Ottar Michelsen

BIODIVERSITY INDICATORS AND ENVIRONMENTAL PERFORMANCE EVALUATIONS

Outline of a methodology

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Outline of a methodology

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2003

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Preface

This report was originally prepared as a part of the compulsory organised training programme in my doctoral studies in environmental management. The report was the outcome of a subject called 'Biodiversity indicators and environmental performance evaluations' which was organised as an individually adapted list of set literature (DI-LSF01).

After the evaluation and due to fruitful discussions with after the presentation of the final report, the report has revised and to some degree extended.

The intention has been to go through a set of methods for biodiversity impact assessment and both evaluate to what degree these methods are built on valid ecological assumptions and also if they can be used to include biodiversity considerations in environmental performance evaluations. An outline for a new approach to include biodiversity considerations is presented.

Chapter 1 is a presentation of the problem that is addressed. Chapter 2 is focusing on loss of biodiversity and why this is a problem. Here it is also argued that biodiversity should be the main focus when addressing land use changes in environmental performance evaluations. In chapter 3 different measures of biodiversity is presented, and chapter 4 give a brief introduction to the link between present forestry methods and loss of biodiversity. In chapter 5 different existing methodologies to include biodiversity considerations are presented and a new approach is outlined.

I want to thank Professor Håkan Hytteborn, Department of Biology at Norwegian University of Science and Technology, and Professor Annik Magerholm Fet, Department of Industrial Economy and Technology Management at Norwegian University of Science and Technology, who were responsible for the subject.

I also want to thank Dr. Bjørn Åge Tømmerås at Norwegian Institute of Nature Research and Hans Blom at Skogforsk/Norwegian Institute of Forest Research for comments on the preliminary report and suggestions for improvements.

Trondheim, 7th October, 2003

Ottar Michelsen

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1 Introduction

There is a growing focus on documentation of environmental performance and environmental impact from products, both for internal use for producers to be able to identify areas for improvement, and for external use through environmental product declaration of different kinds. The development of Life Cycle Assessment (LCA) methods has been important for this exploration and documentation.

LCA is a tool for analysing the environmental burden of products at all stages in their life cycle – from the extraction of resources, through the product of materials, product parts and the product itself, and the use of the product to the management after it is discarded (Guinée 2002). LCA has however so far mainly been used to study consumption of raw materials and energy, emission of pollutants and generation of waste, but there is a growing concern that also land use and biodiversity aspects should be included. However, there is still a trend that performed LCA studies omit land use totally (Weidema and Lindeijer 2001 and i.e. Broers 2002) and when land use is included, it is not linked to biodiversity aspects (i.e. Schuurmans et al. 2002).

In Europe the natural vegetation in most regions is forest. In south the natural vegetation is Mediterranean evergreen broad-leaved forest, in Central and Western Europe deciduous forest is the dominating natural vegetation, and in Fennoscandia there is boreal coniferous forest.

Even though the forested areas in Europe have increased during the last decade with more than 9 millions hectares (UNEP 2002), most of the natural forest vegetation is transferred to agricultural and urban areas, and most of what is left is strongly influenced by forestry. The remaining areas are strongly influenced through fragmentation and other human introduced disturbances such as hunting of large animals and introduction of domestic herbivores, and the forests as we know them today is hence partly a result of human activities (Angelstam 1998, Larsson 2001). For example, it is estimated that only 0,2 % of the Central European deciduous forests remain in a relatively natural state (Hannah et al. 1995).

The situation in other areas is even more alarming. World wide 93,4 millions hectares forest disappeared during the last decade, and in Africa, where the situation is worst, about 7,5 % of the existing forest in 1990 was gone ten years later (UNEP 2002). In addition, these numbers do not include the areas of natural forest vegetation that are transferred into plantations. This global situation should also be taken into consideration when dealing with the situation in Europe since timber and timber products are sold world wide. Tropical timber is for instance used for furniture and building materials (doors, flooring etc.) in Europe as a replacement for locally grown timber.

Changes in land use have led to dramatic losses of biodiversity (i.e. Pimm et al. 1995, Chapin et al. 1998) and there is thus a need for considering the consequences for biodiversity in all kinds of decision making. Biodiversity is now included as an own aspect in the Sustainability Reporting Guidelines developed by Global Reporting Initiative (GRI 2002).

‘Biodiversity loss’ is not an appropriate impact category in LCA analysis since there are a lot of reasons why biodiversity is lost, i.e. pollutants, climate change and changes in land use and different impact categories in LCA should be as exclusive as possible (Udo de Haes et al. 1999). Thus the focus will be on biodiversity consequences of land use since a range of other impacts on biodiversity, such as emissions of pollutants, are already included in LCA.

There is however no single ‘authorised method’ for assessing the impacts of land use in terms of loss of biodiversity (Guinée 2002). There is thus an obvious need to develop methods to include these aspects in environmental documentation and there are done several attempts (see Lindeijer 2000 for a review). These methods are however still immature, and with only one known exception unable to diverge between different forestry regimes. Most LCA studies that include forest activities mainly focus on the energy consumption and related emissions (Athanassiadis 2000, Harjanne 2001, Korhonen et al. 2001).

There has however been an increased focus on sustainable forestry and different forest certification systems are developed (such as FSC and PEFC). The disadvantage is that these do not provide data on forestry that easily can be used in LCAs of products where wood is a raw material. In addition these certification systems only give a threshold value and not information about the absolute environmental pressure that LCA studies intend to provide. The certification systems do also vary between different counties with more or less the same forest types and some of the systems are in addition highly criticised from environmental organisations (i.e. Liimatainen and Harkki 2001).

The main focus will be on boreal forest. This is chosen since boreal forest undoubtedly is an important biome since it covers large areas and a considerable quantity of threatened species live in boreal forest. In Norway almost half of the species in the Norwegian Red List are forest species (Direktoratet for naturforvaltning 1999b). In addition forest provides raw material for a number of important product chains (in Norway timber industry is present in 313 of 434 municipalities (Miljøverndepartementet 2001)), the forest industry is in a process focusing on environmental certification and the need to document sustainability in the sector is stressed by the authorities (Landbruksdepartementet 1998). The European Union has also signed all the international and European initiatives on sustainable forest management (Glück 2000) so there is no doubt that documentation of environmental impact from forestry is a topic that will be focused in the years to come.

The intention here is to give an outline of a methodology to include land use impact on biodiversity in forestry in LCA. The intention is to develop a method that gives different score according to forestry practice. Here focus is on boreal forest, but the method should with modifications be useful also for other forest types and for non-forest natural ecosystems.

2 Loss of biodiversity

In the Convention on Biological Diversity it is stated that '*Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*' (UNEP 1992).

The definition includes diversity not only between species, but also within species and between ecosystems. Never the less, loss of biodiversity is most commonly recognised as loss of species. Extinction of species is a phenomenon that probably is as old as life it self and at the moment the earth is in the sixth major known extinction event (Chapin et al. 1998). The present situation differs however in two major points. First, while earlier extinction events probably were a result of changed physical environment, the current event is biotically driven (Chapin et al. 1998). The changes are due to human impact on land use and species invasion in addition to climatic changes. Second, the present extinction rates are 100 to 1000 times their pre-human levels and already as much as 20 percent of the species in some groups of organisms are extinct (Pimm et al. 1995). In addition, Pimm et al. (1995) claim that if all species that currently are regarded as threatened become extinct, the future extinction rates will be even 10 times faster than recent rates. As a consequence as much as 50 percent of the species in some regions might vanish during this century (Soulé 1991). Also within those species that do not get extinct, specific characters will be lost due to reduced genetic diversity.

The situation is more or less the same for several taxonomically diverse groups from widely different environments (Pimm et al. 1995). For instance are 24 per cent of all mammals and 12 per cent of bird species regarded as globally threatened (UNEP 2002). In Norway the national red list includes more than 20 per cent of the species in those taxonomically groups that are included (Direktoratet for naturforvaltning 1999b). The number of species classified as endangered is 292 out of 14.637 evaluated species. In addition 103 species are known as extinct in Norway.

Diaz and Cabido (2001) state that conservation of species richness deserves the highest priority in ecological agendas and OECD states that as much biodiversity as possible should be safeguarded (OECD 2002). One of the targets in the 6th environmental programme in EU is to stop biodiversity loss by 2010[1].

The single most important factor for loss of terrestrial biodiversity on a global scale is changes in land use (Chapin et al. 2000, Sala et al. 2000, Schenck 2001). In some biomes, other impacts are more important, as in boreal forests where climate change is believed to have a greater impact on loss of biodiversity in the present century (Sala et al. 2000). However, this is mainly a consequence of huge changes in land use have already occurred and the following impact on biodiversity is already an old phenomenon in these biomes.

[1] http://europa.eu.int/comm/environment/newprg/

2.1 Why maintain biodiversity?

> How different would the world be without the fly to help in the decomposition of wastes and carcasses? Without the fly as an experimental animal, how much less would we know about population cycles, about nervous functions, about heredity? What is a fly worth, an oak tree, a crow, a wisp of thistledown? By how much would life be diminished if Shelly had not written his *Ode to a Skylark*, if Emerson had not penned *The Rhodora* or Lanier *The Marshes of Glynn*? How many persons would not be alive today if we had not discovered penicillin, the improbable product of a lowly green mold? If it is true that half our new drugs are being produced from botanical sources, how can we afford to neglect or destroy any portion of the earth's green mantle? Who can say what obscure plant or animal may someday be precious to us? Are not all precious, since in fact we understand so little about the interdependence of living things, since life itself is the most precious of all? The earth has spawned such a diversity of remarkable creatures that I sometimes wonder why we do not all live in a state of perpetual awe and astonishment.
>
> H. E. Evans. 1966. *Life on a Little-known Planet*

Biodiversity defies easy description and quantification and hence conservation of biodiversity have often low priority (OECD 2002). However, there are several reasons to maintain biodiversity, and Kunin and Lawton (1996) list up five main arguments:
1. Humankind has a moral and ethical responsibility to care for life on earth.
2. Many organisms bring pleasure to countless people and enrich our lives.
3. Species can be useful as drugs, food, etc.
4. Organisms provide essential 'ecosystem services' and maintain life support systems of the planet.
5. Species are the touchstones of whether we are using the planet sustainable.

The problem is that some of these arguments hardly can be investigated scientifically and most attention here will be drawn to the maintenance of ecosystems. It is however worthwhile to have the other arguments in mind. Some authors claim for instance that it is not possible to achieve sustainability without recognising the intrinsic value of nature (i.e. Høyer and Aall 1997).

There is also no doubt that species can be extremely useful and valuable for humankind. Bengtsson et al. (2000) claims that the main motivation for the Convention on Biological Diversity in 1992 actually was economic factors. Principe (1991) has calculated the value in terms of medicinal potential of each unexplored species of plant at approximately US$ 1.6 million, and even though this is probably an overestimate (Kunin and Lawton 1996), it gives a rough figure of the economic motivation.

There is also beyond any doubt that a lot of the species are useful to humans. Kunin and Lawton (1996) pinpoint for instance that almost 10% of the 300.000 existing flowering plants are edible, but most of the food comes from only two dozens widely grown species. The potential that may exist among other plants are demonstrated by plants as soybean and oil palm that during the last century grow from minor regional specialities to major global crop species. In addition there is a genetic source for disease resistance of currently domesticated species, species for biological pest control and medicines. The problem is that it is in most cases impossible to know which species that might be useful without extensive testing, and every species is hence a potential 'treasure'.

Some authors claims that in theory all values can be transferred into monetary terms, and using monetary terms for biodiversity would ease valuation (i.e. Groot et al. 2002). Institutions as OECD (2002) also focus on such values to ease decisionmaking and hence perform 'cost effective' conservation of biodiversity. There is however no straight forward method of doing this today and this will not be discussed any further here.

2.1.1 Biodiversity and ecosystem processes

There is an ongoing discussion if it is the species diversity as such or rather the functional diversity that really matters (i.e. Diaz and Cabido 2001). According to Diaz and Cabido (2001) there is a growing consensus that functional diversity rather than species number per se, determines ecosystem functioning. Some authors use plant species richness as a surrogate for functional richness (i.e. Lawton et al. 1998a, Tilman 1999), but this is again criticised by authors that point out that there is no universal connection between species richness and functional diversity (Diaz and Cabido 2001).

There is however by no means straight forward to use functional diversity. Martinez (1996) points out that there is no commonly accepted definition on 'function', but there is acceptance that function has something to do with ecological processes. The question is hence to maintain the ecological processes. This does however not solve any problem, since then the question is what process should be focused. Martinez (1996) claims that functional diversity can be defined as the variety of interactions with ecological processes, and Gitay and Noble (1997) argue that there are no universal functional groups of species. They use the term for a '*group of organisms that respond in a similar way to a syndrome of environmental factors*', which means that different species constitute different functional groups dependent on what function is focused. Species can even be in different functional groups at different life stages. Hence what functional group that is present is a result of what functions that is focused and according to Martinez (1996), this leads to the 'uniqueness hypothesis' that assert that whenever a species is lost, a particular function of an ecological system is largely eliminated.

One of the roughest divisions of species possible is to divide between different trophic levels, and Lawton (1994) point out that if we remove all the species in an entire trophic level, this would obviously reduce the ecosystem processes dramatically. The example is rather hypothetical, but it is however interesting to ask the question how many species are actually needed to perform the ecosystem functions. In other words – how many species can be removed or be extinct without altering the level of ecosystem functions.

Lawton (1994) set up four different hypotheses for the relationship between species richness in an ecosystem and the rate of an ecosystem process (i.e. primary production):

1. redundant species hypothesis
2. rivet hypothesis
3. idiosyncratic response hypothesis
4. null hypothesis

In addition, Johnson et al. (1996) enlarges the list with a hypothesis that actually is a variant of the rivet hypothesis:

5. diversity-stability hypothesis

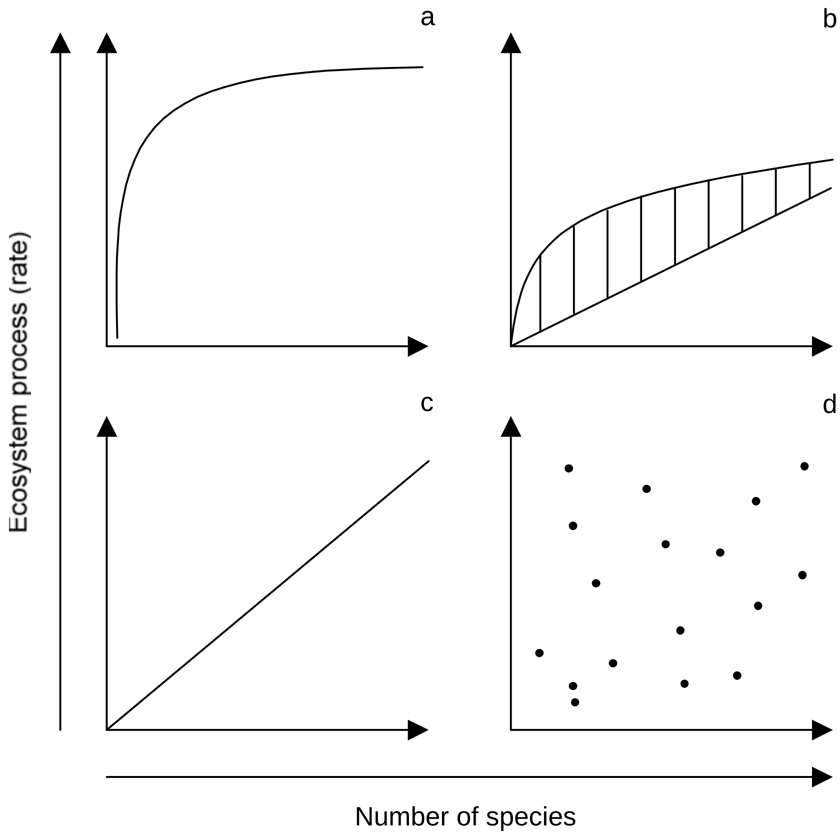

FIGURE 1 – HYPOTHETICAL RELATIONSHIP BETWEEN THE RATE OF AN ECOSYSTEM PROCESS AND SPECIES RICHNESS (a REDRAWN FROM JOHNSON ET AL. (1996), b-d REDRAWN FROM LAWTON (1994)).

<u>The redundant species hypothesis</u> (Figure 1a) suggests that there is a minimal diversity necessary for proper ecosystem functioning. Hence most species are redundant in their roles (Walker 1992).

<u>The rivet hypothesis</u> (Figure 1b) gives a contrasting view and suggest that all species make a contribution to the performance of the ecosystem and any removal of a species from the ecosystem will affect the ecosystem processes in a negative way (Ehrlich and Ehrlich 1981). Here species are compared to rivets holding together a complex machine and suggests that the functioning will be impaired as its rivets (here: species) fall out. In Figure 1b a range of different responses are indicated. Here some species are considered as more important than others, but which species that are most important is a result of which other species that is present.

<u>The diversity-stability hypothesis</u> (Figure 1c) is a variant of the rivet hypothesis and predicts a linear relationship between species richness and ecosystem processes (Johnson et al. 1996). Here all species are more or less equal.

The idiosyncratic response hypothesis (Figure 1d) suggests that ecosystem function changes when diversity changes, but the magnitude and direction of change is unpredictable since the roles of individual species are complex and varied (Lawton 1994). The last hypothesis, the null hypothesis, simply suggests that ecosystem functions are insensitive to changes in species composition.

One advantage with these rather schematic hypothesises, is that they are testable. Lawton (1994) refers to experiments carried out under controlled conditions. The results points in different directions according to what process that is studied and do not give any clear answer. Uptake of CO_2 and plant productivity did however follow the predictions from the rivet hypothesis.

These experiments were carried out under rather artificial conditions and with species richness far below what exists in natural ecosystems, but also other studies have shown a connection between plant productivity and biodiversity (Naeem et al. 1996, Hooper and Vitusek 1997, Tilman et al. 1997, Chapin et al. 1998, Symstad 1998, Yachi and Loreau 1999, Chapin et al. 2000, Loreau et al. 2001). A study performed by Wardle et al. (1997) did however show a decrease in productivity with increased species richness, and a similar phenomenon is found in alpine regions several places in the world, i.e. New Zealand and Peru (Stephan Halloy[2], pers. comm.). To make the picture even more fuzzy, Funaki and Morin (2003) have shown that also the community assembly is influencing the relation between biodiversity and productivity.

The results from these studies do not give any clear answer to how species diversity affects ecosystem functioning in general. One question is if the presence of a specific function group is more important than species diversity per se. Hooper and Vitousek (1997) show in their study that number of species present is more important for net primary productivity that number of functional groups present, but Diaz and Cabido (2001) concludes that most studies indicates that the most important factor is what functional groups are present. Also Tilman et al. (1997) have shown that not all species are equal in their study.

Loreau et al. (2001) pinpoint that even if studies indicate a connection between diversity and ecosystem functions, the mechanisms will still not be clear. One possibility is of course that processes such as niche differentiation and facilitation result in higher rate of ecosystem processes as number of species increases. However, it is also possible that a high number of species present, simply increases the chances that species with high influence on ecosystem processes are present. In at least some studies the results show complementary between species, but other hypothesis can from their point of view still not be rejected (Loreau et al. 2001).

The problem is however still that universal functional groups don't exist (Gitay and Noble 1997) and the connection between diversity and ecosystem process is dependent on what process that is focused. Martinez (1996) shows that the correlations between

[2] Researcher at Invermay Agricultural Centre, New Zealand

diversity and ecosystem process can be both positive and negative, dependent on what process that is studied.

One last important question is what function the species actually have in the ecosystem and if loss of particular species seriously will affect ecosystem functions or if their functions easily either might be carried out of other species or the function loss is actually negligible for the ecosystem functioning. The term ‘keystone species’ is here used.

Lawton (1994) also uses the term ‘engineering species’. Autogenic engineers change the environment via their own physical structures, while allogenic engineers change the environment by transforming living or non-living materials from one physical state to another. Trees are good examples of autogenic engineering species, while beavers can serve as an example of an allogenic engineering species that transform living trees to dead trees in a beaver dam and thus greatly influence the ecological conditions around the dam. The consequences of loss of such species is more easy to predict, but at the same time it is not possible to foresee what species actually have key roles in ecosystems (Cushman et al. 1995, Kunin and Lawton 1996, Aarts and Nienhuis 1999) – this will often not be noticed before the species is extinct.

Bengtsson et al. (2000) state that present ecological theory simply is not ready to address the effects of changes in species diversity and ecosystem functions. The only possible conclusion is hence that it is not possible to predict what will happen with the ecosystem processes when a species get extinct in the ecosystem. Martinez (1996) also argues against the use of the term redundant and claims that this is a value laden word. He claims that experiments that conclude that something is redundant more or less are a result of the experiment conditions. If for instance grasshoppers are removed in a pollination experiment, the result will indicate that the grasshoppers are redundant since they are not pollinators. This does however not mean that the grasshoppers are redundant since they might be important in other ecosystem processes.

2.1.2 Biodiversity and ecosystem stability

It is not only the level of ecosystem processes that is important, but also the stability of the processes. MacArthur (1955) asserted that populations of species would be more stable in communities with more species because these communities provide a greater variety of trophic resources. Even though this is challenged as a general rule, it is according to Lawton (1994) becoming increasingly obvious that loss of plant species may reduce the ability of ecosystems to withstand, and recover from, extreme events.

Even if universal functional groups don’t exist, it is possible to divide species into different functional groups for specific processes (Gitay and Noble 1997). Examples might be the ability for nitrogen fixation and provide food for different species. When focusing on one such specific function, experiments have shown negligible changes in ecosystem functioning when a species is removed as long as there are other species present within the same functional group (Chapin et al. 1998). Hence it is argued that what matters for ecosystem stability is the presence of specific functional traits (Diaz and Cabido 2001).

There is however a growing concern that such examples of redundancy don't show the long-term effects of reduced species diversity. A number of authors emphasise that species that seems redundant actually is an insurance against sudden changes in the rate of ecosystem processes in the event that species are lost (Cushman et al. 1995, Naeem and Li 1997, Chapin et al. 1998, Aarts and Nienhuis 1999, Yachi and Loreau 1999, Bengtsson et al. 2000, Diaz and Cabido 2001), either this is caused by stochastic processes, habitat destruction or any other reason.

Chapin et al. (1998) line up several hypothesises based on research results that follow this argument. They postulate that high species diversity reduces the risk of large changes in ecosystem processes in response to directional or stochastic variation in the environment or in response to invasion of pathogens and other species. This is supported by others, and Bengtsson et al. (2000) claim that this 'insurance hypothesis' is one of the best arguments for maintaining biodiversity.

2.2 Biodiversity and impact from changes in land use

There are as already mentioned a lot of causes to the present threat to biodiversity, i.e. global warming, nutrification, acidification and release of toxic substances, and as pointed out by Ude de Haes et al. (1999), these should be treated independently in LCA. There is however reasons to argue that the greatest threat is caused by changes in land use (i.e. Chapin et al. 2000, Sala et al. 2000).

There is also a well developed theoretical framework and empirical evidence for the relationship between area and biodiversity that is useful also when considering changes in land use.

One important relationship is the relationship between number of species (S) and the size of an area (A) that follows the formula

$$S = cA^z$$

where c depends on the groups of species chosen and z=¼ (MacArthur and Wilson 1967). The value of z is however not fixed and varies between 0,15 and 0,8, but it gives an rough estimate and indicate that an area 16 times as large as another one should hold twice as many species (Hengeveld et al. 1995). This alone indicates that any decrease in the size of a habitat, will result in reduced species diversity.

One important theory is MacArthur and Wilson's (1967) 'equilibrium theory of island biogeography' that predicts number of species on an island as a balance between immigration and extinction rates. It is based on very simple predictions. First the extinction rate will be higher on a small island than on a bigger island since the population sizes are smaller and hence more vulnerable to changes in environmental conditions and natural fluctuations. Similar, the immigration rate will be higher if the neighbouring island is closer, simply since it by chance is easier to migrate between close islands than between remote islands. The last assumption is that the emigration rate from a big island is higher than from a small since there are more species and the populations are bigger. These assumptions are shown graphically in figure 2 and give the number of species on an island as a function of island size, distance to other islands and size of these islands. It is important to have in mind that this is not only valid for 'real' islands in an ocean, but for all kinds of islands, i.e. forest patches in grassland.

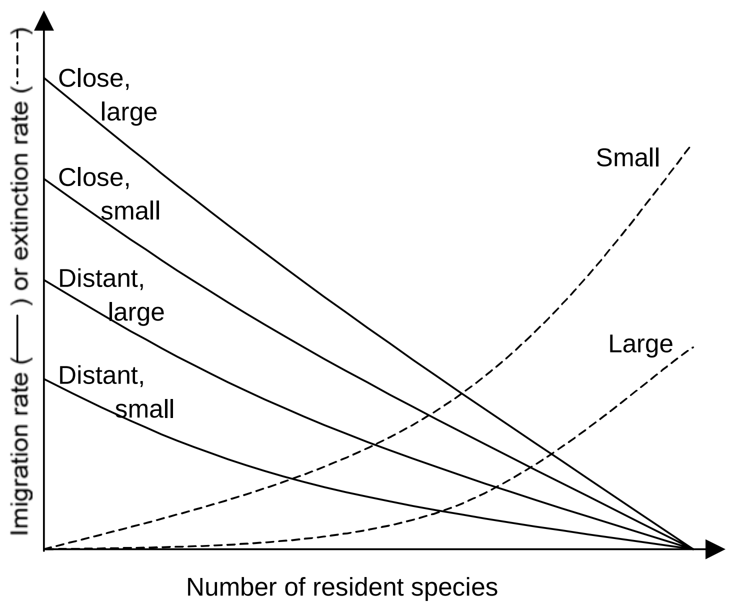

FIGURE 2 – THE NUMBER OF SPECIES PRESENT AS A RESULT OF ISLAND SIZE AND DISTANCE TO OTHER ISLANDS (AFTER MACARTHUR AND WILSON 1967).

This has strong implications for land use and species richness. Size of natural habitats must be taken into consideration together with the distance between related habitats if species richness is to be maintained.

Different studies give support to this theory. For instance Andrén (1994) has shown greater species loss of birds and mammals than can be explained by loss of habitat alone. He concludes that pure habitat loss, patch size and isolation of patches all have an effect on population sizes and extinction rates. He also recognises that there normally is linearity between the size of a population and the habitat, but at a given threshold, the populations start to decline faster than the loss of habitat. Thus at a given point radical and unpredicted changes might occur. Other studies have shown similar results, such as Didham et al. (1996) who showed a decrease of diversity of insects due to fragmentation, and Laurance et al. (1997) who showed a decrease in biomass in a rainforest as a result of habitat fragmentation.

Other aspects that are important are relations between biodiversity and productivity, biodiversity and successional stage, biodiversity and evolutionary time and biodiversity and spatial heterogeneity. There is evidence that all these aspects play a role in determining the biodiversity in an area, but they will not be discussed here. Here we do with stating there is a strong evidence that spatial heterogeneity is important in its own right (see i.e. Begon et al. 1990 p. 826) which is important for forestry planning and indicates a loss of biodiversity in more homogenous managed forests. This is also known as the ‘mosaic concept’.

3 Measures of biodiversity

3.1 Number of species and indexes of biodiversity

Biodiversity is a concept with a wide content, but still the most frequently used measure of biodiversity is number of species (Gaston 1996). Gaston (1996) claims there are four obvious reasons. First species richness is thought by many to capture much of the essence of biodiversity, and many authors use the two terms more or less as synonymous. Second, species richness as term is widely understood. Third, species richness is considered in practice to be a measurable parameter, and at last, much data on species richness already do exist.

A measure of number of species within an area (of a given size) is known as alpha (α) diversity (Hengeveld et al. 1995). Another measure of within-area diversity is gamma (γ) diversity. Gamma diversity usually refers to diversity within a large region (Hengeveld et al. 1995) and its comprehension has direct connotation with dealing with biodiversity at the landscape level. It has hence no upper limit, but most often refers to a whole region or a country (Hengeveld et al. 1995). Regional species list can thus be regarded as a lower bound on gamma diversity for the area.

Beta (β) diversity was introduced by Whittaker (1960) to designate the degree of species change along a given habitat or physiographic gradient. Hence it is a measure of between-area diversity and cannot be expressed in numbers of species since it is a rate. Normally it is represented in terms of the similarity index or of a species turnover rate. The simplest definition of beta diversity is the ratio of the gamma diversity of a region to the average of alpha diversity of local areas within the region (Hengeveld et al. 1995).

Since diversity is a lot more that number of species, a number of diversity indexes that not only measure the absolute number of species but also take account of their relative abundance are developed. One of the simplest is Simpson's index (Simpson 1949) that is given by the formula

$$D = \frac{1}{\sum_{i=1}^{S} P_i^2}$$

where S is the number of species in a community and P_i is the relative proportion for the ith species (i.e. in biomass or number).

Another frequently used index is Shannon diversity index (Shannon and Weaver 1949), given by the formula

$$H = -\sum_{i=1}^{S} P_i \ln P_i$$

A major problem with these indexes is that assumptions made about sampling may be difficult to meet (Hengeveld et al. 1995). In addition this type of combination indexes can be difficult to interpret.

Neither these indexes take all aspects of biodiversity into account. Hence there is a need to find other measures of biodiversity. Some have tried to focus on functional diversity rather than species diversity (see i.e. Diaz and Cabido 2001), but this raises several new questions since there as already mentioned not even exist an accepted definition on ‘function’ (Martinez 1996).

3.2 Indirect measures of biodiversity

Biodiversity as defined by UNEP (1992) cannot be measured directly. It is especially hard to see that genetic diversity for a community can be measured in near future, and there are done a number of attempts to find indirect measures of biodiversity with use of indicators that are possible to measure.

In lack of better measures, some authors have suggested to use diversity of different species groups as measures of overall biodiversity. An example is Duelli and Obrist (1998) that have studied correlations between different species groups and overall species diversity and concluded that use of *Coleoptera* gives the best result. They have also taken time effort into consideration, and then they conclude that *Heteroptera* and flowering plants are the best species groups.

This result is used as an assumption for the ‘Species-pool effect potentials method’ (SPEP) that is a method for evaluating land-use impact on biodiversity (Köllner 2000) and included in the life cycle impact assessment method “Eco-indicator 99” (Goedkoop and Spriensma 2000). This method will be described in section 5.1.

Also other taxonomic groups are used, i.e. birds (Järvinen and Väisänen 1979) and tiger beetles (Pearson and Cassola 1992), but Duelli and Obrist (1998) claim that the choice of taxonomic group usually is influenced by the expertise of the people involved and not correlation to overall biodiversity.

It is not possible to draw general conclusions from these studies. For instance Duelli and Obrist (1998) made their investigation in cultivated areas, and there is no indication that the results can be used elsewhere. In addition there is an overwhelming number of studies that show no correlation between species richness in one taxonomic group and species richness in other groups (i.e. Prendergast et al. 1993, Hengeveld et al. 1995, Gaston 1996, Dobson et al. 1997, Lawton et al. 1998b, Moalu and Alatalo 1998, Chapin et al. 2000, Larsson 2001). For instance Lawton et al. (1998b) conclude that on average only 10-11 percent of the variation in species richness of one group could be predicted by the change in richness of another group.

There is hence no wonder that several authors have been focusing on the possibility to measure biodiversity without measuring either species or functional groups, but other aspects that are related to biodiversity. Hansson (2000) states that a biodiversity indicator might as well be a structural component, a process or some other feature of the biological system that insures maintenance or restoration of the most important aspects of biodiversity when present. From this point of view Larsson (2001) separates between two ways of choosing biodiversity indicators. The first way is to find parameters of a particular component of biodiversity. In practical use this is the same as some kind of measure of species present as described above.

The other way is based on an analysis of the key factors affecting biodiversity. Larsson (2001) states that in the forest context, this involves acknowledging that biodiversity is dependent on the structure of stands and landscapes, the species in them and the management and disturbance regimes they experience. These indicators can again be divided in three groups of indicators, and Noss (1990) shows it is possible to develop a hierarchy of indicators for all three groups from gene to landscape level.

Structural indicators are motivated by the assumption that more complex habitats will support greater variety of species. Larsson (2001) list several studies that support this assumption. Examples of structural indicators are canopy structure and openness and dead and dying wood.

The second group is compositional/species indicators. These are actually species measures, but instead of species diversity it is the present or absence of species that either are functionally important or in other way give functional information. This can be functional important species (i.e. keystone species or engineering species – see 2.1.1) and species that that are sensitive to and thus indicates disturbance regimes, isolation, crucial resources etc.

The last group is functional indicators. These are indicators of which abiotic and biotic disturbance factors and management regimes that are present, such as fire frequency, wind and snow, grazing impact etc.

Based on these assumptions Larsson (2001) lists 17 key factors for assessing biodiversity in European forests. These indicators will be further described in section 5.2.2.

4 Forestry and impact on biodiversity

Since the UN Conference on Human Environment in Stockholm in 1972, forest politics on all levels has been characterised with the dualism of both economic interest in timber production and ecological interest in maintaining environmental values such as biodiversity (Glück 2000). These two aspects are also obvious in governmental documents, such as the Norwegian white book on 'Value creation and environment – possibilities in the forestry sector' (Landbruksdepartementet 1998).

Focus in forestry systems have been timber production which in many cases have been at the expense of biodiversity (Johnson and Jonsson 1995), and there is no doubt that present forestry regimes do reduce biodiversity (Noble and Dirzo 1997). This is mainly due to loss or at least a decline of features that exists in natural and old forests such as dead wood, old trees, fire and natural grazed areas (Bengtsson et al. 2000). Siitonen (2001) estimates that the average amount of dead wood in Fennoscandian boreal forests at the landscape level has been reduced by 90-98 percent due to forest management. In addition ditching and introduction of new tree species have been common, and it is also a problem that some succession stages have become very rare, such as old growth stands of deciduous trees in areas dominated by coniferous forests (Baumann et al. 2001a). Duffy and Meier (1992) have found evidence that even 50-85 years after clear-cut logging in a temperate hardwood forest in United States, the herbaceous plant-species diversity of affected areas had yet to recover.

Bengtsson et al. (2000) claims it is naïve to argue that splitting forests into reserves and high-intensity forestry areas will preserve even a fraction of the biodiversity in European forests, and hence it is necessary to include biodiversity consideration also in forest that is harvested. Protection of species rich areas ('hot spots') is also shown to be insufficient to protect biodiversity (i.e. Prendergast et al. 1993, Dobson et al. 1997). It is not the amount of forest but the changes of composition and structure that have caused the decline in biodiversity in boreal forests (Angelstam 1998), and to be able to perform a sustainable forest management it is important to understand the natural forest dynamics and mimic these in forestry. According to Bengtsson et al. (2000) there is still a lack of knowledge, but this makes the need to implement the knowledge that do exist even higher.

The challenge is to both take precautions to maintain biodiversity and still perform an economically profitable forestry. Bengtsson et al. (2000) claims that many forest owners have had problems accepting that biodiversity should affect how they manage forests, but especially due to marked pressure this is now changing. One result is the range of different forest certification systems, such as FCS and PEFC.

Boreal forests are also relative species poor and might hence not have gained much attention in biodiversity conservation. However, Pastor et al. (1995) pinpoint that boreal forests have high functional diversity, but a small number of species within each functional group. Hence the maintenance of biodiversity is probably very important for proper ecosystem functioning. Cushman et al. (1995) also emphasise that relative species poor areas (such as boreal forests) are more vulnerable for alien species.

5 Biodiversity and environmental performance measures

5.1 Land use and biodiversity in present LCA techniques

There are performed a number of LCA studies where impact from forestry is included, but these mainly focus on energy consumption and to some extent mass balance, and not biodiversity aspects in forestry (Athanassiadis 2000, Harjanne 2001, Korhonen et al. 2001). The only known exception is a study performed as a joint project between Axel Springer Verlag, Stora and Canfor (1998), which will be described later. This is somewhat in contradiction to the fact that during the 1990s there was a growing concern that land use should be included in LCA analysis and a number of methodologies appeared (see Lindeijer 2000 for a review).

A fist very simple approach was to use area multiplied with time of land use as an indicator [$m^2 \times$year] (see i.e. Müller-Wenk 1998), but this put the value of all land equal which is obviously not correct. There was hence a need to develop a method to give value to the specific area. Lindeijer (2000) argues that changes of land use and occupation of land area should not be regarded as equal and uses two different formulas to express these impacts, namely:

land occupation impact = area A × time t × quality

land change impacts = area A × quality difference

The question is then how to quantify the quality and quality difference of land area and land use impacts.

Müller-Wenk (1998) goes through a number of impacts changes in land use might have, both for human health, ecosystem quality and resource availability, and concludes that with present methodology, the focus of land use should be on consequences for ecosystems. He thus constructs a valuation method for areas based on ecosystem quality that he defines as number of species present in the area and the percentage of these that are threatened. The species counts are based on vascular plants since studies of Duelli and Obrist (i.e. Duelli and Obrist 1998) indicate that vascular plant might serve as an indicator for overall species diversity. The land valuation is based on conditions in Switzerland, but he claims that it should be possible to expand this to at least other European countries.

In addition he separate between land use that cause transforming of the area from one type to another (i.e. transformation from cropland to urban areas) and use that maintain present state. This is based on a subdivision of land use in a time dimension. The first phase is the natural state. Then come the transformation phase and the use phase. When the area is abandoned there is a re-naturalisation phase before it ends in a re-naturalised state. During these transformations the quality of the ecosystem will change, i.e. as shown in figure 3. However the level of ecosystem quality will be a result of how this is defined and what processes that take place. The shape of the figure can have different forms depending on what processes that take place and how ecosystem quality is defined. There is also no necessity that the re-naturalised state will have lover quality than the natural state.

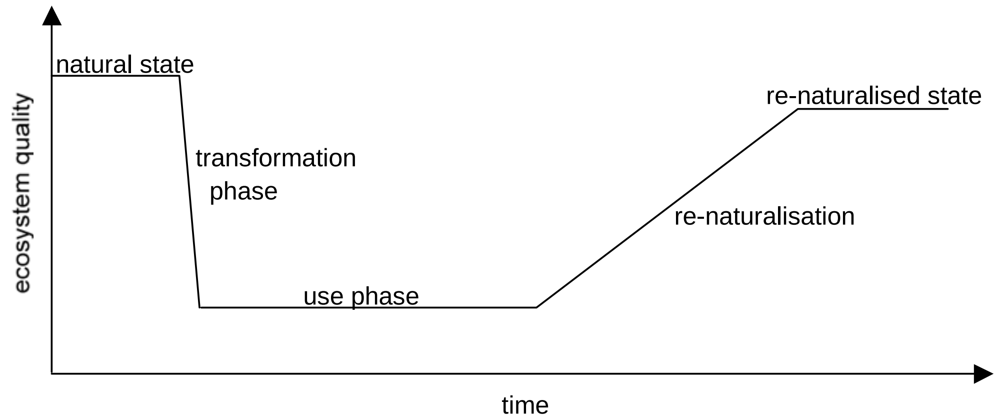

FIGURE 3 – DIFFERENT PHASES IN LAND USE AND LAND TRANSFORMATION AND THE FOLLOWING CHANGES IN ECOSYSTEM QUALITY.

Impact from land use with no transformation is hence given as a product of time and the quality of the area, based on the assumption that the use delays the re-naturalisation process with the time of the activity. When land transformation takes place, also the time of the re-naturalisation must be taken into account. This means that the time include both the time when the area is used and the predicted time for re-naturalisation (Müller-Wenk 1998).

Müller-Wenk (1998) defines quality according to the natural state. There is however no consensus about this (Guinée 2002), and Weidema and Lindeijer (2001) use the re-naturalised state as reference point and argue that this is a better choice independent of the difference in quality of the natural and the re-naturalised state.

Köllner (2000) develops this method further to what he calls the 'species-pool effect potentials method' (SPEP). This is also based on vascular plant diversity and potential loss of diversity in different areas. The method is based on the species-area relationship ($S = cA^z$, see section 2.2). A slightly adapted version of SPEP is also included in the LCA method Eco-indicator 99 (Goedkoop and Spriensma 2000). Also Lindeijer et al. (1998) have developed a method where degradation of ecosystem quality is measures as loss of biodiversity based on plant species measures.

A different approach is represented by the 'Biotopmethod' (Blümer and Kyläkorpi 2001) and by using the 'Hemeroby Concept' (Brentrup et al. 2002). These are in short based on giving different quality measures to different areas in such a way that it is possible to weight land use based on the quality of the area. Using the 'Hemeroby Concept' all areas are given a value based on 'naturalness degradation potentials' (NDP), and in the 'Biotopmethod' it is constructed a quality measure based on occurrence of red list species, key elements that is known to enhance biodiversity and rarity of the ecosystem. Even if none of these methods are capable of measuring different management regimes today, it might be possible to expand the 'Hemeroby Concept' to also include management practice when calculating the NDP values. This is however not further discussed here.

Lindeijer (2000) review different approaches for measuring impacts form land use in life cycle assessment. The SPEP method is among those that use key indicators that in most cases are vascular plant diversity, or in some cases (free) net primary production. As already pointed out (see chapter 2.1.1) there are however no clear links between neither vascular plant diversity and overall biodiversity or a correlation between NPP and biodiversity. In addition, it is clear that these methods will not enable to diverge between different forestry regimes.

Use of NPP is also problematic of other reasons. First, it is possible to have higher NPP in the use-phase than in the natural phase, i.e. in some agricultural areas (Weidema and Lindeijer 2001). This will indicate a positive effect on many transformations, which is not true, at least if the focus is on loss of biodiversity. In addition, it is possible that NPP is high although other life-support functions may be endangered in the periods between harvesting and re-establishment of a new vegetation cover due to changes in runoff, shelter, wind speed etc. (Weidema and Lindeijer 2001).

Use of NPP seems actually more as what Lindeijer (2000) characterise as a 'functional approach'. These methods try to characterise land use based on the level of (ecosystem) function maintenance such as erosion resistance and productivity. The third groups of methods that Lindeijer (2000) identifies divide land use into different land use classes where the 'Hemeroby Concept' belongs.

There are however steadily proposed new approaches, and Weidema and Lindeijer (2001) develop a method where they take species richness (SR) in an area, the scarcity of the actual ecosystem (ES) and the vulnerability (EV) of the ecosystem into account. The method will not be described in detail here, but for species richness they make use of plant species richness and relate this to the least species rich area. This means that they give less value to species poor areas than species rich areas, and all species are regarded as equal. This is however to some degree compensated with use of ecosystem scarcity that gives a higher value for ecosystems that naturally are rarer. The last factor is ecosystem vulnerability that gives higher value for ecosystems where a greater proportion of original distribution is vanished. This factor is made in such a way that the value is increasing exponentially as the area declines.

Weidema and Lindeijer (2001) show how the method can be used for different cases and also make comparisons to use of NPP. However, the data needed to perform the calculations limits the method to only enable comparison of land use activities on biome level.

All these methods do however all have the same limitations since they primarily are developed to diverge between use of different areas for one purpose (i.e. where is it preferable to construct a road) and not to diverge between different managing regimes of an ecosystem, such as different forestry regimes. There is however one exception. A methodology for evaluation land use impact from forestry was developed as a joint project between Axel Springer Verlag, Stora and Canfor (1998) with use of Infras as scientific consultant.

Based on criteria from the Montreal and Helsinki processes for sustainable forest management and recommendations from different forest certification approaches (such as FSC and CSA), they develop a sustainability factor consisting of three components. These are a qualitative sustainability factor, a quantitative sustainability factor and an area factor. In many ways this method represents a mix of the three different methods for land use impact assessment that Lindeijer (2000) identifies.

The qualitative sustainability factor is again composed of 13 different indicators where 7 are on conservation of biodiversity, 3 on maintenance of forest ecosystem condition and soil productivity, and 3 on conservation of water resources. The score for each indicator is given in a scale from A to E where A represents no measurable effect on the ecosystem quality, and E represents total damage. For each forest area these scores are given out from the knowledge of the involved personnel. Then the figures A-E are given relative scores according to the state of the area. C is considered to be current industrial average and is always given the score 1. This means that a B in one area not necessarily gives the same value as a B in other areas for the same indicator, and also that a B in one indicator not necessarily gives the same value as a B in another indicator in the same area. The final score for the qualitative sustainability factor is the mean score for the 13 indicators. None of the biodiversity indicators do necessarily demand direct measurements since the score (A-E) is based on a qualitative description.

The quantitative sustainability factor is representing the current harvesting level as a proportion of the long run sustained yield. If the current harvesting level is equal to or less than the long run sustained yield, the factor is given the score 1. If the current harvesting level is higher, the score will be more that 1.

The area factor is given with the formula $A = \frac{1}{1 - \text{fraction lost}}$ where the fraction lost is the fraction of the forest that is lost to other land use, i.e. roads and buildings. The sustainability factor is given by multiplying the three factors. After construction the sustainability factor, they also add a forest reduction factor and hence also loss of area is taken into account.

Axel Springer Verlag, Stora and Canfor (1998) also try to link the results of the land use impact to the LCA results obtained through the LCA-method Eco-indicator 95. Their results indicate that the land use impact represents 10% of the overall environmental burden from newspaper production and 3% from magazine production. Eco-indicator 95 is however no longer regarded as a useful tool (i.e. Lindeijer 2000), so these results are questionable. Never the less, the results clearly show that land use should not be neglected in these kinds of LCA studies.

5.2 Valuation of impact from different forestry regimes

If a useful methodology for evaluating impact from different forest regimes in LCA is to be developed, it seems obvious that it must be simple, but at the same time be based on ecologically valid assumptions. Of the already mentioned methods, only the method from Axel Springer Verlag, Stora and Canfor (1998) enables comparison between different forestry regimes, but some of the aspects included in Weidema and Lindeijer (2001) seem to be of great value.

A number of the methods mentioned in the previous section diverge between impact due to land occupation and land transformation. It is possible to argue that long time forestry is something between these two. Most methods to evaluate land use impact are constructed to be able to make a decision on where a certain activity should be located. Forestry must necessarily be performed in a forest and the forest will normally be maintained as a forest, i.e. there is an impact from the occupational time. But, in addition forestry will to some degree change the natural conditions in the forest. In the most extreme case a forest is transformed into a plantation (which not even necessarily was a forest in the natural state). On the other extreme, the impacts are so small that it is not possible to detect any changes from the natural condition. There is hence a need to quantify the land change impacts due to forestry.

To be able to construct a useful method it is therefore necessary to make the assumption that the original state of the area is a natural forest and that it will continue to be a forest in infinite time. Then the time dimension is meaningless and leaves the quality change due to forestry activity as the interesting part.

Several authors have argued that vascular plant diversity might be a good measure for characterizing land use impact and ecosystem quality (Lindeijer et al. 1998, Müller-Wenk 1998, Köllner 2000, Schenk 2001, Weidema and Lindeijer 2001). This seems however not to be a good idea. First of all it is as discussed in chapter 3.2 doubtful that vascular plant can serve as a properly indicator on overall biodiversity.

Second, if ecological changes are to be measured through registration of changes in species composition, other groups of species are more useful. Molau and Alatalo (1998) have shown that bryophytes are better indicators than vascular plants for effects of global warming, Hilmo and Holien (2001) have shown that lichens are useful indicators for edge effects and fragmentation, and Bongers (1990) has shown that nematodes are useful indicators for soil conditions.

Third, it is not only important what species that is present, it is also important to maintain areas that enable invasion. Both the equilibrium theory of island biogeography (see chapter 2.2) and the metapopulation concept (see i.e. Schemske et al. 1994) emphasise the need to secure areas that allow invasion and reestablishment of populations, not only areas with populations present. A focus on presence or absence of different species is more or less consciously based on an assumption of static conditions in the ecosystems. According to present ecological knowledge this is simply not true.

Fourth, there might be a tremendous time lag between the change in conditions and actual change in species composition. Saunders et al. (1991) emphasise that this time lag might be on several hundred years for long lived species, such as long-lived trees.

A last argument is that not only what species that is present is of interest, but also the amount. Chapin et al. (2000) emphasises that also abundance matters for ecosystem functioning, and Didham et al. (1996) show that even if a species is present in an ecosystem, the ecosystem might function as if the species is absent if the abundance falls under a certain level. Hengeveld et al. (1995) emphasise that number of species alone is not enough to evaluate diversity, but also i.e. evenness should be taken into account.

This does of course not mean that identification of vegetation is useless. Hengeveld et al. (1995) state that identification of vegetation types have been important for conservation planning, resource management and monitoring of environmental change. This is however not transferable to overall biodiversity evaluation and many of the mentioned methodologies to evaluate biodiversity in LCA fail in one or more ecological assumptions done.

Williams and Humphries (1996) argue it will not be possible to work out direct measures of biodiversity in the nearest future due to both lack of knowledge and limited resources. Lawton et al. (1998b) have figured out that an all-taxa biological inventory for a 'representative hectare' of tropical forest might absorb as much as 10-20 percent of the entire global workforce of about 7000 systematists. For Norwegian conditions, Sætersdal et al. (2002) have calculated that 200 systematists will use 200 year to record all species of vascular plants, macrolichens, bryophytes and wood living polypores in the productive part of the forests.

An additional problem is that it is not possible to predict what does really matter (see i.e. Kunin and Lawton 1996), and hence it is with present knowledge not possible to identify a group of indicator organisms or key species (see section 3.2).

It seems hence obvious that biodiversity measures have to be performed indirectly as indicated in chapter 3.2 and focus must be on maintaining areas with the ecosystem functions intact (see i.e. Soulé 1991, Kunin and Lawton 1996, Spratt 1997, Chapin et al. 2000). Aarts and Nienhuis (1999) goes even further and state that the ultimate goal of nature conservation should not be the preservation of the structure of an ecosystem, but the preservation of the ecosystem's capacity to keep on functioning under a range of environmental conditions.

Based on the understanding of present state of knowledge of measuring impact on biodiversity from land use, it seems as a combination of the methods developed by Axel Springer Verlag, Stora and Canfor (1998) and Weidema and Lindeijer (2001) is the right way to go. Axel Springer Verlag, Stora and Canfor (1998) constructed an index on basis of a qualitative sustainability factor, a quantitative sustainability factor and an area factor. It would also be preferable to construct a measure that would make sense for different ecosystems, not only for forest, and here ideas from Weidema and Lindeijer (2001) are used. A measure for biodiversity impact from land use change could hence be described by the formula

Impact on biodiversity = $CMB \times EV \times ES$

where *CMB* is conditions for maintained biodiversity in the ecosystem, *EV* is ecosystem vulnerability and *ES* inherent ecosystem scarcity.

The quantitative sustainability factor from Axel Springer Verlag, Stora and Canfor (1998) is not included. According to Udo de Haes et al. (1999) it is important to make different impact categories as exclusive as possible, and focus here is chosen to be biodiversity loss due to land use and changes in land use. Hence level of extraction of resources should not be included. Udo de Haes et al. (1999) propose three different land use impact categories where extraction of abiotic resources and extraction of biotic resources are separated from degradation of biodiversity due to land use. Even if overexploitation of biotic resources might result in decreased biodiversity, this is thus not included here.

5.2.1 Inherent ecosystem scarcity (ES)

This term is introduced by Weidema and Lindeijer (2001). The motivation is to have an indicator for the scarcity of different ecosystem since biodiversity linked to scare ecosystems normally would be more vulnerable than biodiversity linked to more widespread ecosystems. Weidema and Lindeijer (2001) express the indicator as the inverse value of the potential area of the ecosystem (A_{pot}), resulting in the formula

$$ES = \frac{1}{A_{pot}}$$

Weidema and Lindeijer (2001) use the indicator value at biome level where data is globally available, but argue that the formula could be used at lower ecosystem levels when data become available. To normalise the value, they use the potential area of the biome that is most common (boreal forests) and multiply with this value. This give values in the range [1, 5.4] (see Weidema and Lindeijer 2001).

To be able to use this factor for different forestry regimes, it is necessary to have area factors for different forest types. The best solution is probably to use vegetation type as classification level. In Norway Fremstad (1997) provides the most comprehensive classification system and should hence form the basis for such registrations. However, no data on potential area of the different vegetation types is available at present. If it is not possible to get data on a lower level than biomes, it will not give any sense in comparing different forest regimes. It will however still be useful if the methodology is to be used for different types of ecosystems.

It will probably be useful to reduce the range to [0, 1] to make comparisons possible independent of scale (biome, landscape, vegetation type etc.).

5.2.2 Ecosystem vulnerability (EV)

Ecosystem vulnerability is introduced to give information about the present total area pressure to an ecosystem type. Axel Springer Verlag, Stora and Canfor (1998) use an area factor given by the formula

$$\text{area factor} = \frac{1}{1 - \text{fraction lost}}$$
for this purpose.

Weidema and Lindeijer (2001) propose another formula given by

$$EV = \left(\frac{A_{exi}}{A_{pot}} \right)^{z-1}$$
where A_{exi} is the existing area of the ecosystem and A_{pot} is the potential area.

This formula is closer linked to the species-area relationship given by the formula $S=cA^z$ (see section 2.2) and seems hence intuitively better that the formula suggested by Axel Springer Verlag, Stora and Canfor (1998). This factor gives values between 1 when there is no loss of the particular vegetation type and infinitely when the remaining areas are approaching zero percent.

Weidema and Lindeijer (2001) use also this factor at the biome level, and in similar way as mentioned with the previous factor, it would be more useful if it was possible to use the factor at a smaller scale such as vegetation type. Theoretically this factor could be used at the level of the amount of forest left on a property, but this makes not much sense for biodiversity considerations. Hence scale is here a problem that must be solved.

Weidema and Lindeijer (2001) are somewhat unclear about the z-value and use both 0.15 and 0.35. As a starting point, it could however be recommended to use 0.25 (see section 2.2).

Even though there is not sufficient data available to use ES and EV at an appropriate level, there exists information about to what degree vegetation types in Norway is endangered. Fremstad and Moen (2001) classify vegetation types in the same scale as is used in species red lists, namely extinct (Ex), critically endangered (CR), endangered (EN), vulnerable (VU), lower risk (LR) and least concern (LC). In addition a category where data is deficient (DD) is used. The vegetation is classified according to Fremstad (1997).

Instead of struggling with deficient data on ES and EV at a suitable level, it is thus possible to use this information as a surrogate for $ES \times EV$. A suggestion could hence be to use the following values for $ES \times EV$:

Critically endangered (CR)	1.00
Endangered (EN)	0.50
Vulnerable (VU)	0.25
Lower risk (LR)	0.12
Least concern (LC)	0.06

Data deficiency should be treated with caution, but data deficiency could be interpreted as a vegetation type with a small distribution and hence threatened. In accordance with the precautionary approach 0.50 should be used as value when data is deficient.

In Annex I all forested vegetation types identified as threatened (CR, EN, VU and LR) by Fremstad and Moen (2001) are presented. None forested vegetation types are classified as DD.

If these values are used, work has to be done to see how future use of *ES* and *EV* could give values in the same range since with the formulas presented here, *ES* is in the range [0, 1] while *EV* is in the range [1, ∞]. The product is hence theoretically in the range [0, ∞].

5.2.3 Conditions for maintained biodiversity (CMB)

The purpose with this factor is to give information about the maintenance of biodiversity in the ecosystem in focus. This is in contradiction to the species richness factor used by Weidema and Lindeijer (2001) which is based on the assumption that species rich ecosystems are more 'valuable' than species poor ecosystems. This might be true if the value of an ecosystem is based on the probability of finding species that might be valuable as food and medicines (see section 2.1), but if value is based on the maintenance of ecosystem function, such an assumption is not possible to support (i.e. Martinez 1996). Hence all natural functioning ecosystems are valued equal and focus is whether biodiversity in the ecosystem is maintained.

This factor should hence be used to quantify the differences of impacts on biodiversity due to different ecosystem management regimes, i.e. forestry. Several authors (i.e. Johnson and Jonsson 1995, Ohlson and Tryterud 1999, Eid et al. 2002) pinpoint that considerations to maintain biodiversity reduces the possibility to harvest as much as could have been done, and hence argue that products from such forests should be acknowledged as more valuable. The use of such a measure in LCA of products makes this possible. This factor is somewhat similar to what Axel Springer Verlag, Stora and Canfor (1998) call a 'qualitative sustainability factor'.

Larsson (2001) identifies 17 key factors for biodiversity (at stand scale) for forests. These are divided in 8 structural factors, 2 compositional and 7 functional of which 3 are due to natural disturbances and 4 from human influences. These are:

- structural factors
    - S1 tree species
    - S2 stand size
    - S3 edge characteristics
    - S4 forest history
    - S5 habitat type(s)
    - S6 tree stand structural complexity
    - S7 dead wood
    - S8 litter
- compositional factors
    - C1 species with specific stand type and scale requirements
    - C2 biological soil conditions
- functional factors
    - N1 fire

- N2 wind and snow
- N3 biological disturbance (incl. pests)
- H1 forestry
- H2 agriculture and grazing
- H3 other land-use
- H4 pollution

The challenge is hence to use these to develop a set of indicators that can be measured/recorded without too much effort and combine these to a factor value for maintained biodiversity.

The scientific challenges for doing this are considerable. The indicators must be easy and precise to use, and still capture what is important for maintenance for biodiversity. Further they have to be scaled – there must be taken a decision for at what level is the impact zero and how the indicator score are to be related to the increase in impact. Last the indicators have to be added to one factor and hence they must be weighted according to each other.

As a first suggestion all indicators should be measured on a four level scale where the scores are given the same meaning as Larsson (2001) use for key factors:

0 – no impact
1 – slight impact
2 – moderate impact
3 – major impact

In addition it is suggested that all indicators should be multiplied with a factor according to the relative impact the condition the indicator capture has for maintenance of biodiversity. This means that an indicator with slight impact actually have the scale [0, 1, 2, 3] while an indicator with major impact have the scale [0, 3, 6, 9].

The valuation of these indicators must necessarily be dependent of what kind of forest that is in focus since the valuation will differ. In addition, if this methodology is going to be expanded to other ecosystems, other sets of indicators will be necessary. The *CMB* must hence be given a maximum value. It is hence recommended that the *CMB* factor value is given as

$$CMB = \frac{\sum_{i=1}^{n} BI_i}{\sum_{i=1}^{n} BI_{i,\max}}$$

where BI_i is the weighted score for biodiversity indicator i. The values of *CMB* will hence be in the range [0, 1].

Primarily based on Larsson's (2001) key factors, the following indicators for boreal spruce forest are suggested:

- fragmentation
- cutting

- tree species composition
- regeneration
- dead wood
- area set aside
- natural disturbances
- ditching

Fragmentation

This is maybe one of the most severe problems for biodiversity today, and several authors stress this aspect (i.e. Noss 1990, Andrén 1994, Cushman et al. 1995, Hengeveld et al. 1995, Johnson and Jonsson 1995, Didham et al. 1996, Laurance et al. 1997, Larsson 2001). A good measure for fragmentation could at least comprise the consideration of factors S2 and S3 in Larsson (2001). Fragmentation is however a result of a range of activities and here it is suggested to separate between cutting and fragmentation due to more or less permanent constructions as roads and power lines in the forested area.

Boreal forests are still rather continuous and a suggested indicator by Norwegian authorities is meter road per km². This should however be expanded to include all constructions such as power lines etc. In Norway the average is about 900 meters road per km², but on some properties there is about 3000 meters road per km², which is about the average in several countries in central Europe (Landbruksdepartementet 1998).

The impact on biodiversity is of course not only dependent on the length of constructions, but aspects such as shape on stands and habitat isolation, but to be able to construct a simple indicator, such aspects are suggested omitted. There is no doubt that the present level of fragmentation of forests are believed to have a negative impact, which means that 900 m/ km² can not be given the value '0'. A suggestion might be that this factor is given the value '0' in the range [0, 500], '1' in the range [501, 2500], '2' in the range [2501, 5000] and '3' in the range [5001, ∞], but these figures are no more than guesses and need further work. It is also possible to use a more descriptive description of the different scores with emphasise on effects. This is done by Axel Springer Verlag, Stora and Canfor (1998). The disadvantage with this is the introduction of the subjective opinion of the person doing the valuation.

The factor should be given weighting factor 3 (Larsson 2001).

Cutting

Johnson and Jonsson (1995) emphasise that large-scale operations have generally demonstrated more adverse impacts on forest ecosystems than small-scale operations. Cutting is altering the physical fluxes across the landscape, such as radiation, wind and water fluxes. Hence, cutting should as far as possible mimic natural forest dynamics (Bengtsson et al. 2000), and clearcutting should ideally be performed in such a way that the species with the lowest dispersal ability still is able to move between their feasible habitats (Chapin et al. 1998). Sometimes it is argued that clearcutting mimic fire and windthrows and hence is almost natural, but this is misleading and maybe completely wrong (Essen et al. 1997, Bengtsson et al. 2000). Johnson and Jonsson (1995) states that it is shown that harvesting practices that mimic natural-stand dynamic and landscape-level dynamic provides the best opportunity to maintain biodiversity in natural forest ecosystem. Cutting regime is hence a factor that should be included and the necessity of protecting old trees (i.e. Ohlson and Tryterud 1999, Bengtsson et al. 2000, Larsson 2001, Eid et al. 2002) could also be taken care of within this factor. Johnson and Jonsson (1995) stress that it is important that these trees are not damaged by forestry machines.

A possible indicator could be based on a combination of size of clearcutted area, trees left per area unit etc. It then seems rather obvious that it has to be made a description of what is necessary for the zero impact level; i.e. no clearcutting over more that 2 decare and at least 5 mature trees left per decare. The definitions for the different scores must be worked out.

This factor incorporates to some degree the factor S1, S4 and S6 from Larsson (2001), in addition to S3 that also is included in the previous factor, and should be given the weighting factor 3.

<u>Tree species composition</u>

It is also important to take care of tree species composition (i.e. Ohlson and Tryterud 1999, Larsson 2001). It is hence not only important to leave a certain number of trees, but also to maintain the diversity among tree species. This is important also for the diversity of other species and also the nutrient cycling (Pastor et al. 1995). According to Ohlson and Tryterud (1999) the deciduous trees should constitute about 15 percent of the mature trees.

A possible indicator should be based on a combination of the conservation of all naturally occurring species and the ratio among them. Use of alien species should also be included since this has a severe impact on biodiversity (i.e. Cushman et al. 1995). In the same way as with the previous indicator, it is necessary with a description of the zero impact level. This should be presence of all naturally occurring species where the percentage reflects natural occurrence and no occurrence of alien species. The definitions for the different scores must be worked out.

This factor incorporate especially factor C1 from Larsson (2001), but to some degree also S8 and C2 since these at least to some extent are dependent on the species composition. Also this factor should hence be given the weighting factor 3.

<u>Regeneration</u>

Regeneration regime should also be taken into consideration since this have a huge impact on i.e. how successions will run (i.e. Bengtsson et al. 2000). This indicator should hence reflect the proportion of the area that is left for natural regeneration and the zero level should hence be no plantations. The limits for the other scores must be worked out.

This indicator is also incorporating several of the structural factors identified by Larsson (2001) and should be given the weighting factor 3.

Dead wood

The amount of dead wood is another factor that is stressed by several authors (i.e. Johnson and Jonsson 1995, Ohlson and Tryterud 1999, Bengtsson et al. 2000, Larsson 2001, Siitonen 2001). Siitonen (2001) has estimated that the amount of dead wood in boreal forests is reduced with as much as 90-98 percent and more than 50 percent of original saproxylic species might disappear in the long run as a consequence. In Norway 22 percent of the red list species in forests are associated with lying dead wood (Baumann et al. 2001b). There is not only the amount of dead wood that matters, but also the quality. This means that it is special important that dead wood from different tree species is present and at different decay stages.

Following Siitonen (2001), all reduction of dead wood will necessarily result in reduced number of saproxylic species due to a general species-area relationships (see section 2.2). There is however not possible to give a fixed number of how much dead wood that have to be present in a managed forest before it can be said to be of no impact since the amount vary naturally both between forest habitats, types and latitude among other factors (Siitonen 2001).

Ståhl et al. (2001) give an overview over different methods to estimate the amount of dead wood. It seems most appropriate to use number of standing dead trees (snags) and lying dead trees (downed logs) as measure, supplemented with tree species when possible.

It seems at present not possible to suggest within which ranges the different scores for this factor should be placed. It is however obvious that the amount of dead wood must be significantly higher in managed forest that what is the average situation today and that dead wood from all naturally occurring species must be present when the score is set as '0'. In Sweden FSC certification demands 3 m^3 per hectare (Swedish FSC Council 2000)

The indicator should be given weighting factor 3 (Larsson 2001).

Area set aside

No matter how many precautions that are taken in the managed forest, there will still be a need to set some areas aside to ensure areas where natural dynamics can occur and also take care of particular important areas, such as 'hotspots', old growth forest and transition borders to i.e. rivers. In boreal forests Gjerde and Sætersdal (2002) estimates that 5 percent of the total area can capture 20 percent of the occurrence of red list species. Baumann et al. (2001b) also identifies different habitats that seem more important to protect, such as ravines and rocky walls, since they are relatively rare. Different certification systems also demand areas set aside, and in Sweden the FSC certification demands in total 5 percent (Swedish FSC Council 2000).

It seems at present not feasible to suggest within which ranges the different scores for this indicator should be placed. The score must however both take the total area in consideration together with the identification and protection of particular important areas if present. Also this indicator should be given the weighting factor 3.

Natural disturbances

Natural disturbances such as fire, wind and snow and biological disturbances (key factors N1, N2 and N3 in Larsson (2001)), is rather difficult to use as indicators since the frequency of this disturbances will wary considerably. The indicator must hence probably be given score based on to what degree natural disturbances are allowed to occur and to what degree the area is allowed to follow a natural succession after the disturbance. This has hence to be some sort of qualitative measure that at least should promote that a proportion of the affected area is allowed to follow natural successions.

Following Larsson (2001) this indicator should be given the weighing factor 3.

Ditching

Ditching strongly influences the water regimes in an area and hence the biodiversity (Baumann et al. 2001a). The amount of area affected by ditches should hence be measured where the zero level is no ditches. The limits of the other scores must be worked out.

This indicator is not treated independently by Larsson (2001), but due to the severe effect of ditches, it seems obvious that it should be given the weighting factor 3.

Other factors

Also other impact factors could be considered, such as grazing, but due to the low impact (see Larsson 2001) it is omitted here. It is however possible to include more indicators with time since the score of the *CMB* is independent of the number of indicators. This makes also the different weighting factors useful even if all are set to 3 so far. This makes it also useful even if it turns out to be impossible to determine reasonable levels for some of the indicators proposed and hence have to be omitted.

Fertilisation is not included here even though this undoubtedly might have a severe impact in biodiversity. Fertilisation should however be included as eutrophication in LCA and should hence not be included here to avoid double counting. This is also the case with pesticides used in the forest and other types of air pollution that the forest is exposed to.

5.3 Biodiversity measures and forest certification systems

The major challenge if this impact assessment method is to be used, is to get relevant data for the timber that reaches the product chains. One opportunity is to use forest certification systems to provide this kind of data. As mentioned in chapter 1, there is however no such data easy available due to the structure and demands in forest certification systems today. It is however possible to use the threshold values in the forest certification systems in lack of better data. Here Swedish FSC demands for instance at least 5 percent of the total area set aside (Swedish FSC Council 2000) while the Norwegian ‘Levende Skog’ (‘The Living Forest’, a certification system linked to PEFC) demands 1 percent (Levende Skog 1998). This does however provide rather unsatisfactory data and hence LCA results since all forestry certificated from a specific certification system will have the same score in *CMB*, and the result is actually a weighting of the different certification systems.

There is however promising work going on in several countries to develop registration forms for forestry planning. In Norway this project is called ‘Miljøregistrering i skog – biologisk mangfold’ (MiS, ‘Environmental registrations in forests – biological diversity’, see i.e. Gjerde and Baumann 2002).

The intention with this work is to develop tools for forestry planning that are supposed to support the foresters in making decisions on how to plan the forestry activities and still maintain biodiversity. Even though this is meant to be a tool for planning, the registration forms could with some modifications also provide input data for the method outlined in the previous section. This do however presuppose that the registration schemes are changed in accordance to the indicators necessary for calculation of *CMB*, and the data must be public available and follow the timber. The use of such registration forms as an obligatory activity for certified forests would be of great value.

6 Discussion and conclusions

The method outlined is based on previous methods for including land use impact on biodiversity in LCA, and is in line with recommendations from i.e. Udo de Haes (1999) and Lindeijer (2000). The method could be developed to be useful for different types of ecosystems.

The quality of an ecosystem is actually defined as a product of inherent ecosystem scarcity (*ES*, section 5.2.1) and ecosystem vulnerability (*EV*, section 5.2.2). The quality is hence defined independent of number of occurring species – the valuation is strictly based on the natural occurring amount of the ecosystem and the percentage left.

As already emphasised, there is at present lack of data on a useful scale. There is however several initiatives taken to improve the knowledge on the distribution on different vegetation types, which in time will increase the amount of available data. Unfortunately the different initiatives in Norway are not using the same taxonomy on vegetation types which reduce the value of these registrations. Fremstad (1997) has presented a comprehensive classification of vegetation types in Norway, but the Directorate for Nature Management has developed another classification system which only to some extent is linked to the vegetation types given by Fremstad (Direktoratet for naturforvaltning 1999a). Local municipalities are requested to use this classification when they are mapping the vegetation in their areas. Furthermore, the MiS-project is using a third classification and the statistics of forest condition and resources in Norway (i.e. Tomter 1999) provides forest data on yet another from. This is severely reducing the value of these registrations. The classification used in the European NATURA 2000[3] habitat classification system is to some extent similar to the classes Fremstad (1997) uses, but on a larger scale. The taxonomy used in the different systems is compared in Annex 1.

An open question that needs to be solved is how to deal with seminatural vegetation types. If values on *ES* and *EV* are used strictly, seminatural vegetation types are regarded as without any value. It is of course possible to argue that only natural occurring vegetation types should be protected, but this is not an common opinion and will undoubtedly result in extinction of a range of species adapted to these habitats through millennia. Ironically this problem is solved as long as values based on how endangered a vegetation type is, are used instead of $ES \times EV$ since seminatural and natural vegetation types are treated the same way by Fremstad and Moen (2001).

In addition there is introduced a factor for conditions for maintained biodiversity in the ecosystems (*CMB*, section 5.2.3). If necessary precautions are taken in management of an ecosystem, this reduces the quality impact of the area use. This is shown in figure 4. In situation I, the quality of the area is given as $ES \times EV$ and t is from the beginning of the land transformation to the re-naturalisation has taken place. The reference state is here the original natural state. In situation II, the quality difference is reduced due to precautions taken, and the quality difference is given as $ES \times EV \times CMD$ ($CMD < 1$, see section

[3] http://europa.eu.int/comm/environment/nature/hab-an1en.htm

5.2.3). In addition, the values must be multiplied with an area factor in both cases (see section 5.1). Müller-Wenk (1998) suggests re-naturalisation times for different ecosystems.

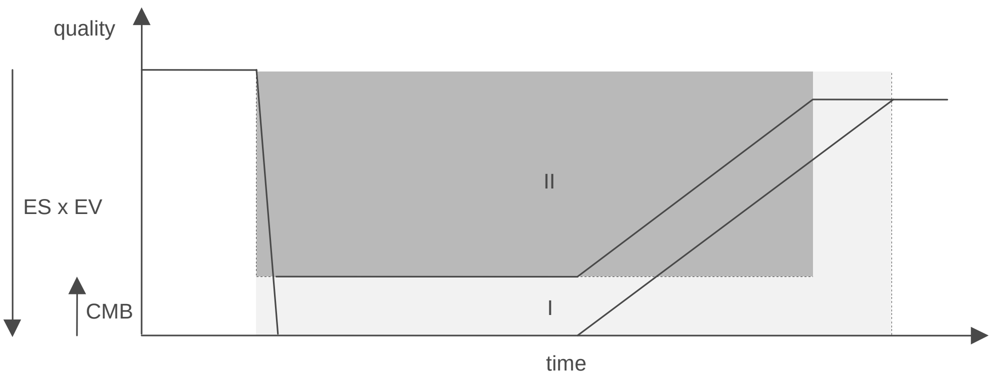

FIGURE 4 – DIFFERENT ECOSYSTEM QUALITY CHANGES DEPENDENT ON CONDITIONS FOR MAINTAINED BIODIVERSIT (*CMB*).

When the land transformation already has taken place, the time *t* will be the time the land area actually is used. As described in section 5.2, the time dimension is meaningless if the forest is maintained as a forest. Hence, instead of using area, the total impact should be calculated as a function of quality change and area needed to provide the necessary amount of timber. If a company uses timber with different values for impact on biodiversity, an average should be used.

The challenges are to develop measures for *CMB* for different ecosystems and develop measures for *ES* and *EV* for other levels than biomes.

For forestry an outline for the *CMB*–factor is presented in section 5.2.3. The advantage with the suggested way of constructing the *CMB*-factor, is that it is actually possible to use the factor as soon as one of the indicators are properly worked out since the range of scores is fixed. The usefulness of the factor will of course increase with increased number of indicators included since the number of considerations taken increases. This will also be the situation for other ecosystems, if *CMB*-factors are to be developed.

The *CMB*–factor for forestry need however further work since none of the indicators suggested here are possible to use as long as the managing requirement for the different scores are not set. Here research activities are needed, but experience from work done as part the ‘Environmental registrations in forests’-project in Norway (i.e. Gjerde and Baumann 2002) and similar projects in other countries, could hopefully provide a staring point.

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Annex 1 – Threatened vegetation types in Norway

This is an overview over the forested vegetation types that are regarded as threatened in Norway according to Fremstad and Moen (2001). The nomenclature given in bold is the name used by Fremstad and Moen (2001). Nomenclature used on the identified vegetation types by Fremstad (1997) (given both in Norwegian and English), Direktoratet for Naturforvaltning (DN, 1999a), 'Miljøregistreringer i Skog' (MiS) and Natura 2000 are given as identified by Fremstad and Moen (2001). Also the information on level of threat, to what degree the vegetation types are threatened by forestry, changes in distribution and other remarks are from Fremstad and Moen (2001).

Purpurlyng-furuskog

Nomenclature Fremstad	A3 Røsslyng-blokkebærfuruskog / Heather – bog bilberry – Scots pine wodland; *Calluna vulgaris – Vaccinium uliginosum – Pinus sylvestris* woodland A3d Purpurlyng-utforming / *Erica cinerea* subtype
Nomenclature DN	Kystfuruskog
MiS	Not mentioned
Natura 2000	No parallel
Level of threat	VU - Vulnerable
Threatened by forestry	Yes, among others. Main threat due to inherent scarcity
Changes in distribution	?
Remarks	Only present in Scotland in addition to Norway

Kalkskog

Nomenclature Fremstad	B2 Kalklavurtskog / Calcareous low-herb woodland B2a Xerofil furu-utforming / Xerophilous *Pinus sylvestris* st. B2b Mesofil furu-utforming / Mesophilous *Pinus sylvestris* subtype B2c Bjørk-utforming / *Betula pubescens* ssp. *pubescens* st.
Nomenclature DN	Kalkskog
MiS	Kalklågurtskog
Natura 2000	No parallel
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, among others. Urban development and lime pits important factors
Changes in distribution	Declining
Remarks	Further subdivision possible, different subtypes have different levels of threat

Høystaudegranskog

Nomenclature Fremstad	C2 Høystaudebjørkeskog / Tall-herb, downy birch and Norway spruce forest C2b Høystaudegran-utforming / Tall-herb – *Picea abies* st.
Nomenclature DN	Urskog/gammelskog

	Kystgranskog
MiS	Høgstaudeskog
Natura 2000	9050 Fennoscandian herb-rich forests with *Picea abies*
Level of threat	LR – Lower risk
Threatened by forestry	Yes, important
Changes in distribution	Declining

Nordlig høystaudeskog

Nomenclature Fremstad	C2 Høystaudebjørkeskog og høystaudegranskog / Tall-herb, downy birch and Norway spruce forest C2d Lappflokk-storveronika-bjørk-utforming / *Polemonium acutiflorum – Veronica longifolia* st.
Nomenclature DN	Bjørkeskog med høystauder
MiS	Not mentioned
Natura 2000	9040 Nordic subalpine/subarctic forests with *Betula pubescens* ssp. *czerepanovii*
Level of threat	LR – Lower risk
Threatened by forestry	Vulnerable to all changes in land use due to inherent scarcity
Changes in distribution	?

Blåbær-bøkeskog

Nomenclature Fremstad	D1 – Blåbær-edelløvskog / Bilberry deciduous woodland D1b – Blåbær-bøkeskog / Bilberry – beech subtype
Nomenclature DN	Gammel edelløvskog
MiS	Not mentioned
Natura 2000	9110 *Luzulo-Fagetum* beech forests
Level of threat	LR – Lower risk
Threatened by forestry	Yes, among others
Changes in distribution	Increasing

Lavurt-eikeskog

Nomenclature Fremstad	D2 – Lavurt-edelløvskog / Low-herb deciduous woodland D2a – Lavurt-eikeskog / Low-herb – oak subtype
Nomenclature DN	Rik edelløvskog
MiS	Lågurt-eikeskog
Natura 2000	9020 Fennoscandian hemiboreal natural old broad-leaved deciduous forest (*Quercus*, *Tilia*, *Acer*, *Fraxinus* or *Ulmus*) rich in epiphytes
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, among others
Changes in distribution	Decreasing

Lavurt-bøkeskog

Nomenclature Fremstad	D2b – Lavurt-bøkeskog / Low-herb – beech subtype (low-herb deciduous woodland)

	D3 – Myske-bøkeskog / *Galium odoratum* – beech woodland
Nomenclature DN	Gammel edelløvskog Rik edelløvskog
MiS	Lågurt-bøkeskog
Natura 2000	9130 *Asperulo-Fagetum* beech forests
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, among others
Changes in distribution	Increasing?

Rikt hasselkratt

Nomenclature Fremstad	D2 – Lavurt-edelløvskog / Low-herb deciduous woodland D2c – Rike kysthasselkratt / Rich hazel thickets, coastal st. D2d – Rike hasselkratt, østlig utforming / Eastern st.
Nomenclature DN	Gammel edelløvskog
MiS	Not mentioned
Natura 2000	No parallel
Level of threat	EN – Endangered
Threatened by forestry	Yes, as new areas for spruce
Changes in distribution	?

Alm-lindeskog

Nomenclature Fremstad	D4 – Alm-lindeskog / Wych elm – small-leaved lime woodland
Nomenclature DN	Rik edelløvskog Gammel edelløvskog
MiS	Alm-lindeskog
Natura 2000	9020 Fennoscandian hemiboreal natural old broad-leaved deciduous forest (*Quercus*, *Tilia*, *Acer*, *Fraxinus* or *Ulmus*) rich in epiphytes
Level of threat	LR – Lower risk
Threatened by forestry	Yes, as new areas for spruce
Changes in distribution	Decrease stopped?

Gråor-almeskog

Nomenclature Fremstad	D5 – Gråor-almeskog / Grey alder – wych elm woodland
Nomenclature DN	Rik edelløvskog Gammel edelløvskog
MiS	Not mentioned
Natura 2000	9020 Fennoscandian hemiboreal natural old broad-leaved deciduous forest (*Quercus*, *Tilia*, *Acer*, *Fraxinus* or *Ulmus*) rich in epiphytes
Level of threat	LR – Lower risk
Threatened by forestry	Yes, as new areas for spruce
Changes in distribution	Decrease stopped?

Or-askeskog

Nomenclature Fremstad	D6 – Or-askeskog / Alder – ash woodland D6a – Or-ask-utforming (østlig) / Grey alder/alder – ash st. D6b – Svartor-ask-utforming (vestlig) / Alder – ash subtype
Nomenclature DN	Rik edelløvskog Gammel edelløvskog
MiS	Or-askeskog
Natura 2000	9020 Fennoscandian hemiboreal natural old broad-leaved deciduous forest (*Quercus*, *Tilia*, *Acer*, *Fraxinus* or *Ulmus*) rich in epiphytes
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, as new areas for spruce
Changes in distribution	Decreasing?

Rik sumpskog

Nomenclature Fremstad	E4 – Rik sumpskog / Rich swamp woodland
Nomenclature DN	Rikere sumpskog
MiS	Gran- og bjørkesumpskog Lauv- og viersumpskog
Natura 2000	9080 Fennoscandian deciduous swamp wood
Level of threat	EN – Endangered
Threatened by forestry	Yes, as new areas for spruce after draining
Changes in distribution	Decrease stopped?

Varmekjær kildeløvskog

Nomenclature Fremstad	E5 – Varmekjær kildeløvskog / Thermophilous spring woodland E5a – Snelle-ask-utforming / *Equisetum – Fraxinus excelsoir* subtype E5b – Slakkstarr-svartor-utforming / *Carex remota – Alnus glutinosa* subtype
Nomenclature DN	Rikere sumpskog
MiS	Lauv- og viersumpskog Varmekjær kildelauvskog
Natura 2000	9080 Fennoscandian deciduous swamp wood 9020 Fennoscandian hemiboreal natural old broad-leaved deciduous forest (*Quercus*, *Tilia*, *Acer*, *Fraxinus* or *Ulmus*) rich in epiphytes
Level of threat	CR – Critically endangered
Threatened by forestry	Yes, among others
Changes in distribution	Decreasing

Svartor-strandskog

Nomenclature Fremstad	E6 - Svartor-strandskog / Alder seashore woodland

Nomenclature DN	Rikere sumpskog
MiS	Not mentioned
Natura 2000	9080 Fennoscandian deciduous swamp wood
Level of threat	EN – Endangered
Threatened by forestry	Probably as new areas for spruce
Changes in distribution	Decreasing

Gråseljekratt

Nomenclature Fremstad	E2 – Lavland-viersump / Lowland willow swamp E2a – Gråselje-urt-utforming / *Salix cinerea* - herb subtype E2b – Gråselje-høystarr-utforming / *Salix cinerea – Carex* st
Nomenclature DN	Included in delta areas
MiS	Viersump
Natura 2000	No parallel
Level of threat	VU – Vulnerable
Threatened by forestry	No
Changes in distribution	?

The following vegetation types are not natural woodland vegetation, but different types of cultivated and seminatural vegetation with tree layer. There is not given any reference to nomenclature in MiS since this vegetation types are not treated here.

Løveng

Nomenclature Fremstad	Several vegetation types in G (anthropogenous grassland), together with tree layer
Nomenclature DN	Kulturlandskap
Natura 2000	6530 Fennoscandian wooded meadows
Level of threat	CR – Critically endangered
Threatened by forestry	No
Changes in distribution	Decreasing

Hagemark

Nomenclature Fremstad	Several vegetation types in G (anthropogenous grassland), together with tree layer
Nomenclature DN	Hagemark
Natura 2000	No parallel
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, plantations
Changes in distribution	Decreasing

Beiteskog

Nomenclature Fremstad	Several vegetation types in G (anthropogenous grassland) (2, 4, 5, 12, 13)
Nomenclature DN	Naturbeitemark
Natura 2000	No parallel

Level of threat	VU – Vulnerable
Threatened by forestry	No
Changes in distribution	Decreasing

Høstingsskog

Nomenclature Fremstad	Not mentioned as separate vegetation type (forest where foliage is harvested)
Nomenclature DN	Skog
Natura 2000	No parallel
Level of threat	EN – Endangered
Threatened by forestry	Yes, plantations
Changes in distribution	Decreasing

Rik (inkl. intermediær) skog-/krattbevokst myr

Nomenclature Fremstad	L1 – Skog-/krattbevokst intermediær myr / Intermadiate wooded and scrub-covered fen M1 – Skog-/krattbevokst rikmyr / Rich wooded and scrub-covered fen Transition types to E3 – Gråor-bjørk-viersumpskog og kratt / Grey alder – downy birch – willow swamp woodland and scrub E4 – Rik sumpskog / Rich swamp woodland E5 – Varmekjær kildeløvskog / Thermophilous spring woodland E6 – Svartor-strandskog / Alder seashore woodland
Nomenclature DN	Rikmyr Rikere sumpskog
Natura 2000	7230 Alkaline fens
Level of threat	VU – Vulnerable
Threatened by forestry	Yes, drainage and plantations
Changes in distribution	Decreasing
Remarks	Level of threat is varying remarkably between different vegetation zones

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