. 2002; Valanko 2003);

(1) Integrating nature and society. The CBD itself has three goals, the conservation of biological diversity, the sustainable use of its components, and the fair and equitable sharing of benefits. The objectives for management are now regarded as a matter of societal choice, and the importance of building democratic, just, and prosperous societies as a prerequisite for successful nature conservation has been emphasized.

(2) Moving management towards the local level. There has been widespread focus on decentralization and devolution of natural resource management to various local levels, in an attempt to create a more effective and just management of resources. For example, we have the principle of subsidarity ("decisions within a political system should be taken at the lowest level consistent with effective action"; Jordan & Jeppesen 2000), which is also codified in European Union legislation.

(3) Towards holism. There has also been a move away from viewing resources in isolation from each other. Now the dominant paradigm is to focus on whole ecosystems, not only from the point of view of a wider range of ecological processes, but also from that of a far wider range of stakeholders and interest groups. The Ecosystem Approach (Korn et al. 2003; Smith & Maltby 2003) that is advocated by the CBD and the parallel field of Ecosystem Management (Grumbine 1994, 1997; Brussard et al. 1998) which is emerging among North American management (the US has not signed the CBD) agencies are frameworks designed to embrace this holism.

Despite the dramatic increases in public and political awareness, in scientific knowledge, in practical experience with various management systems, and with the development of conceptual models such as the Ecosystem Approach there is still a great deal of controversy and uncertainty about how these ideas work in practice. At least part of this controversy is due to the belief that some of the joint goals of the CBD and the new paradigms enshrined in the Malawi Principles actually conflict with each other, even to the point of working towards mutually exclusive goals.

This report aims to examine one of these issues in detail, namely the spatial scale at which resources are managed. The second Malawi principle states that "Management should be decentralized to the lowest appropriate level" - so the vital question is how local is appropriate? The potential contradiction lies in moving responsibility for resource management down to a local level at the same time as expanding the view to include the ecosystem which inevitably will include ecological or social processes that operate at very large (non-local) spatial scales. In effect, a "local ecosystem" is a contradiction in terms.

1.2 Study approach

Our report is motivated by ongoing discussions about the appropriate scale at which Norwegian natural resources should be managed. This discussion has traditionally focused on the harvest management of valued game species such as the wild ungulates (Danielsen 2001; Bråtå 2003). However, our goal is to examine this through the lens of the Ecosystem Approach and therefore we will focus on the whole Norwegian boreal forest ecosystem, and beyond. Central to this is the inclusion of species such as the large carnivores (whose populations are recovering throughout Scandinavia) which strongly interact with large ungulates, in addition to other factors such as climate. We have selected the roe deer (*Capreolus capreolus*) and the Eurasian lynx (*Lynx lynx*) for illustrative focus. In addition, the project has analyzed data on wolves (*Canis lupus*), moose (*Alces alces*), wild reindeer and caribou (*Rangifer tarandus*) in Norway and Greenland, and muskox (*Ovibos moschatus*) in Norway. The report addresses a number of issues, including;

(1) At what spatial scales do ecological processes operate?
(2) Are scales constant?
(3) Examples of the direct application of scale data in management.
(4) International experience with managing resources at different scales.
(5) Do our present paradigms collide?
(6) Principles and blueprint for the future.

The first half deals with some concrete examples where the results of data analysis is used to illustrate the importance of spatial scale, and where these results are applied to current management situations. The second part deals mainly with concepts and principles.

This review has been developed within the project entitled "Biological and administrative perspectives on defining the spatial scale for management of interacting resources" funded by the Research Council of Norway (NFR), the Directorate for Nature Management (DN) and the Norwegian Institute for Nature Research (NINA). However, most of the data collection and the statistical analysis of existing data, has been conducted under the auspices of various projects also funded by the NFR "Changing landscape" research program, especially that of "Large carnivores and human societies" (ROSA). In addition, we have reviewed a wide range of literature on the subject with a special focus on picking up the way the issue is viewed from different disciplines. These include anthropology, common-pool resource management, landscape ecology, human demography, population dynamics, population genetics, political ecology, political geography, sociology, human-dimensions, economics, philosophy and conservation biology. In addition, we have examined literature from oceanic, coastal and terrestrial habitats, from arctic, tropical and temperate zones. While the questions examined may overlap greatly, these disciplines and regions traditionally maintain unfortunately rigid borders between each other. We hope that our work will help practitioners within each discipline see how achieving real world results in the field of conservation requires interdisciplinary thinking. Our work differs from much of what is already published because we focus heavily on the multi-use, and often privately owned, landscape rather than protected areas (**Linnell et al. 2001a,b**). All references in **bold** text are products of this project. We have deliberately not analysed the relative success of Norwegian management systems. Rather we have chosen to present ecological data and perspectives relevant for Norway against a wide international evaluation of the topic.

2 At what scales do ecological processes operate?

A central aim of this project has been to identify at which spatial scales various ecological processes operate. However, even the different aspects of a single species ecology can operate at widely different scales. In this example we look at the roe deer.

Many ecologists state that the most interesting thing a species can do is to die! A carcass provides carrion for many scavengers and decomposers and a very strong nutrient pulse to a localized area. We have conducted studies on roe deer carcasses placed in the forest (as simulated lynx or wolf kills) and studied the scavengers, the decomposers, and the vegetation. Video monitoring has revealed that many species utilize a carcass, including red fox, pine marten, crows, ravens, magpies and tits – indicating that a carcass has an impact on an area of at least several kilometers. However for beetles and vegetation we have documented that the effect of the carcass is confined to some tens of metres (**Melis et al. 2004; Teurlings et al. unpublished**).

While alive, roe deer occur in a wide range of habitats throughout Norway, from boreal forest to agricultural areas. However, habitat can influence roe deer on a fine scale. We have shown that the proportion of habitats available to a roe deer within its home range (a scale of 10-100 ha) can influence its reproductive success. For example, in agricultural areas without predators, individuals with greatest access to forest within their home range produce larger litters that achieve greater winter weights (**Nilsen et al. 2004**). In boreal forest habitat where predators occur we have shown that fawns born in home ranges with high proportion of edge habitat are exposed to more red fox predation (**Panzachi et al. in prep.**).

Although adult roe deer tend to occupy stable home ranges, or at most partake in relatively short seasonal migrations, juvenile roe deer disperse over very large distances (up to 120 km). However, dispersal distances and rates vary widely between populations. The conventional wisdom among early roe deer researchers claimed that animals should disperse more at high density, keeping the population size constant. Although this idea of so called “social regulation” was attractive, the last 25 years of data indicates that the opposite actually happens. In low density populations, many animals disperse, and disperse far, while in high density populations, very few animals disperse (**Linnell et al. in prep**.). The same pattern can be seen in the historical spread of roe deer across Scandinavia. Starting in southern Sweden in the mid 19th century, roe deer have currently spread to the shores of the Barents Sea. During this expansion, they spread slowly in the rich productive southern habitats, but expanded rapidly in the more northern, low productivity habitats (**Andersen et al. 2004**). (A similar pattern of decreasing dispersal frequency with increasing density has also been seen in the muskox *Ovibos moschatus* population on Dovrefjell, **Asbjørnsen et al. 2005**).

Roe deer are also the most common prey of lynx and are also killed by wolves, and are therefore affected by factors operating on much larger scales than individual roe deer move over. Individual lynx and wolves have home ranges that cover hundred, or thousands of square kilometers, and Norwegian large carnivore populations are clearly influenced heavily by management policy in neighboring Sweden and Finland (**Andersen et al. 2003; Linnell et al. 2001, 2005a**) from where individuals can easily disperse into Norway.

Finally, on the largest scales of all (continental) we have shown how roe deer populations are negatively effected by snow depth, and described how large scale climatic phenomena such as the North Atlantic Oscillation actually synchronize roe deer population dynamics over scales of several hundred kilometers (**Grøtan et al. 2005**).

In other words roe deer ecology operates on scales varying from 20m to the entire North Atlantic. Incorporating this knowledge into roe deer management is a challenge, but in some ways the complexity also provides inspiration. It should no longer be a question of “at what administrative level should roe deer be managed”, but rather “at which levels should each of the many factors influencing roe deer be managed". In other words rather than agonizing over which scale responsibility should lie with, we must view wildlife management as being multi-scalar, with different decisions being made at different levels (table 1).

Table 1. *Relating ecological and administrative scales for an interacting predator prey system.*

		<1	**<10**	**<100**	**<1000**	**>1000**
Roe deer						
	Home range (km^2)	X	X			
	Migration (km)	X	X			
	Dispersal (km)	X	X	x		
	Climate effects (km)	X	X	X	X	
Lynx						
	Home range (km^2)				X	x
	Dispersal (km)			X	X	
Administrative						
	Private property (km^2)	X	X	x	x	
	Municipality (km^2)			x	X	x
	County (km^2)				x	X

3 Can we predict intraspecific variation in scales?

As we have seen home range size is important for matching ecological scales with administrative scales. Not surprisingly home ranges of animals vary widely depending on their ecology, body size, habitat and geographic distribution. Earlier studies focusing on inter-specific comparisons (Harestad & Bunnell 1979; Gittleman & Harvey 1982; Kelt & Van Vuren 2001) have confirmed general patterns that home range size increases with body size and the degree of carnivory. However, there is considerable variation in home range size within a species. Our studies of Eurasian lynx have shown that home range size can vary by a factor of 10 within Europe (Jedrzejewski et al. 1996; **Linnell et al. 2001a**; **Herfindal et al. 2005**). Similarly, roe deer home ranges can also vary by a factor of 10 to 20 (Danilkin & Hewison 1996). Clearly, transferring data from one area to another is fraught with risk.

In an attempt to explain this variation we have investigated the extent of intraspecific variation in home range size among mammalian carnivores, a well studied group for which much variation exists. Our first analysis was designed to demonstrate the importance of intraspecific variation when exploring the influence of other life history traits such as body size (**Nilsen & Linnell submitted**). Using the same species as Harestad & Bunnell (1979) we analysed how using home range data from different populations of a species would influence the home-range size body size relationship. The results of this resampling analysis revealed that the choice of which populations were selected when representing a species could have dramatic results on the overall allometric relationship. The exponent varied from 0.3 to 1.54 depending on the population selected. A further analysis where the choice of species to include in the analysis was varied led to even greater variation, with the exponent varying from 0.18 to 2.76.

Our next step was to try and explain intraspecific variation in home range size for one of our key study species, the Eurasian lynx (**Herfindal et al. 2005**). Within individual home ranges in southeastern Norway we found that home range size varied with an index of roe deer density (based on harvest density). Lynx home ranges with a higher roe deer density index were smaller. On a European scale we related mean home range sizes in 10 different study areas with remote sensing derived estimates of environmental variation. The Fraction of Photosynthetically Active Radiation (FPAR) was used to calculate indices for both overall environmental productivity and seasonality. For lynx, there was a clear pattern with lynx in more productive, and less seasonal, areas having smaller home ranges (**Herfindal et al. 2005**).

To test the generality of this finding further, we compiled home range size and FPAR data from 199 studies of 12 carnivore species (**Nilsen et al. 2005**). For eight of the species we found significant relationships between home range size and FPAR derived measures of either seasonality, or productivity, or both. However, the relationships were often complex. For example, Canadian lynx (*Lynx canadensis*) showed a negative relationship between home range size and seasonality, while bobcat (*Lynx rufus*) showed a positive relationship.

Our focus on intraspecific variation in home range size may appear to make it harder to predict home range size, and therefore the appropriate size of management units. However, the fact that we have found simple indices of environmental variation that can explain a large part of this variation provides hope. Future directions should try and incorporate these environmental variation indices with basic life-history data (body size) and qualitative measures that reflect ecology (such as mean prey size). We are confident that such analyses will greatly increase our ability to predict home range size for different species in different environments given some basic knowledge of their ecology.

4 Application of scale data

During this project we have developed a number of direct management applications of spatial scale data. The following sections provide a brief illustrative overview of some of these.

4.1 Using movement data to develop population monitoring methodology for lynx

The Norwegian management system for large carnivores (Miljøverndepartement 2003) places heavy emphasis on population monitoring. As lynx are managed as a game species and are subject to relatively heavy harvest (quotas are very close to the maximum population growth rates), there is a continual need for monitoring data so that quotas can be adjusted each year. Monitoring large carnivores is always a difficult and expensive activity (Linnell et al. 1998), however we have developed some cost-effective methods based on knowledge of movement data. The main method used for monitoring lynx is based on minimum counts of reproductive units, or family groups. Each winter observations of tracks of two or more lynx seen together are collected and verified. Using knowledge of the size and shape of lynx home ranges, the territorial nature of adult animals, and of movement rates, we were able to produce a set of distance rules that can be used to identify tracks that are so far apart that they are unlikely to be from the same family group (**Linnell et al. in press a**). The result is a conservative minimum count of the number of reproductive females present in the population. The availability of telemetry data has allowed us to standardize the methodology, with locally adapted distance rules scaled to the populations space use (longer distances are used in less productive habitats). National level application of this methodology has revealed that the Norwegian lynx population has declined by 40% during the last 9 years (**Linnell et al. submitted a**).

Because of large variation in lynx reproduction (Andrén et al. 2002; Andersen et al. 2003) it has also been desirable to develop an index to monitor the total lynx population rather than just the reproductive portion. Using movement data we were able to simulate the probability of lynx movements intersecting transect lines placed in different densities and configurations with respect to the landscape (**Linnell et al. in press b**). This modeling has allowed the development of a network of transects that form the basis for a track count index which compliments the family group count. Together these methods form the basis of the National Large Carnivore Monitoring Program's lynx component (**Linnell & Brøseth submitted**).

4.2 Where did the wolves come from?

Since wolves reappeared in southern Scandinavia in the 1980's, there has been constant controversy about their origins. Researchers have demonstrated that the genetical profile of the wolves indicates that they are of Finnish-Russian origins. However, debate centers on how they got to southern Scandinavia, with skeptics claiming that the animals have been released rather than having arrived on their own. In an effort to clarify what is biologically possible for wolves we summarized all existing data on dispersal movements from North America (little published European data exists), and related this to the distances they would need to travel. We also considered the Baltic Sea ice data to evaluate how often an ice crossing would have been possible to shorten the distance. The review indicated that the distance is possible for wolves to travel, although it is at the limit of what has been documented. The ice route would greatly shorten the distance, and wolves in other populations have made similar ice crossings. The conclusion is that it is possible for wolves to have recolonised south Scandinavia on their own, and that there is no need to invoke conspiracy theories or illegal reintroductions to explain the return of the wolves (**Linnell et al. 2005a**).

4.3 Planning for large carnivore recovery

Extrapolating from the ecological studies of the movements of individual large carnivores it is apparent that their populations operate on very large spatial scales. Accordingly, it is important to look at similar scales when planning for their conservation. A first step was to model the availability of potential habitat, which indicates the potential limits for their recovery. The area that we choose to examine was the Scandinavian peninsula, consisting of all Norway, Sweden, and Finnish Lapland. Using data from radio-collared adult females to describe suitable habitat we used a Mahalanobis distance statistic within a Geographic Information System to identify areas of potential habitat for lynx, bears, wolves and wolverine throughout the peninsula (**Lande et al. 2003; May et al. in press**). The results indicated that >90% of the Scandinavian peninsula is potential habitat for wolves, lynx and bears, while around 50% is potential wolverine habitat. This potential habitat is also more or less continuous, indicating that fragmentation is not presently an issue. These results indicate that the Scandinavian peninsula can host several thousand individuals of each species, and that managers have a great deal of freedom in deciding where to conserve the various species.

The second step was to combine the maps of habitat suitability with maps of material conflict potential (domestic sheep, semi-domestic reindeer, beehives) in Norway. The results of these analyses indicate that there are enormous differences in the conflict potential between different parts of the country (**Linnell et al. 2003; Lande 2004**). However, when considering the large home range sizes of large carnivores there were very few areas big enough for a single individual home range without some material conflicts being present. This implies that adopting a geographically differentiated management will only influence the degree of conflict rather than the presence or absence of conflict. The difficulty in totally avoiding conflicts is enhanced by the fact that adult males and juveniles of both sexes of all four species range over larger areas than the reproductive part of the population. This creates a zone of influence surrounding the known breeding distribution within which depredation on livestock can occur. This zone of influence extends up to 200 km, and is greatest for wolves, followed by bears, lynx and wolverines (**Linnell & Brøseth in prep. a**). The implication is that management zones will need to be large, and that it will not be possible to maintain sharp boundaries or steep density gradients.

4.4 Conserving carnivores in protected areas or the multi-use landscape?

A simple comparison of home range sizes of species such as large carnivores and the area of habitat available within national parks or nature reserves revealed that the Norwegian protected area network has very little contribution to make to the conservation of bears, wolves and lynx. Only a few individuals can potentially live exclusively within protected areas. Wolverines are a partial exception in that a relatively larger portion of their potential habitat and present day distribution is found in protected areas (**Lande et al. 2003; Linnell et al. 2001a, 2003**). In addition, livestock are often grazed within national parks indicating that even protected areas are not free from sources of material conflict. A consequence of this is that the conservation of large carnivores will have to take place in the arena of the multi-use landscape, a landscape where much of the land is private property. Fortunately, large carnivores seem to be able to survive well within multi-use landscapes with relatively high human densities as long as legislation is favourable to their conservation (**Linnell et al. 2001b**). However, this multi-use landscape approach places serious constraints on the ambition of conservation objectives. Achieving population viability for carnivores, and restoring some ecosystem processes and selective forces may be possible. However, it is highly unlikely that it will be possible for population densities of carnivores and ungulates to establish any form of equilibrium densities through trophic interactions, or for carnivores to exert their full keystone potential (**Linnell et al. 2005b**). In other words it is unlikely that we will ever see the range of cascade effects in the multi-use landscapes of Norway that have been seen in wilderness settings like Yellowstone National Park (Smith et al. 2003; Ripple & Beschta 2003).

4.5 Principles for geographically differentiated management

Norwegian management of large carnivores is based on a system of zoning, or geographically differentiated management. A central element of zoning is that the zones correspond to the appropriate ecological scale at which their target species or habitats operate. When large carnivores are considered it is apparent that different zones must be large, and that the maintenance of sharp borders between zones will be impossible. We have developed a series of basic principles that need to be considered when planning management zones for large carnivores (**Linnell et al. 2003, 2005c**), although the principles should apply to a range of other situations.

● It is necessary to coordinate the zoning of large carnivores with the zoning of actions intended to mitigate conflicts.
● The size and distribution of zones should conform to the scales at which large carnivores use the landscape.
● Different carnivores are associated with different conflicts.
● Different conflicts receive different benefits from zoning.
● Management must be coordinated between the different zones.
● Management must be predictable.

It is also important to consider that zoning must also be acceptable to the human inhabitants that occupy the area (Brosius & Russell 2003). This will often require a compromise between the purely ecological and purely social optimal designs. For example, while a strict zoning policy may make the adaptation of livestock husbandry to large carnivore presence easier and cheaper, it will cause far larger social conflicts from people who feel they are being forced to live in a "reservation" (**Linnell et al. 2005c**). Strict zoning, with minimal population goals may also reduce the possibility for adopting conflict reduction measures such as opening for large carnivore hunting.

5 How can we define management units?

5.1 What is a population?

Before we can determine at what scale biologically sound management should occur we need to define what we actually mean conceptually, and especially what we mean by the idea of population. Text books routinely define a population loosely as a group of interbreeding individuals. This open definition is because no other more robust and universally accepted definition exists in ecology. Ecologists hotly debate both the conceptual and operational definitions (Camus & Lima 2002; Berryman 2002; Baguette & Stevens 2003). The wide range of potential patterns of distributions and spatial structures that a species can adopt make global definitions difficult (Thomas & Kunin 1999). Conceptually, most ecologists view a population as embracing an area within which animals interbreed, and where population dynamics are mainly governed by birth and death rather than immigration or emigration. The problem comes in defining this operationally or quantitatively.

In the face of a lack of theoretical consensus managers are forced to resort to *ad hoc* approaches. Unfortunately, coming up with an alternative definition is beyond the scope of this project. Our goal is here to review the approaches that have been used, and to point out some vital directions for future thinking on the subject.

Different disciplines have tended to define both populations and biologically important management units in different ways. These include taxonomic approaches, ecosystem approaches, distributional and geographical approaches, demographic approaches, economic approaches and behavioral approaches.

Taxonomic approaches. Approaches based on genetics and taxonomic identity have developed to help prioritize limited conservation resources on units where the greatest degree of genetic diversity can be conserved, The United States Endangered Species Act (ESA) has gone furthest in applying this approach through the use of "Evolutionary Significant Unit" (ESU) designations. The basic logic is that there must be significant genetic differences between two areas of a species distribution before they can receive special treatment under the ESA. While this may help ensure that the most genetic variation found within a species is conserved, the approach has received widespread criticism. Firstly, there is the subjective issue of how to define a “significant” genetic difference (Vogler & Desalle 1994; Fraser & Bernatchez 2001). Secondly, it is possible for very low levels of gene flow to even out genetic differences while making no significant contribution to population demographics - in which case what would appear to be one population from a genetic point of view, could be two from a demographic point of view (Taylor & Dizon 1999). Thirdly, the focus on static genetics states completely ignores the fact that modern conservation goals focus much more on the conservation of ecological and evolutionary processes than just on genetic diversity (Bowen 1999). Although the taxonomic approach is mainly useful for large scale conservation prioritizing, it has been operationalized in the harvest management of Canada geese in North America. Canada geese with such clear genetic and morphometric differences that they are given separate subspecies designation from several breeding areas mix on migration and wintering areas where they are harvested. In order to ensure that harvesting does not only focus on one or a few subspecies, studies have been conducted to determine the relative presence of different subspecies in certain harvest areas (Pearce et al. 2000; Scribner et al. 2003).

Ecosystem approaches. Another view of a biologically meaningful management unit is the ecosystem, which has been earlier defined as "an area within which energy flow is balanced" (Odum 1969 in Cianneli et al. 2004). If interpreted strictly this definition is unworkable as most energy comes ultimately from the sun, and the use of this energy is influenced by long range transport of nutrients (Matson et al. 2002). This implies that all energy / nutrient models ultimately exist on a global level. However, the concept can be operationalized by focusing on the area used by the most mobile species that has a significant ecological function. In many cases the most mobile species may be a predator, although there are many other cases where it will be a prey (e.g. wildebeest in the Serengeti ecosystem). This approach has been attempted in two marine systems, the Bering Sea (Ciannelli et al. 2004) and Antarctica (Constable & Nichol 2002), and one terrestrial, the Greater Yellowstone System (Keiter & Boyce 1991). The main conceptual strength of these approaches is that the focus on a multi-species approach where the widest ranging sets the limits.

Distributional and geographical approaches. The most common approach historically has been to base management units on geography. Different geographical regions (watersheds, valley systems, mountain ranges, climatic zones) have been assumed to contain different populations of the species in question. Such an approach may work in some cases with species that have discontinuous distributions or are habitat specialists. However, with mobile species that occupy many, or seasonally distinct, habitats a number of false assumptions can be made.

Demographic approaches. A commonly used approach in the setting of conservation units, for example protected areas, is to define areas that are needed to conserve minimum viable populations (MVP) of the species in question (Wielgus 2002). Usually only demographic viability is considered, as opposed to genetic viability. In recent threshold harvest models that integrate a viability analysis, it is assumed that harvest units equal or exceed the area needed to contain an MVP. Although this approach may be objective up to a point, different analysis approaches can produce widely different estimates of the MVP threshold (Tufto et al. 1999 vs Wielgus 2002). Another little used approach is to examine the spatial scale at which population dynamics lose their synchrony (Grøtan et al. 2005).

Economic approaches. Although it has never been operationalized a logical management unit from the socio-economic and human-dimensions point of view is one that embraces the full range of costs and benefits associated with a resource. This represents the scale at which costs and benefits are internalized.

Behavioral approaches. Determining the spatial scales at which animals use space is intuitively a crucial first step when examining the scale at which they should be managed (Wiens et al. 2002). For mammals at least there are four crucial measures that describe an individual's use of space and its relationships with conspecifics. Firstly, the *home range* is a well developed concept within ecology to describe the area within which adult individuals live their lives during a given season or year. Secondly, in some seasonal environments individuals may have spatially separated seasonal home ranges. In these cases the *migration distance* between seasonal ranges is an important parameter to quantify. Thirdly, there are different ways in which the home ranges of different individuals relate to each other. These patterns of *social organization* vary from "social" where many individuals overlap in their use of an area (wild reindeer, moose), to "territorial" where same sex animals (e.g. lynx) or different social groups (wolves) occupy adjacent, but non-overlapping territories. Fourthly, juveniles often settle at some distance from their parents. This process of *natal dispersal* has important consequences for demography and management.

What may seem surprising is that few attempts have been made to develop ecological theory that link this movement data with applied issues of management scale. In contrast there are many examples where this data has been applied to real world management systems in an *ad hoc* manner. Before telemetry was developed individuals of many species were marked with rings or ear-tags to trace movements. Ever since the development of radio-telemetry technology in the 1960's, animals have been radio-collared and tracked using ever more advanced equipment, from VHF, to Argos, to GPS. Determining patterns of seasonal movement and group membership of individuals has been the objective, with a view to identifying biologically meaningful units for management.

For example, in Alaska, Canada, Greenland and Norway, individual caribou / wild reindeer have been marked in most geographic regions. These marked individuals have revealed that *Rangifer* normally consist of multiple large, more or less discrete units with coordinated (multiple individuals move together in herds) and predictable (strong fidelity in seasonal range use) movements. The transfer of marked individuals between adjacent units appears to be low, and on this basis a number of discrete populations have been recognized. Very little is known about natal dispersal in *Rangifer*, but it is assumed to be very low. Under these circumstances it is usually fairly simple to define operational management units or populations (Hall 1989; Fancy et al. 1990; Ferguson & Gauthier 1992; Valkenburg 1998), although long-term temporal changes in movement need to be considered (Ferguson & Messier 2000). In southern Norway, the discontinuous nature of wild reindeer habitat makes this operational definition of management units even easier. An exception to these patterns are caribou in western Greenland (Cuyler & Linnell submitted), wild reindeer on Svalbard (Tyler & Øritsland 1990), and woodland caribou throughout North America (Rettie & Messier 2001) where animals move individually, and a high degree of substructuring of populations is expected.

Moose, red deer (*Cervus elaphus*) (including North American elk), white-tailed deer (*Odocoileus virgianus*) and mule deer (*Odocoileus hemionus*) have also been studied in this way, and again operational definitions of management units have been made based on the identification of clusters, or discontinuities, of adult movements (Edge et al. 1986; Mackie et al. 1998; Hjeljord 2001). Also in these studies data on natal dispersal is often absent or poor, and is assumed to be low. In migratory populations it has usually been easy to define units with little ambiguity, however, in non-migratory populations occupying continuous habitat it is hard to identify non-ambiguous criteria for grouping individuals into different population management units. Telemetry studies often reveal that the social structuring of these species can be much more complex than is often assumed, for example with the existence of maternal clans in white tailed deer (Aycrigg & Porter 1997; Nelson & Mech 1999; Oyer & Porter 2004).

Global polar bear (*Ursus maritimus*) populations have been intensively studied to identify patterns of population structuring. Early studies used topography and distribution to identify management units (Taylor & Lee 1995). These have been further refined with studies of movements using satellite collars (Bethke et al. 1996; Armstrup et al. 2000; Taylor et al. 2001; Mauritzen et al. 2002; Lunn et al. 2002), and genetics (Paetkau et al. 1999), to the extent that 20 management units are currently recognized throughout the holarctic region. Again it has been the clustering of movements of partially overlapping adult females that has provided the basis for unit designation.

One of the few quantitative approaches that has been proposed to relate movement parameters to an area of management is Wright's neighborhood area estimator (Crawford 1984). Based around known dispersal distances this area is proposed to contain an effective population. The only proposed application of it in management has been for mountain lions (*Puma concolor*) in western North America (Laundré & Clark 2003). The model makes some assumptions about the distribution of dispersal distances and may not be applicable in many cases. However, the approach offers an interesting line of enquiry to develop some objective ways to turn movement data into management units.

5.2 A hierarchy of scales

These approaches do not necessarily compete with each other, because in effect they measure different things. These lie within a hierarchical structure, with genetic considerations residing at a large scale than demographic considerations, and behavioural considerations residing at the smallest scale. All aspects need to be considered at some stage and point in a management process. However, management issues are also hierarchical. Overall policy issues must occur at the largest spatial scales. Below this lies the area of action planning, and below this again lies the area of action implementation. For a game species the policy issues are to determine if harvest is a goal or not, and to develop broad frameworks concerning which methods are acceptable, who should have access, and in what seasons. At the action planning stage, specific hunting quotas can be set for the various "populations", and locally adapted seasons can be set. Finally, even smaller scales can be used to distribute the effort of hunters, or the potential for economic income, perhaps even down to the level of individual landowners. Effective management therefore does not lie at one scale, but at all scales.

5.3 Does it matter if we get it wrong?

Very few spatial models have been applied to management scenarios. Exceptions include Jonzén et al. (2001) and Milner-Gulland et al. (2000) - who have shown that harvesting one population as two, or two as one, can have serious effects on the outcome. For example, making decisions on too coarse a scale can on one hand lead to low harvest yields as locally dense populations may not be harvested as heavily as they could have been. However, it can also lead to local extinction if a quota that might have been defensible on average across a large area, falls disproportionally on a small sub-population. In reverse, making decisions on too fine a scale can also lead to clear errors. This type of problem can also arise if decision making power becomes too fragmented or decentralized. In such a situation it is possible for a single biological unit, such as a population, to fall within the jurisdiction of more than one authority. If management is not coordinated between jurisdictions it is highly likely that severe problems can arise for either jurisdiction to achieve their goals. Effective management must find an appropriate balance of scale. However, at present we just do not have the necessary theory to determine operational and objective guidelines.

6 International experience with management at the local scale

From the arctic to the tropics a wide range of different management systems have been tested where control, influence and responsibility have been transferred to varying degrees from central authorities to local levels (table 2) (Decker et al. 2000). This brief review attempts to draw some of the main conclusions out of the ongoing discussions about the success and failure of these experiments.

Table 2. *A continuum of local involvement in natural resource management. Simplified from Decker et al. 2000.*

Government centralized management
Informing
Consultation
Communication
Advisory role
Collaborative partnership
Community self-governance
Central → → → → → → → → → **Local**

6.1 The rise and fall of community based conservation

Up until the 1970's the dominant paradigm within protected area management was largely the top-down authoritarian model where local people were excluded and prevented from extracting resources. In response to conflicts between parks and people living around them (among other shortcomings of this system) the 1980's and 1990's saw a dramatic growth in attempts to establish bottom-up conservation. A vast range of community-based conservation (CBC) and integrated conservation and development (ICD) projects have been initiated, especially within the tropics. The basic rational of these projects was to link conservation with rural development by allowing local communities to benefit from some of the natural resources contained within protected areas. However, after more than 20 years there is little evidence that the approach has helped biodiversity conservation in the target areas. A number of very critical reviews have been written in recent years pointing out the weakness and false assumptions of the approach (Agrawal & Gibson 1999; Songorwa 1999; Terborgh 1999; Kellert et al. 2000; Newmark & Hough 2000; du Toit et al 2004; Adams et al. 2004). The critiques of these authors focus on a number of issues, including (1) lack of capacity at local scales, (2) presence of local level corruption and the inability for local authorities to resist the domination of local and / or global power players (= elite capture), (3) lack of interest in many communities, (4) increased economic well being often leads to increased environmental impact through increased consumption or by attracting immigrants, (5) project aims are too broad and too long term to be achievable or measurable, (6) local communities are often internally divided, and (7) local communities have often already outstripped the resource base. Furthermore, biodiversity conservation and long-term sustainability are rarely, if ever, more economical in the short term than "resource mining", habitat conversion and intensive land-use. Therefore, the central problem lies in the false expectation that local communities will give up lucrative development opportunities in favor of relatively abstract goals (Newmark & Hough 2000; Nesbit & Weiner 2001; du Toit et al. 2004; Adams et al. 2004). This is especially true for many biodiversity components that are not resources, especially those which are actually sources of economic losses, and may even be dangerous (elephants, large carnivores etc) (Bostedt 1999; Songorwa 1999).

Faced with the many problems of community-based conservation, and the growing urgency of the tropical biodiversity crisis there has been a widespread backlash against this approach

(Kramer et al. 1997; Terborgh 1999). These authors have called for a return to authoritarian protectionism. Not surprisingly a counter reaction has formed against this protectionism, with a range of authors trying to identify the reasons why CBC programs have failed (Brechin et al. 2002; Wilshusen et al. 2002; Berkes 2004). These authors have tried to focus on ways to keep the central tenants of CBC (combined environmental and social goals) while identifying ways to get around the common obstacles to successful CBC.

6.2 Decentralization and devolution

A parallel set of literature exists from the field of political geography where the focus has been on evaluating the success of decentralization and devolution of control over natural resource management to more local levels, again mainly from tropical areas. The goals have also been to increase environmental management efficiency and improve equity and social justice. Typically this literature only evaluates the effect on resource management and not on other biodiversity components. Even when focusing on these economically beneficial resources the experience among published accounts is rather negative (Wyckoff-Baird et al. 2000; Larson 2002; Ribot 2002; Lane 2003; Namara & Nsabagasani 2003; Post & Snell 2003). Problems such as lack of capacity, lack of resources, local level corruption and elite capture, lack of incentive, and the unwillingness of local communities to give up alternative development opportunities associated with non-sustainable resource use also appear in this literature. A common problem is that when the controlling power of central government is removed, the local resources can be easily dominated by local (or global) power elites, making the access to and control of resources less democratic than it was originally (Lane 2003). Decentralization advocates point out that a common problem is that not enough power has been decentralized, or responsibility has been decentralized without the power, authority or resources to back it up (Ribot 2002). The problem is that in the face of widespread abuse of the limited powers which have been decentralized, few governments are willing to decentralize even greater powers. Many authors have pointed out that the role of central government is needed to provide a non-local point of view and to represent a broader range of values associated with a given resource than those that occur locally (Larson 2001).

Furthermore, the experience with common-pool resources provides insights into the difficulties of excluding non-local resource users (Ostrom et al. 1999). In tropical forests much of the current overexploitation of wildlife resources (the bushmeat crisis) is conducted by non-local actors for non-local markets (Robinson & Bennett 2000). Even if local communities wanted to exclude these non-local actors it is unclear to what extent they would have the enforcement capability. Deterring meat hunters might be hard enough, but deterring people seeking to exploit exceptionally high value products like elephant ivory, tiger bones and rhino horn without para-military style resources has proven to be impossible.

6.3 Co-management

Throughout Canada and the United States natural resource management issues are often intertwined with the rights of first nation people. In attempts to balance power between the central and local levels widespread use has been made of the co-management approach (Caulfield 1997; Decker et al. 2000; Zachrisson 2004). These concepts have been expanded to a wide range of other situations involving the management of fisheries and wildlife. While each case has its own design, the general pattern involves creating a forum in which local and central actors meet, and where some decision making power is delegated to this forum. The IUCN defines co-management as "a partnership in which government agencies, local communities and resource users, non-governmental organizations and other stakeholders negotiate, as appropriate to each context, the authority and responsibility for the management of a specific area or set of resources". Experience has been mixed (Dion 2003), but has been far more positive than the CBC and decentralization results from tropical regions (Decker et al. 2000). Unfortunately it is difficult to assess if this is due to management design, or the context (rich countries, higher institutional capacity, greater access to scientific data, lower human densities), or both (Kellert et al. 2000). When extending the concept of co-management to include any form of participatory management there are many examples of the use of this form of management to secure local peoples access to natural resources (see Borrini-Feyerabend et al. 2004), and even several examples of its application to the management of conflict species such as large carnivores. These case studies describe very mixed results (Nie 2003). However, of the international experience that exists, these participatory / power sharing / co-management systems offer some of the best models of incorporating a degree of local level involvement with sustainable outcomes. However, one factor that may have contributed to the relative success of these models may also be a weakness in terms of their general application. This is due to the fact that these systems tend to focus on a single-species or geographically-defined resource (e.g. a single caribou population, salmon within a single watershed). How these co-management systems will work when faced with a multi-faceted ecosystem remains to be seen (Zachrisson 2004).

6.4 Can we transfer experience from the developing world to the developed?

By far the greatest amount of published experience on participatory management, decentralization, devolution, and other community based approaches comes from the developing world and especially from indigenous people (Borrini-Feyerabend et al. 2004). The question is to what extent this can be transferred to the developed world, for example to a country like Norway. There are few objective criteria that can be used to answer this question, however, there are a number of potential differences that need to be considered before experience can be transferred.

(1) Norway is predominantly a Christian country, and therefore is likely to be heavily influenced by the Judao-Christian doministic viewpoint where humans are separate from nature – whereas many developing countries have other religious / philosophical views that often view humans as being integrated into nature (Gardner 2002).

(2) The focus in the developing world has been on community participation processes. In Norway, the sense of community is likely to be weaker because of a high mobility of people between communities, and a general western focus on the individual.

(3) In the developing world literature, the local context within natural resource management tends to be associated with the public (at the community level), whereas the non-local view is focused on the private (e.g. big business / power elites). In Norway, the local context tends to focus on private property (local landowners have harvesting rights), whereas the non-local interest tends to focus on the public interest (at national or international levels).

(4) In the developing world, the focus on equity is often due to the fact that there is a net flow of resources away from local level (i.e. resources extracted locally often bring little local benefit), however, in Norway there is often a net flow of resources from the central to the local level through rural subsidies.

(5) In the developing world there is a focus on maintaining traditional land-use practices and resource management systems. In Norway, the historical context of many resources has been over-exploitation (for example forests and wild ungulates) and even state sponsored extermination (bounties on predators from the 18th century until 1980).

(6) There is generally less direct dependence on local use of natural resources and primary extraction in Norway than in many developing countries.

6.5 Not all biodiversity is a resource

Probably the single biggest problem with the application of many community based management systems are the false assumptions that all biodiversity is a resource, and that sustainable exploitation of resources equates with biodiversity conservation. This is far from the truth. In many cases, biodiversity can be a resource, and it may be possible to harvest many species in a relatively sustainable manner. However, if we consider the modern interpretation of biodiversity to include not only genes and species, but also communities, landscapes, species interactions and ecological processes (Redford & Richter 1999) it is apparent that all exploitation has an impact on biodiversity. In some cases exploitation will have only subtle effects on species and ecosystems, such as changing the age and sex structure of populations, or altering the relative abundance of some species. However, in many cases exploitation may have severe effects on species and ecosystems, potentially leading directly or indirectly to extinctions (Redford & Sanderson 1992; Redford & Feinsinger 2001). The conservation of some habitats, such as old growth forests, may be virtually incompatible with exploitation of any significant resources.

Another class of biodiversity is that which causes conflicts with humans. Many species cause direct economic losses for humans, by depredating livestock (**Linnell & Brøseth 2003**), destroying crops (Hoare 1999), or by transmitting diseases to humans or their livestock. Even when these conflicts can be mitigated, the costs can be very high (Breitenmoser et al. 2005). Some species such as large predators and a wide range of snakes and insects also directly kill people (**Linnell et al. 2002**). In the last few years following the high profile outbreaks of Ebola, SARS, and avian 'flu, together with the spread of Lyme disease and tick-born encephalitis there has been a greater focus in the western world on the role that wild species play as a reservoir and source for zoonosis, a fact long appreciated in the developing world.

Therefore, it is clear that ensuring that the exploitation of a few valuable resources is sustainable need not necessarily lead to the conservation of the rest of the biodiversity living within an ecosystem. Likewise, it is hard to expect local people to carry the costs of living with some ecosystem components just so they can exploit others, or to voluntarily accept lost development opportunities without some forms of incentive or compensation.

7 Democracy, environmental justice and scales

A central scale related issue within the field of political geography is the issue of environmental justice (Williams 1999; Meadowcroft 2002; Kurtz 2003; Gough 2004). The basic issue is that the costs and benefits of certain activities or policies are often unevenly distributed in space, potentially leading to an unjust situation for some social groups. The main focus of research in the field has been on the placement of (locally) undesirable facilities such as toxic waste dumps, polluting industries and nuclear power plants. Early research indicated that a disproportionate number of these facilities were located in regions dominated by social groups with low income and of certain ethnic (non-white) backgrounds. However, later research indicates that simple economics and market forces can explain the observed distributional patterns as opposed to deliberate bias against certain social groups (Williams 1999). Regardless of mechanism, the result is that some people have to live close to facilities that are viewed as being negative. In other words the cost is carried locally for a facility that provides a benefit for a more widely dispersed, or distantly located, public (Singleton 2002). Accordingly local communities adopt a **N**ot **I**n **M**y **B**ack **Y**ard attitude (NIMBYism).

The issue is somewhat similar to that of the conservation of those ecosystem components that cause conflicts with humans, or whose conservation is associated with lost development opportunities. Things that represent a "public good" on a large scale may represent "public bads" locally (Bostedt 1999). Large carnivores are a classic example. Because the costs of material and social conflicts resulting from their presence are felt locally, attitudes to these species are often significantly less positive in the areas where they occur rather than in distant areas and cities (Nesbitt & Weiner 2001; Williams et al. 2002; Bjerke et al. 2003; Ericsson et al. 2003). However, the opposite situation may also occur. For example, in the harvest of Norwegian moose populations where the benefits (recreational opportunities, sale of licenses and meat) of harvesting a "public good" fall to the local landowner while the costs (the bill for compensating forest damage and vehicle collisions, and investment in infrastructure to mitigate vehicle collisions) fall on society as a whole (Danielsen 2001; Storaas et al. 2001).

Callicott (2002) points out that human societies, and their social values, are organized into a nested hierarchy (table 3), where different spatial scales are often associated with different values and social norms (McNeill & Lichtenstein 2003). Conflicts of values and interests between these scales are inevitable, and occur in all fields, not just the environmental (Nie 2004).

Table 3. *The nested hierarchy of human organization, within which different values are associated with different scales.*

Global human community
International neighborhood - continental
Nation or autonomous state
Region or county
Municipality
Community - town or village
Family
Self
Global → → → → → → → → → **Individual**

Callicott recognizes that there is as yet no perfect mechanism to solve cross scale conflicts. However, based on a range of philosophical ideas, including those of Arne Næss, he proposes that simple issues of preference or lifestyle should give way to issues associated with fundamental values and matters of livelihood, and that widely held values (e.g. at the human or national scale) at higher scales should have priority over local values (e.g. at the ethnic or community scale). According to this rational the internationally held values in favor of biodiversity conservation should have preference over more local level values that conflict with conservation, although a difficult value conflict can arise in cases where local people are totally dependent on non-sustainable practices for survival.

In order to minimize the need for values held at one level to overrule those held locally, Callicott (2002) calls for the creative use of win-win solutions - and fortunately there are many good examples of these (Rosenzweig 2003). This is especially true in human-dominated landscapes where conservation goals are set at a more pragmatic level than those set in designated wilderness areas (**Linnell et al. 2005b**). There is also a growing focus on using participatory processes to negotiate outcomes that are viewed as being acceptable. Many formats for participation exist, each with their own advantages and disadvantages (Escobar 1998; Flores & Clark 2001; Nie 2002, 2004; Meadowcroft 2002; Robertson & Hull 2003; Singleton 2002; Borrini-Feyerabend et al. 2004). Building a forum for mutual trust and exchange of ideas is a vital part of the process, as is ensuring that a wide range of stakeholders and interest groups is included. This process of broadening the constituency that is involved can especially help in reducing the tendency of conflicts to polarize (Brox 2000). Peterson et al. (2004) have reviewed a number of cases where the community approach has been used and conclude that we should not expect consensus in many cases where conflicts occur, and that we need to develop a format which allows participatory processes to produce results even in the absence of consensus. Although we are developing some experience with different models in Norway (e.g. **Andersen et al. 2003**; Andersen & Hustad 2004) and elsewhere (Nie 2002, 2004) there is clearly a need for much more research on this topic.

Another way to soften the impact of a value conflict is through economic compensation. The development of community-based conservation has been one broad attempt to compensate for the costs of conservation by allowing some resource use and subsidizing development programs using external funding. However, as we have seen, CBC has had rather variable success. The emerging idea from environmental economists is that it is better to adopt a conservation performance approach where funds are used to pay directly for specific desired outcomes (Ferraro 2001; Ferraro & Kiss 2002). The logic is that the "cheapest way to get something that you want is to pay for what you want, rather than pay for something indirectly related to it" (Ferraro & Kiss 2002). For example, this has been the principle behind such programs as European agricultural and agri-environmental subsidies and the conservation easements practices in the United States (property owners give up some of their property rights in return for payment or tax reductions). Several good examples exist of where these focused economic incentive schemes have increased the local acceptance of a costly, conflict-causing species such as the snow leopard (*Panthera unica*) in India and Mongolia (Mishra et al. 2003; Jackson & Wangchuk 2004). The challenge is to expand these small scale initiatives into larger scale actions, and to expand them from the single species approach into habitat or ecosystem wide approaches.

However, it remains unclear to what extent economic incentives or compensation can influence attitudes and outcomes when the conflicts contain a strong social component in addition to a material / economic component (Skogen & Haaland 2001). In one of the few studies to actually evaluate the relationship, Naughton-Treves et al. (2003) have found that compensation did not improve the attitudes of livestock owners to wolves. Rather, their negative attitudes were founded in far deeper and more fundamental values. However, the payment of compensation was regarded as being successful because it eased the conscience of the wider public.

This field of political ecology must be one of the most under-recognized fields within the entire natural resource management / conservation debate. It is also one of the most crucial as it provides the link between environmental conflicts and the democratic and economic processes that structure and drive our societies. The future development of interdisciplinary approaches must expand the range of disciplines that are brought into play.

8 Balancing the local and the global

Having presented ecological data on animal movement patterns and reviewed a wide range of literature on the issues of local and decentralized management is it possible to find some common threads and draw conclusions? Based on our research and experience we feel that it is possible to draw some general conclusions and make recommendations. Some of these are based on empirical data and review, but a good deal is also based on our own informed and reflected ideas, interpretation, inspiration and creativity. Only further experience and experiments in management will determine if these ideas are valid or not.

It is clear that the former centrally planned, top-down, authoritarian forms of management have many weaknesses, especially when it comes to gaining legitimacy in an increasingly democratic society. The future will inevitably require a greater degree of public participation in the management of natural resources (Borrini-Feyerabend et al. 2004). However, it is also clear that there are often some fundamental problems with the local level management of natural resources as has been tested to date under a wide range of situations.

The bottom line is that natural resources differ greatly in the extent to which they can be managed at different levels, with some resources being more suitable for local level management than others. Characteristics of resources that may be successfully managed locally include (Ostrom et al. 1999; Zachrisson 2004);

● **The resource is of low to medium value**. If resources have exceptionally high value (rhino horn, elephant ivory, tiger bones, wool from Tibetan antelopes) it will be hard to regulate their exploitation, because people will be willing to risk a great deal in return for the high potential gains of violating the regulatory system. The history of market hunting of wildlife and present day fisheries also underlines the problems when resources have exceptionally high value. Local level institutions are only likely to be able to effectively regulate the exploitation of low to medium value resources.

● **The resource is not currently depleted.** If a resource is currently abundant it will be much easier to institute local level management than if the resource is depleted.

● **The resource occurs within small, recognizable borders and has predictable behavior.** Resources must exist locally if there is any chance of managing them locally. If the resources range spreads across several administrative units, these units must cooperate. Management units must at least embrace the annual home ranges of individuals.

● **When exploitation efficiency is low such that there are some intrinsic negative feedback loops**. This will increase the stability of the resource and will help prevent overexploitation even in the absence of accurate monitoring data.

● **When the costs and benefits exist at the same scale**. For example, with wild ungulates many of the costs of damage to agriculture and forestry are felt at the same scale as the benefits from recreation and meat value. This favors local level management as both sides of the conflict are felt within the same community, potentially even by the same landowners. In contrast, large carnivores have high costs locally, while the benefits are felt by the national or global communities, making it very hard for local communities to negotiate with their counterparts in the conflict.

● **Users must have an interest in the sustainability of the resource**. If they have no obvious benefit from it, it is unlikely that they will sacrifice development opportunities associated with its conservation. Unfortunately, the vast majority of biodiversity has an obscure, indirect, intangible, unquantifiable or abstract value.

● **There are well established local norms for resource conservation with punishments.**

● **The resource causes few conflicts with other human activities, and is not heavily influenced by human land-use.**

● **The natural history of the resource is well studied.**

In the Norwegian context these characteristics describe such species as small game, freshwater fisheries, moose, red deer and roe deer which have been managed at a local level for decades with relatively successful results (Danielsen 2001), at least within the terms of the goals that have dominated until the present. How this will function in the future as goals shift is unclear. However, the question remains about the general applicability of local level management to other species or resources with different characteristics. In this context it is necessary to point out a few issues that are often forgotten in the gray zone when "natural resources management" principles and experience are extrapolated to "biodiversity conservation".

● **Not all ecosystem components are resources.** While many ecosystem components can be harvested to produce economically or recreationally important products, there is a great deal of biodiversity that has no quantifiable economic value. In fact, many species are the cause of economic conflicts with humans (Bostedt 1999). These species destroy crops, kill livestock and pets, compete with hunters for game, serve as reservoirs for diseases that affect humans and their livestock and even endanger human life.
● **Not all conflicts associated with biodiversity are material**. A wide range of social/cultural conflicts are also associated with resource exploitation and biodiversity conservation. These are non-economic in nature and include resentment of outside involvement in local affairs and differences in traditions, values and knowledge systems (e.g. Peterson et al. 2002; Skogen & Haaland 2001).

● **Not all exploitation is compatible with conservation**. The debate about the nature of sustainability is exhaustive and ongoing. However, while some natural resources can be harvested with minimal negative effects, there are a wide range of conservation objectives that cannot be achieved in the face of exploitation. Conserving these species or habitats will imply high opportunity costs.

● **The constituency of biodiversity conservation has gone global**. Stakeholders today include most world citizens. Their involvement is sanctioned by a range of international agreements and by their economic ties through donor activities and globalized trade. This makes the definition of stakeholders and community very difficult.

● **The currency of biodiversity conservation is more than economic**. Today there is a growing realization that biodiversity conservation is motivated by ethics and aesthetics as much as the economic (Ghilarov 2000; Norton 2000; Jepson & Canney 2003).

● **Some ecosystem components operate on very large spatial scales.** Some species and ecological processes simply do not exist at a local level - this includes species with large home ranges (wild reindeer, large carnivores) or those that migrate seasonally (many birds, whales, fish), or else they are strongly influenced by large scale geo-physical processes like weather.

In Norway these characteristics describe large carnivores, old growth forest, and to some extent wild mountain reindeer. Large carnivores are associated with costly and controversial conflicts (**Andersen et al. 2003**), use large areas, have little economic value, and their management is bound-up by a range of international agreements. Even though it may well exist on a small scale, old growth forest conservation automatically represents an opportunity cost. Wild reindeer move over large areas in most, but not all areas (**Cuyler & Linnell submitted; Linnell et al. in prep**) and although they represent a valued resource, their conservation requires habitat conservation that prevents the development of other, more lucrative, landuses and activities such as second home development (Bråtå 2003; Andersen et al. 2005). These species / habitats are clearly much less suitable for local level management.

However, if we return to the ecosystem way of thinking it is not possible to just say that different resources / species should be managed independently from each other at different levels because;

- **Few, if any, ecosystem components can be considered in isolation.** All ecosystem components interact with other components, even in human dominated ecosystems where many processes are suppressed. It is therefore impossible to consider the exploitation of a resource in isolation from the ecosystem.

- **Different ecosystem processes occur at different scales**. As we have seen, different ecological processes associated with a single species can occur over scales ranging from meters to hundreds of kilometers.

To return to our large carnivore – large ungulate example: The presence of large carnivores will influence the number of ungulates that can be harvested, while the conservation of the large carnivore populations depend on the ungulates being managed at such a level that there is enough prey. How then should the ecosystem be managed when its interacting components differ in the ecological scale at which they operate? In addition, the political influences on large ungulate management are mainly local or national, but large carnivore issues are usually international - with Norwegian carnivore populations being influenced by immigration from Sweden and Finland, whose populations are in turn heavily influenced by EU policy from Brussels, and by Russia. This issue also occurs in many other systems, such as with cormorants and fisheries in Denmark (Nielsen et al. 2001) and in southern ocean fisheries (Constable et al. 2000; Constable & Nichol 2002) - although in these examples it can sometimes be the prey that operates at larger scales than the predator.

The answer is that there is never a simple question of deciding if management should be local or central. Control of any resource should therefore never be totally delegated. All management must be multi-scalar. The real question is how much power for the management of each ecosystem component should be placed at each level. In other words we have to view management as having a hierarchy of scales with some decision making power lying at each level (Giampietro 1994; Midmore & Whittaker 2000; Crow 2002; Vogt et al. 2002; Wiens et al. 2002; Degnbol et al. 2003). Different degrees of power over various species or resources will lie at different levels depending on their ecological, social and economic characteristics. This hierarchy must be nested, in that the upper levels place limiting frames on the next layer, and so on. The higher levels should simply define overall goals, and general limitations, with each successive layer adopting more specific and more locally adapted rules and policies. As long as the more local levels operate within the framework of the upper level's overall goals the system should function.

Three main problems arise to prevent the implementation of this system. Firstly, deciding on the overall goals may be controversial, especially for conflict species, or species whose conservation will incur significant opportunity costs for the local level. In these cases the local level and central levels will often argue for significantly different overall goals (Callicott 2002). There is no magic solution to solving these conflicts. Present day thinking calls for a focus on the process behind decision making, as much as on the decisions that are made (Nie 2002; Peterson et al. 2004). In effect this question strikes at the heart of how we perceive democracy (Arblaster 2003), and how it balances the protection of minority rights with the majority getting their will. As pointed out by Peterson et al. (2004) there is a fundamental paradox in democracy between freedom granted through individual liberty, and equity granted through popular sovereignty. Just as every country has its own version of democracy (Arblaster 2003), it is likely that each situation will require locally adapted decision making processes (Borrini-Feyerabend et al. 2004). The key words here are ensuring bottom-up representation and participation.

A second problem lies with coordinating this nested hierarchy of scales (Perry & Ommer 2003). Within an ecosystem context, with a wide range of species and processes interacting at a range of scales it is inevitable that a wide range of institutions existing at many different administrative levels will need to coordinate their activities. Agencies and government institutions at different levels are notorious for being territorial over the powers that rest at their level. When recognizing how diverse and complex both the ecological and the social / cultural aspects of ecosystems are, it is highly unlikely that many local level administrations will have the human capacity to deal with these new demands. Duplicating capacity is also unlikely to lead to effective use of limited funds. Degnbol et al. (2003) argue for the creation of a new form of professional whose job is to cross scales, and to ensure that information flows in both directions. It is also important that some form of upward and downward accountability exists. Designing and initiating a functional hierarchical system is likely to need inspired leadership from the upper level (Callicott 2002).

Finally, there is the issue of developing mechanisms to address the fact that the costs and benefits often occur at different scales. In the language of economists there is a need to internalize the external costs - in other words there is a need for economic incentives that reduce the differences between the value (or cost) of biological diversity to the private individual and its value (or cost) to society as a whole (Folke et al. 1996). As costs and benefits often lie at different spatial scales, there is a need to establish a revenue flow from the global to the local (Adams unpublished) which far exceeds current rates from activities such as ecotourism.

These last three points underline the need to involve even more disciplines into the fields of natural resource management and biodiversity conservation. There is a clear need for political scientists to add their skills to the existing teams of ecologists and social scientists working on the issue, despite the challenges that this integration represents (Gibson et al. 2000, Vogt et al. 2002). We also need to transfer experience between different ecosystems and contexts - especially between aquatic, coastal and terrestrial realms. Experience with integrated coast zone management seems to be particularly far advanced.

As both ecosystems and societies change it is inevitable that we will need to constantly adapt the management of ecosystems. This implies that there is no single correct way of doing things that once identified will endure for long. Rather we must accept change, and focus on designing decision making processes that can constantly respond to changing circumstances. These are difficult tasks, however, the issues are fundamental. They deal with how as a society we make decisions, how our democracies function, and how we take care of the local in an increasingly globalised world.

A final twist about the Norwegian context

The tensions between local and central are likely to increase further within the Norwegian context as there are several trends that are on a collision course with each other. On one hand, the conservation agenda is putting even more focus on halting the loss of biodiversity by 2010. On the other hand the last decade has seen a growing trend towards the commercialization of nature as land-owners are forced to look for novel ways to earn money from natural resource management. In the present Norwegian context "local" is increasingly becoming "private". These two trends are likely to lead to far greater conflicts in the future as increased resource use will often be incompatible with the desire for biodiversity conservation. Likewise, the social / political trend towards greater individualism is likely to clash with the increased focus on large scale global ecological processes such as climate change. Because the desire for biodiversity conservation and the economic need to exploit resources operate on different scales (regional, national and global vs. community and individual) the conflict between local and global management scales is likely to increase in the future in Norway.

9 Final words

If we are to satisfy diverse societal goals within the limited space that our planet provides there is going to be a need to cast aside dogma, naïvity and ideology on all sides of the debate and develop a set of pragmatic compromises that tackle difficult issues, make difficult decisions and come up with long term strategies. The paradox is that local participation and influence can probably be best achieved if clear frames are set by central authorities (Peterson et al. 2002). Fortunately, the principle of subsidarity (Jordan & Jeppesen 2000) that is at the heart of European management philosophy, and much of the present day democratic tradition, is able to include this apparent paradox. The move towards a "Europe of the Regions" (Gough 2004) offers the potential to see how a central authority (in this case trans-national) can provide frames for more local (regional or sub-national) levels to find locally adapted solutions. The challenge here however, is in maintaining the lines of communication and dialogue open from the local to the trans-national, such that the local level can influence the frames that are placed from above. Although Norway is not currently a part of the EU, it is clear that it is affected indirectly by ideas, and directly by policy and trade agreements that originate from the rest of Europe.

What about the Malawi principles with which we opened this discussion? We have documented that a number of potential contradictions exist between the individual principles. We hope that we have outlined some potential solutions to these problems. However, the core of the solution lies with viewing the principles in their entirety, rather than individually (Prins 1999). In this context there is some way to go before we see their successful implementation on a large scale, although some small scale successes are encouraging (Smith & Maltby 2003). Given the enormous effort that has gone into getting the Ecosystem Approach onto the international agenda it seems logical to proceed by developing and adapting its structure, rather than beginning from scratch.

Finally, while human management and political structures can be debated and adjusted we must not loose sight of one clear fact - and that is the ecological processes that we seek to exploit, manage, or conserve operate on scales that are not subject to human influence. This may often cause inconvenience when biological scales do not fit the political scales that we have constructed - however, in these cases it is us humans who must adjust. When managing natural resources and conserving biodiversity we must accept that nature sets the ultimate limits.

10 The future

Four main areas of research and development remain for the future:

From the point of view of ecology there is a need to develop far better spatial models of how management units with different sizes and distributions perform with natural resources and ecosystems that have different spatial characteristics. Finding ways to formally utilize existing or readily obtainable movement data (be it on home-range, migration or dispersal) should be a priority.

For the political and social sciences there is a need to develop better models for multi-scale public participation in decision making. Existing models need to be critically evaluated from the points of view of several disciplines to identify the successes, failures, and trade-offs. The various co-management systems appear to offer the most successful formats that have been tried so far.

Political scientists and social-economists need to develop equitable, effective, and publicly acceptable models for redistributing costs and benefits. The systems of conservation easements or direct payments appear to offer some of the best starting points.

Greater public involvement requires an informed public. There is a clear need to communicate many of these complex issues to the public. Most importantly is the need to communicate that there are many trade-offs associated with natural resource management systems. In effect we must view them as a range of "set menus" rather than a buffet, or "a la carte" menus. Although still in its infancy, some of the scenario methods may be highly suitable for these exercises.

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