Land management meeting several environmental objectives

Minimizing impacts on greenhouse gas emissions, biodiversity and water

Knowledge compilation and systems perspectives

MATS OLSSON, PETRA ANDERSSON, TOMMY LENNARTSSON, LISETTE LENoir, LENNART MATTSSON & ULRIKA PALME

Which mitigation options in land-use management do meet the goals for greenhouse gas emissions, biodiversity and water security?

This report makes a systems analysis of land-use and its implications for greenhouse gas emissions, biodiversity and water.

The main conclusions are that:

• Most land management strategies can meet the goals for biodiversity and water in an adequate way, except intensive forestry, although trade-offs between different environmental values will be necessary.

• It is important that there is an understanding of how the prerequisite for impacts of land-use changes in a changing climate.
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– minimizing impacts on greenhouse gas emissions, biodiversity and water

Knowledge compilation and systems perspectives

By Mats Olsson, Petra Andersson, Tommy Lennartsson, Lisette Lenoir, Lennart Mattsson & Ulrika Palme
Preface

This report presents a systems perspectives of interactions and feedbacks in land-use and its implications for greenhouse gas emissions, biodiversity and water. The aim of the report is to serve as a basis for land-use planning and management nationally but it could also give a common background for Swedish participants in the negotiation in international conventions.

The report analyses the mostly used management options in Sweden and the implications for greenhouse gas emissions, biodiversity and water. In the final chapter most of these management options are tabled showing implications specifically for terrestrial biodiversity.

The report is part of a coordinated work which Swedish EPA initiated to focus on cross cutting issues within recommendations for Kyoto Protocol and its land use sector (LULUCF) and biodiversity.

A research team representing different disciplines has compiled the report and made the analysis. The research team are responsible for the content in the report.

The results in the report has been discussed with stakeholders at Swedish Environmental Protection Agency dealing with negotiations in the Climate and Biodiversity Conventions UNFCCC and UNCBD.

The report has also been discussed in two following workshops together with external stakeholders, one on agricultural land “Biodiversity and 0-emissions for carbon” and one on forest land that has to provide biomass and bioenergy, sequester carbon and meet the goals for biodiversity and ecosystem services.

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Land management meeting several environmental objectives
Introduction

Land use and management measures have strong implications on environmental properties such as the quality of soil, water and air, and biodiversity, as well as on social and cultural values. This is also clearly indicated in the Swedish National Environmental Quality Goals where measures to meet quality criteria have been set. However, there are several question marks concerning relations between land management and environment, and in particular there are conflict situations where a specific management form might be positive for the environment in one respect, but negative in another. Although it is not the mandate of science to balance different needs, it is definitely its role to describe the environmental, social and cultural impacts, as well as the economic viability, including the use of environmental and sustainability assessment tools. In this way science enables decisions towards sustainable land use.

The environmental, social and cultural impacts from land use might in the future be even bigger as the needs for food, feed, fibre and fuel are supposed to increase substantially. At the same time climate change and loss of natural resources will further limit our ability to meet the demands for food, feed, fibre and fuel. In short, we need to produce more under more difficult circumstances, with less available resources and with less (preferably not any) environmental negative impact. Thus, there is an urgent need to more accurately understand relations between environment and a number of land management measures. Towards this background, the aim of this project was to describe the state of art concerning land management and environment and to elucidate urgent knowledge gaps in order to enable prioritization of further research. The focus is on Swedish conditions, although globalization due to increased global trading and increased global environmental concerns necessitate a certain outlook beyond national boundaries.

There is almost an unlimited amount of land use and management varieties. For this reason the study was restricted to some management forms that either concerns a large part of Sweden or, according to the present knowledge, may provide big consequences and/or big uncertainties. It was also restricted to terrestrial land use including wetlands, i.e. the use of water bodies, and fisheries are excluded. Included are complicated questions in forestry such as harvest of biomass in production forestry (c. 60% of all Swedish land), use of harvest residues, cutting forms, nitrogen fertilization, liming, choice of tree species and drained peat-land management. In agriculture we focused on fertilization, liming, cropping systems and tillage and crop-residue management. We decided not to evaluate the use of genetically modified organisms neither in agriculture nor in forestry as the large political and environmental uncertainties involved motivate a report by itself. Finally we also assess methods and consequences for energy forestry and, briefly, for reindeer grazing since about 40% of the Swedish land-area is used for reindeer grazing. If reindeer production is used as an alternative for intensive meat production
it will be a measure to decrease emissions of greenhouse gases. Grazing by reindeer affects biodiversity, often positively, especially in areas that suffer from increased abundance of broad-leaved vegetation due to climatic changes. Conflicts are possible in future: the area that is suitable for reindeer grazing may decrease due to a warmer climate, but also due to demands for agricultural development.

The report is organized in such a way that the management forms are discussed one by one, followed by a systems perspectives approach. We begin with summarizing conclusions.

**Systems perspective – How to read figures in the report**

For most chapters in the report there are one or two summarizing figures drawn from an environmental systems perspective. For most options described in the figures there is a reference state given in the figure caption (although not illustrated in the figure). When so, the figure must be read with the reference state in mind as e.g. an increase or decrease in biodiversity depends on that reference state. In the figures, boxes represent activities, and arrows either represent flows, or simply “leads to”, when connecting two activity boxes. Green colour signifies avoided activities and related resource use and emissions. Grey colour signifies activities or flows that are likely to be of minor importance in the specific scenario. Oval shapes with dotted boundaries and open arrows at both ends represent activities which life cycles should be included in a systems perspectives for a full picture, but which are either beyond the scope of the report, or link to an earlier figure which is then given. In two figures life cycle data from the CPM (Center for environmental assessment of product and material systems) LCA database, 2011, is included. These date from 2005 and are included to give the reader an idea of the size of resource use and emissions involved.

In the summaries below the pictures, the various effects, goal conflicts and the knowledge gaps discussed refer to environmental effects and ecosystem services. Conclusions on economic and social aspects are beyond the scope of this report.
Summary and discussion

Table 1 is a very condensed summary of the report. It must be read with the comparisons made in mind, i.e. a specific action is not necessarily positive or negative with regard to the chosen parameters generally speaking – only as compared to the reference states used in this report. The effect on climate change is either direct (source or sink of carbon dioxide) or indirect (via a substitution effect). In the case of fossil fuel substitution there is a delay in climate change mitigation; whereas the emission of CO₂ from biomass burning is immediate, the uptake of CO₂ in the trees that are replacing the cut trees is taking place over decades. Generally speaking, substitution for a construction material is more effective than substitution for fuel. Notably, the table says nothing of the size of the impacts discussed; for this we refer to the special chapters and the literature cited. Neither does the table, nor the report, say anything about how to measure the impact of the different actions. Let alone the report says something about how the various effects can be compared to each other. Most plausible, the answers to these questions will vary from case to case, but also between different actors in the field, depending on what is ascribed the highest importance – or value – in different situations (Haider & Jax 2007).

Critical trade-offs

It can be seen from table 1 that many activities that have a positive effect on climate change through a stock or sink mechanism also have positive effects on biodiversity, whereas an increased substitution effect tend to conflict with biodiversity. Similar patterns are there for eutrophication and water regulation (when relevant). These patterns give rise to complex choices as it has to be considered how important harvest of biomass (substitution effect) is as compared to e.g. biodiversity or eutrophication. Except local and case specific aspects – social as well as ecological – there is also a time aspect involved. Our obligations to future generations also needs to be taken into consideration in management of natural resources (de-Shalit 1995; Dobson 1999).

Notably, biodiversity, the nitrogen cycle and climate change (in that order) have been pointed out by Rockström et al. (2009) as the three most critical out of nine so called planetary boundaries. Crossing these boundaries is, according to the authors, associated with a risk of deleterious, possibly disastrous consequences for humans. This is pointed out to underline how critical land use measures are, and that the trade-offs between climate change, biodiversity and nitrogen cycle impacts are far from obvious. How do we determine what degree of climate change that corresponds to a given change of biodiversity? It can be argued that increased climate change will in the end affect biodiversity negatively, but on the other hand it can also be argued that higher biodiversity generally means more resilient ecosystems, and more resilient ecosystems cope better with climate changes.
Table 1. Overview of environmental impacts from the actions discussed in the report. Please note that only environmental and so called cultural ecosystem services are included and that the actions and their effects must be regarded in relation to a reference state – in this case given in the figures referred to in the table. “Subst” = substitution effect. See text for more nuanced descriptions of the various actions and their effects.

<table>
<thead>
<tr>
<th>Action/Effects (Fig)</th>
<th>Climate change</th>
<th>Biodiversity</th>
<th>Other</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest reservation; short term (1)</td>
<td>Increased sink</td>
<td>Increase</td>
<td>Cultural and regulating ecosystem services</td>
</tr>
<tr>
<td></td>
<td>No subst</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Forest reserve; long term (2)</td>
<td>Preserved C stock</td>
<td>High</td>
<td>Cultural and regulating ecosystem services</td>
</tr>
<tr>
<td></td>
<td>No sink No subst.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clearcutting; short term (3)</td>
<td>Loss in C stock</td>
<td>Decrease</td>
<td>Increased runoff and mobility of N sp (mainly temporary and local effects), potential leakage of mercury; acidification</td>
</tr>
<tr>
<td></td>
<td>No subst</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Clearcutting; long term (4)</td>
<td>Subst.</td>
<td>Comparatively low</td>
<td>Potential leakage of mercury; acidification</td>
</tr>
<tr>
<td>Selective cutting (5)</td>
<td>Subst.</td>
<td>Increase</td>
<td>Potential leakage of mercury; acidification</td>
</tr>
<tr>
<td>N fertilization (6)</td>
<td>Subst. Increased sink</td>
<td>Decrease</td>
<td>Eutrophication</td>
</tr>
<tr>
<td>Deciduous trees (7)</td>
<td>Several possibilities</td>
<td>Increase</td>
<td>Less acidification</td>
</tr>
<tr>
<td>Wetland restoration (8)</td>
<td>Increased sink</td>
<td>Increase</td>
<td>Flood and erosion control, denitrification, methane emissions, methylation of Hg</td>
</tr>
<tr>
<td></td>
<td>No subst.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Removal of harvest residues (9)</td>
<td>Subst.</td>
<td>Decrease</td>
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</tr>
<tr>
<td>NPK fertilization (10)</td>
<td>Subst. Increased sink</td>
<td>Decrease</td>
<td>Eutrophication, use of pesticides</td>
</tr>
<tr>
<td>Wetland restoration (11)</td>
<td>Increased sink</td>
<td>Increase</td>
<td>Flood and erosion control, denitrification, nutrient retention, methane emissions</td>
</tr>
<tr>
<td></td>
<td>Subst.</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Liming (12)</td>
<td>CO₂-emissions</td>
<td></td>
<td>Less acidification; improved soil structure</td>
</tr>
<tr>
<td>Ley production (13)</td>
<td>Increased sink</td>
<td>Increase</td>
<td>Less eutrophication</td>
</tr>
<tr>
<td>Energy forests (14)</td>
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<td>Positive impact if subst. for intensive meat production</td>
<td></td>
<td>Counteracts increased broad-leaved vegetation due to climatic changes Overgrazing sometimes occur</td>
</tr>
</tbody>
</table>
A few of the land use measures investigated are positive from climate change point of view as well as from a number of other perspectives. These measures include forest reservation (in the short term), wetland restoration, livestock production with ley (if compared to livestock production with only arable crops), and energy forests (if compared to agriculture). A switch to deciduous tree species may also fall into this category, although here there’s a lack of knowledge regarding productivity as well as emissions associated with many tree species. Similarly, certain kinds of selective cutting may be positive from many points of view, but again there are uncertainties with regard to actual emissions. Such (potential) win-win solutions are usually only possible on small areas compared to the area subject to, e.g., conventional forestry, but may be highly significant for the preservation of threatened biodiversity and a number of other ecosystem functions. A national land use strategy aiming for (environmental) win-win options only will however not be possible. Trade-offs between different environmental values will be necessary.

Many of the parameters discussed through the report depend on site specific characteristics. Occurrence of species and site conditions such as soil properties, geology, hydrology, climate, deposition vary from one place to the other. In addition to this, people have different preferences, both at the individual level and at the cultural level. All of this, on top of the scientific difficulty of saying what is “best” when it comes to trade-offs between e.g. climate change and biodiversity, makes it impossible to recommend a “best land management option” on a general level; it will vary from one place to the other and over time, and a variety of options will be needed. A variety of options can be seen as a means of safeguarding a variety of values and ecosystem services, meeting different needs and preferences of people, and as a way of precautionous risk spreading.

The issue is further complicated when social and economic aspects, in terms of cultural ecosystem services are added. Briefly, cultural ecosystem services are “The non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience, including, e.g., knowledge systems, social relations, and aesthetic values.” (Millennium Ecosystem Assessment 2003, p 58). As an example, productivity may be somewhat lower for selective cutting than for clear-cut forestry as well as for deciduous forests compared to spruce stands. On the other hand, selective cutting and deciduous forests may enable cultural, aesthetic and recreational values that production forestry misses. At the same time, it is more plausible that the selective cutting-forests and the deciduous forests enable cultural and recreational activities such as fishing, picking wild berries and hunting. In the case the economic value of these activities is limited (such as in many high-income countries), it is reasonable to include them in the cultural ecosystem services as they contribute to the high cultural, aesthetic, social and health values of a biodiverse landscape (Norling 2001). The cultural ecosystem services are most plausible difficult to replace (Lisberg Jensen 2008). In many cases, then, trade-offs seem to be unavoidable not
only between environmental aspects, but also between environmental aspects on the one hand and social and economical aspects on the other, especially if including the global situation in the reasoning (Dobson 2007). In these trade-offs, science can give advice, but the decisions remain political, and dependent of valuations and preferences.

Concerning preferences, there is e.g. a risk that many preferences that people have, are monotonous, short sighted, temporary or just unrealistic to an extent that will challenge environmental decision making, environmental policy and/or environmental ethics (Minteer, Corley & Manning 2004; Minteer & Miller 2011). Furthermore, there is an extensive discussion in environmental ethics about the importance of natural landscapes (Callicott 2001; Hettinger 2002). On the other hand, empirical studies show that naturalness is not crucial. On the contrary, cultivated landscapes obviously have the social, cultural, aesthetic and spiritual values that many people appreciate (Norling 2001).
Basis of the Synthesis

1. Forestry

1.1 Management versus unmanaged

We conclude that:
An unmanaged forest in average holds more carbon both in the standing biomass and in the soil, i.e. the carbon stock is larger in unmanaged than in managed forest ecosystems.

Harvesting and using forest biomass in the long run gives an emission reduction that is larger than the loss in sink, and hence, that managed forestry with use of the harvested biomass significantly decreases the emissions of greenhouse gases to the atmosphere.

Large, continuous areas of natural forest must be left untouched to maintain populations of forest specialists. However, if natural processes such as wild fires are not sufficient due to e.g. fragmentation, management measures for conservation are necessary. Heterogeneity in dead wood substrates, such as different degree of decay and, different sizes and types, increases biodiversity. Intensifying forestry will decrease availability and heterogeneity of dead wood and is thereby negative for biodiversity.

Introduction

Management refers here in general to measures undertaken in productive forestry, and in particular to the harvest and removal of biomass at thinnings and final harvests. Harvest strategies have a significant impact on several ecosystem services. It affects directly the biomass stock and the annual growth rate, and thereby also CO$_2$ fluxes in forest ecosystems and the soil carbon pool. In addition they also indirectly affect CO$_2$ fluxes from the soil due to their impact on soil climate and hydrology. Management practices may also affect biodiversity, soil acidity and run-off water properties.

Greenhouse gases

Removal of biomass has an instant but short-termed impact on the amount of standing biomass in a forest. Remaining trees and understory vegetation will soon fill the gap and compensate the losses. From this follows that individual stands that are unmanaged may be very different from those that are managed, e.g. in terms of standing volume. However, this is not the case in a long-term perspective (a full rotation or more) or in a landscape perspective where stands with all ages occur. Therefore, it is necessary to pay attention not to current conditions but to the mean conditions or accumulated production during an entire rotation cycle. Unthinned stands tend to give the largest mean standing biomass and, therefore, the largest biomass carbon stocks (Eriksson 2006). Unmanaged forests (without thinning) can
store more carbon than managed forests (Cooper 1983; Thornley & Cannell 2000; Maclaren 2000; Kirschbaum 2003). Long-term simulation studies have indicated that the total ecosystems C pools in unmanaged boreal forests are bigger than in managed forests (Bengtsson & Wikström, 1993; Karjalainen 1996). Also Svensson (pers. comm.) using the processed COUP-model shows that unmanaged forests have in average over the entire rotation cycle a higher biomass stock. Eriksson (2006) found no significant differences in biomass growth between thinning regimes for Norway spruce. Other studies have found minor differences in growth between thinning regimes (Hynynen & Kukkola 1989; Mielikäinen & Valkonen 1991; Eriksson & Karlsson 1997). In unthinned stands a substantial part of the growth will however die off through self-thinning.

Prolonged rotation results in a larger carbon stock in the biomass (Eriksson 2006; Liski et al. 2001; Kaipainen et al. 2004). Harmon et al. (2009) showed through simulation that the accumulation of carbon in living biomass in Douglas-fir/Western Hemlock forests, if major disturbances were excluded, increased for the first 200–300 years and reached a long-term steady-state store after 600–700 years of age.

As the production of above and below ground litter from trees may be considered as a function of the standing biomass (Ågren et al. 2006) it is suggested that the annual C input, as tree litter, to the soil decreases with thinning intensity, and a shortened rotation length. This is consistent with a simulation study by Eriksson et al. (2007) where management with a longer rotation and increased mean biomass C stock resulted in about 10% more soil organic carbon (SOC). Consequently an unmanaged forest will hold more SOC. The COUP model indicates higher soil organic content in unmanaged forest (Svensson, pers. comm.). However it should be stressed that stand data, only, might not reflect the input of C input from the whole forest ecosystems since the forest floor vegetation may provide a substantial amount of litter as showed by Berggren Kleja et al. (2008). They found that the field layer vegetation constituted 8-30% of the total litter input. Stendahl et al. (2010) showed that litter production from less dense stands with more field vegetation may offset the differences in tree-litter. Larger accumulation of carbon in soil due to prolonged rotation was found by Eriksson (2006) and Kaipainen et al. (2004). However, Liski et al. (2001) reported the opposite, finding that when the rotation length is shortened, the soil carbon stock increases.

Harvest and use of forest biomass may play an essential role to decrease greenhouse gas emissions through substitution of either fossil fuels or construction materials. However, unmanaged forestry with no biomass harvest will severely exclude this possibility. The substitution effect is large, and in some cases the decrease in C emission as CO₂ is larger than the content of the carbon in the biomass (Eriksson et al. 2007; Sathre et al. 2010). As the substitution effect is larger for construction materials than for biofuels, the effect on emission decrease is dependent on forest conditions such as tree species and tree dimensions. Larsson et al. (2009) calculated the average substitution
Land management meeting several environmental objectives

Biomass removal will enable more biomass to be produced and harvested, and the loss in emissions through substitution will grow by time. The net emission decrease through managed forests including substitution and sink potential may thus significantly exceed the net effect of unmanaged forestry including only the sink effect. This can be exemplified with data from a modelling experiment with the COUP model (Svensson, pers. comm.). After a period of 200 years since establishment the accumulated sink in biomass and soil amounted to 13200 g C per m² in a managed forest but 15400 g in an unmanaged forest. At the same time the total harvested biomass in the managed forest amounted to 11900 g C per m², equal to an emission-reduction potential of in average 11300 g C. During unmanaged forest there was no substitution possibility. Thus, the net effect amounted to 15400 g C per m² without management, but 24500 g C under management. The modelled annual net average effect of managed forestry under these conditions corresponds to 0.45 ton C per ha forest land (about 10 Mton C per year for Swedish productive forest land).

Biodiversity

Effects on threatened biodiversity are discussed in Appendix A for a number of specific land use-change scenarios (2.3: Stand continuity in forestry). In summary the impact of climate change may either increase or decrease the ambitions for long-term protection of the remaining c. 10 % of the forest landscape in which natural or semi-natural processes are still dominant. Protection of the remaining c. 1 million ha of known but unprotected forest value-cores for biodiversity is essential for halting the present loss of more demanding forest species and ecosystem functions connected to natural forest. Considerations to biodiversity at logging are not sufficient for decreasing the negative effects considerably and the loss of biodiversity is therefore more or less proportional to the loss of area of value-cores. Protection of production forest (forest that is formed by forestry rather than by natural processes) from logging is also necessary in order to increase patch area and connectivity and thereby the long-term loss caused by fragmentation.

One of the most important differences between managed and unmanaged forests is the scarcity of coarse woody debris in managed forests (Laiho & Prescott 2004). Andersson & Hytteborn (1991) reported that the average volume of decomposing wood is 73±65 m³/ha in unmanaged forests whereas it is only 11±2.8 m³/ha in managed forests. Natural forests or older stands can have values of several hundreds to thousand m³/ha of lying decomposing wood. The amounts of dead wood in Finnish forests are reduced by 90-98% compared to natural levels and also in other European countries dead wood is reduced (Verkerk et al. 2011 and ref. therein). The types of dead wood ranges from slash, logs, snags, low and high stumps, and vary in size-dimensions and degree of decay, tree species, age and whether it is standing or lying. Most of the logs in managed forests have a diameter smaller than 10cm whereas logs...
are much thicker in natural forests. Also in natural forests dead wood is present in more stages of decay than in managed forests (Andersson & Hytteborn 1991). Another important difference between managed and unmanaged forests is that unmanaged forests have a heterogeneous age structure and a greater genetic diversity and thereby greater resilience after environmental changes (Evans & Perschel 2009).

In natural forests dead wood results from mortality or natural disturbances such as wind felling, diseases, beaver activity, insect outbreaks and wild fires. For instance, wild fires generate large amounts of snags (Hegeman et al. 2010). In managed forests dead wood results mainly from harvest practices. Storm-felled stems are usually taken out from managed forests (Andersson & Hytteborn 1991).

Dead wood is an important resource for fungi, lichens, mosses, arthropods, mammals and birds. Natural forests harbour more saproxylic moss species than managed forests (Andersson & Hytteborn 1991). Many species of mammals and amphibians are related to older, more decayed and larger sized woody debris (Riffell et al. 2011) that is present in larger amounts in unmanaged forests. Birds use snags for nesting, perching and foraging (Riffell et al. 2011 in press and ref. therein). Coarse dead wood is used by reptiles as refugia, foraging substrate, basking and for mating activities. On the other hand, coarse dead wood may attract birds-of-prey and thereby affect some reptile species negatively (Riffell et al. 2011 in press and ref. therein), possible through competition for food. Amphibians use moist coarse dead wood as habitat. Removal or addition of coarse dead wood and/or snags had negative effects on the species richness of amphibians indicating that these animals may be sensitive to disturbance per se and may be restricted to unmanaged forests (Riffell et al. 2011 in press and ref. therein).

Managed forests may have a higher abundance and species diversity of Carabidae, Araneae and Formicidae than natural forests (Johnston & Holberton 2009) but some arthropod species are exclusively restricted to undisturbed forests (Halme & Niemelä 1993). Only a part of the saproxylic beetle community is able to inhabit old forests. Many saproxylic organisms need sun-exposed habitats and will benefit from forest management (Kaila et al.1997). Halme & Niemelä (1993) suggested that small and open forest fragments have higher plant biomass than large and closed forest areas and thereby greater amounts of prey items leading to increased carabid abundances and diversity. Johnston & Holberton (2009) suggested that creation of canopy gaps or thinning may increase abundance of ground-dwelling arthropods and thereby, increased numbers of ground-foraging birds. Different saproxylic beetle species are depending on stumps of different sizes and types (Jonsell et al. 2007). Lying dead wood such as logs are often more species rich than standing dead wood but there are organisms that are restricted to standing dead wood. Biodiversity will thus increase if different sources of dead wood are present (Verkerk et al. 2011 and ref. therein) and if different intensities of management are applied. Halme & Niemelä (1993) suggested that large, continuous and natural forest areas must be left untouched to maintain populations of forest specialists.
Water quantity and quality

The effects of management on water quantity and quality are essentially the same as those discussed in the next section on clearcutting vs selective harvesting.

Systems perspectives Forest reservation versus production forestry

Towards steady state after X years
Effects decreasing with time in italics

Fig. 1. Forest reservation vs production forestry: short time perspective

In all comparisons between different kinds of forest management, including forest reservation, the time perspective is decisive.

Fig. 1 illustrates the years following forest reservation, before the processes in the forest reserve has reached steady state (a number of hundred years), as compared to business as usual, i.e. production forestry. During these years, the carbon stock will increase in biomass as well as in soil, i.e., the forest will act as a carbon sink. Moreover, biodiversity and ecosystem functionality in the area will be maintained or increase (depending on what kind of forest is reserved). The area can also be expected to strongly increase its capacity with regard to water purification and water retention.

Positive effects: Maintenance or increase of biodiversity, capacity for purification and retention of water, cultural ecosystem services, and increase and maintenance of the carbon stock.

Negative effects: No biomass harvested (i.e. no substitution potential or storage of C in wood products).

Potential win-win effects: Biodiversity – carbon sink/stock – water quality. Not investigated: Effects on the nitrogen cycle. The reserve will most likely have a positive effect as no fertilization will take place and the leakage of N will decrease.

Identified knowledge gaps: Effects of isolation of conservation forests and loss of natural dynamics on old forest communities in a changing climate and a changing landscape.
Fig. 2 illustrates the situation when, after some hundred years, the forest has reached a mature stage characterized by steady state processes. The forest does not function as a carbon sink any more, but constitutes a substantial carbon stock. The area will contribute little to any kind of direct economic production, except tourism, but can on the other hand be expected to supply a number of ecosystem services, including water and climate regulation, water purification, pollination, seed dispersal and cultural services (having e.g. spiritual, recreational and scientific values).

Positive effects: Ecosystem services, including water and climate regulation, carbon stock, water purification, and cultural services.

Negative effects: No biomass harvested (i.e. no substitution effect)


Potential goal conflict: No substitution potential vs effects listed under win-wins.

Identified knowledge gaps: As for forest reservation (fig 4). C-dynamics, in particular the impact of natural disturbances.

Climate change and gaps of knowledge
Climate changes as changes in temperature may affect the mobility and thereby the dispersal patterns of insects, including saproxylic insects. Unmanaged forests are usually relatively small and at large distance from each other. Dispersal ability of species is important to ensure vigorous populations. Little is known about the mobility and dispersal patterns of saproxylic organisms and Andersson & Hytteborn (1991) reported that variation between species may be great.

1.2 Clearcutting versus selective cutting
We conclude that:
In a long-term perspective, or in a landscape perspective, there are no general major differences in mean forest production between selective cutting and
clearcutting. This means that there is no general long-term difference for the CO$_2$ sink in forest biomass.

There are no general differences in mean CO$_2$ uptake in trees as well as mean CO$_2$ emission from the soil leads to the conclusion that the sink strength of the forest ecosystem for the entire rotation cycle over about 100 years or more, or in a landscape perspective, is not dependent on the management form. However, for individual stands with different age-class distribution among trees there might be substantial differences.

In summary, the highest biodiversity in a managed landscape can be expected when a high variation in forestry methods is applied. Leave islands and different levels of tree retention will favour different organisms. Selective cutting is not necessarily a key-method for preserving biodiversity but will favour certain groups, e.g. shade and continuity depending species.

All management practices (and natural disturbances like e.g. storm falling) lead to increased mobility, and hence leakage, of a range of substances, whereof leakages of various N species are best documented. All findings above should however be regarded as potentially serious environmental problems. However, the processes involved are complex and other parameters, than intensity of forestry, contribute to the actual effect of any management practice. Important such parameters are local climate, soil structure, topography and atmospheric deposition.

Introduction

Clearcutting is since long a widely used logging practice, meaning the felling and removal of the entire standing crop of trees from a given tract of forest, normally followed by regeneration of a new even-aged forest. Clear-cut management includes one or several thinning operations prior to final cutting. Selective cutting represents a kind of uneven-aged forest management where no clearcuts are made, and refers to the cutting of single trees or small groups of trees (partial cutting) at a time, leading to a more or less continuous forest cover in space and time.

Greenhouse gases

The growth of a tree or the production in a stand is very much dependent on its age and follows a s-formed function. Thus it is obvious that the current production in individual stands with different management forms and age-class distributions may be very different. A young even-aged stand, e.g. a new plantation following clearcutting, has a much lower current production than an uneven-aged stand where many age-classes occur. Likewise, the current production in an older even-aged plantation might be higher than the production in an uneven-aged stand. The age impact disappears in a landscape perspective, because all age classes will be presented under clearcut management, but in different stands, whereas under selective cutting all age classes are occurring in one or the same stand. For this reason it is important to discuss the long-term mean production during 100 years or more, i.e. equal to the rotation length in forestry.
Cedergren (2008) made a comprehensive review on experiences and potential of clearcutting versus selective cutting with focus on mean forest production and diversity in Sweden. He lists some reports that show lower mean long-term forest production for selective cutting (e.g. Andreassen 1994, Elfving et al. 2006), but also some reports that show no major difference (e.g. Lundqvist 1989, 2005). Cedergren (2008) concludes in his review (2008) that the mean forest production with clearcut management with start point at the clearcut area is of the same size as forest production with selective cutting with start point at the existing stand. Also internationally there seems for boreal/temperal ecosystems to be no clear evidence for lower production with selective cutting than with clearcutting. Harmon et al. (2009) show that partial harvest may lead to higher average C stores in forest biomass than clearcut management, especially when the interval between harvests is short. Deal et al. (2002) studied the effects of partial cutting on conifer growth in southeast Alaska. Their analysis of the data did not detect any significant changes in stand growth due to the partial cuts. They concluded that silvicultural systems using partial cutting could provide a sustainable timber resource including more valuable spruce trees, while also maintaining stand structural diversity and old growth characteristics. Neither could Long & Shaw (2010), investigating stand structure and growth in ponderosa pine stands in western USA, show that stand growth is strongly influenced by structural diversity.

As pointed out by Cedergren (2008) site properties may however be decisive for the outcome of management forms, e.g. tree species composition and soil fertility. Selective cutting is likely to be more successful in spruce-dominated stands than in pine stands, and on fertile soils. Thus, site adapted selective cutting is a challenge.

Uneven-aged stands are needed for selective cutting. Loss in production and sink strength due to the conversion of an even-aged forest to an uneven-aged forest suitable for selective cutting might be high. Cedergren (2008) draws the conclusion that only about 1.8 Mha of Swedish forests (of a total of 23 Mha) are presently suitable for selective cutting. It should also be noted that the economic output under certain circumstances is higher for clearcutting than for selective cutting (Cedergren, 2008). However, selective cutting may enable better wood quality.

It has been suggested that clearcutting, in contrast to selective cutting, results in increased CO₂ losses from the soil to the atmosphere. The data support for such a statement originates from eddy-flux measurements of net-ecosystem exchange. Lindroth et al. (2009) and Magnani et al. (2007) show that a clear-cut area is a net source of CO₂ to the atmosphere and concluded that it takes 15% of the full rotation time until the new regenerated forest will have a positive CO₂ balance. However, Bjarnadottir et al. (2009) reported a strong sink for atmospheric CO₂ after twelve years after site preparation and afforestation, based on three years (2004-2006) of measurements of net ecosystem exchange (NEE) in a young Siberian larch plantation on Iceland. Magnani et al. (2007) show that after disturbance events, such as harvest, the forest is typ-
ically a net source of carbon over the first years, followed by a broad peak in C sequestration in maturing forests. Thus, the eddy flux measurements show that, following the loss in photosynthesis after removal of living biomass, the ecosystem may during a certain period, until new vegetation is established, turn into a net source of carbon dioxide to the atmosphere.

The high emission rates following clearcutting may suggest that cuttings enhance decomposition of litter and soil organic matter. Decomposition rates are affected by temperature, litter quality and site conditions. Clearcutting may increase soil temperature which in turn might result in increased microbial activity and higher decomposition losses (Houghton et al. 1987; Hyvönen et al. 2005). However, this effect may be overestimated. Warming experiments have indicated that the loss of soil C is a small and short lived effect, because only the labile soil C pool is exhausted (Jarvis & Linder 2000; Melillo et al. 2002; Berggren Kleja et al. 2008). A number of experiments have shown that mass loss in litter from clear-cuts actually is lower than in adjacent uncut stands (e.g., Johansson 1984; Blair & Crossly 1988, Yin et al. 1989, Prescott 1977; Prescott et al. 2000). The lower decomposition rate in clearcuts in these studies was attributed to drier conditions in the surface soil. Thus, temperature was not the major factor controlling rates of decomposition. The high net flux following clearcutting is for these reasons not caused by enhanced soil respiration rates, but entirely due to low photosynthetic rates. Consequently, there is no support for the hypothesis that clearcutting increases CO₂ emissions from the soil in relation to selective cutting.

For a complete analysis of the impact of cutting strategies it is essential to pay attention not only to the sink strength of the ecosystem but to carry out an environmental systems perspectives including the emissions during forest establishment, harvest and transports, and the substitution effect of forest biomass. Management affects the potential to use tree biomass for substitution of either fossil fuels or construction materials and thereby the greenhouse gas emissions to the atmosphere. As the substitution effect is larger for construction materials than for biofuels (Eriksson et al. 2007; Sathre et al. 2010) it is essential how management regimes affect the proportion of big dimensions of harvested trees. No environmental systems perspectives of clearcutting versus selective cutting were found and we stress that there is an important research gap in this field. It is likely, however, that emissions during harvest and transport will be higher with selective cutting due to less harvest per ha in average, and larger areas needed to retrieve a certain volume of wood.

**Biodiversity**

Effects on threatened biodiversity are discussed in Appendix A for a couple of land use-change scenarios (2.3: Stand continuity in forestry). In summary, many species groups and ecosystem functions of forest ecosystems depend on other ecological variables than stand continuity. Continuous cover forestry (CCF) *per se* is therefore not automatically favouring biodiversity connected
to dead wood and old-growth trees, and may even be negative compared to clearcutting forestry for species depending on light influx and high temperatures. The effects on biodiversity depend on which forest type that is used, which land use that is replaced by CCF, and on how CCF is performed. CCF may favour species groups depending on (a) small-scale continuity of trees, e.g. mycorrhizal fungi, and species with low dispersal capacity utilising substrates of young and medium-aged trees; (b) small-scale continuity of thin wood (as a result of more frequent logging); (c) species favoured by shade continuity, e.g. drought-sensitive cryptogams on ground, trees, and wood, e.g. in moist forest; (d) species favoured by stand continuity, e.g. forest tits and branch-living lichens. In general, species typically occurring in moist forest and other fire refugia can be expected to benefit from stand continuity in CCF. In the traditional agricultural landscape, forests were used for a multiple of purposes, including grazing. This often resulted in semi-open forest with long continuity of trees. CCF will probably show little resemblance with such forest use, mainly because of focus on high stand density. CCF as an alternative to protection in value-cores will be negative for threatened biodiversity, but some forms of CCF, e.g. coppicing, may be developed based on knowledge of traditional land use, that are highly favourable for biodiversity.

For both clearcutting and CCF biodiversity depends mainly on how the silvicultural practice is done, for example regarding size of clear-cuts, choice of tree species and considerations to biodiversity. The latter include leave islands and retention trees, which are elaborated below.

**LEAVE ISLANDS**
Leave islands, defined as uncut areas on clear-cuts, are thought to support biodiversity. Likens et al. (cit. in de Graaf & Roberts 2009) found that the species composition of the understory vegetation in forests differed from the pre-harvest vegetation 60-80 years after clear-cut indicating that leave islands is an important tool to preserve sensitive plant species that can not recover between harvest periods. Plant species that prefer closed canopy habitats or that are sensitive to disturbances declined whereas species that prefer open or disturbed habitats increased at clear-cut sites. In a short-term study De Graaf and Roberts (2009) showed that most plant species persist in leave islands 3 years after clearcutting.

**GREEN-TREE RETENTION**
Green-tree retention attempts to mimic natural disturbance by leaving some live trees in clear-cut areas. The canopy affects microclimate, soil conditions and quality and quantity of litter input. Herbaceous ground vegetation is depending on the forest density and the level of tree-retention has great influences on the herbaceous cover and plant species composition. Low levels of retention results in vegetation that resembles clear-cuts (Rosenvald & Lõhmus 2008) whereas vegetation in higher retention levels does not change compared to un-cut forests (Craig & MacDonald; 2009; Matveinen et al.)
Craig and MacDonald showed that changes in vegetation cover and species composition occurred between 10-20% retention for all herbaceous species. Plant species associated with 20, 50 and 75% retention levels resembled that of un-cut forests. Cover of grasses increases with harvest intensity due to increased light availability and disturbance. After eight years however, herbaceous cover and plant species richness was rather related to tree density than to retention treatments (Zenner et al. 2006; Craig & MacDonald 2009). Greater harvest intensity, thus lower level of tree-retention, has been shown to increase plant species richness but mostly through increase of generalist species. Plant species that are adapted to mature forests, such as shade and moisture depending mosses and ferns, are often vulnerable to intensive forestry (Burke et al. 2008). Tree retention affects also other organisms than plants. Rosenwald et al. (2008) found that ectomycorrhizal fungi increase in abundance and species richness when retention trees were left. Birds and salamanders may also be affected positively by retention trees. In timber-harvesting areas in North America communities of some threatened bird species may be completely depending on the presence of retention trees (Rosenwald et al. 2008). Furthermore, retention trees produce coarse woody debris in regenerating forest stands. Forest-interior saproxylic beetles need high levels of retention (Rosenwald et al. 2008). On the other hand abundance and species diversity of bees was found to be highest in areas with lowest retention level due to increased abundances of flowering herbs and nesting places (Romey et al. 2007).

Gap of knowledge
Rosenwald et al. (2008) found 214 studies on the effects of retention trees on biodiversity or other ecological parameters. Only 22 of the studies on biodiversity were carried out in the boreal and 3 in the temperate zone of northern Europe. For many species it is poorly investigated whether tree retention has short-term or long-term effects on plant- and insect communities. Also, the selection of retention trees according to age, size, shape of the crown, density and configuration of the trees has usually not been taken into account in biodiversity studies (Matveinen et al. 2006; Rosenwald et al. 2008). Matveinen et al. (2006) showed that the effects of different sized retention-tree-groups sometimes had quite unexpected effects on the abundance of spiders and carabidae. They suggested that the studied sized range might not have been relevant for these arthropods. The appropriate level for tree retention is different for different organisms. For instance, species that are depending on shade and moist habitats need a higher retention level than species that are favoured by sun-exposed habitats. Open habitats such as clear-cuts may have winter temperature beneath the supercooling point of for instance some spider species (Suggitt et al. 2011). It has not investigated if such species can survive in clear-cuts with high understore vegetation or if leave islands or retention trees are needed. The effects of fluctuations in summer and winter temperature due to forest management have not been investigated on the level of whole communities. Global
change in terms of altered precipitation regimes or changes in temperature may interact with responses of organisms to forestry managements, including tree-retention and leave islands. This has poorly been investigated.

**Systems perspectives clearcutting versus selective cutting**

![Diagram of clearcutting effects](image)

When clearcutting is executed for the first time in an area, i.e. the reference state is virgin forest the consequences are very large for the first years: emissions of carbon, leakage of nitrogen and mercury, and a substantial loss of biodiversity and cultural and regulating ecosystem services (see previous section) (Fig. 3). Repeated clearcuttings give similar effects, although of more or less diminishing scale. Biodiversity effects, in particular, are less prominent after repeated cuttings since much biodiversity disappeared after the first cutting. The various activities related to clearcutting require a certain input of resources (e.g. diesel) and causes emissions (CO₂, NOₓ,...). These most likely represent minor flows as compared to the carbon in biomass in this case, but should be included for a full picture.

**Positive effects**: Large potential for substitution of fossil fuels and construction materials. Habitat or dispersal-corridors for organisms that need sun-exposed habitats.

**Negative effects**: Large release of carbon (both through use of fuels and through leakage) and loss of biodiversity as well as cultural and regulating ecosystem services. Some acidification and leakage of nitrogen and mercury.

**Potential goal conflict**: Substitution potential vs negative effects listed above.

**Identified knowledge gaps**: Impact of clearcutting on soil C dynamics, cycling of mercury and weathering of minerals.
Selective cutting versus clearcutting

When comparing selective cutting to clearcutting (Fig. 4), the exact meaning of “selective” is decisive. “Selective” can mean anything from “small” clear-cuts (X x X m) to so-called continuous cover forestry. With regard to carbon emissions, no clear and consistent differences between selective cutting and clearcutting (irrespective of the definition of “selective”) have been documented. Less efficient use of machinery in the case of selective cutting may lead to increased resource use and emissions related to harvest as compared to clearcutting.

Leakage of mercury and nitrogen appear to decrease with selective cutting due to operations on a smaller scale, but how large this effect is, is likely to be determined by site-specific characteristics and the exact practices applied. The major difference between the two forms of management that have been documented concerns the effects on biodiversity. Most kinds of selective cutting have less deleterious effects on some aspects of biodiversity than clearcutting, although more demanding forest species suffer equally from selective as from clearcutting because of the deficit of old-growth trees, dead wood, mosaic exposure conditions, mixed stands, etc. The specific biodiversity response to harvest will be site-specific (e.g., due to different biodiversity) and depend on the silvicultural practice applied. Generally speaking, a variety in management forms will improve the chances of saving forest biodiversity, although much of the biodiversity and ecosystem functions rely on natural or traditional-anthropogenic disturbance regimes that are not met by any silvicultural practice so far developed.

Positive effects: Large potential for substitution of fossil fuels and construction materials while the impacts on biodiversity are less severe than those resulting from clearcutting.

Negative effects: Leakage of mercury and nitrogen

Potential win-win effects: Biomass production with maintenance of some ecosystem functions.
Identified knowledge gaps: A systems perspectives study, including the use of machinery, of selective cutting vs clearcutting. Effects of selective cutting on leakage of mercury and nitrogen. Moreover, the effects of direct carbon emissions after clearcutting and selective cutting appear to need further study.

Water quality and quantity

Tree harvesting affects the hydrology as well as the water chemistry in the affected area in a range of more or less interconnected ways. Below, the most important impacts will be described in brief.

The close connection between deforestation and increased runoff is well documented, although the variations between catchments are large, pointing to the importance of other parameters such as pedological conditions and climate (see e.g. Bosch & Hewlett 1982 and Andréassian 2004 for reviews). Large scale impacts on runoff are mitigated by felling on only a fraction of the catchment (ca 10% according to Brandt et al. 1988; Tetzlaff et al. 2007).

Several studies have shown that clearcutting in wet areas result in elevated water tables (Bliss & Comerford 2002; Huttunen et al. 2003; Pothier et al. 2003; Mäkitalo & Hyvönen 2004). An elevated water table leads to an increase of the volume of soil with anaerobic conditions, which influences the microbial processes related to emissions of N₂O and CH₄. Notably, the effects on the water table may be seasonal so that the water table after clearcutting is considerably higher as compared to the usual condition in the non-growing season, but only insignificantly higher (Hubbart et al. 2007) or even lower (Bliss & Comerford 2002) in the growing season. In dry areas, on the other hand, clearcutting results in lowered water tables as a result of interception loss (see e.g. Ganatsios et al. 2010).

Water chemistry is affected in several ways by tree harvesting and other forest management. Gravelle et al. (2009) showed how clearcutting as well as selective cutting resulted in significant increases of NO₃⁻ and NO₂⁻ concentrations in forest streams. Other studies on the effects of clearcutting report similar results with regard to increased nitrogen leakage over a period of one to 11 years after harvest (Akselsson et al. 2004; Rothe & Mellert 2004; McBroom et al. 2008; Hope 2009; Futter et al. 2010). Just as for the effects of tree harvesting on the water table, nitrogen leakages have been observed to show a seasonal pattern (Jost et al. in press). The overall impacts are however often regarded as small and clearcutting not a threat to water quality when carried out on a reasonable scale and with good management practices (Mannerkoski et al. 2005, McBroom et al. 2008 and Futter et al. 2010). In a larger perspective, the overall contribution of nitrate leakage from forestry (stem only harvest) was estimated to about 3% of the total Swedish nitrogen load to the Baltic by Futter et al. 2010, while Akselsson et al. (2004) estimated the contribution of nitrogen from clearcuts to vary between 1% in south Sweden (where agriculture is the dominating source of nitrogen) to 11% in the forested central part of southern Sweden. Notably, nitrogen leakages are larger from areas...
with high levels of soil nitrogen, which means that the problem increases with fertilization, and from northern to southern Sweden due to increasing levels of atmospheric deposition further south (Nordin et al. 2009).

Besides nitrogen leakage, leakage of base cations and subsequent acidification (Watmough et al. 2003; Baldigo et al. 2005; Akselsson et al. 2007; Fukushima & Tokuchi 2008), and leakage of aluminium (McHale et al. 2007) as well as mercury (Garcia et al. 2007; Kreutzweiser et al. 2008) are documented effects from tree harvesting. The loss of base cations and hence the acidifying effect is potentially counteracted by increased mineral weathering, but whether this will be enough for recovery of nutrient pools after tree harvesting is not clear from the studies performed due to large uncertainties in the assessment of weathering rates (Akselsson et al. 2007; Klaminder et al. 2011). As in the case of leakage of nitrogen after forest harvest operations, the effect on leakage of mercury appears to depend largely on scale and management practices. Swedish studies have shown comparatively small effects on leakage of mercury from afforested areas as compared to the results from North America (Bishop et al. 2009; Sørensen et al. 2009; Eklöf & Bishop 2010), and, according to Bishop et al. (2009) at the same level of magnitude as that from processes in naturally occurring wetlands (see chapter 2.5).

There are few studies available on the effects of selective cutting compared to clearcutting with regard to the parameters mentioned above, but Weis et al. (2001) report on less leakage of nitrogen and base cations after selective cutting.

1.3 N fertilization

We conclude that:
Fertilization with N gives a strong and immediate C sink in both soil and forest biomass, including removal of CO₂ from the atmosphere, particularly if the extra biomass production is harvested and used for substitution.

In summary, application of fertilizer will change species composition of plants and soil biota. Plant diversity will decrease due to fertilizing. Changes in plant and soil biota communities depends however on soil type, environmental conditions and fertilizer type and application dose and a conclusive statement on the effects of fertilizing on forest plants and soil animals can not be made. The effect of fertilization on birds and mammals, including big grazers, is poorly investigated.

Introduction
Fertilization of forests with nitrogen has been practiced in Sweden since the beginning of the 1960s. During the 1970s almost 200 000 ha was annually fertilized but presently, mainly for environmental reasons, only about 60 000 ha are annually fertilized. Traditionally 150 kg N per ha is applied one time as ammonium-nitrate in mean aged or old conifer stands.
Greenhouse gases
The fertilization increases the photosynthetic needle biomass and because of this the production increases by 25–50% during about 10 years. In practice this means an increased production by 10–20 m$^3$ during about 10 years (Fahlvik et al. 2009). Roswall et al. (2004) investigated the potential to increase forest production and draw the conclusion that fertilization is the only measure in forestry that gives an immediate increased production response. They report that an application of 150 kg N per ha gives an increased production by 13–18 m$^3$. The fertilization response depends on site and stand conditions and is e.g. higher for spruce than for pine. Fertilization may lead to unwanted environmental effects and for this reason it is important that fertilization is site adapted and carried out with consideration to environmental values and time for application. The Swedish Forest Agency has suggested guidelines for fertilization. The significance of fertilization in Sweden for increasing growth rate and uptake of CO$_2$ has recently been suggested by Fahlvik et al at 2009 and Larsson et al. 2009 in a Governmental commission. They emphasize the potential of stand-demand adapted fertilization (Swedish: BAG= BehovsAnpassad Gödsling), described as frequent addition of both macro- and micronutrients in amounts and composition that is based on needle analyses. This type of fertilization should start in young stands and is estimated to be able to increase production by 10 m$^3$ per ha and year. Specific site and stand conditions should be met for optimal results and minimized negative environmental consequences, and the available forest area for stand-demand adapted fertilization in young forests was estimated to 300 000 ha (Fahlvik et al. 2009).

Also Gode et al. (2010) in a synthesis report on bioenergy in Sweden conclude that fertilization can increase forest production, although it might also result in negative environmental implications such as nitrogen leaching and eutrophication in the sea. They suggest, however, that correctly performed fertilization should not lead to nitrogen leaching.

Hyvönen et al. (2008) analyzed literature data and data from novel studies in 15 long-lasting experiments in Sweden and Finland and draw the conclusion that addition of a cumulative amount of 600–1200 kg N ha$^{-1}$ resulted in a mean increase in tree C stock of 28 kg C per kg N added (“N-use efficiency”). This is consistent with an increased production of 10–20 m$^3$ per ha. They also found a mean increase in the soil of 12 kg C per kg N added. The increase in soil carbon stocks following fertilization was also confirmed by Fahlvik et al. (2009) in a literature review. Also Johnson & Curtis (2001) found in a meta-analyses of 26 North American studies that fertilization significantly increases carbon accumulation in the soil. The effect on soil carbon is believed to be due to partly increased litter production, partly decreased decomposition of organic matter in the soil (e.g. Fahlvik et al. 2009). Fertilization with N and NPK caused increased litter production, reduced long-term decomposition rates and increased C storage in the soil (Franklin et al. 2003).
Nitrogen fertilization might result in formation of dinitrogenoxide ($N_2O$) during denitrification and nitrification processes in the soil. This is particularly the case in soils where the C/N ratio is below 20 (Klemedtsson et al. 2005). Dinitrogenoxide is a greenhouse gas that is about 300 times stronger than CO$_2$. In a literature review Fahlvik et al. (2009) came to the conclusion that a maximum of 0.5 – 1.0% of added nitrogen to forest soils might be emitted as N$_2$O. The maximum nitrogen emission from the application of 1 kg nitrogen may for this reason be estimated to 10 g nitrogen or 15.7 g N$_2$O corresponding to 4.71 kg CO$_2$eq or 1.3 kg C. This is a substantially lower amount than the extra accumulation of C in trees and in soil. In conclusion only about 3% of the extra sink strength in trees and soil following fertilization is lost through extra N$_2$O emissions. This does not exclude that emissions might be much higher on soils with an exceptionally low C/N ratio, however it is not likely that such stands will be fertilized.

For a complete analysis of the impact of N fertilization it is essential to pay attention not only to the sink strength of the ecosystem but to carry out an environmental system analysis including the emissions during forest establishment, harvest and transports, and the substitution effect of forest biomass. Such a study was conducted by Eriksson et al. (2007) and showed that a strong net impact of fertilization in intensive forestry on the removal of CO$_2$ from the atmosphere.

**Biodiversity**

**PLANTS**

Fertilization in forests changes the species composition of vascular plants, bryophytes, fungi, and lichens (Bernes 1994; Högbom & Jacobson 2002; Zetterberg et al. 2006; Strengbom & Nordin 2008; Forsum 2008, Dahlberg et al. 2010). For example, broad-leaved grasses and plants like Rubus idaeus, Chamaenerion angustifolium may increase after fertilization. Broad-leaved plants may out-compete other plant species such as bilberry (Vaccinium myrtillus). Thus, fertilising may support plant species that are fast-growing whereas species that need a long recovery time, like lichens decrease after fertilization. Although the effects on understory vegetation are often rather large (e.g. Strengbom & Nordin 2008b), little is known about whether species are threatened by nitrogen fertilization in forest ecosystems. For mycorrhizal fungi, fertilization is mentioned as a threat in the latest red-list, especially on nutrient-poor soils (Dahlberg et al. 2010). Plant seeds that are adapted to germinate after fires may be trigged to germinate after fertilization (Bernes 1994). The seedlings have, in that case, to compete for light with trees and may die.

**BIRDS AND MAMMALS**

The effect of fertilization on birds and mammals is poorly investigated (Nohrstedt 2001). N-fertilized areas were avoided by reindeer for grazing (Nohrstedt 2001 and references therein).
SOIL BIOTA
The effects of forest fertilization on soil organisms depend on soil type, environmental conditions and fertilizer type and application dose. Application of fertilizer in forests can have positive, negative or neutral effects on soil microbial biomass and heterotrophic soil respiration (Forge & Simard 2000). Application of fertilizer may affect the quality and quantity of litter and thereby the substrate quality for soil organisms. Fertilization affects the species composition of lichens and mosses as well, and this may affect the abundance and species assemblages of soil fauna. Fertilization of coniferous soils may induce a shift from fungi to bacteria driven systems mainly due to negative effects on mycorrhizal fungi (Verhoeft & Brussaard 1990). In addition, Forge & Simard (2004) showed that fertilization of clear-cuts changed the composition of nematode species, possible due to changes in the proportion of bacterial and fungal biomass. Changes in the composition of microbes may affect fungivorous soil fauna. Lindberg & Persson (2004) showed that N-fertilization affects many mites and springtails species negatively. However, some species increased in abundance in fertilized plots compared to control plots and species richness of mites and springtails was thereby not significantly affected by fertilization. The number of species and total abundances of soil fauna showed interaction effects between fertilization and irrigation as the application of solid fertilizer had negative effects whereas fertilization in combination with irrigation had positive effects on soil fauna. Negative effects of fertilization are thought to be more serious under dry conditions (Minor & Norton 2004 and ref. therein; Lindberg & Persson 2004). The interaction effects of changed moisture conditions and application of fertilizer on soil fauna are poorly understood.

Climate change and application of fertilizer, lack of knowledge
Climate change and the increasing demand for renewable energy sources may require fast-growing tree species and application of fertilizer. The nutrition status of forest sites may decrease if slash and stumps are harvested (Olsson 1996). It was therefore suggested that application of fertilizer in young forests should increase (Gustafsson et al. 2009). The effects of fertilization on soil animals are poorly understood. It has been shown in choice experiments that arthropods often select food with an optimal combination of proteins and carbohydrates. It has also been shown that food choice depends on CO2-levels and N-content of litter (Lavy 1996 and ref. therein). It is poorly investigated how application of fertilizer affects the diet and performance of soil organisms and how these effects interact with climatic changes in the environment.

Water quality and quantity
Intensive forestry has been suggested and investigated as a means of decreasing the greenhouse gas emissions in Sweden. This section rests on two reports from that investigation: one with focus on environmental consequences
Intensive forestry has been shown to have a great potential for climate change abatement through increased uptake of carbon in forest biomass, but as it requires nitrogen fertilization, it will also lead to increased leakages of nitrogen to watersheds. Increased leakage of N mainly takes place upon fertilizer distribution and upon harvesting. The usual leakage of nitrogen observed upon tree felling (see previous section) also increases with fertilization (Forge & Simard 2000; Franklin et al. 2003; Futter et al. 2010). Forge & Simard (2000) suggested that microbial biomass is greater in soils of mature forests than in clear-cuts. The microbial biomass in mature forest soils can, thus, take up a larger amount of applied N than the microbial community in clear-cut soils. According to Nordin et al. (2009), the total leakage of nitrogen from intensive forestry in Sweden should be possible to limit to 850 tons per year, which would correspond to a 2% increase of the total yearly leakage of nitrogen from forest land.

Nitrogen fertilization is also known to contribute to acidification and leakage of base cations, but this may be avoided through utilization of a fertilizer substrate which is correctly balanced with regard to other nutrients, and distributed in several small doses rather than few big ones. The latter would also contribute to limiting the nitrogen leakages.

**Systems perspectives N fertilization in forestry**

*Ex of life cycle inventory (LCI) data for a) Production of 1 kg N fertilizer: 918 g CO₂, 4.6 g N₂O, 0.87 g CH₄ b) Transport by truck 1 km: 200-1000 g CO₂*

Fig. 5. N-fertilization vs no forest fertilization
Forest fertilization with nitrogen (Fig. 5) results in increased production and hence increased assimilation of carbon. The carbon sink strength of fertilized forests increase both in terms of carbon stored in biomass and in soil. The extra biomass produced can be used to substitute fossil fuel or construction material. Fertilization will lead to formation of some N$_2$O, but only about 3% of the sink strength gained through fertilization will be lost this way. The exact figure, however, depends on soil characteristics and the fertilization practices applied. For a full picture of the environmental impact from fertilization, resource use and emissions related to production, transport and spreading of nitrogen should be considered. Fertilization is likely to result in leakage of nitrogen compounds to water. Again, the exact amount is site and practice dependent. The effects on biodiversity are negative for plants and fungi but poorly investigated for most other species.

**Positive effects:** Increased biomass production.

**Negative effects:** Nitrogen emissions to air and water, and biodiversity, e.g. bryophytes, lichens and mycorrhizal fungi.

**Potential goal conflict:** Substitution potential and carbon sink – eutrophication; substitution potential and carbon sink – biodiversity.

**Identified knowledge gaps:** Effects on biodiversity.

### 1.4 Tree species

We conclude that:

Norway spruce is the tree species in Sweden that gives rise to the biggest C sink in biomass and soil. Scots pine gives a bigger sink in tree biomass than silver birch, but silver birch gives a bigger soil carbon sink than Scots pine.

**In summary the choice of tree species is important for biodiversity.** Many arbourious organisms that use living or dead trees are confined to single or taxonomic groups of tree species and to specific microclimatic conditions. The effects of choice of tree species on soil biota are very complex and poorly investigated.

**Introduction**

The choice of tree species in forestry, indigenous or exotics, is an important decision since it affects e.g. the biomass production quantity and quality, soil properties such as acidity and carbon content, biological diversity and run-off water properties. The choice of tree is mainly based on production economy, in turn the result of productivity potential in specific forest areas combined with market conditions. Soil- and region-specific growth rate (including damage from moose browsing etc) combined with quality-specific market price has so far been important drivers for the choice of tree species. Deciduous tree species have systematically been suppressed in production
forest in large parts of Sweden, in favour of coniferous trees including exotic species as Contorta pine. As a result, the forest landscape has been drained of deciduous trees which has drastically decreased the potential for preserving forest biodiversity (Axelsson et al. 2002), and most likely with profound impact also on the ecosystem function of forest ecosystems (cf. Bengtsson et al. 2000; Johnson et al. 2003). Deciduous trees are usually absent or sparse in older production stands, and in younger stands, before selective thinning, only birch is still rather common. Coniferous trees, especially Norway spruce, has also been favoured through extensive planting, both within and south of the spruce’s natural distribution area in Sweden (SOU 2007:60, Appendix B18). In southern Sweden, also pine has decreased drastically on more productive soils due to the forestry’s focus on spruce (Classon 2008). This is considered a serious negative change and support programs for pine production has been suggested (Bergquist 2009).

The tree composition in Swedish forests is also affected by grazing and browsing of moose, roe deer etc, which is strongly decreasing the proportion of surviving saplings and shoots of deciduous trees, and the economic output of pine cultivation. However, studies in Uppland has shown that the density of several deciduous tree species is considerable until the first and second thinning in spite of hard browsing pressure (Upplandsstiftelsen unpublished data).

Greenhouse gases

Of the total forest volume in Sweden Norway spruce (Picea abies) constitutes 42.2%, followed by Scots pine (Pinus silvestris) 38%, silver birch and European white birch (Betula pendula & Betula pubescens) 11.1%, trembling aspen (Populus tremuloides) 1.4%, European alder and grey alder (Alnus glutinosa & Alnus incana) 1.3% and oak (Quercus robur) 1.0% (Swedish Statistical Yearbook of Forestry 2010). Each one of all other tree species constitutes considerable less than 1%. The differences within the country are however big. In northernmost Sweden pine constitutes 57% and spruce 23%, whereas in southernmost Sweden spruce constitutes 45%, beech 17% and pine 11%. Each species has specific site demands such as on nutrient and water availability and climate. A high production therefore requires a site adapted choice of tree species. The production range between the most common species is spruce > pine > birch (Fahlvik et al. 2009). On average pine production was 60% of spruce production in southern Sweden and 95% in northern Sweden. Corresponding figures for birch and spruce were 40% and 60%. Birch was expected to produce around 60% of pine in northern Sweden and about 70% in southern Sweden (Fahlvik et al. 2009).

The most common exotic tree species are Lodgepole pine (Pinus contorta), European larch (Larix decidua), Japanese larch (Larix leptolepis) and their hybrid (Larix x eurolepis), and Sitka spruce (Picea sitchensis). Other species that are introduced in a very small scale are Douglas fir (Pseudotsuga menziesii) and white fir (Abies grandis). The production potential, site needs and sensitivity to various kinds of damages are reviewed by Fahlvik et al. (2009).
For afforestation or reforestation of abandoned nutrient-rich farm land the following species are suggested, ranged according to their production potential (Fahlvik et al. 2009): hybrid populus > hybrid aspen > grey alder > European alder > silver birch > European white birch.

The impact on greenhouse-gas emissions is strongly dependent on production rate, i.e. the higher production the bigger the sink is. However also tree quality is an important factor since it is decisive for the substitution potential. Highly productive species with short rotation will be important as bioenergy for substitution of fossil fuels but in a longer perspective slow growing species with long rotation may provide construction wood and outcompete the fast growing species. It is for this reason important to carry out environmental systems perspective for different species in which the substitution potential as well as emissions during establishment, management, harvest and transport is included.

Tree species affect the soil carbon content in different ways; 1) the amount of litter fall, 2) the amount and vertical distribution of root litter, litter quality, soil climate and understory vegetation. A review on how trees affect the soil C stock was made by Nordin et al. 2009 in a Governmental commission. They concluded that conifers, particularly spruce, give rise to much higher soil C stocks than deciduous trees with reference to Vesterdal et al. 2008. This study was however only able to show differences for the uppermost soil layer (O horizon) and it should be stressed that also the mineral soil should be accounted for. In a tree species comparison Oostra et al. (2006) showed that the amount of SOC in the O-horizon plus mineral soil 0–20 cm the order of species was spruce > elm > oak > beech > ash > hornbeam. The fact that the species with the highest SOC stock did not have the highest biomass production, e.g. the relation between spruce and beech, suggests that there are large differences in decomposition or leaching rates between species. Stendahl et al. (2010) showed that, based on field data, the national mean SOC stock was 9.2 kg per m² in spruce dominated stands and 5.7 kg per m² in pine dominated stands. They also showed that the differences were larger in northern Sweden than in southern Sweden. This was explained as an effect of much more litter supply from understory vegetation in comparatively open spruce stands in northern Sweden than from dense stands in southern Sweden. Another investigation in southern Sweden confirmed the differences between spruce and pine but also showed that birch takes an intermediate position (Hansson, pers. comm.).

**Biodiversity**

Effects on threatened biodiversity are discussed in Appendix A for a couple of land use-change scenarios (2.1: Choice of tree species in forestry). In summary, climate change may affect the forestry’s choice of tree species both in directions being positive for threatened biodiversity and in negative directions, e.g. increased use of exotic tree species. Most exotic tree species harbour only restricted proportions of the tree-depending biodiversity compared
to indigenous trees, and exotic tree species thus contribute little to the maintenance of ecosystem functions. On the contrary, some exotic tree species are locally invasive and may therefore have negative effects on native tree species. Also in stands formed by indigenous species the tree species composition is essential for the diversity of substrate for species, both regarding wood and tree substrates, and regarding soil conditions, litter quality, and light influx. However, considerable proportions of the forest biodiversity and ecosystem functions require specific qualities of the trees, for example concerning age, growth form, and light conditions. Therefore, forestry that aims at dense stands and short rotation may restrict the positive effects of increased tree species diversity.

In order to meet the demands for renewable energy sources, forestry may intensify. Increased production of spruce, Lodgepole pine (*Pinus contorta*) and hybrid aspen (*Populus tremula*) have been suggested to increase forest production (de Jong and Lönnberg 2010). De Jong & Lönnberg (2010) argued that the effects of conversion of agricultural land to production forest on biodiversity are unknown. Gustafsson et al. (2009) concluded that intensive cultivation of spruce had the greatest negative effects on biodiversity compared to cultivation of Lodgepole pine and hybrid aspen. Dispersal of Lodgepine seeds and hybrid aspen seeds and pollen is one of the risks named in this rapport. Effects on threatened biodiversity are discussed in Appendix A for a couple of land use-change scenarios (choice of tree species in production forest). Some general effects are discussed below.

In all types of forestry, based on clearfelling as well as on stand continuity, the choice of tree species is important for biodiversity. One reason for this is that large proportions of the forest species of e.g. insects, fungi, lichens, and bryophytes use tree structures as habitat and are more or less confined to single or taxonomic groups of tree species (Ehnström & Axelsson 2002; The Swedish Species Information Centre’s Data sheets on threatened species; www.artdata.slu.se). Tree structures are, e.g., leaves, wood and bark, and live tissue in bark/stem and the differences are due to, e.g., bark structure, pH in bark and trunk runoff, wood chemistry, and growth form. Some tree species are more species rich than others but because of the tree-specificity a species-rich tree species cannot replace a species-poorer tree species. For each tree species the value for biodiversity is highly depending on the growth conditions of the trees (in particular in terms of productivity, light conditions, and moisture conditions), and on the age of trees (Ehnström & Axelsson 2002). Thus, trees do not automatically provide habitats for species, but the proper conditions are needed. Most of the tree-connected biodiversity is demanding or is favoured by sun-exposed conditions and old-growth trees whereas forestry aims at dense stands with little exposure and at early harvest of trees. Therefore, increased variation in tree composition in production forest is not necessarily fully matched by increased variation in biodiversity.

Another reason is that different tree species create different habitat conditions in terms of soil conditions (e.g. the difference between Podzol and Cambisol), temperature, precipitation etc.
After tree harvest, stumps may be important habitats for many organisms that live on dead wood. Different stump species harbour different organisms. Lindhe & Lindelöw (2004) found that stumps of *Populus* harbored more red-listed saproxylic beetle species than stumps of other trees but we know very little about the diversity of other insect species on these tree stumps (Gustafsson et al. 2009). No studies have been carried out on plant- or insect diversity on stumps of Lodgepole pine (Gustafsson et al. 2009). Tree species affects also litter quality and quantity and thereby the abundance and species composition of soil fauna. For instance, Schuldt et al. (2008) found that the activity of ground-dwelling spiders was related with litter depth. Litter depth in their turn was dependent on the dominant tree species, with beech being composed slowly compared to ash or lime and thus resulting in a deeper litter layer. Moreover it has been shown that different litter quality may inhabit soil fauna with different stress tolerance. Mite species in coniferous forest soils are less tolerant to drought than species in beech forests (Taylor & Wolters 2005). It is poorly investigated how tree species affects stress tolerance of soil fauna, including stress due to climate changes.

**Lack of knowledge**

In south-eastern Sweden the risk for summer droughts will increase whereas other areas will get more precipitation. Also the annual temperature will increase (SMHI 2000). Climate change in itself might cause changed tree species distributions. Koca et al. (2006) predicted that the dominance of Norway spruce and to a lesser content Scots pine will reduce in the boreo-nemoral zones in Sweden during the 21 century in favour of deciduous broad-leaved trees. Increases in temperature and longer growing season allow oak, beech, silver birch and lime to expand northwards. Soil organisms in different tree species stands seemed to have different traits such as drought tolerance and they might respond different to changes in precipitation or temperature. It is poorly understood how changes in climate will affect abundance and species composition of soil fauna (Lindberg et al. 2003). Few, if any, studies have investigated how changes in climate will interact with the diversity and performance of soil animals in stands of different tree species.

It should be noted that there are few straightforward relationships between climate change and the choice of tree species by the forestry (cf. SOU 2007:60b). There are usually different ways of dealing with a certain climate change effect (see appendix). Two aspects in particular are subject to alternative approaches, in turn leading to different policies and forestry practices: (1) the trade-off between long-term economic security and higher but less secure profit; (2) whether we focus on (short-term) carbon release from old forest, carbon storage in young forest, or long-term potential for substitution. Since none of these approaches is obviously correct from all aspects, the choice of approach differs between, for example, organisations and society sectors, and are often the result of of opinion-based decisions. For example, in the Climate Policy of the Swedish Forest Agency (Skogsstyrelsen 2009c) a 100-yr per-
perspective is considered more important than the more short-term perspectives applied in the Kyoto Protocol and by the Swedish Government (an emission goal for 2020 and a vision for 2050). Potential for substitution in a 100-yr perspective is considered more important than the maintaining and increasing of stored carbon by the present forestry. Long-term production security is considered more important than short term growth in most cases.

**Systems perspectives Choice of tree species**

An increased use of deciduous trees and pine in forestry would favour biodiversity and also contribute less to acidification of soil and water. Production stands of both pine and most deciduous trees have often lower carbon stock in biomass and soil, but many deciduous trees may be harvested with shorter rotation periods than pine and spruce, which implies faster re-storage of released carbon. Deciduous trees can also produce biofuels based on frequent harvest of shoots; in Sweden this is currently used only in *Salix* plantations.

*Positive effects:* Increased proportions of deciduous trees and Scots pine favour biodiversity and deciduous trees contribute less to acidification of soil and water. Mixed stands have lower vulnerability to storms and large-scale pest outbreaks. Deciduous trees have potential for short-rotation biofuel or fibre production, i.e. high substitution potential with fast re-storing of carbon and potentially positive effects on biodiversity.
Negative effects: Reduced production (and hence substitution potential) of some types of production stands compared to monocultures of spruce or pine. Lower carbon stock in some cases.

Potential win-win effects: Biodiversity – acidification; substitution potential with rapid re-storing of carbon (shooting-based forestry) – biodiversity, acidification

Identified knowledge gaps: Characteristics of different deciduous species and production methods with regard to substitution potential as well as emissions during establishment, management, harvest and transport.

1.5 Forest draining

We conclude that:
In a short-term perspective forest production gives rise to a small net source of greenhouse gases to the atmosphere due to the drainage and oxidation of peat. But in a longer perspective, i.e. more than one rotation, the accumulated substitution effect results in a net uptake of greenhouse gases from the atmosphere.

Introduction
The area peat-covered land with forest production is significant in Sweden and amounts 5Mha which is about 10% of all land in Sweden, or 20% of Swedish forest land. When the peat thickness exceeds 30cm the land is categorized as peatland the area of which amounts to 64% of the area peat-covered land. Drainage of peat-covered land to enable forest or agricultural production and peat cutting has occurred since the middle of the 1800 century and today a total of 1.5–2 Mha peat-covered land is drained for the purpose of forest production (Fahlvik 2009). Forest draining has repeatedly been subject to governmental subsidies (Eliasson 2008), beginning already around 1840. As late as in the 1980s c. 15 000 hectares were drained, c. 30 000 hectares were subject to protective drainage after clearcutting, and c. 30 000 hectares subject to ditch restoration (Löfroth 1991).

Drainage is today regulated by 11 Kap. MB and the interpretation of this legislation is comprehensively treated in Naturvårdsverket (2009). In southern Sweden drainage is forbidden since 1994 and in northern Sweden permit is required. In 1994–1996 c. 400 hectares were drained annually, based on dispensation in the south (Swedish Statistical Yearbook of Forestry 2010). No national data are available thereafter. In Västerbotten in northern Sweden 5–10 permit applications were treated annually 1998–2002 (Naturvårdsverket 2007).

Protective temporary draining after clearcutting is allowed and since 1994 c. 50 000 hectares have been treated by protective draining (8 500 hectares in 2009, Swedish Statistical Yearbook of Forestry 2010). Although protective
draining is not allowed for wetland it is unclear how large areas that ecolog-ically constitute wetland habitats are affected.

The peat-covered forest land provides several ecosystem services and their management has implications for several of Sweden’s environmental quality goals.

**Greenhouse gases**
The prerequisite for forest production is drainage, but this also means oxidation of peat and emission of CO$_2$, and at certain conditions also emissions of N$_2$O and CH$_4$. Today we are facing a management problem; should drainage be sustained to maintain high forest production, or should the wetland be restored through closed ditches in order to improve biological diversity.

Emissions of CO$_2$ due to the oxidation of peat (heterotrophic respiration) vary according to von Arnold (2004) between 70 and 303 g C per m$^2$ and year depending on fertility. For N$_2$O, emissions amounted to 0.5, 0.1 and 0.02 g N$_2$O per m$^2$ and year for high, moderate and low fertility, respectively (Klemmedtsson et al. 2004). This corresponds to 40, 8 and 2 g Ceq (N$_2$O is 296 times as effective a greenhouse gas as CO$_2$). The emissions of CH$_4$ are low and negligible on drained peat-covered land. The total emissions, i.e. CO$_2$ plus N$_2$O, is in average 263, 197 and 127 g Ceq per m$^2$ and year for rich, moderate and low fertility, respectively.

The average uptake of CO$_2$ in biomass during the entire forest rotation through photosynthesis varies between less than 70 to more than 290 g C per m$^2$ and year. These values are somewhat lower than the emissions which suggest that the forest ecosystems on drained peatland in most cases are small greenhouse-gas sources. However, there are also several cases where there is a net ecosystem basis is an increased uptake of CO$_2$ following drainage and forest plantation (Minkkinen et al., 2009; Maljanen et al., 2009).

During a forest rotation the uptake of CO$_2$ almost balances the emission of greenhouse gases from the peat. This is also the case if the substitution effect is accounted for. Larsson et al. (2009) calculated the average substitution effect to 600–800 kg CO$_2$ per m$^3$. This is approx. equal to 0.8 – 1.1 kg C per kg C in biomass. During two or more rotations the accumulated substitution potential will exceed emissions from the peat.

**Biodiversity**
Effects on threatened biodiversity are discussed in Appendix A for wetland draining in the forest landscape. In summary, climate change may change current draining practices leading to either increased or decreased forest draining, depending on which priority that is given to different effects. Threatened biodiversity is affected negatively by all draining of wetland, including renewed draining of formerly but unsuccessfully drained areas which often still have high wetland biodiversity. The magnitude of the negative effect depends mainly on the species richness and ecosystem functionality of the drained area. Also the recovery of biodiversity and wetland ecosystems following ceased
ditch maintenance or wetland restoration varies in a similar way. Biodiversity is affected by draining through both hydrological changes and by the forestry that follows after draining. Hydrological systems constitute structural networks in the forest landscape which may provide connectivity corridors and stepping stones. Drainage may therefore also increase fragmentation.

Wetland habitats are characteristic for the Fennoscandian forest landscape and range ecologically from slightly moist and temporarily wet forest to bogs and fens that lack forest because of the hydrologic conditions. The Swedish flora and fauna is rich in species specific to wetland habitats, although rather few to one single habitat. The wetland species depend on specific combinations of soil conditions, air moisture, habitat structures etc. that are generated by the water. Wetlands also indirectly influence the habitats for species, for example by affecting tree species composition (e.g. alder, ash, and willow), the dynamics of forest stands, and the local climate. This implies that hydrology alone is not forming wetland habitats but natural processes in, for example, the tree dynamics are required as well.

The wetland biodiversity has declined due to drainage in both forest and the agricultural landscape. Hydrology often connects these two land use sectors. For example, large area of forest wetland was drained when lowland lakes and streams were regulated for agriculture in the 19th and early 20th centuries. The drainage is reflected in the red-list; for example, c. 40% of the red-listed birds, 15% of the beetles, and 13% of the bryophytes occur in wetland habitats (Gärdenfors 2010). The proportion of drained wetland varies between regions (Strömgren 2006) as well as between habitats. Wetlands close to agricultural areas, in lowland areas, and wetland habitats that are easily transformed to production forest, have in particular declined. On average, c. 15 % of the forested wetland area has been drained (Hånell 1990). Of the remaining wetland c. 80% of the objects are affected by draining (Gunnarsson & Löfroth 2009).

In the old agricultural landscape wetland comprised a resource and a variety of wetland habitats were mown for winter fodder, also in the forest landscape. Ceased traditional land use causes habitat changes that threaten biodiversity. This implies that hydrology alone is not forming wetland habitats but anthropogenic processes in related to traditional agriculture are required as well.

Since 1850 c. 1.5 Mha of wetland have been drained for forestry, of which c. 300 000 hectares are still considered too wet for production (Strömgren 2006). Also successfully drained wetland often return to wetland as the organic soils break down and the ground level thereby sink closer to the groundwater level. Also reduced efficiency of ditches contributes to the back-succession to wetland. This creates a need for restoring the drainage systems and cleaning of ditches in large areas of more or less wet drained areas. Ditch maintenance is allowed in order to maintain, but not increase drainage effects but as for protective draining ditch maintenance probably drains consider-
able areas that are ecologically wetland habitats (Naturvårdsverket 2007). In the province of Gästrikland c. 0.5–1 km of ditches are cleaned annually in the forest landscape, in Norrbotten coastal area c. 3 km, in Västmanland c. 10 km and in Kalmar c. 60 km. It is not likely that biodiversity considerations restrict the ditch maintenance significantly. For example, on the property of Sveaskog 16 % (c. 50 000 ha) of the formerly drained wetland area are considered in the need of ditch maintenance (Hägglund 2009). Only c. 6 500 ha are considered to be less suitable for drainage for environmental reasons (here related to the FSC, Hägglund 2009).

**Water quality and quantity**
Altered drainage of forest land will affect hydrology and water chemistry in the catchment. Less drainage will increase the capacity of the area to function as a water magazine as the average water table level will increase. Moreover, the water table level and the water flow fluctuations over the year will even out which leads to less erosion and flooding (see e.g. Ghermandi et al. 2010 for a review of wetland values and services). Effects on water chemistry are caused primarily by the fact that less drainage will cause larger and more stable zones of anoxic layers in the soil profile, and an increased rate of precipitation as the speed of water flow slows down. This will, among other effects, lead to increased nutrient retention (primarily N and P), decreased fluxes of DOC, and hence decreased eutrophication in the catchment. For more details on these aspects of wetlands, see chapter 3.5 on Wetlands in the agricultural landscape.

**Systems perspectives Decreased drainage in forestry**

![Diagram](image)

**Fig. 7. Forest wetland restoration vs continued drainage**
In the temperate and boreal forest landscape, a negative effect of wetlands is increased methylation of mercury (St.Louis et al. 1994; Selvendiran et al. 2008), primarily by sulphur reducing bacteria. In boreal forests, the highest methyl mercury production appears to take place in wetlands with intermediate nutrient status (Tjerngren et al. 2010). There is hence a potential goal conflict between the restoration of wetlands for nutrient retention and flood control on the one hand, and reduction of mercury in the aquatic food chains on the other.

Drainage is a prerequisite for productive forestry, but on peat land drainage implies oxidation of peat and emissions of CO₂, and at fertile sites also N₂O. Without drainage, anaerobic conditions lead to decreased mineralization of carbon compounds, and hence accumulation of carbon (peat formation) and decreased emissions of CO₂ and dissolved organic carbon (DOC). The higher water table also leads to increased denitrification, but not necessarily to increased emissions of N₂O, due to the greater reduction of N₂O to N₂. On the other hand, anaerobic conditions also lead to increased production of methane and methylation (and demethylation) of mercury. Moreover, no biomass will be harvested and hence there is no substitution potential. In a short time perspective, wetland restoration results in a lower climate change impact, and furthermore contribute strongly to biodiversity, water purification and water regulation. In a long term perspective, loss of harvested biomass, and hence lost substitution potential, may be negative.

**Positive effects:** Biodiversity, water purification, water regulation, nutrient retention and nitrogen circulation.

**Negative effects:** No harvest of biomass (wood or peat) that can give a substitution potential; methane production and methylation of mercury leading to increased levels of bio available mercury

**Potential win-win effects:** Biodiversity – water quality – water regulation – carbon sink and stock.

**Potential goal conflict:** Substitution potential – biodiversity; substitution potential – eutrophication; substitution potential – water regulation; substitution potential – carbon stock; reduced levels of MeHg – biodiversity; reduced levels of MeHg – eutrophication; reduced levels of MeHg – water regulation; reduced levels of MeHg – carbon stock.

**Identified knowledge gaps:** Uncertainties regarding emission factors; net emissions of greenhouse gases, based on environmental system analyses, with consideration to various site conditions.

### 1.6. Removal of harvest residues

We conclude that:

**Removal of harvest residues such as branches and tops most likely induced a small decrease in soil organic carbon content, and hence, is a source of CO₂.**

On the contrary there is no support for decreased soil organic carbon due to extraction and removal of stumps, particularly if this happens on an area
that anyhow is scarified. Environmental systems perspectives with substitution effects included shows that residue removal and use substantially reduces emissions to the atmosphere.

In summary, slash and stumps are important sources of dead wood in the managed forest landscape. Also, slash and stumps regulate the micro-environment and provide organisms with hiding and nest places. Although most saproxylic species disappear already at logging, the remaining species largely depend on sun-exposed slash and stumps. Removal of stumps and slash may affect diversity of birds, mammals and other animals negatively.

**Greenhouse gases**
Removal of tops, needles, leaves, branches, stumps and roots may result in decreasing C supplies to the soil in a rate that depends on what type and how much residues are removed. According to a review by Egnell (2007) modeling showed that removal of branches and tops (70 % of potentially available) reduced the amount of residues that was returned to the soil by 40%. If in addition also stumps were harvested (70 % of potentially available) the returned amount of residues was reduced by 70%. If residues are left on/ in the ground most of it would decompose almost entirely within one rotation. Consequently, in the long run or in a landscape perspective (with all age classes) the effect of removal on soil C stocks would be small, but in individual stands and with a short-term perspective the impact might be significant.

Stumps, coarse and large roots contain approximately 30 ton C/hectare (Palviainen 2005). Removal of stumps will thus result in decreases in the amount of dead organic carbon in the forest soil. In addition, stumps have 20 to 30 times higher C/N ratio than fungi. This implies that fungi growing on stumps need to take up N from the soil adjacent to stumps, thereby immobilizing N from an area around the stump. Stump harvesting may therefore lead to N leakage from the soil. However, Lenoir et al. (2010) showed that C-storage and N-pools were not affected by stump harvest nor was N-mineralisation. C mineralisation, however, was slightly higher in control sites. De Jong & Lönnberg (2010) concluded that decomposition and mineralisation processes will hardly be affected by stump harvest.

Ågren & Hyvönen (2003) estimated, based on forest inventory data and modeling, that removal of needles, branches and tops would decrease soil C stocks with 10% over a 150 year period compared to a reference scenario without residue removal. There were relatively small changes in the accumulation of new soil carbon when the needles were left on-site compared with when they were removed (e.g. by Bengtsson & Wikström 1993; Ågren & Hyvönen 2003). Leaving the needles on-site has a negligible effect on the soil carbon stock because the removed needles only represent a small proportion of the total amount of residues produced during the course of a rotation (Ågren & Hyvönen 2003). Lundblad (2009) in a Governmental commission concluded that if removal of branch and top residues increases by 100%, then the amount of soil carbon would decline by 2.5%.
Another possibility of removal of harvest residues is reduced long-term supplies of litter due to nutrient removal and loss in biomass production, and hence, less annual litter supplies to the soil. However, Wall (2009) found that, out of 58 scientific reports with studies from 227 experiments at different places in the world, as much as a bit more than 80% did not show any significant changes in growth rate. In only 13% of the studies a loss in growth was detected. This result is in accordance with experiences in the Nordic countries with tendencies to lower production, however not statistically verified (de Jong & Lönnberg 2010).

The short-term effects of forest clearing and whole-tree harvesting were studied in the northern hardwood forest ecosystem at the Hubbard Brook Experimental Forest in central New Hampshire, USA. There was an immediate increase in soil C content, but eight years after harvest in the Hubbard Brook experiment the soil C pool in the O horizon had decreased significantly (Johnson et al. 1995). Also in the mineral soil the C content tended to decrease.

Extraction and removal of stumps may affect 1) the amount of organic matter, 2) decomposition rates of litter and old organic matter, and 3) stand establishment, growth rate and litter supplies. Johansson (1994) investigated decomposition effects of site preparation by analyzing the mass loss in litter-bag experiments with pine and spruce needles during incubation in a boreal forest soil during three years. She found that the accumulated effect during three years for mounding, ploughing and disk trenching was an increased mass loss by about 30%. This suggests that mixing of soil material strongly promotes decomposition rates, and, hence, results in carbon losses. De Jong & Lönnberg (2010) conclude from ongoing stump-extraction experiments in Sweden that stump extraction and removal seems to have little or at least no short-term impact on CO₂ emissions in comparison with scarified areas.

Experiences from soil scarification experiments may be used to assess the impact of stump removal. Örlander et al. (1996) studied the long-term effects of spade inversion or ploughing on the soil carbon content 40 – 70 years after the site preparation. They found for two sites that the carbon content was reduced by 6–21% to 100cm depth, and for one site by 41% to 25cm depth. They also recorded increased long-term growth. The results show that the increased biomass production did not, with the long-term perspective, increase the supplies of carbon to the soil through litter enough to compensate for the increased decomposition losses. Nordborg (2001) came to contrasting results. He investigated the effects of intensive site preparation on carbon stocks, plants survival and nitrogen leaching. Intensive site preparation was defined as deep soil cultivation to a depth of more than 40 cm in 50% or more of the regeneration area. His results did not support the hypothesis that the loss of C from the soil increases with increased intensity of the disturbance. One possible explanation to the contrasting results could be that soil organic matter was buried deeper into an environment less favourable to microbial activ-
ity. Another explanation might be chemical stabilization at oxide surfaces in material from spodic horizons.

For a complete analysis of the impact of residue harvest it is essential to pay attention not only to the sink/source processes but to carry out an environmental systems perspectives including the emissions during e.g. harvest and transports, and the substitution effect of forest biomass. Such a study was conducted by Holmgren et al. (2007) and showed a strong net impact of the use of forest residues on the removal of CO$_2$ from the atmosphere.

**Biodiversity**

Effects on threatened biodiversity are discussed in Appendix A for some land use-change scenarios (2.4: Considerations to biodiversity in forestry). In summary, the effects on threatened biodiversity of use of slash are very restricted because most of the more demanding species are lost at logging. There are however important biogeographic and habitat exceptions, in particular regarding slash of broad-leaved deciduous trees. Systematic use of logging residuals (LR), in particular stumps, may reduce the benefits of general considerations to biodiversity in forestry.

The increasing demand for renewable energy sources, whole-tree harvest and stump harvest is becoming more common (Bengtsson et al. 1997) and may decrease the dead wood supply in managed forests (Caruso et al. 2008). Information on the quantities of slash- and stump that is removed is poor (De Jong & Lönnberg 2010; B. Olsson pers. comm.). Around 65% of the total slash volume was removed on 23 clear-cuts in eastern central Sweden (Rudolphi & Gustafsson 2005). Approximately 40,000 hectare is subjected to slash removal yearly. This equals 1% of the area of production forests (Naturvårdsverket 2005; Bengt Olsson, SLU pers comm.). Slash removal is more common in southern than in northern Sweden (Naturvårdsverket 2005, Skogsstyrelsen web page www.skogsstyrelsen.se/Myndigheten/Statistik/Amnesomraden/Trädbransle/Tabeller--figurer/). Removal of slash and stumps from forest stands leads to decreases in substrate for soil organisms and extreme changes in microclimate conditions (Sohlenius 1996). Slash (i.e. branches, twigs and needles) and stumps are the most important sources of dead wood in production forests and they are an important substrate for many wood-living organisms, including fungi, lichens, mosses and arthropods. Increasing forest production during the last 100 years in Sweden has caused increasing amounts of slash. Production forests produce nowadays 10 million m$^3$ slash per year due to thinning, clearcutting and by natural disturbances like wind. On the other hand, the amounts slash from deciduous trees in the Swedish forests is decreasing. Slash and stumps may be important sources of dead wood in these forests (De Jong & Lönberg 2010) and may support many wood-living organisms (Lindhe & Lindelöw 2004; Naturvårdsverket 2006; De Jong & Lönberg 2010). However, thin woody residuals can not replace coarse wood stumps and logs because many red-listed wood-living organisms are depending exclusively on coarse wood (De Jong & Lönberg
The microclimatic conditions on clear-cuts will be changed if slash and stumps are removed. Soils on clear-cuts without slash will be warmer and dryer than clear-cuts with slash. In addition, stump harvesting also destroys most of the remaining dead wood such as logs (Hjältén et al. 2010). Thus, slash and stump harvest may have negative effects on the wood-living flora and fauna.

DEAD-WOOD HETEROGENEITY
Species composition of lichens (Caruso et al. 2008) and saproxylic beetles (Hjältén et al. 2010 and ref. therein) differs between slash and low stumps, thus, a variety of dead-wood substrates including logs, low and high stumps and slash, is necessary to maintain biodiversity in production forests. On the other hand many organisms that use slash as a substrate or as a hiding place are often generalists and not specific depending on slash. They can live under or on stones, at the base of trees and in the soil. About 20–30% of all wood-living organisms are depending on dead wood and many of these organisms are red-listed. Red-listed species are usually using coarse dead wood such as stumps as substrate (De Jong & Lönnberg 2010 and ref. therein). Thus, thin harvest residues generally harbour only few demanding and red-listed species (Dahlberg & Stokland 2004; Junnien et al. 2006; Jonsell et al. 2007; Caruso et al. 2008), even though thin wood dimensions on clearcuttings are somewhat more species-rich than in forests (Jonsell et al. 2007; Jonsell 2008). Therefore, removal of harvest residues can be expected to mainly affect more generalist species on thin wood. There are however important exceptions for certain tree species in certain regions, e.g. aspen in Uppland and oak and other broad-leaved deciduous trees in eastern Småland (Jonsell et al. 2007; Hedin et al. 2008; Cederberg et al. 2001; De Jong & Lönnberg 2010). The importance of harvest residuals for the diversity of wood-living Coleoptera depends on the diversity of the coleopteran fauna in the surrounding landscape (Naturvårdsverket 2006). Jonsell (2005) showed that the diversity and the number of red-listed saproxylic beetles tended to be greater on oak- and aspen stumps in landscapes that have a rich coleopteran fauna compared to landscapes that are impoverished in beetle diversity. For stumps of birch and spruce tree such relation for red-listed species was not found whereas for the beetle diversity on spruce stumps a contradicted result was found: the total number of beetle species was greater on spruce stumps in landscapes that are impoverished in beetle diversity. For some rather generalist species on wood and litter, e.g. species of bryophytes, removal of residues has been shown to be negative because of reduced substrate abundance, reduced protection against dry-out on clearcuttings, and increased disturbance frequency (Åström 2006; Gustafsson 2004). Considering that almost all of a forest’s biomass is removed at clearcutting, further substrate reduction through removal of harvest residues may even break the last remaining substrate continuity in the production forest. The piles of extracted forest fuel are attractive for wood-living insects, especially if the cut area is cleaned of dead wood, and often function as eco-
logical traps (Jonsell & Hedin 2009; Hedin et al. 2008). When extracting residues from the clearcutting also other wood is frequently collected, including wood left at logging as biodiversity considerations, “consideration trees”, that have fallen after the logging (Andersson 2000; Gustafsson 200; Rudolphi & Gustafsson 2005).

SLASH HARVEST EFFECTS ON PLANTS
Slash is an important habitat for organisms that are sensitive for desiccation such as some moss species (de Jong & Lönnberg 2010 and ref. therein). Effects of slash harvest on the abundance of mosses and dry sensitive plants are greater on southern slopes, in the centre of clear-cuts and in dry habitats than on northern slopes or near forest edges. Some plant species such as Rubus idaeus may decrease on clear-cuts whereas plants that need sun-exposed habitats like Calluna vulgaris may increase. De Jong & Lönnberg (2010) suggest that changes in vegetation due to slash harvest are mainly caused by microclimatic changes whereas decreases in nutrient content after slash removal have less effect on plant species composition. Slash from deciduous trees, especially oak is an important substrate for wood-living fungi (Norden et al. 2004) whereas many fungi species are more common on spruce stumps than on slash from spruce.

SLASH HARVEST EFFECTS ON WOOD-LIVING INSECTS
Many wood-living insect species may be attracted to and may lay eggs in residue piles. The development from egg to imagines takes 2 till 3 years for some insect species. This means that the insects are not able to complete their lifecycle before the slash is taken to the energy plans. Residue piles may therefore function as so called ecological traps for many insects, including red-listed insect species (Mats Jonsell, SLU pers comm.; de Jong & Lönnberg 2010 and ref. therein).

RESIDUE HARVEST EFFECTS ON SOIL BIOTA
Harvest of residues and stumps will decrease the amounts of dead wood and thereby decrease the amounts of substrate available for wood-living organisms. Slash is an important habitat for soil-living arthropods (de Jong & Lönnberg 2010 and ref. therein). Bååth (1980), Lundgren (1982) Sohlenius (1982) and Lundkvist (1983) showed that fungi, bacteria, nematodes and potworms, were negatively affected by the removal of residues. Bengtsson et al. (1998) showed that residue harvest had negative effects on the abundances of gamasid mites, spiders, beetle larvae. The effects of residue harvest on microbi-detrivorous fly and mosquito larvae and fungivorous springtails differed between sites. The removal had no effect at a more productive spruce site whereas the abundances of fly and mosquito and springtales decreased after residue harvest in a pine site. Bird & Chartarpaul (1986) showed that the abundances of mites and springtales were lower in plots were residues were removed compared to plots that were stem-only harvested. Persson
et al. (2010) showed that removal of residues in general result in long-term decreases in abundances of almost all soil fauna groups, including springtales and mites, soil-living beetle, fly and mosquito larvae but no changes in functional groups of soil fauna were detected. The effect of removal of residues was most pronounced in the first twenty years after clearcutting but diminished after 30 years. Invertebrates that prefer open sun-exposed habitats such as spiders may be favored by slash removal (Gunnarsson & Nitterus 2004 cit in Naturvårdsverket 2006).

STUMP HARVEST
Dead wood, including stumps, is essential for biodiversity in forests and is often investigated in conservation surveys (Norden et al. 2004) or in pest management studies (Kelsey & Joseph 1999). Stumps are the main source of dead wood in production forests and are important for biodiversity whereas dead wood present in natural forest reserves are of limited value for many species. Few nature reserves are subjected to wild fires and other disturbances and the canopies become closed. Sun-exposed habitats are relative scarce in these forests and species that are favoured by shade take over (Lindhe & Lindelöw 2004). Stumps on clear-cuts benefit insect and plant species that need sun-exposed habitats. High stumps in clear-cuts harbour many common and red-listed insect species that are not found in closed forests (Lindhe & Lindelöw 2004; Jonsell et al. 2005). Low stumps represent ca 80% of the dead wood on clear-cuts (Caruso et al. 2008) and are for that reason an important substrate for many organisms on clear-cuts. Low stumps are, for instance, used by saproxylic insects such as beetles and parasitic wasps (Abrahamsson & Lindblad 2006).

EFFECTS OF STUMP HARVEST ON PLANTS
Stumps are important as substrate for mosses and lichens (Caruso et al. 2008) and plants that are sensitive for desiccation may decrease on stump-harvested sites. It has been shown that de production of blueberry (Vaccinium myrtillus) was 60–70% lower in stump-harvested sites compared to control sites 22–28 years after harvest (de Jong & Lönnberg 2010) but this was mainly due to soil damages caused by heavy vehicles in the stump-harvested sites (Tryggve Persson, SLU pers. comm.).

EFFECTS OF STUMP HARVEST ON SOIL BIOTA
Stumps are suitable habitats for soil fauna that is involved in decomposition processes. For instance, springtails (Setälä & Marshall 1994), millipedes (Persson et al. in prep) and oribatid mites (Astrid Taylor, SLU, pers. comm.) are found in large numbers in stumps. Tree species and size of stumps are affecting the composition of soil arthropods in the stumps (Persson et al, in prep). On the other hand, Lenoir et al. (2010) reported that no differences in soil fauna abundances or diversity could be found in sites that were stump harvested compared to stem-only sites 20–30 years after harvest.
The importance of low stumps for biodiversity is under discussion. It has been suggested that organisms common in low stumps also can use other types of dead wood as substrate. For instance, stumps harbour many frequently found fungi species that can use also slash and logs as substrate and red-listed fungi are often depending on logs (de Jong & Lönnberg 2010). Although logs are more important for the diversity of fungi, a massive increase of stump harvest may have negative effects on fungi species that at the present are common.

Some further effects on threatened biodiversity are discussed in Appendix A (Considerations to biodiversity in forestry).

**Lack of knowledge**

Naturvårdsverket (2006) published a report on current and lack of knowledge on the impact of slash and stump removal on among others biodiversity. Some of the questions are addressed in the research program Sustainable Supply and Refining of biofuels funded by the Swedish Energy Agency (Gode et al. 2010; De Jong & Lönnberg 2010). Most studies on the effects of slash removal are carried out in intensively managed forests and landscapes, on relatively small scale and during short periods and the consequences of slash removal for biodiversity at larger scales are difficult to predict from the results. More assessments are needed to extrapolate results from field experiments to landscape levels and on longer time scales (Naturvårdsverket 2006). The effects of slash removal on small mammals and other vertebrates including birds of prey have hardly been investigated (Naturvårdsverket 2006). Several studies have been published on the effects of slash removal on soil fauna (Lundkvist 1983; Bird & Chatarpaul 1986; Mann et al. 1988; Sohlenius 1996; Bengtsson et al. 1997; Bengtsson et al. 1998; Persson et al. 2010). However, it is still poorly understood how harvest of different sources of dead wood will affect population dynamics of organisms that are common at the present but that may decrease due to harvest or to changes in the microclimatic environment. Changes in precipitation and temperature, due to climate change, may affect population dynamics and migration patterns of organisms, including invasive species. Knowledge about this is scarce. The impact of ecological traps on red-listed species is not well investigated. The preference of stump size for many wood-living insects and lichens is poorly investigated. Studies on the importance of slash and stumps for small mammals, reptiles, birds and predators is scarce.

**Water quality**

A way of using the forest more efficiently as an energy resource is to harvest not only timber, but also slash (i.e. branches, twigs and needles) and stumps. How this is to be achieved without risking the environment has being investigated in a study financed by the Swedish Energy Agency. This following section is based on a report from that project including an extensive literature review (de Jong & Lönnberg 2010).
The effects from slash and stump harvest on the environment is far from clear due to uncertainties in research investigations and a vast variation in natural conditions and management processes. On land with high nitrogen loads, slash and stump harvest may relieve the area of excess nitrogen so that the net effect is a decrease in nitrogen leakage, but the actual results from studies of leakages of nitrogen vary. The effects on eutrophication are hence not clear.

Stumps, coarse and large roots contain approximately 30 ton C/ha (Palviainen 2005). Removal of stumps will thus result in decreases in the amount of dead organic carbon in the forest soil. In addition, stumps have 20 to 30 times higher C/N ratio than fungi. This implies that fungi growing on stumps need to take up N from the soil adjacent to stumps, thereby immobilizing N from an area around the stump. Stump harvesting may therefore lead to N leakage from the soil. However, Lenoir et al. (2010) showed that C-storage and N-pools were not affected by stump harvest nor was N-mineralisation. C mineralisation, however, was higher in control sites. De Jong & Lönnberg (2010) concluded that decomposition and mineralisation processes will hardly be affected by stump harvest.

Increased removal of biomass from the forest ecosystem leads to increased acidification and loss of base cations. Studies indicate that the effect on pH diminishes at about 15 years after slash and stump harvesting, while the negative effect on the supply of base cations seemed to disappear 32–35 years after harvest (Zetterberg & Olsson 2011). Finally, to compensate for the loss of nutrients caused by slash and stump harvest, ashes are often returned to the harvested area. This is, theoretically, unproblematic from an environmental point of view – it can even be positive as it has a pH increasing effect – but if not only pure forest biomass ashes are used, the practice may lead to contamination with heavy metals and organic pollutants that end up in the waters of the catchment. Effects of application of fertilizer are discussed in paragraph 2.3. The chemical aspects and effects of mixing ashes of different origin and/or quality are not discussed in this report.

Slash has often been used to protect the forest floor from damages through heavy vehicle driving. Removal of slash increased damages of the forest floor by heavy vehicles with 60% (Naturvårdsverket 2005) and also slash that is left on the clear-cut and logs may be damaged leading to poor substrate quality for wood-living insects (Hautala et al. 2004; Naturvårdsverket 2005; De Jong & Lönnberg 2010) may even affect water quality (De Jong & Lönnberg 2010).

Removal of harvest residues (Fig. 8) increases the amount of harvested biomass, which increases the substitution potential. Efficient biomass removal entails efficient nutrient removal. This is compensated for through fertilization. More intense management leads to increased mobility, and hence leakage, of a range of substances. Removal of harvest residues also has some negative impact on biodiversity, although the effects are rather restricted as most biodiversity was lost already at clearcutting. The impact on biodiversity includes also effects of fertilization. However, removal of slash from broad-
Leaved deciduous trees may reduce biodiversity seriously. The exact impacts depend on site specific characteristics as well as on the amount of residues removed and methods applied.

**Positive effects**: Increased biomass harvest and hence substitution potential, as well as decreased N load on soil and water.

**Negative effects**: Potential biodiversity loss, eutrophication of water, acidification, leakage of mercury.

**Potential goal conflict**: Substitution potential – eutrophication; substitution potential – non-toxic environment.

**Identified knowledge gaps**: The impacts of changed precipitation patterns and temperature on the population dynamics and dispersal of many organisms that prefer or are depending on dead wood, sun-exposed habitats and clear-cuts are still poorly understood. Impact of stump extraction on soil C turnover, particularly in old organic matter. Uncertainties regarding the rate of weathering of minerals that could potentially compensate for loss of base cations and acidification.

### 1.7 Liming

We conclude that:

**Liming has long-lasting effects on soil living organisms and mineralisation processes. A general conclusion on the effects of liming on biodiversity can not be made and liming should only be applied with caution.**
Biodiversity

PLANTS

Hallbäck et al. (in Staaf et al. 1996) showed that liming has long-term (68 years) positive effects on the species richness of mosses. Lichens are negatively affected by liming, especially in nutrient poor *Pinus sylvestris* forests. In these forests liming increased the abundance of *Vaccinium vitis-idaea* and *Calluna vulgaris* whereas *Vaccinium myrtillus* was negatively affected. Liming on nutrient rich spruce forests increases the abundance of broad-leaved plants like *Urtica dioica*, *Rubus idaeus*, *Chamaenerion angustifolium*. In short, liming has long-term effects on the composition of plant species, including mosses and lichens. These changes in species composition are site specific.

SOIL BIOTA

Persson et al. (1989) showed that liming did not affect abundance of fungi. However, liming tended to reduce the amounts of FDA-active fungal hyphae in soils that have a high CN ratio compared to soil with a low CN ratio. Also Erland & Andersson (in Staaf et al. 1996) showed that liming did change the composition of mycorrhizal fungi. It was shown that the effects of liming on mycorrhiza are site specific and may be related to nutrient and heavy metal content of the soil. Persson et al. (in Staaf et al. 1996) found that abundance and composition of functional groups of mushrooms was affected by liming. Mycorrhizal fungi like milk-cup (*Lactarius* spp.) and russulas decreased after liming whereas decomposing litter mushrooms increased.

Liming sometimes increases the number of nematodes in the field. Some studies have shown that the number of nematodes may increase initially after liming, after which the number decreases and stay at a low level for a long time (de Jong & Lönnberg 2010 and ref. therein). Persson et al. (1989) showed that the composition of trophic groups within the nematode community changes after liming but that these changes are site-depended. The abundance of fungivorous /root-feeding, bacterial feeders and omnivorous nematodes was reduced at one site (Fexboda) after liming. The nematode community was less affected by liming in another site (Jädraås) although the bacterial feeders decreased after liming and so did the total number of nematodes. In short, the effects of liming on nematodes are site-depended. High pH and competition with earthworms may affect the species composition in limed plots.

The potworm *Cognettia sphagnetorum* has often low abundances in limed forest plots. This species is an acid tolerant species and is often dominant in coniferous forest. The species can maintain high population levels even in limed forests. However, competition with earthworms may sometimes decrease the abundance of *Cognettia sphagnetorum* after liming (Persson et al. 1989). Liming increases the abundance of snails, earthworms and protozoa and decreases the abundance of many oribatid species (Persson et al. 1996).

Changes in the abundance of soil fauna may have implications for C and N mineralisation processes. Increases of earthworms affect the activity of
bacteria positively, thereby possibly increasing C mineralisation and decreasing C storage. Furthermore earthworms feed on fungal hyphae and thus N-immobilisation by fungi may decrease. Increases in bacteria may affect the population of bacterivorous protozoa which may increase mineralisation of N. Ravenek (2009) showed that liming has long-lasting effects such as increases in C and N mineralisation rates, reduced organic layer and decreased C and N pools and higher abundances of earthworms still after 25 years after application of lime. However, data on liming effects on C and N pools in coniferous forest are contradictory and effects may depend on the N concentration of the soil. Liming may have a greater effect on C and N mineralisation rates in soil with high N-content (Rudebeck 2000 and ref. therein).

Lack of knowledge
As shown above effects of liming on soil fauna and on the relation between different soil organism groups and mineralisation processes are complex, site specific and not fully understood.
2. Agriculture

Conflicts and/or synergies

For the production of biomass there is a need of land. In this there is a basic conflict with respect to biodiversity since the introduction of management inevitably affects and changes the habitats for all species. The intensity of management matters. Low input of nutrients conflicts with the production level and sustaining high soil carbon stocks but decreases gaseous emissions and nutrient leaching.

2.1 Fertilizer use

Introduction

Most of the Swedish arable land is subject to some type of fertilization, regardless if the production is intended for food, feed or industrial products. The supplied amount of nutrients 2009 was in average 107 kg N, 24 kg P and 107 kg K per ha (SCB 2011). Average rates of mineral fertilizers, on fertilized acreage, were 84 kg N, 7 kg P and 15 kg K per ha. These are quantities of great importance for yields and maintenance of soil fertility, but they also have environmental effects including impacts on biodiversity. The following discussion focuses on nutrient application without consideration of fertilizer form. Use of organic or mineral fertilizers has similar purposes i.e. to improve yields from both quantitative and qualitative points of view and to avoid soil nutrient impoverishment and fertility degradation.

Biomass effects

Supply of nitrogen (N) has a profound impact on yields, confirmed in numerous field experiments (Mattsson 2004, 2006). For cereals recommended N-rates will increase yields with at least 50% compared with no N-supply. The general form of the production curve implies that the yield increase per unit N added diminishes as the N rate increases (Fig. 9). The contrary, decreasing N rates, results in larger and larger yield losses per unit N.

A one year reduction of N-application with say 50% from recommended N-rates will reduce the yield with 15%. A continuing N rate reduction with 50% from optimum rates over several years will result in yield reductions of 25%. It must be stressed that this estimate starts at the economic optimal N-rates. Many fields are not fertilized with optimum rates and on those the yield losses will be larger than 25%. But on the other hand, on fields that are fertilized above economic optimum rates yield losses will be less due to the flat phase of the production curve.

Biomass effects of N applications in other crops are similar to those for cereals. In sugar beets a 50% reduction of the N rates from the recommended rates (120 kg) reduces the average yield of roots from 42.5 t ha\(^{-1}\) to 38.6 t or 9%. Increasing the N rate with 50% to 180 kg N ha\(^{-1}\), gave a root yield of
45.6 t ha\(^{-1}\) or 7% increase (Mattsson 2011a). Since sugar content and processing quality is negatively correlated with rising N levels, recommendations are usually limited to 120 kg N per ha. In potatoes yield reductions with the same order of magnitude \textit{i.e.} 10% on dry matter basis are expected (Ericsson 1988). In oil seeds 15% yield reductions might be expected (Mattsson 2011b). In ley production there is the choice of using N-fixing legume plants. Hence dry matter yield losses from reduced N applications can be avoided or significantly reduced.

In 2009 cereals were grown on 1.05 Mha, ley and green fodder crops on 1.18, oil seeds on 0.10, potatoes on 0.03, and sugar beets on 0.04 Mha. Fallow and unspecified land use made up 0.16 Mha and pastures and meadows occupied 0.4 Mha (SCB 2010). Sugar beets, potatoes, and oil seeds are high valued crops for the individual growers but they are grown on a minor acreage than cereals and leys.

Estimates of the biomass effects of the nutrients P and K are not as evident as for N. Soil nutrient status, soil types, cropping systems, and different crop responses are factors that must be considered. However, as adding P and K can’t be excluded and for an acceptable and sustainable production they must be applied.

**Soil C effects**

The average soil carbon content in Swedish arable top soils is 2.6±1.2 % (Eriksson et al. 2010). Typically, on mineral soils, the C is concentrated to the upper soil layers. The top-soil layer (0–20 cm) with a bulk density of 1250 kg m\(^{-3}\) (common used value) thus contains 65ton C ha\(^{-1}\). A mean estimated C stock for all arable land to 0.6 m depth is 125 ton ha\(^{-1}\) (Eriksson et al. 2010). Generally we consider Swedish arable top soils to be in a steady state condition concerning C, \textit{i.e.} the input of C is balanced by losses through mainly decomposition. For arable peat soils this is not the case and is discussed later.
Fertilizer use has an impact on biomass production and soil processes, and because of this, the C dynamics in soils are affected. Increased yields give more raw material for humus formation, and more C is introduced into the soil (Carlgren & Mattsson 2001; Körschens et al. 2002; Christensen et al. 2006; Johnston et al. 2009; Mattsson 2009). The contrary, small yields and minor amounts of crop residues result in less C introduced and less humus formation. While fertilization measures normally have immediate effects on yields their net effects on humus formation and humus contents are only measurable in the long perspective.

Differences in top-soil C content between optimum N supplied treatment and half this N-rate is 0.1 percentage units as measured in the Swedish soil fertility experiments (Mattsson 1999). A change with 0.1 percentage unit means a long-term C-stock change of 3125 kg ha⁻¹ in the 0–0.2 m top soil layer (provided a bulk density of 1250 kg m⁻³). In Fig. 10 an example is shown on soil C changes over time at different fertilizing regimes.

![Fig. 10. Top soil C content in treatments without (IIB0) and with mineral N (IIB3), respectively.](image)

The differences between the treatments, and also the trends, are strongly dependent on the starting C-level. Larger changes will occur if the starting C-content is different from the steady state level. Types of fertilizers and cropping systems also have effects. This is discussed below in the section Cropping systems.

We claim, based on experiences exemplified in Fig. 10, that 50 years is a reasonable time span for steady state conditions. The assumption used is crucial and will affect the conclusions considerably (see further down). With a 50 year time span the change rate will be 3125/50=63 kg C ha⁻¹ yr⁻¹. The top soil loses C, i.e. there is an emission to the atmosphere of the greenhouse gas CO₂. The estimated value can be related to the N application which was reduced with 50% from 107 to 54 kg N ha⁻¹ yr⁻¹. Consequently 1.2 kg soil C is lost per kg reduced N application. This value might be compared with 15 kg C per kg N for forest land (see section Forestry above).
Gaseous N emissions from the use of N fertilizer

Globally, 60% of the anthropogenic nitrous oxide emissions have an agricultural origin, of which 40% units are caused by N processes in the soil (Berglund et al. 2009, citing IPCC 2007). Jordbruksverket (2009) concludes for Swedish conditions that 35% of total emissions are \( N_2O \) from N use in agriculture. Likewise, globally 50% of the methane emissions are considered to be of agricultural origin of which 30%-units originate from ruminants digestion. The Swedish estimate is 20% of total emissions.

Other estimates of nitrous oxide emissions from N-fertilizer use vary from 0.05 to 0.97% of total N applied (McKenney et al. 1980; Chadwick et al. 2000). In a compilation by Muñoz et al. (2010) estimates from 0–44 kg \( N_2O-N \) ha\(^{-1}\) yr\(^{-1}\) were found considering all \( N_2O \) emissions from arable land, not only from direct N-fertilizer use. This amount (44 kg \( N_2O-N \) ha\(^{-1}\)) is equivalent to 69 kg \( N_2O \) which is equal to 20 ton CO\(_2\)-Ceq (carbon dioxide equivalents) ha\(^{-1}\) yr\(^{-1}\). Nitrous oxide (\( N_2O \)) as a greenhouse gas is around 300 times stronger than CO\(_2\).

Average values of \( N_2O \)-emissions for fertilized cereal production are 1.77 to 4.77 kg \( N_2O-N \) ha\(^{-1}\) yr\(^{-1}\) or 0.02–0.04 kg \( N_2O-N \) kg\(^{-1}\) N applied (Berglund et al. 2009). A standard value given by IPPC (2006) is 0.01 kg \( N_2O-N \) kg\(^{-1}\) N. Given this latter estimate, \( N_2O \)-emissions for Swedish conditions with an average N-rate of 107 kg N ha\(^{-1}\) will be 0.11 kg \( N_2O-N \) ha\(^{-1}\) yr\(^{-1}\) or 52 kg CO\(_2\)-Ceq. This value is dependent on temperature, soil water content, decomposable organic matter, nitrate-N and oxygen pressure, and hence very variable.

Gaseous emissions from N-fertilizer production

During the production of N fertilizers \( N_2O \) and CO\(_2\) are emitted. Eriksson et al. (2007) estimated total emissions of CO\(_2\) and \( N_2O \) from fertilizer production to 2.6 kg CO\(_2\)-Ceq kg\(^{-1}\) N produced, when considering a global warming potential over a time horizon of 100 years. Berglund et al. (2009) concluded in their literature review values between 4–11 kg CO\(_2\)-Ceq kg\(^{-1}\) N depending on type of N-fertilizer related to the fertilizer production. The most frequent used N form was ammonium-nitrate with an emission factor of 6.8 kg CO\(_2\)-Ceq kg\(^{-1}\) N.

The annual average use of 84 kg N per ha in mineral fertilizers would for this reason give rise to an emission of around 571 kg CO\(_2\)-Ceq ha\(^{-1}\) yr\(^{-1}\).

Net impact on greenhouse-gas emissions

It has been pointed out that the biomass production curve for N for many of the agricultural crops is rather flat. A reduction of the N-rate with 50% might reduce the yields with 25% in a long-time perspective. This will also have implications on the emissions of greenhouse gases. As discussed above reduced N rates cause the soil C to decline. According to calculations above 63 kg C ha\(^{-1}\) yr\(^{-1}\) is lost from the soil during a 50-year period, equal to the same amount C emitted to the atmosphere. Reduced N rates will decrease gaseous emissions from the production of N fertilizer. A 50% decrease would reduce emissions by 364 kg CO\(_2\)-Ceq ha\(^{-1}\). With less N fertilizer used, the gaseous...
losses of N₂O are reduced. For 107 kg N ha⁻¹ the N₂O-N losses are calculated to equal 52 kg CO₂-Ceq ha⁻¹ yr⁻¹ and for 54 kg N they are estimated to 26 kg CO₂-Ceq ha⁻¹ yr⁻¹. The fluxes of greenhouse gases at average N application rate, and half this rate, are listed in the table below.

<table>
<thead>
<tr>
<th>Greenhouse gas emissions at two different levels of N application rates. Calculated for a 50-year period.</th>
<th>107 kg ha⁻¹ yr⁻¹</th>
<th>54 kg ha⁻¹ yr⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil C, annual change, Ceq ha⁻¹ yr⁻¹</td>
<td>0 (steady state assumed)</td>
<td>63</td>
</tr>
<tr>
<td>Gaseous losses from fertilizer production, CO₂-Ceq ha⁻¹ yr⁻¹</td>
<td>571</td>
<td>286</td>
</tr>
<tr>
<td>N₂O from denitrification, CO₂-Ceq ha⁻¹ yr⁻¹</td>
<td>52</td>
<td>26</td>
</tr>
<tr>
<td>Sum greenhouse gas emission, CO₂-Ceq ha⁻¹ yr⁻¹</td>
<td>623</td>
<td>375</td>
</tr>
<tr>
<td>Wheat production, kg ha⁻¹ yr⁻¹</td>
<td>6000</td>
<td>4500</td>
</tr>
<tr>
<td>Emission per produced unit, kg CO₂-C per kg</td>
<td>0.104</td>
<td>0.083</td>
</tr>
</tbody>
</table>

To the right the effects on gaseous emissions given as CO₂-Ceq of a 50% reduction of N-rates are listed and to the left the corresponding emissions, provided steady state is valid for soil C, i.e. no annual changes. Reduced N-fertilization causes a C loss from soil to the atmosphere. A loss of production will also occur. We can express the net result as emissions per produced unit.

The figures indicate that a decrease in N application rates would reduce not only the total amount of greenhouse gases emitted to the atmosphere, but also the amount of emission per unit production. It should be kept in mind that the conclusion is only based on emission changes due to changes in soil C, emission caused by production and N₂O emissions from applied N. No other factors are included. One could argue that the use of N and high production rates increases the need of pesticides which might further increase greenhouse gas emissions, and thus make reduced N application rates even more favourable. It should also be noted that the soil C reduction caused by reduced N rates was calculated based on the assumption that steady state is obtained after a 50 years change in N-application rates. This implies that, in a longer perspective than 50 years, the decrease in emissions following decreased N fertilizing rates would be even higher. However, decreased production implies needs for more land to relocate lost production, e.g. deforestation with increased emissions, or increased import from other regions with different emission conditions. To ensure future food security with an increasing world population it is not only a question of maintaining present production but also to increase total food production. In this context lowering production per ha is a doubtful action.

From our data we conclude that it is efficient with respect to greenhouse gas emissions to reduce the use of mineral N fertilization. However, if there is a switch from mineral N to organic N the higher application rate might be
better with respect to greenhouse gas emissions. This is due to the fact that a large part of the emissions are related to the production of mineral N.

**Organic soils**

There are 268 000 ha of cultivated organic soils in Sweden which is 7.6% of the agricultural land (Berglund 2011). About 25% of this area is intensively cultivated with annual crops. The rest is used for pastures and ley production. Greenhouse gas emissions from organic soils were estimated to be 3.1 to 4.5 Mt Ceq yr⁻¹ or 6–8% of total national emissions (Berglund 2011).

**New crops and varieties**

A switch of crops or introduction of new varieties does not necessarily change the need for nutrient application. It is the crop's productivity and ability to use present nutrients that is determining. Sugar beets e.g. with a long vegetation period are not heavily N fertilized today compared to many other crops (Jordbruksverket 2010). Winter wheat on the other hand, with an even longer vegetation period (one year) from seeding to harvest is significantly heavier N fertilized. The reason to these differences is that fertilizer rates are adapted to the yield and quality of the products. Shifting cropping conditions allow for other crops. But the choice is not only depending on agroecological conditions. Markets and political factors must also be considered. Legume plants are able to utilize N₂. An e.g. well established lucern stand can fix 200 kg N ha⁻¹ and could reduce mineral fertilizer consumption considerably. However, if this is a relevant option there must also exist a demand for the product in this case herbage, either for feeding or for energy production. Measurements and experiments show that humus contents in agricultural soils increase from south to north (Eriksson et al. 2010; Mattsson 1999). This is mainly, but not completely, attributed to differences in temperatures. More of ley production in northern regions compared to southern plays a role.

*Possible development:* A prolonged vegetation period will allow for a change in choice of crops. The frequency of annual crops will increase northwards. Present humus contents in arable soils in these parts, which today are higher than in the south, will slowly decrease with some %-units to typically estimated 3.5% (Andrén 2007).

**Effects on water quality**

A humid climate with precipitation exceeding evapotranspiration means a need of efficient drainage of arable land. Certain losses are inevitable because the systems are not completely closed (Carlsson et al. 2003). According to Engström et al. (2007), leakage of nitrogen and phosphorus from arable land contribute almost half of the eutrophication impact from Swedish sources. Eutrophication is also pointed out as one of the most important impacts on the environment from agriculture.

Generally, losses increase when more nutrients are circulated in comparable crops or production systems but efforts to minimize leaching in relation
to income and costs are always undertaken. A shift in precipitation will also change fertilization strategies towards increased efficiency, optimizing yields, and minimizing losses. A shift in climate towards longer and periodically wetter conditions will focus the need to undertake measures against leaching especially nitrogen (SOU 2007:60).

Leaching losses from Swedish arable land are measured continuously in various programs (e.g. Carlsson et al. 2003; Datavärdskaftjordbruksmark, 2011). Local climate, precipitation, soil type etc. are strong influential factors. Average annual transports of N, P and K obtained for a catchment in central Sweden over 35 years was 13 kg N, 0.2 kg P and 1 kg K per ha. Average run-off was 223 mm. Carlsson et al. (2003) reported average N transports of 30–40 kg ha⁻¹ in 2001/2002 in the southern situated production areas and 10–15 kg ha⁻¹ in more central Swedish areas.

There are several ways to decrease leakages of N and P from arable land, except, of course, reduction in nutrient load. They include the use of winter crops (N and P) (Gustafson et al. 2000; Ulén and Jakobsson 2005), activities that reduce erosion, no or limited tilling (P) (Bergström et al. 2007; Laurent et al. 2007), tramline management (Deasy et al. 2010) and riparian buffer zones (N and P, especially particulate P) (Hoffmann et al. 2009; Jeppesen et al. 2010), and the introduction of ponds and wetlands in the agricultural landscape (primarily N) (Gustafson et al. 2000; Lee et al. 2009; Jeppesen et al. 2010). It is however also clear from the literature that the processes involved in retention of nutrients are extremely complex, and that the outcome of any one activity to ameliorate water quality is uncertain (Braskerud 2002; Cherry et al. 2008; Sharpley et al. 2009; Withers & Hodgkinson 2009). Climate-change effects add to this complexity.

It might be assumed that a longer and warmer vegetation period implies a switch in the choice of cultivated crops (Eckersten et al. 2008). Crops can be seeded earlier in the spring and the growth will last longer in the autumn. Firstly a switch in the choice of cultivars will take place and secondly new varieties and species might be introduced. Here breeding and adaption to changing environmental conditions walk hand in hand. A good example is the cereal triticale nowadays frequently cropped in Central Sweden and southwards. Previously this was not possible because the winter hardiness was not good enough and perhaps also because the winter temperatures were lower than today.

Modern agriculture includes, along with the use of fertilizers, the use of various pesticides. These substances leak into ground as well as surface waters, see Wivstad (2005) for a review of use and occurrence of pesticides in Swedish agriculture. The risks identified by Wivstad include human end environmental health. Due to Swedish legislation the use of pesticides is comparatively low in Sweden. Large quantities of pesticides are however imported with food and fodder. Furthermore, the use of pesticides, especially fungicides, may increase with increased intensity of agriculture (ibid.).

Human exposure to contaminants and pathogens from agriculture may increase with increasing effects of climate change, primarily through increased
runoff and temperature (Svenskt Vatten 2007; Boxall et al. 2009), but uncertainties in this field of study are large (Bloomfield et al. 2006)

**Fertilization effects on biodiversity**

The effects of the use of fertilizers on N-leakage and N\textsubscript{2}O-emmissions depend on climate, soil type and crop (Gode et al. 2010). The benefits of short rotation coppice include uptake of waste nutrients and heavy metals, habitats for birds, mammals and plants, and carbon sequestration in the soil (see also 4. Energy forests). Converting cropland into bioenergy plantations can reduce the loss of nitrate and phosphorus in the soil and water (Mann & Tolbert 2000; Baum et al. 2009). Berndes et al. (2004) found that Cd concentrations originating from P fertilizers were a 100 times more removed by Salix than by straw. Energy crops may be grown on marginal cropland (set-asides) such as fields with poor productivity or poor drainage (Mann & Tolbert 2000; Wiens et al. 2011). Although energy crops may need fertilization to ensure high yield, this does not necessarily lead to intensification of agriculture.

**EFFECTS ON SOIL FAUNA**

Agricultural practices such as tillage, fertilization, removal of plant biomass and plant residues and the use of pesticides affect soil organisms. The effects of fertilizing are not consistent within studies and between studies, and effects depend on soil type, crops and local climatic conditions. Bünemann et al. (2006) have summarised the current knowledge on the effects of organic and mineral fertilizers on soil biota. It was concluded that the main effect of organic fertilizer was an increase in biological activity with increasing plant productivity, crop residue inputs and soil organic matter. Postma-Blaauw et al. (2010) found that mineral fertilization increased fungal biomass and the number of nematodes but had negative effects on earthworm abundances.

**EFFECTS ON NEMATODES**

An increase of the number of nematodes after NPK application has been shown in many studies (Ruess et al. 2002, and ref. therein). Fertilizer may increase the number of general opportunists (Ruess et al. 2002) and change the composition of trophic groups in the nematode community (Freckman & Virginia 1997). Ruess et al. (2002) showed that NPK application can change the ratio of fungivorous to bacterivorous nematodes in a mountain field but not in alpine heath grassland. Yeates et al. (1997) found that the abundance of nematodes (in particular of fungal-feeders) was higher in organically fertilized grasslands compared to conventional farms. Ruess et al. (2002) suggested that increased number of nematodes after NPK-fertilizing was related to increased plant biomass and less to microbial biomass.

**OTHER SOIL BIOTA-ORGANIC VERSUS MINERAL FERTILIZATION**

In addition, according to Bünemann et al. (2006) organic fertilizers are highly variable in characteristics such as dry matter content, pH, salinity, plant mate-
rial and C content etc. The effects of application of an organic fertilizer on soil organisms may therefore vary from positive to harmful. Yeates et al. (1997) found that the abundance of tardigrades and mites was higher in organically fertilized grasslands whereas earthworm abundance and biomass was lower compared to conventional farms. Kromp (1990) showed that abundance and species richness of carabid beetles was higher in organic managed fields compared to fields where mineral fertilizer was applied. Osler et al. (2008) found a higher number of mites in organic agricultural fields compared to fields that were fertilized with phosphorus. However, when the study was repeated the year after, the result was the opposite. A decreased number of soil animals after application of P- fertilizer may be due to the toxicity of metal contaminants such as cadmium, mercury and lead (McLaughlin et al. 2000). N- and K fertilizers in contrast contain low levels of toxic metals. In addition to the characteristics of the fertilizer applied, the effects on microbes and soil fauna depends also on several other factors including the crop in the field (Postma-Blaauw et al. 2010; Osler et al. 2008), soil type (Yeates et al. 1997) and climatic conditions (Foissner 1992). Foissner (1992) showed that conventional farming compared to organic farming reduced soil fauna seriously in arid regions whereas the negative effects of conventional farming on soil fauna are less severe in regions with sufficient precipitation. The effects of fertilizing on soil organisms are thus interacting with local factors such as microclimatic conditions.

KNOWLEDGE GAP

The impact of changes in precipitation and temperature on how fertilization affect soil organisms remains unknown. For a full picture of the environmental impact from fertilization, resource use and emissions related to production, transport and spreading of nitrogen and phosphorus should be included. Effects on biodiversity vary from positive to harmful and depend on fertilizer characteristics, application dose, soil type and climate, and on which species groups that are evaluated.

Positive effects: Higher production, increased soil carbon stock, more food per time and area unit.

Negative effects: Eutrophication, biodiversity loss, potentially increased use of pesticides (indirect effect), phosphorus consumption.

Potential goal conflict: Substitution potential and carbon sink – biodiversity; substitution potential and carbon sink – eutrophication; substitution potential and carbon sink – resource use (P); substitution potential and carbon sink – non-toxic environment.

Identified knowledge gaps: Effects on biodiversity of N fertilization. N-transformations under different environmental conditions.
2.2 Liming effects

Conflicts and synergies

Liming for correction of soil chemical status to neutral or near neutral acidity has a positive effect on the abundance of earthworms and bacteria. There is a conflict because the use of carbonaceous minerals directly leads to CO₂ evolvement from the supplied material to the atmosphere, but also indirectly because of the enhancement of the mineralization of organic material. Liming for improvement of soil structure reduces nutrient leaching.

Introduction

Liming arable land is undertaken for adjusting the soil pH. This has impacts on the microbiological processes, and usually a positive effect on yields is observed. Indirectly, the dynamics of organic matter are affected. The use of liming material results in CO₂-production when carbonate compounds are oxidized to CaO. According to IPCC (2006) a maximum of 0.44 t CO₂ per t calcium carbonate and 0.48 t per t dolomite is evolved.
Continuous acidification
Swedish agricultural soils are exposed to a continuing acidification typical in a humid climate. Soil respiration, fertilization, product harvesting and depo-sitions are all acidifying processes. Roots and microflora respiration produce carbonic acid. Nitrification of ammonium containing mineral fertilizers produces acidifying H⁺ ions. Base cations are removed with the crop. Altogether this is calculated to correspond to a CaO demand of 150 kg ha⁻¹ yr⁻¹ (Haak 1991). Present consumption of liming materials equals 90 kg CaO ha⁻¹ yr⁻¹ (SCB 2010) equal to 79 kg CO₂-Ceq ha⁻¹. If all this is dissolved and CO₂ produced we end up with total emissions from liming of arable land in the size of 205 Mt.

Nutrient availability and plant uptake
Liming is a measure to keep agricultural soils in a satisfactory soil chemical condition. Liming arable fields with low acidity increases pH and is shown to increase crop yields (Lupwayi et al. 2008, and ref. therein). Top soil pH measured in water is recommended to be in the interval 6–6.5 (Jordbruksverket 2010). In this interval the solubility and plant availability of phosphorus is at its maximum. Also the availability of many micro nutrients is fairly good although the interval is a compromise. For some nutrients e.g. manganese, solubility decreases drastically with increasing pH while for some such as boron it increases. At low pH (<5) the aluminium concentration will be hazardous to plant roots. Solubility and plant uptake of Cd, a non biogene element also increases at low pH-values.

Impact on organic matter
Bacteria dominate over fungi at pH-values above 6. For the decomposition and mineralization of organic matter this is favourable and results in an effective nutrient circulation. The pH is lowered when crop residues are returned and decomposed (Simán et al. 1993). Liming stimulates the decomposition of organic matter. Data from running liming experiments started during the 1960ies indicate this. This implies that if liming is not undertaken an increase in soil C could be expected due to incomplete decomposition. However, to our knowledge there are at present no such estimates available.

Soil structure
A soil with a good liming status has a structure which is more resistant to degrading than an acid soil. Both soil physical and soil biological processes are factors behind this. Liming also has a direct physio-chemical effect on soil structure and is a measure to improve and strengthen the soil structure (Simán et al. 1982).

Possible development: Rising temperatures will increase the decomposition of organic matter and the release of CO₂. To counteract the acidification this will induce increased liming needs as will higher precipitation because of increased nutrient leaching including base cations.
Liming effects on biodiversity

The effects of liming in arable fields on soil fauna are poorly investigated. Application of lime in arable fields increased microbial biomass and activity (Lupwayi et al. 2008). Rousk et al. (2010) studied the microbial abundance and species composition in soils collected across a long-term experiment at the Rothamsted research station. Chalk was applied one time in the mid of the 19th century and created a pH gradient from 4.0 to 8.3 within 200 m. The abundance of bacteria increased fourfold across this gradient whereas the abundance of fungi was not related with pH. The composition of bacterial and fungal communities, however, were both affected by pH. It is poorly investigated how changes in microbial composition affects food web systems and soil fauna diversity in agricultural soils. Global change may increase the demands for renewable energy sources, including energy crops. Application of lime, may thus become important.

Systems perspectives Liming of arable land

Liming arable land (Fig. 12) is undertaken to increase soil pH and normally results in increased yield. Liming, however leads to direct emissions of CO$_2$ from the carbonaceous mineral, and indirect emissions from increased mineralization of organic substances caused by the increase in pH. Liming also leads to reduced leakage of nutrients.

Positive effects: Increased production, reduced acidification, nutrient retention, improved soil structure.

Negative effects: CO$_2$ emissions.

Potential win-win effects: No identifiable.

Potential goal conflict: Carbon source – acidification; carbon source – eutrophication

Identified knowledge gaps: Effects on biodiversity
2.3 Tillage and crop residue management

Conflicts and synergies

Tillage operations for high production generally accelerate the mineralization of organic material and enhance the CO₂ evolvement (conflict). Simultaneously, the microflora is favoured especially if crop residues as straw are left on the field (synergy). Ley producing systems are more C efficient than annual arable production systems.

Introduction

A significant proportion of the nutrients taken up by a crop is left on the field in crop residues at harvest. Often, but not always, the residues are returned and incorporated in the soil. In other cases they are taken away and used for feeding and/or bedding and returned in the form of FYM (farm yard manure) or compost material. Certainly some residues are permanently removed as in the case of bio-fuels.

A simple C-balance

A good cereal crop produces annually 6.5 t ha⁻¹ of grain. As an approximation the sum of biomass in straw, stubble and roots, which is returned as reduces to the soil, is of the same size. As the C content of the organic matter is around 50%, about 3.2 t C ha⁻¹ yr⁻¹ is returned to the soil. If the straw is removed only 1.6 t C ha⁻¹ is returned to the soil. Within a couple of years 90% of the C is transformed to CO₂ by the heterotrophic microflora in the soil, whereby the long-term accumulation of organic matter in the soil after straw removal is around 0.2 t C ha⁻¹ (Persson & Kirchmann 1994). It is well known since long that the heterotrophic microflora is very efficient in decomposing fresh plant material leaving only minor amounts of slowly decomposable organic material that will be added to the bulk C pool every year (Jansson 1986).

With an average of 2.6% C in the top 20 cm layer, the C stock size is 65 t ha⁻¹ (provided a bulk density of 1250 kg m⁻³). Mineralization of stabilized organic material in agricultural soils is 1–2% per year of the C-pool (Jansson 1968; Persson 2003), i.e. 0.65–1.3 t C is annually lost through decomposition. Thus, the annual long-term supply of residues after straw removal (0.2 t C) is not enough to compensate for the annual decomposition losses, and consequently, straw removal is resulting in a loss of soil organic carbon.

Supply of organic material e.g. manure has a significant effect on soil C content. After 50 years live-stock rotation with ley and manure, as shown by long-term fertility experiments, the C content was 0.2 %-units higher than for a system without ley and manure (Fig. 13).

In the live-stock rotation a 6 year period included 2 years with ley. Manure was applied once during the same period and all straw was removed. In the rotation with arable crops only (no leys) all crop residues were chopped and incorporated in the soil and no manure was applied.
A switch to cropping systems from arable crops to systems with live-stock as described would result in an increased C incorporation in the soils with 100 kg C ha\(^{-1}\) yr\(^{-1}\) over a 50 year period. A comparison based on produced unit per kg Ceq in such a system is complicated due to the complex mixture of products, meat among others, that are produced. Berglund et al. (2009) compiled Ceq as life cycle analysis values per kg product for a number of animal products. They reported 0.96–1.1 kg Ceq kg\(^{-1}\) milk, 11–15 C kg\(^{-1}\) meat (dairy cows), 17–19 kg Ceq kg\(^{-1}\) meat (ecological production), 2.6–3 Ceq kg\(^{-1}\) pig meat and 1.3–1.4 Ceq kg\(^{-1}\) chicken meat.

Residue management

Tillage and crop residue treatments are two factors that can influence the balances significantly. Tillage can be deep or shallow. Ploughing or non ploughing or reduced tillage are optional. Traffic intensity may vary. In close connection with tillage strategies is the handling of crop residues such as chopping and incorporation in the soil ploughed under or removal for feeding, bedding or fuel purposes. Many possible combinations are at hand with impacts on the mobilization/immobilization of C.

Ploughing or non-ploughing

There are two primary reasons for ploughing – to prepare the soil for sowing and for weed control. In non ploughing systems other types of sow-bed preparations are necessary and also strategies for control of perennial weeds which commonly increase in those systems. The choice of methods and strategies is to a large extent ruled by other factors than a changing climate with one important exception namely the frequency of freezing/thawing of the clay.
soils. For the maintaining of optimum soil structure a period of frost is very helpful. Ley producing systems are non ploughing systems but discussed in another section.

Not only ploughing, but also the intensity or frequency of other tillage operations have effects, particularly on clay soils (Feiziene et al. 2010). Soil temperature and soil water content are often more important factors than the above-soil biomass production (Feiziene et al. 2010; Sainju et al. 2010). Timing of residue incorporation affects leaching losses (Aronsson et al. 2003). Delayed autumn residue incorporation reduces leaching losses.

In non-ploughing systems or systems with reduced tillage, carbon will mainly be incorporated in the top layer of the soil compared to conventional ploughing where C will be allocated to a thicker and deeper soil layer (Rydberg 1987; Hernandz et al. 2010). This might consequently result in elevated contents of C in the top soil in non-ploughing systems. Etana et al. (1999) concluded, based on Swedish field experiments, that the soil organic C stock did not change significantly when methods with deep (27 cm) and shallow (14 cm) ploughing were compared. Shallow ploughing increased the C concentration in the surface but decreased it in deeper layers. Muñoz et al. (2010) concluded for tropical soils that the mean rate for C accumulation in no-till systems was 0.35 Mg ha\(^{-1}\) yr\(^{-1}\) (1 Mg = 1 \times 10^6 g) higher than for other evaluated systems.

Possible development: A changed climate will most likely affect the tillage and crop residue management. Generally, the choice of cropping system e.g. ley production or annual crops is ruled by the climate. In many northern situ- ated locations ley production is the only option. A warmer climate may result in a change from ley production to more annual crops. This will increase tillage operations and consequently, as mentioned above, reduce the humus content to a new steady-state at a lower level. There is a potential for win-win situation: Soil management with increased C sequestration in order to mitigate climate change will also increase soil fertility (primarily through decreased erosion).

Residue removal and biodiversity
Collecting of plant residues after harvest decreases the amount of food resources and increases temperature fluctuations and moisture loss of upper soil levels (Gergócs & Hufnagel 2009). Organic matter is an important substrate for microbes and soil fauna. Plant cover and high soil organic matter content, moderate soil moisture conditions and temperatures without high fluctuations and neutral pH is usually beneficial for most soil animal groups, including earthworms (Benado et al. 2005).

2.4 Cropping systems

Conflicts and synergies
Systems with animals are generally less C effective with respect to energy consumption. However, the use of manure is more positive for the C stock than
fresh organic matter. Steady state C levels are generally higher in systems with ley than in arable crop systems. The soil C effects of leys which are harvested are small. Integrated systems for fodder and meat production are favourable for soil C stock maintenance.

Introduction
Systems of nutrient supply, rotations, cropping for food, feed or fuel are factors which can be used to characterize cropping systems. Nutrient supply can be accomplished with organic fertilizers or with inorganic ones. Rotations are chosen to optimize the agro-ecological conditions and systems are specialized for types of products.

Organic or inorganic fertilizers
One major grouping of cropping systems distinguishes between systems with and systems without manure applications. The latter is based on mineral or inorganic fertilizers only, while the former uses manure or organic fertilizers possibly completed with mineral fertilizers. A shortage of plant available nutrients and inoptimal timing of nutrient supply with respect to plant demand is common in systems with organic fertilizers only. The origin of nutrients, organic or inorganic sources, is irrelevant for the plant. Agricultural crops take up nutrients in simple inorganic molecules or ions and not in complicated organic substances. But types of fertilizers have different effects on the soil C dynamics. Organic well decomposed fertilizers like farm yard manure or compost have a greater impact on the humus formation than inorganic types (Persson 1987; Carlgren & Mattsson 2001; Körschens et al. 2002; Christensen et al. 2006; Johnston et al. 2009). Manure and compost also contain all micronutrients that are essential for plant growth, and which might be absent in mineral inorganic fertilizers.

Rotational effects
The choice of rotations i.e. the types of crops used and the sequence in which they are grown is determined by how the nutrient supply is designed, the purpose of the production and the agro-ecological conditions. The type of crops included in specific system affects the soil C content. In ley production the absence of tillage means reduced C mineralization and a significant annual input of litter as well, and explains why the C stock might increase (Berglund et al. 2009). It is clearly shown in Swedish experiments how the soil C concentrations and the C stocks in top-soils are higher in ley production systems than in systems for arable crops (Jansson 1986; Persson & Kirchmann 1994; Ericson & Mattsson 2000; Carlgren & Mattsson 2001; Mattsson 2002; Mattsson & Persson 2006). The reduced tillage operations in ley systems is one reason. The potential possibility of farm yard manure application is another. It is worth mentioning that on intensively cropped grasslands the evidences of C stock increases are rare. In rotations where harvest residues are left and incorporated in the soil the humus content is higher than where
all residues are removed (Carlgren & Mattsson 2001). In extreme situations where all straw is removed a long term difference in soil C concentration of 0.1–0.2 percentage units is expected (Mattsson & Larsson, 2005).

A comprehensive report about the future Swedish agricultural production systems with respect to anticipated climate changes is given by Eckersten et al. (2008). They conclude that the potential for food production probably will be more favourable in terms of productivity, but drawbacks will also appear in the form of attacks of new pathogens, extreme weather and increased nutrient leaching.

Biodiversity
Effects on threatened biodiversity of changed use of ley are discussed in Appendix A (2.5: Choice of crop on farmland). In summary, climate change may lead to either increased or decreased use of ley. The main biodiversity effect of lay is its contribution to habitat variation. This implies that the effects are landscape dependent: positive effects in landscapes dominated by annual crops but negative effects in more forested areas in which deficit of open land and annual crops are already a threat to biodiversity. Indirect effects of ley cultivation on biodiversity can be expected through effects on the grazing of semi-natural grassland.

Systems perspectives Ley production

![Diagram showing the benefits of ley production](image)

Fig. 14. Ley production systems vs annual arable production systems for live-stock

Live-stock production, generally speaking, has a larger environmental impact than arable crops due to a considerably higher input of resources and energy per unit output. The actual impact, however, depends on the cropping system. With a ley production system (2 years of ley in a rotation period of 6 years), the environmental impact is smaller than with a rotation with arable crops.
only (Figure 14). Ley production entails fewer tilling operations, and hence decreased mineralization and a typical C steady state level above the one with arable crops only. Increased areas of ley favours biodiversity in relation to arable crops. Agriculture without ploughing is potentially positive compared to systems with ploughing with regard to climate change as well as biodiversity and water quality (due primarily to less erosion) – but may result in increased use of herbicides except in the case of ley production, where the use of herbicides is less frequent. Ley production is normally accompanied by the use of organic fertilizers, e.g. farm yard manure, which further increases the positive effect on soil C as these fertilizers favour humus formation to a higher extent than inorganic fertilizers do in relation to the amounts of nutrients applied.

**Positive effects:** Increased biodiversity in some landscapes, increased soil carbon stock, decreased nutrient loss (through decreased erosion).

**Potential win-win effects:** Carbon sink – eutrophication; carbon sink – biodiversity.

**Identified knowledge gaps:** How to integrate animal and crop production in order to reduce nutrient losses, facilitate use of ecosystem services from livestock management, and increased biodiversity. Effects on soil C accumulation of long-term ley systems and effects on necessary consumption behaviour.
3. Wetlands

Overall conclusion
Potential win-win: climate change mitigation through wetland restoration – eutrophication and biodiversity, although the most effective wetlands from a nutrient retention perspective are not necessarily those that contribute most effectively to increased biodiversity (Hansson et al. 2005)

Introduction
Wetlands are important in regulating hydrology, nutrient cycling and climate, and are moreover highly biodiverse ecosystems. In the future, harvest of biomass from wetlands is expected to increase as a means of nutrient removal and as a source of biofuel. With increasing frequency of heavy rains (related to climate change), there will also be an increasing need for wetlands in catchments with surface waters for drinking water production to protect these waters from increased runoff of pathogens, organic material and pollutants. While the importance of wetlands will increase with increasing effects of global warming, they will also be increasingly threatened by the expected rise in temperature.

The level of the water table influences the level of oxygenation of soil and sediments, which in turn regulates the microbial processes involved in carbon and nitrogen cycling. With the high water table of wetlands come, in brief, anaerobic conditions that lead to decreased mineralization of carbon compounds, and hence accumulation of carbon and decreased emissions of CO$_2$ and dissolved organic carbon (DOC). Moreover, the cycling of nitrogen is efficient in wetlands. In the aerobic zone, nitrifying bacteria convert nitrogen compounds to nitrate, while the anaerobic zone is home to denitrifying bacteria that reduces nitrate to nitrogen gas. On the other hand, anaerobic conditions also lead to increased production of methane.

Effects on greenhouse gas balance
The carbon sequestering capacity of natural wetlands is well documented (Kennedy & Mayer 2002; Holden 2005; Chivers et al. 2009; Lund et al. 2009; Moss 2010; Reddy & Jawitz 2010), but exactly how big that capacity is, is highly uncertain. In northern peatlands, where carbon sequestration is regarded as particularly important, carbon sequestration has been estimated to 8.7 ± 12.2 g CO$_2$ m$^{-2}$ year$^{-1}$ (Lund 2009). Like many other biological processes, those involved in carbon cycling are sensitive to a range of parameters. In this case, water table level and soil temperature appears to be of particular importance. Carbon sequestration in wetlands and forests is positively correlated with increased soil water content, and negatively with soil temperature (Chivers et al. 2009; Lund 2009; Lundblad et al. 2009; Ju et al. 2010). The same is valid for arable land (Vleeshouwers & Verhagen 2002; Freibauer et al. 2004), although here carbon sequestration is also heavily influenced by a range of other management practices.
Methane emissions from wetlands are also strongly correlated with temperature and water table level. In this case positively for both parameters (Turetsky et al. 2008; Long et al. 2010; Ringeval et al. 2010; Tian et al. 2010; Sha et al. 2011). In their study of a Canadian peatland, Long et al. (2010) estimate that the net effect of that peatland has been one of negative radiative forcing (cooling) as the effects of carbon sequestration have dominated those of methane emissions. What will happen to the balance between positive and negative radiative forcing in wetlands with increasing effects of climate change is, however, uncertain. Lund (2009) e.g. shows that a wetland can switch, at least temporarily, from being a carbon sink to a carbon source due to changes in the climate.

The third major component in the climate regulation capacity of wetlands is nitrous oxide (N$_2$O). Nitrous oxide is primarily produced as an intermediate in denitrification and, to a lesser extent, through nitrification processes. An increased water table leads to increased nitrification (as an effect of longer retention times) and denitrification (due to increased anoxic zones), but not to increased emissions of nitrous oxide, due to the greater reduction of N$_2$O to N$_2$ (Elmi et al. 2004; Elmi et al. 2005). Similar results are reported from studies of drained versus undrained forest soils (Von Arnold et al. 2005). Saari et al. (2010), state that unmanaged wet peatlands are minor sinks or low sources of nitrous oxide. Parameters that increase nitrous oxide emissions include fluctuation of the water table (Yu & Ehrenfeld 2009), increasing temperature, and episodes of freeze-thaw processes (Yu et al. 2007), and the N, P and Ca content and pH of the soil (Regina et al. 1996).

**Effects on biodiversity**

Effects on biodiversity of some land use-change scenarios are discussed in Appendix A (2.6: Changed draining and use of wetland in the agricultural landscape). In summary, climate change may lead to either an increase or a decrease in drainage and water regulation in the agricultural landscape. Restored hydrology of lowland wetlands is strongly beneficial for threatened biodiversity, but the effects differ depending on how much of the original wet ecosystem that is left. Most wetland ecosystems in the agricultural landscape are however also depending on traditional land use, e.g. mowing and grazing, and harvest of wetland grass for biofuel or food provides a potential for restoring biodiversity and ecosystem functions given that the harvest methods show ecological resemblance with the traditional land-use that have formed the ecosystems.

**Effects on water quality and quantity**

Wetlands work as water stores and hence even out fluctuations in water level and stream velocity downstream in the catchment. This leads to less erosion and fewer and less severe flooding events. In dry areas it also contributes to a decreasing risk of severe draughts in the dry season.
Effects on water chemistry are caused primarily by the fact that lower water velocity will cause larger and more stable zones of anoxic layers in soil and sediments, and an increased rate of precipitation of suspended solids in the aquatic zone of the wetland. Other important processes in wetlands that contribute to water quality amelioration are adsorptions to soil particles and assimilation by plants. These processes lead to increased nutrient retention (primarily N and P), decreased fluxes of organic matter, and hence decreased eutrophication in the catchment. Oxidized microsites around plant roots enhance the wetland’s capacity for a range of aerobic bacterial processes, including decomposition of pollutants.

The capacity of wetlands to retain nitrogen, phosphorus and various fractions of organic material has been documented for natural wetlands (e.g. Ready et al. 1999; Saunders & Kalff 2001; Kennedy & Mayer 2002; Knox et al. 2008), and constructed wetlands (e.g. Sakadevan & Bavor 1999; Braskerud 2002; Fink & Mitsch 2004; Vymazal 2007; Chavan & Dennett 2008; Weisner & Thiere 2010). The retention capacity for nitrogen is often higher and more stable than that of phosphorus although both show wide variations. Under certain conditions, wetlands can also be transformed from sinks to sources of phosphorus, especially summertime, due to seepage from the sediments (Bergström et al. 2007). The nutrient retention capacity of wetlands depends on many variables, but plant community, nutrient load, temperature, hydraulic load and age of the wetland appear to be the most important ones.

**Systems perspectives on wetland drainage (in the arable landscape)**

![Diagram showing effects of wetland drainage](Image)

**Effects decreasing with time in italics**

- Flood mitigation
- Build up of C, P and N in biomass
- Harvest of biomass
- Use of biomass
- Production and use of alternative biomass

**Increased methane production**

**Nitrification and denitrification**

**Increased biodiversity**

**Potential win-wins:**
- Subst. potential and C sink – eutrophication
- Subst. potential and C sink – biodiversity
- Subst. potential and C sink – water regulation

Fig. 15. Wetland restoration vs continued or increased drainage on arable land
Just as in the forest landscape, wetland restoration in the agricultural landscape (Figure 15) leads to anaerobic condition, which in turn leads to decreased mineralization of carbon compounds, and hence accumulation of carbon and decreased emissions of CO$_2$ and dissolved organic carbon (DOC). The higher water table also leads to increased nutrient retention, but also denitrification, often without increased emissions of N$_2$O, due to the anaerobic conditions. On the other hand, anaerobic conditions also lead to increased production of methane. Wetlands generally lead to increased biodiversity, both in general and in particular for threatened biodiversity. In the agricultural landscape, wetlands on former arable land, along streams and lakes etc will be productive, due to the high nutrient load, and could be harvested for biomass which could substitute fossil fuels or grazed which would substitute meat or milk production on arable land or based on fodder crops.

**Positive effects:** Nutrient retention, nitrogen recirculation, water regulation, water purification, increased biodiversity, biofuel production, food production without active fertilization

**Potential win-win effects:** Substitution potential and carbon sink – reduced eutrophication – biodiversity – water regulation.

**Identified knowledge gaps:** Emission factors.
4. Energy forests (Salix)

**Introduction**

The use of biomass derived from fast-growing trees grown on former agricultural land is a common practice in Sweden (Hoffmann & Weih 2005). Energy forestry is considered to be an alternative source of energy that can be produced with less disruption to the environment compared to other energy sources like coal and oil (Gustafsson 1987). Plantations of spruce, willow, poplar and other hardwood species are possible alternatives (Energimyndighetens program, 2000–2004). In Sweden Salix coppicing for bioenergy has been used for several decades, and the existing area was approximately 15 000 ha around 2005 (Agrobränsle 2005), but this has now decreased till 12 000 ha (Pår Aronsson, pers. comm.). Only 300 ha poplar plantations are established (Karacic et al. 2003). The Swedish Energy Agency wanted to increase the area for energy forests to 100 000 ha in 2010 (Weih 2006) but this goal was not reached.

It has been suggested that energy plantations on former agricultural land have the potential to affect biodiversity positively since this type of cultivation increases structural diversity in agricultural landscapes that are dominated by traditional crops like cereals and sugar beet (Sage & Robertson 1994). However, research on the biodiversity in energy forests resulted in contradictory conclusions (Hartley 2002) and the impact of energy forests on biodiversity seems to depend on the landscape context, the size of the plantation and the management intensity (Hoffmann & Weih 2005).

**Effects of energy forests on biochemistry, C storage**

The production of agricultural crops has caused depletion of soil organic matter (Mann & Tolbert 2000). Converting cropland and set-aside fields into energy crops with rotations of 3–10 years will increase soil organic matter and sequestration of carbon in the soil (Baum et al. 2009). Non-tillage management, litter input and increased abundance of ectomycorrhizal fungi and fine roots may change C-storage. C-storage in the topsoil of tree plantations on former agricultural fields increased whereas nutrient losses decreased in poplar and willow plantations (Baum et al. 2009). On the other hand, C-storage in soils did not change when agricultural fields or forests were transformed into willow plantations but more studies are needed to make general conclusions about this topic (Ågren cit. in Gode et al. 2010).

Hardwood energy forests have the ability to take up heavy metals and nutrients in excess and are, therefore, useful as multifunctional systems for treating waste products as municipal wastewater and sewage sludge (e.g. Landberg & Greger 1996; Aronsson & Perttu 2001; Dimitriou et al. 2005). Applying sewage sludge is suggested to have a lower effect on global warming than inorganic fertilizers because it is less energy-intensive to produce (Gilbert et al. 2011). In Sweden, about 2/3 of the willow plantations are fertilized with red sewage sludge. The use of sewage sludge is regulated in
EU directive 86/278 and includes rules for the content of heavy metals and application rates (Berndes et al. 2004). Salix has a great potential for cleaning moderately contaminated soil from heavy metals and can for instance clean former agricultural fields from Cd-contamination caused by the application of P-fertilizers (Baum et al. 2009).

**Biodiversity**

*Effects on the diversity of plants*

Poplar and willow plantations established on abandoned agricultural fields will gain in floristic diversity (Weih et al. 2003), whereas plant diversity will decrease in intensively managed plantations on former forest sites (Berg 2002; Halpern & Spies 1995). To establish Salix plantations, herbicides are often applied (Energimyndigheten 2011). Application of herbicides has negative effects on the diversity of plants.

*Effects on the diversity of birds and mammals*

Poplar and willow plantations were shown to increase diversity of birds (Göransson 1994; Skärbäck & Becht 2005; Gustafsson et al. 2009), game animals (Göransson 1998) and field hares, rabbits, weasels and shrews (Skärbäck & Becht 2005) in a landscape that was dominated by agriculture. On the other hand, birds that prefer open land may decrease when arable fields are transformed into willow plantations (Skärbäck & Becht 2005).

*Effects on soil biota*

It has been suggested that energy forests in the otherwise monotone agricultural landscape may act as refuges and dispersal corridors for predatory arthropods that are often useful for the control of pest insects (Björkman et al. 2004). The number of earthworms and the abundance of ectomycorrhizal fungi increased whereas carabid and spider density decreased in willow plantations on former arable land (Baum et al. 2009). Lagerlöf et al. (2010) found higher abundance and biomass of earthworms in *Salix* plantations (66–324 ind./m²) than in cereal fields (0.06–28 ind./m²). This result was constant over more than ten years, irrespectively of crop rotation in the cereal fields or harvest in the energy forests. Minor & Ciunciolo (2007) found that diversity of oribatid mites was lower in willow plantations than in abandoned fields and deciduous forests but slightly higher than in corn fields. Minor & Norton (2003) carried out a comprehensive experimental study on the effects of different forms of fertilizer in willow (*Salix x dasyclados*) plantations on mites. Different types of fertilizer had different effects on the two groups of mites. Effects on abundance and species richness could be positive, negative or neutral. In general, chicken manure increased the numbers and species diversity of mites compared to control treatments. Application of biosolids had positive effects on predaceous mites whereas the number and species diversity of oribatid mite decreased when biosolids were applied.
Lack of knowledge

It is poorly known how climate changes and increased demands for renewable energy will influence decisions on tree-choices for energy production. At present, SLU is working on the cultivation of Salix varieties that are adapted to a warmer and dryer climate (Martin Weih, Dept. of Crop Production Ecology, pers. comm.). Soil biota assemblages may differ in willow and Salix plantations and soil microbial communities differ even between willow varieties (Baum et al. 2009). Very little is known about the relation between tree species or variety, soil biota and changing climatic conditions.

Systems perspectives Energy forests

Biomass from energy forests, normally Salix spp. can be used to substitute fossil fuels and hence potentially decrease the rate of climate change (Fig. 16). The impact on biodiversity depends entirely on former land use, i.e. a switch from cultivation of arable crops to energy forest normally has a positive effect on biodiversity, whereas a switch from e.g. a pasture would have negative effects. For a full picture of the environmental impact, resource use and emissions related to production of fertilizers should be included, as well as those related to any machinery used in the cultivation. Both resource use and emissions, however, can be expected to be lower per unit output than in the case of arable crops. Some deciduous energy-forest species have the ability to assimilate not only nutrients, but also heavy metals, and are therefore useful as multifunctional systems for treating waste products such as municipal wastewater and sewage sludge.
Positive effects: Biomass production, increased soil carbon stock, nutrient and heavy metal assimilation, positive biodiversity impacts in large-scale agricultural landscapes.

Negative effects: Negative biodiversity impacts in heterogeneous landscapes, disturbance of recreational values (most people do not like energy forests).

Potential win-win effects: Climate change – biodiversity; climate change – eutrophication.

Potential goal conflict: Substitution potential and carbon sink – recreational values.

Identified knowledge gaps: Effects of large-scale rotation coppice production on dispersal of organisms, including pest species and predators, invasive species and biodiversity in general. Little is known about whether Salix can act as an invasive species.
5. Reindeer grazing and biodiversity

In summary; grazers, including reindeer may inhibit climate-driven expansion of deciduous shrubs and may counteracts the effects of warming on the changes in vegetation. The effects of grazing on vegetation and fauna diversity is very complex and depends on grazing pressure, habitat type and climate and may differ between winter- and summer-grazing.

Introduction

The mountain ecosystems in northern Europe have been grazed by reindeers since the end of the last glacial period and the mountain vegetation is a result of a long grazing history (Moen et al. 2009). Reindeer grazing occurs at 16.6 MHa, that is 40% of Swedish land-area (Bernes 1994). In general, grazing may affect plants and soil organisms through three mechanisms: herbivory, production of dung and urine and trampling (Sørensen et al. 2009). Grazing may increase or decrease plant and invertebrate diversity depending on grazing intensity, timing of grazing, productivity of the site and biome (forest or tundra).

Biodiversity

Plants

It has been observed that trees and shrubs have expanded in some arctic regions. This may be due to warming or to increase precipitation during the last century. Increased plant biomass followed warming treatments reduced cover of lichens (Cornelissen et al. 2001). Grazers, including reindeers may inhibit climate-driven expansion of deciduous shrubs (Olofsson et al. 2009), and other changes in vegetation in Northern Europe and may thus be important for preserving the vegetation in some parts of northern Sweden. In short, grazing may counteract the effects of warming on the changes in vegetation.

During winter, reindeers mainly graze on lichens whereas grass and other green vegetation are important food resources during summer (Bernes 1994). Reindeers are locally kept in high densities and overgrazing and trampling have affected plant richness negatively. For instance, 90% of lichens cover in Lövhög, Härjedalen is disturbed by trampling (Bernes 1994). Trampling by reindeer is reported to reduce also the abundance of shrubs, dwarf shrubs, grasses and forbs. On the other hand Kullman (2009, and citations in that paper) showed that plant species richness in alpine ecosystems in Scandinavia increased from 1950 until now, possible in response to climate warming and despite intensive reindeer grazing.

Grazing often reduces the abundance of dominant plant species and increases the overall plant diversity (Suominen & Olofsson 2000) and, thereby, reindeer grazing can preserve open tundra by inhibiting the expansion of deciduous shrubs. In a long-term experiment (50 years), reindeer grazing during summer decreases the abundance and height of willows (Salix lappo-
num) and cover of vascular plants compared to winter-grazing only (Olofsson et al. 2009). The effects of grazing on bryophytes may be positive, negative or neutral. Van der Wal (2006) suggested that reindeer grazing in the tundra shifts the vegetation from lichens-dominated to bryophyte-dominated. Kitti et al. (2009) found that the total cover of bryophytes was not affected by timing but the species composition was changed by grazing. The effects of grazing on vegetation in different forest or tundra types are poorly investigated (Pajunen et al. 2008; Olofsson et al. 2009).

Soil biota
The effects of reindeer grazing on other organisms than plants are poorly investigated (Suominen & Olofsson 2000). But Sørensen et al. (2009) showed that simulated reindeer-grazing affected the abundance or biomass of several organism groups, including microbes, springtails, nematodes and potworms. However, the effects were not consistent between the treatments (simulated grazing, trampling and fertilization). Suominen & Olofsson (2000) showed that the abundance of most invertebrate groups was higher in grazed plots. But the effect of grazing was modified by vegetation type. Cockroaches, money spiders, mites and snails were negatively affected by grazing in plots with Cladonia cover. High grazing pressure may decrease the abundance of Cladonia (Göran Thor, SLU pers. comm.) and thereby some arthropod groups may negatively affected. The microclimatic conditions in grazed sites may be warmer and dryer and may be hostile for arthropods that are sensitive for desiccation.

Effects of reindeer grazing on nutrient cycling
Reindeer grazing has a positive effect on N availability in the soil, mainly due to the production of faeces and urine. Activity of bacteria is usually high in dung, thereby stimulating mineralisation processes (van der Wal et al. 2004).

Climate change and grazing by reindeer, lacks of knowledge
In short, reindeers may be important to conserve plant diversity in northern Europe and to counteract the changes in vegetation caused by climate change. Grazing effects of reindeer may by complex and should be studied across different habitat types (Pajunen et al. 2008; Moen et al. 2009) and across different grazing regimes such as summer and winter grazed areas (Olofsson et al. 2009) and grazing intensity. Data on summer grazing effects are scarce (Moen et al. 2009). Cornelissen et al. (2001) showed that experimental warming may decrease the abundance of macrolichens, staple food for reindeers during winter. This might be a serious problem because reindeer are limited by winter forage (Moen et al. 2009). On the other hand, increased summer precipitation may have positive effects on the growth rates of lichens (Cornelissen et al. 2001). Greater snow depth hampers reindeer from reaching lichens (Bernes 1994). Decreased snow depth and elevated temperature may lead to increased grazing pressure on lichens during winter whereas increased summer precipi-
tation may have positive effects on lichens. Cornelissen et al. (2001) suggested that environmental change may alter the foraging behaviour of reindeers and thereby the competition between lichens and vascular plants. The relations between reindeer grazing, climate change and plant and insect diversity in northern Europe are not yet understood (see e.g. Olofsson et al. 2009; Kullman 2009).

Grazing by reindeer affects biodiversity, often positively, especially in areas that suffer from increased abundance of broad-leaved vegetation due to climatic changes. Conflicts are possible in future: the area that is suitable for reindeer grazing may decrease due to a warmer climate, but also due to demands for agricultural development.
6. Climate change
– related changes in land use – implications for terrestrial ecosystem functions and threatened biodiversity

6.1. Summary
This part of the synthesis project aims at analysing effects of a number of climate-change related land use measures on biodiversity indicators that are linked to ecosystem function and favourable conservation status. The study is based on a brief literature review.

6.1.1 Definitions and delimitations
For this study we define biodiversity as it relates to the functionality of ecosystems, including biotope-specific ecological processes, species interactions, microclimatic conditions, and the biotopes’ and landscapes’ contents of habitats for species and of viable populations of typical species.

We use specialist, red-listed, and declining species as assumed indicators of ecosystem functionality and conservation status. Such species can be expected to be particularly dependent on, for example, high abundance, connectivity, and continuity of highly specific or clusters of habitats (in turn generated by specific processes, disturbances and conditions at the biotope or landscape level), and of high interdependence in species interaction webs. Such species may also be connected to biotopes which have already become rare and which, thus, indicate a need for preservation and restoration measures at the national level.

6.1.2 Method
The analysis has included three major steps:
- Identification of plausible land-use changes at a degree of accuracy that can be readily linked to certain ecosystems, biotopes, species groups etc. The accuracy is however not high enough to quantify the biodiversity effects. The identification was done through screening of reports and other documents from national and regional sector authorities, NGOs, and production companies, of governmental directives etc. Focus has been on Swedish conditions and Swedish land use.
- Evaluation of the effects on biodiversity based on a brief review of scientific and non-scientific literature dealing with biodiversity effects of land use.
- Discussing patterns in the effects on ecosystems and threatened biodiversity in relation to greenhouse gases.
In order to fit the sub-project to the budget, some general land-use-measures were omitted, e.g. nitrogen fertilization in forestry and agriculture, liming in agriculture, and reindeer husbandry.

Reindeer husbandry would to CBM’s opinion require and deserve a thorough separate synthesis both of the links between biodiversity and reindeer grazing (cf. Linkowski & Lennartsson 2006a,b), and of how reindeer husbandry may be affected by and adapt to climate change. The latter should include climate effects on the ecology of the grazing habitats, on the economy and practice of reindeer husbandry, and on other land-use which affects reindeer husbandry during summer, winter, and migration, such as forestry and tourism.

6.1.3 Results
The following land-use measures were identified as plausible in a climate-change perspective, based on literature.

Choice of tree species in production forest
Phasing-out of Norway spruce in southern Sweden
  Increased use of Scots pine and deciduous trees
  Increased use of non-indigenous tree species
Changed use of exotic tree species in forestry
  Increased planting of exotic tree species
  More restricted use of exotic species
Changed use of mixed tree species composition
  Increased use of mixed stands
  Increased use of homogenous stands

Wetland draining in the forest landscape
Changed practices for wetland drainage
  Stopped draining of undrained wetland
  Stopped new draining and reduced protective draining and ditch cleaning
  Increased draining of formerly drained, but still impediment wetland
  Increased draining of undrained wetland

Stand continuity in forestry
Increased use of forestry without clearcutting (Continuous cover forestry, CCF)
  Some of the clearcutting is replaced by CCF
  CCF instead of protection in some continuity stands and value-cores
Changed length of rotation period
  Clearcutting in production forest is delayed by 1–2 decades
  Clearcutting is shortened in production forest
  Logging is delayed 1–2 decades in old-growth forest
Changed policies for protection of value-cores for biodiversity
Higher proportion of old-growth stands and other value-cores protected
Lower proportion old-growth stands and other value-cores protected
Increased use of temporary reserves

*Considerations to biodiversity in forestry*

Changed routines for removal of dead wood, predominantly of Norway spruce
No change of removal of fallen spruce wood
Increased removal of spruce wood, even in some protected areas
Use of logging residues as biofuel
Continued and further increased use of LR as biofuel

*Choice of crop on farmland*

Changed use of ley on arable land
Ley replaces other crops on an increasing proportion of the arable land
Increased use of (new) annual crops on the cost of ley

*Changed draining and use of wetland in the agricultural landscape*

Changed use of grass from wetland habitats
Increased use of wetland grass for biogas production
Changed draining in the agricultural landscape
Increased transformation of wet arable land to wetland habitats
Increased draining of arable land and wetlands.

The effects of the identified land use measures on biodiversity are presented in paragraph 2 at the end of this appendix.

### 6.1.4 Conclusions and discussion

**LAND USE CHANGES RELATED TO CLIMATE CHANGE**

The identification of plausible land use changes, more or less caused by climate change, showed that there are few direct links between climate change and land use change. On the contrary, *(1) land use may in many cases change in either of two opposite directions.* Furthermore, we can in many cases *(2) expect totally different effects on threatened biodiversity also of land use change-alternatives that are not totally different.* This implies that it is in general difficult to predict effects of climate change on biodiversity.

One example of the *first type* *(1)* is changed draining in the agricultural landscape, which is expected to be the result of climate change. Both a dryer landscape (more draining to counteract climate effects on hydrology) and a wetter landscape (some areas become too difficult to drain) are plausible future scenarios. Some of the climate change-related drivers affect agricultural practices directly, e.g. changed precipitation patterns and less (in time and space) frozen ground. Other drivers act indirectly, either through market econ-
omy (e.g. intensified production/drainage due to higher production potential) or through economic incentives generated by policy measures (e.g. more wetland due to subsidies for nutrient retention and biogas production). The final choice of land use will depend mainly on political decisions and the following policy measures, and to some extent on policy decisions by the land use actors. Greenhouse gas balance, nutrient leakage, and biodiversity effects will be important foundations for such decisions regarding draining in the agricultural landscape.

One example of the second type (2) is the phasing-out of Norway spruce in southern Sweden which may be performed, for example, by increased use of deciduous trees and Scots pine, or by increased use of exotic tree species, e.g. Logdepole pine and Sitka spruce. The first alternative will favour threatened biodiversity compared to today’s situation, the second alternative will have strong negative effects.

The identified land-use changes represent both adaptation to climate change and mitigation of either further global warming (through measures to reducing levels of greenhouse gases), or of secondary effects of global warming (e.g. mitigation of nutrient leakage to aquatic ecosystems). Adaptation measures will be performed in one way or another, driven by production economy, whereas most mitigation measures depend on policy measures.

ECOSYSTEM APPROACH – HOW CAN ECOSYSTEMS CONTRIBUTE TO REDUCING THE ATMOSPHERIC LEVELS OF CARBON DIOXIDE?

As mentioned above, some of the identified land-use measures may affect the levels of greenhouse gases. For some ecosystems, for example some wetland types, also other greenhouse gases than carbon dioxide are highly relevant, and should be considered in system analyses, e.g. of LCA type. Here, only the carbon balance is discussed in terms of ecosystem services in relation to land use and biodiversity.

It is obvious that effects on carbon dioxide levels are based on three types of ecosystem services.

1. Carbon stock, i.e. long-term lock-up of carbon stored in ecosystems. This ecosystem service contributes to reducing the emission of carbon to the atmosphere both in a short-term perspective, e.g. the time perspective agreed in the Kyoto protocol (emission goal for 2020 and vision for 2050), and a long-term perspective (for most ecosystems there is no time limit for how long carbon can be stored).
2. Carbon storage (sink), i.e. increase of ecosystem carbon stocks which can in turn be long-term maintained (see 1, above). This ecosystem service contributes to compensating for the release of carbon to the atmosphere by other sources and by earlier release of carbon from ecosystems, e.g. earlier logging. The time perspective of this ecosystem service includes two aspects. First, in some ecosystems, e.g. peat-forming wetlands, the storage is continuous, while in other
ecosystems the storage is temporary, although sometimes of long
duration (as in forest ecosystems). Second, storage can be trigged by
biomass harvest or other disturbances and ecosystems differ in how
fast they start storing carbon after the disturbance and how large the
storing-peak is.

3. Provision of a renewable carbon sources for the society, i.e. supply of
carbon that sooner or later becomes re-stored in the ecosystems in a
form that can be repeatedly harvested. This ecosystem service con-
tributes to offering a potential for substituting fossil carbon sources
for fuels and materials. The time perspective for this ecosystem
service differs between ecosystems in terms of how fast the released
carbon is re-stored in the ecosystem. Harvest of annual tissue, e.g.
grass, is compensated by re-storage already the following season. In
boreal forest there is a considerable time-lag between release and
re-storage of carbon, normally a rotation period (c. 50–120 years).
Use of renewable carbon sources only does not automatically lead to
substitution of fossil carbon.

LAND-USE PREREQUISITES FOR THE CARBON-RELATED
ECOSYSTEM SERVICES

The three types of ecosystem services require different land-use.

1. Carbon stock (1) requires land-use to maintaining the biomass in
carbon-rich ecosystems, for example in old-growth forest and peat
wetland. The latter ecosystem continues storing carbon (2) whereas
old-growth forest eventually reaches a release-storage balance.
Wetland requires intact or (for drained wetland) restored hydrology,
old-growth forest requires protection from logging.

Theoretically, some proportion of the release component of this
balance, the dead trees, could be utilised as a carbon source, thereby
adding a substitution potential service (3) to the stock service. In
old-growth forest such an extraction of biomass would have a strong
negative effect on biodiversity (see 1.4.3). It would also affect the
storage component through changed recruitment and growth condi-
tions (e.g. Arnborg 1942, Hofgaard 1993, Hörnberg et al. 1995,
Sernander 1936). The possibility of creating production forest which
both maintain high carbon stock and provide a potential for substi-
tution of fossil fuels is discussed in 1.4.4., below.

2. Carbon storage (2) requires land-use to allowing biomass-accumulat-
ing ecosystems to continue their carbon storage function, and to
maintaining the carbon that become stored in the ecosystem (1) in a
time perspective relevant for societal climate goals. Examples of
ecosystems are wetland that requires intact or restored hydrology,
and growing forest which requires protection from logging.
This ecosystem service may be decomposed into amount of stored carbon and rate of carbon storage, respectively. The latter is relevant for the time perspective of carbon storage. It adds one land-use prerequisite to the two (continued storage and maintenance of the stock) mentioned above, namely land-use measures that increases the storage/release ratio. This implies that carbon storage (2) is favoured by measures to increase productivity and may be favoured by some carbon harvest (3) if the harvest increases the carbon storage rate without negatively affecting the total amount of stored carbon.

3. Provision of renewable carbon (3) requires land-use to extracting biomass and to facilitating the regrowth of new biomass. The shorter the harvest intervals, the faster the re-storage and potential substitution of fossil carbon. Examples of ecosystems are grassland (rapid re-storage of released carbon), semi-natural coppice woodland (intermediate rate of re-storage), and boreal production forest (slow re-storage).

Substitution also requires policy measures to encouraging substitution of fossil fuels by biofuels. Further to counteracting substitution of other renewable energy sources (for example renewable energy that is expensive to produce), and increased use of fossil fuels (for example if biofuels contribute to reducing the oil price or if profit from biofuels is invested in use of fossil carbon sources).

LAND USE MEASURES FOR GREENHOUSE GASES – EFFECTS ON THREATENED BIODIVERSITY
General patterns
The land use measures analysed in this sub-project are here grouped according to their influence on greenhouse gases, mainly based on the categories in 1.4.2., above. Also the reduction of nutrient leakage is added as a group. For each group the biodiversity effects are discussed.

a) Threatened biodiversity is positively affected by measures which minimise the release of carbon stored in ecosystems, mainly in wood, soil, litter, and peat. There is thus a positive correlation between threatened biodiversity and maintaining of the carbon stock (1). There is also a positive correlation between biodiversity and the carbon lock-up component of carbon storage (2).

The correlation is due to the strong connections between biodiversity and the two habitat groups wetland and old-growth forest. The effect is strongly depending on age and continuity of the habitat in question. Preserving of carbon stored in old forest and continuity forest has larger positive effects than carbon preservation in younger forest and forest originating from clearcutting. The effect is also strongly depending on the ecological functionality of the ecosystems,
in terms of natural processes and the habitat structures that are formed by the processes. This implies that there is only restricted room for production in such habitats if biodiversity is not to be lost. For example, in old-growth forest the release-storage balance discussed above is an expression of the basal ecosystem processes that form all aspects of biodiversity in this forest ecosystem.

It should be noted that the release-storage balance in an old-growth forest is heterogeneous and dynamic, i.e. that pulses of biomass decay (carbon release) occur at different spatial scales following, for example fires, storm, insect outbreaks and flood. Such processes are different from harvest of carbon as in b) below.

b) Threatened biodiversity is affected negatively by measures which release carbon which has been stored during long time periods in ecosystems, mainly wood, litter, and peat. There is thus a negative correlation between threatened biodiversity and provision of renewable carbon from “old” wood tissue, litter, and peat. Concerning “young” biomass, see point d) below.

The effect is an inverse of the previous point, i.e. the harvest breaks the release-storage balance, the hydrological continuity, etc, to which biodiversity is strongly connected.

c) Threatened biodiversity is affected negatively by most measures which increase the rate of carbon storage through increased growth rate of the vegetation, for example of trees. There is thus a negative correlation between threatened biodiversity and the rate component of carbon storage (see point 2).

Increased growth rate per area unit can be achieved through fertilization, draining, use of fast-growing exotic tree species, and by keeping a high proportion of the trees and stands at a fast-growing (ecologically young) age. The latter measure is affecting biodiversity because of the age-dependence of biodiversity, as described in a). The other measures can be considered to change the basic ecological conditions of the ecosystem, i.e. to replace the original processes and structures by new ones, to which few of the ecosystem-specific species are connected.

d) Threatened biodiversity is affected positively by frequently repeated biomass harvest which release carbon that has been stored through annual or short-term regrowth of plants (not recruitment of new individuals as in point e) in traditional semi-natural ecosystems (see below). There is thus in certain habitats a positive correlation between threatened biodiversity and provision of renewable carbon if the carbon emanates from “young” regrowth biomass, i.e. herbaceous plant tissue and young wood tissue emanating from short-rotation cutting based on shooting.
In this analysis only grassland use is evaluated (see changed draining and use of wetland in the agricultural landscape, below), i.e. the harvest of herbaceous biomass. Regular harvest of woody biomass is not evaluated. The combined use of herbaceous and woody biomass in coppice meadow is rather well known regarding biodiversity, whereas coppice woodland is less known.

Semi-natural is here defined as man-made or strongly man-influenced biotopes and landscapes which through long history of traditional land-use have formed characteristic biodiversity, built up by spontaneous colonisation of “wild” species.

In order to form biodiversity-rich semi-natural ecosystems, biomass harvest needs to be frequent enough to provide habitats that are ecologically characterised by dynamic stability rather than succession. The harvest of herbaceous biomass needs to be more or less annual, the harvest of wood to having a frequency of 1–2 decades (Helldin et al. unpublished, Lennartsson 2010). Such frequent harvest requires biomass production to being based mainly on shooting of perennial plant individuals rather than on mortality and replacement of individuals as in the normal Swedish forestry. Herbaceous perennial plants and shooting deciduous trees and shrubs fulfil those requirements. The positive effects on threatened biodiversity are due mainly to the specific competition conditions that occur in frequently disturbed habitats, and the effects are probably larger if harvest of wood biomass is combined with harvest of the field layer.

e) Threatened biodiversity is affected negatively by frequently repeated harvest which release carbon that has been stored through recruitment (not regrowth as in point d) of new young trees, for example through short-rotation forestry based on conifers. There is thus a negative correlation between threatened biodiversity and provision of renewable carbon if the carbon emanates from young wood tissue based on short-rotation forestry and mortality/recruitment of tree individuals.

The effects on threatened biodiversity are estimated based on literature dealing with high-productive short-rotation forestry (see stand continuity, below). The effects are due to reduction of age- and continuity-dependent ecological processes and structures, without providing dynamic stability as in traditional semi-natural ecosystems (point d).

f) Threatened biodiversity is affected positively by some and negatively by some measures which even out pulses in the release and storage of carbon in ecosystems. Examples of measures being positive for biodiversity are increased species diversity of trees (reduces risks of large-scale felling by storm and insects) and more varied tree age
distribution in stands (reduces risk of storm-felling and provides a potential for continuous cover forestry as an alternative to clearcutting). One example of measures having a negative effect on biodiversity is prevention of naturally induced forest fires.

g) Threatened biodiversity is usually affected positively by measures which reduce the leakage of nutrients to streams, lakes, and seas. For some measures this positive effect is landscape-dependent (see choice of crop on farmland, below).

Specific land-use measures

Description of biodiversity effects as sketched for groups of measures in the previous paragraph is usually difficult to apply on specific land-use measures, for two reasons:

- In order to evaluate, qualitatively and quantitatively, effects of land-use measures on threatened biodiversity, the measures need to be defined in a degree of detail that matches the scale at which species are linked to their habitat, mainly in terms of their affiliation to habitat processes, structures and element, components of fragmentation etc. For example, in order to perform a complete evaluation of the effects of increased proportion of deciduous trees in forestry stands, it is necessary to define tree species composition, age distribution, biotope, region, logging regime etc. Changes of each of these variables can potentially change the outcome of the measure from positive to negative, which makes it difficult to present a clear statement of the measure’s biodiversity effect.

  The analyses performed in this sub-project have necessarily been less detailed, thus providing a general picture rather than a complete measure-response description.

- The effect of a certain land-use measure depends largely on the starting point, i.e. on the state which is changed by the land-use measure. For example, continuous cover forestry is positive for threatened biodiversity if applied as an alternative to large clearcuttings, but strongly negative if applied as an alternative to protection of continuity forest.

SYNERGIES AND CONFLICTS – CHALLENGES AND SUGGESTIONS FOR FUTURE LAND USE

Types of synergies and conflicts

There are two types of synergies and conflicts that need to be considered.

1. As shown in the above summaries, synergies and conflicts obviously occur between effects on greenhouse gases and effects on biodiversity.
2. There are also, irrespective of biodiversity impact, synergies and conflicts between different goals for the greenhouse gases, and between different production goals.

Both aspects of the type 2 conflicts contribute to the fact that there are few straightforward relationships between climate change and the choice of land-use measures by the forestry and agriculture. This can be exemplified by the choice of tree species by the forestry (cf. SOU 2007:60b). As discussed in 1.4.1., above, there are usually different ways of dealing with a certain climate change effect. Two aspects in particular are subject to alternative approaches, in turn leading to different policies and forestry practices: (1) the trade-off between long-term economic security and higher but less secure profit; (2) whether we focus on (short-term) carbon release from old forest, carbon storage in young forest, or long-term potential for substitution. Since none of these approaches is obviously correct from all aspects, the choice of approach differs between, for example, organisations and society sectors, and are often the result of opinion-based decisions. For example, in the Climate policy of the Swedish Forest Agency (Skogsstyrelsen 2009c) a 100-yr perspective is considered more important than more short-term perspectives (e.g. the Kyoto Protocol’s emission goal for 2020 and vision for 2050). Potential for substitution in a 100-yr perspective is considered more important than the maintaining and increasing of stored carbon by the present forestry. Long-term production security is considered more important than short term growth in most cases.

Synergies and conflicts identified in his sub-project, and implications for future land use.

1. Stopped logging of old-growth “natural” forest and old continuity forest, and stopped draining of wetland:

   Synergy between
   Maintenance of the ecosystems’ carbon stock in a short and long time perspective
   and
   Threatened biodiversity

Comments and implications for land use: This measure is essential for preserving forest biodiversity, and for the reduction of carbon emissions from forest ecosystems in both a shorter time perspective, e.g. as suggested by the Kyoto protocol, and a longer perspective. Wetland is all types of ecologically functional wetland, also wetland that are affected by unsuccessful drainage.

Less than 10 per cent of the Swedish forest area function as long-term carbon stock today (more than 90 % is part of a “release now, re-store later” rotation cycle), and as functional forest ecosystems. It therefore seems reasonable that the remaining value-cores for biodiversity are protected from logging
and draining as combined biodiversity hot-spots and carbon stocks, together
with some old forest with potential for developing ecological functionality (see
also 2, below). Note that protection does not exclude conservation measures
such as selective cutting, grazing, and burning.

2. Considerably delayed cutting of old forest:

   *Synergy* between
   Maintenance of the ecosystems’ present carbon stock in a short and long
time perspective, as well as continued carbon storage in a medium time
perspective
   and
   Threatened biodiversity

*Comments and implications for land use:* See previous point. By leaving old
forest which is still in a phase of increasing its biomass both the present carbon
stock will be maintained (as in the previous point) and will the net uptake
of carbon continue instead of being released as in logging. In a shorter time
perspective this is an efficient carbon-storing function compared to storing in
young stands which have a low or negative net uptake because of carbon loss
following clearcutting (see next point). The effect on biodiversity is temporary
at the stand level but may contribute to enhancing effects of general considera-
tions to biodiversity in forestry through longer recovery time. It would also be
an important measure for restoring forest biodiversity as suggested by national
forest strategies, e.g. as a way to counteracting fragmentation.

   Given the synergy described here, the need for biodiversity restoration, and
the climate goals, delayed logging of some old forest may be motivated. The
delay period could for example correspond with the climate goals, i.e. delay
until 2020 or 2050. The measure should focus on stands in which biomass,
growth, and biodiversity potential are as high as possible (see previous point).

3. Logging of old-growth and old forest:

   *Conflict* between
   Release of carbon that provide a potential for substitution in a medium or
long-term time perspective
   and
   Threatened biodiversity
   together with
   Maintenance of the carbon stock and in some cases the ongoing storage of
carbon

*Comments and implications for land use:* This represents the general conflict
between the two approaches “stop release-stop storage” and “release now-
re-store later”. The explanations for the long delay of re-storage (a full rota-
tion cycle for full re-storage and a number of decades for net storage) is that
the production is based on long rotation cycles and that there is a consider-
able loss of carbon from the clearcutting through decay of above- and below-
ground biomass.
Given that more than 90 per cent of the Swedish forestry is already functioning as carbon provider and given the negative effects of logging of old forest on threatened biodiversity, it seems motivated to not further increasing the area of logged forest on the cost of unlogged continuity forest. For the area used for forest production, measures to bolster the conflicts with short-term carbon emission goals and with biodiversity are needed. General considerations to biodiversity at logging solve only a smaller part of the biodiversity problem. Shorter rotation periods would reduce the time-lag problem but enhance the conflict with biodiversity (see point 4) unless new methods (see point 5) or proper areas for such short-rotation, are found. Improvement of substitution-based forestry should probably focus on both method development and on finding areas in which intense forestry could be acceptable from an ecosystem point of view. Such areas could be such in which biodiversity is already severely deteriorated.

4. **Short-rotation forestry:**
   
   **Conflict** between
   
   Faster re-storage of carbon following release of the carbon stock as a potential for substitution
   
   and
   
   Threatened biodiversity

   **Comments and implications for land use:** See previous point. The effects on threatened biodiversity depend on which biodiversity that is present in the stands, on the need for restoration of biodiversity etc. It is probably possible to find areas in which short-rotation forestry is acceptable, especially if compensated by increased protection of value-cores for biodiversity.

5. **Short-rotation harvest of wood for biofuel in semi-natural coppice woodland:**
   
   **Synergy** between
   
   Faster re-storage of carbon following release of the carbon stock as a potential for substitution
   
   and
   
   Threatened biodiversity
   
   **together with**
   
   Reduction of carbon loss following logging

   **Comments and implications for land use:** This synergy contributes to solving three of the conflicts in the previous points: the problem with delayed re-storage which creates a conflict between logging for potential substitution and short-term emission goals (point 3); the problem with carbon release due to decay of dead stumps and root systems which further delays the re-storage function of growing stands following clearcutting (point 3); and the negative effects on threatened biodiversity of logging for substitution in general and short-rotation forestry to speed up the re-storage in particular (points 3,4).
Biofuel production based on coppicing has so far focused entirely on *Salix* on arable land. In order to develop biodiversity-rich ecosystems with fair production potential, a considerable method development is needed. Important components of such development are area potentials, plantation and harvest methods, tree species choice, bio-historical background, production economy, and carbon-nutrient budgets in a LCA perspective. Combinations with food production may be possible, in particular production based on grazing animals.

6. Harvest of grass for biofuel or other production in semi-natural grassland:

*Synergy between*
Immediate re-storage of carbon following release of the carbon stock as a potential for substitution
*and*
Threatened biodiversity
*together with*
Reduction of nutrient leakage

*Comments and implications for land use:* Grass from semi-natural grassland is produced without input of fossil fuels and fertilisers. It can be used for biofuel and meat production, the latter delivering manure as a by-product which can in turn be used for production on arable land. All semi-natural grass thus contributes to reducing nutrient leakage compared to grass produced on arable land. Through harvest of grass from alluvial wetland also nutrients in the water can be retained. Grass from ley provides the same immediate regrowth but suffers from problems with nutrient leakage and absence of obvious positive biodiversity effects.

The potential for use of semi-natural grass is regionally high in Sweden. The basic methods for harvest and use are known, but further development of methods is needed in order to optimise biodiversity and economy, and in order to identify the most important areas from a nutrient retention point of view.

7. Fertilization and other measures to increase growth rate in forestry and agriculture:

*Conflict between*
Increased carbon storage rate
*and*
Threatened biodiversity

*Comments and implications for land use:* Fertilization in forestry affects biodiversity in the fertilised area and to some extent in recipient waters. Fertilization in agriculture mainly affects water and wetland since the arable land that is subject to fertilization is already deteriorated from a biodiversity point of view. Fertilization is necessary in order to compensate for nutrient losses following collection of logging residues.
Both in forestry and agriculture fertilization should be performed in order to minimise negative effects, which in forestry implies that some areas should be excluded from fertilization. Conversely, it may be possible to find areas in which fertilization has little negative effect, and which may be used for more intense production (cf. point 3).

8. Continuous cover forestry as an alternative to clearcutting:

Synergy between

Slight reduction of carbon loss from decaying biomass in the litter following logging

and

Threatened biodiversity

Comments and implications for land use: The effect on biodiversity would be negative if CCF replaces protection of continuity forest. The effect on carbon release is probably limited.

The potential for CCF is currently restricted by limited availability of suitable stands. Transformation of stands to suitable state will take time and have negative effects on production economy, but may at least locally be motivated by positive effects on biodiversity.

6.2 Detailed results: effects of climate-change related land-use changes on threatened biodiversity

6.2.1 Choice of tree species in production forest

In all types of forestry, based on clearcutting as well as on stand continuity, the choice of tree species is important for biodiversity. One reason for this is that large proportions of the forest species of e.g. insects, fungi, lichens, and bryophytes use tree structures as habitat and are more or less confined to single or taxonomic groups of tree species (Ehnström & Axelsson 2002; The Swedish Species Information Centre’s Data sheets on threatened species; www.artdata.slu.se). Tree structures are, e.g., leaves, wood and bark, and live tissue in bark/stem and the differences between tree species are due to, e.g., bark structure, pH in bark and trunk runoff, wood chemistry, and growth form. Some tree species are more species rich than others, but because of the tree-specificity a species-rich tree species cannot replace a species-poorer tree species. For each tree species the value for biodiversity is highly depending on the growth conditions of the trees (in particular in terms of productivity, light conditions, and moisture conditions), and on the age of trees (Ehnström & Axelsson 2002). Thus, trees do not automatically provide habitats for species, but the proper conditions are needed. Most of the tree-connected biodiversity is demanding or is favoured by sun-exposed conditions and old-growth trees whereas forestry aims at dense stands with little exposure and at early har-
vest of trees. Therefore, increased variation in tree composition in production forest is not necessarily fully matched by increased variation in biodiversity.

Another reason is that different tree species create different habitat conditions in terms of soil conditions (e.g. the difference between podsol and brown earth), temperature, precipitation etc.

Most exotic tree species harbor only restricted proportions of the tree-dependent biodiversity compared to indigenous trees, and exotic tree species thus contribute little to the maintenance of forest ecosystem functions and biodiversity. On the contrary, some exotic tree species are locally invasive and may therefore outcompete or have other have negative effects on native tree species. Also active replacement of native species by exotic, e.g. replacement of spruce and pine by planting of lodgepole pine, gives these types of effects on biodiversity. Exotic tree species are thus considered negative for biodiversity because they constitute a more or less serious threat to forest biodiversity and ecosystem function, not because of their non-native origin per se. For practical reasons, however, exotic tree species are usually defined by origin rather than by threat level (e.g. Swedish Board of Agriculture 2010).

The choice of tree species in forestry is mainly based on production economy, in turn the result of productivity potential in specific forest areas combined with market conditions. Soil- and region-specific growth rate (including damage from moose browsing etc) combined with quality-specific market price has so far been important drivers for the choice of tree species. Deciduous tree species have systematically been suppressed in production forest in large parts of Sweden, in favour of coniferous trees including exotic species as Lodgepole pine. As a result, the forest landscape has been drained of deciduous trees which has drastically decreased the potential for preserving forest biodiversity (Axelsson et al. 2002), and most likely with profound impact also on the ecosystem function of forest ecosystems (cf. Bengtsson et al. 2000, Johnson et al. 2003). Deciduous trees are usually absent or sparse in older production stands, and in younger stands, before selective thinning, only birch is still rather common. Coniferous trees, especially Norway spruce, has also been favoured through extensive planting, both within and south of the spruce’s natural distribution area in Sweden (SOU 2007:60, Appendix B18).

The tree composition in Swedish forests is also affected by grazing and browsing of moose, roe deer etc, which is strongly decreasing the proportion of surviving saplings and shoots of deciduous trees, and the economic output of pine cultivation. However, studies in Uppland has shown that the density of several deciduous tree species is considerable until the first and second thinning in spite of hard browsing pressure (Upplandsstiftelsen unpublished data). In southern Sweden, also pine has decreased drastically on more productive soils due to the forestry’s focus on spruce (Classon 2008). This is considered a serious negative change and support programs for pine production has been suggested (Bergquist 2009).

Recently, also some drivers related to climate change have been discussed, of which three are being evaluated here.
Important gaps of knowledge

• A review is needed, probably combined with some new data, of which aspects of biodiversity that will be favoured by the tree composition *per se* in production forest, compared to the aspects that require specific qualities of trees, wood, stand structure etc.
• Browsing may be a threat to economic production of several tree species, but where is it also a threat to ecologically sufficient contents of those tree species?
<table>
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<th>Relation to climate change</th>
<th>Land use until today</th>
<th>Alternative land use changes</th>
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</thead>
<tbody>
<tr>
<td>Phasing-out of Norway spruce in southern Sweden</td>
<td>Norway spruce is assumed to be less economic in southern Sweden due to a set of interlinked processes: (1) increased drought-damage, (2) more favourable conditions for bark-beetles, (3) increased storm frequency.</td>
<td>Spruce is normally favoured and planted in boreal and boreonemoral regions. Deciduous tree species have systematically been suppressed. In southern Sweden, also pine has decreased drastically on more productive soils due to the forestry's focus on spruce. In spite of the anticipated risks in a warmer climate there are at present few indications on more cautious use of spruce in southern Sweden.</td>
<td>Alternative 1: Norway spruce is phased-out through increased use of planted and self-recruited Scots pine and deciduous trees in southern Sweden.</td>
<td>1° Positive effects on biodiversity connected to pine, deciduous tree species, deciduous forest stands (e.g. spring flowering field layer) and non-podsolated soils. The latter include in particular fungi. Shade-tolerant species and species on smaller wood and tree dimensions are likely to be favoured more than the majority of the red-listed species which need larger dimensions and sun-exposed wood and trees. Increased variation in production forest increases the potential for considerations to biodiversity in forestry.</td>
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<tr>
<td>Changed use of exotic tree species in forestry</td>
<td>Recently, the economic incitements for using exotic tree species to increase production has been complemented with arguments related to climate change. Increased growth rate increases storage of carbon and increased logging increases substitution in a certain time perspective. Further, a warmer climate is assumed to increase the interest in using new tree species in forestry.</td>
<td>Since 1980 an area of c 430 000 hectares has been regenerated with <em>Pinus contorta</em>, which now occurs on c. 550 000 hectares. There has been a strong decrease since the maximum in 1984 (c.40 kha), but an increase from the minimum in 2002 (1.7 kha) until today (6.2 kha in 2009). Other exotic species are of little, but increasing use. Planting of exotic tree species is allowed in exceptional cases (9§ svfo). Some changes of legislation has been suggested by the Swedish Forest Agency, e.g. restricted proportion of contorta plantation per land owner (25%) but increased acceptable national and altitudinal distribution.</td>
<td>Alternative 1: Increased planting of exotic tree species, e.g. <em>Pinus contorta</em>, <em>Larix spp.</em>, <em>Picea sitchensis</em>, hybrid <em>Populus</em>, and <em>Acer pseudoplatanus</em>. No control of spontaneous dispersal.</td>
<td>2° Reduced quantities of wood, trees and other substrates of indigenous tree species, and thus of the quantities of habitats for species connected to such substrates. Reduced potential for considerations to biodiversity in forestry and for the potential to create a biologically rich matrix between value-cores. See further 3° below.</td>
</tr>
<tr>
<td>Changed use of exotic tree species in forestry</td>
<td></td>
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<td>Alternative 2: More restricted use of exotic species which are known or suspected to disperse spontaneously and which therefore may be invasive in a warmer climate. Particular caution in habitats that may be susceptible for invasion, e.g. sand, high altitudes, and broad-leaved deciduous forest. Active control of exotic species in some habitats.</td>
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<tr>
<td>General land use change</td>
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<td>Changed use of mixed tree species composition in forestry</td>
<td>Increased damage risks of forest stands in a warmer climate may lead to recommendations for and use of more mixed stands. On the other hand, a focus on potential for substitution in a longer time perspective combined with increased growth potential in a warmer climate may motivate measures to increase the production by using homogenous stands of site-optimal tree species. The natural regeneration of several tree species, e.g. pine and oak, may be favoured by climate change. On the other hand, climate change may increase the populations of browsers, in turn decreasing the regeneration of trees. In the end, however, both browsers populations and stand composition are largely the result of policies and decisions for hunting and forestry, respectively.</td>
<td>The Swedish forestry has largely been based on homogenous stands with one or very few (normally only spruce and pine) tree species. Also in larger geographical scale large parts of Sweden have been homogenised, i.e. most stands in an area have been planted with the same or a few tree species. See further phasing-out of Norway spruce, above.</td>
<td>Alternative 1: Increased use of mixed stands (several tree species in a stand) or a mixture of stands (homogenous stands but different tree species in different stands). Alternative 2: Increased use of homogeneous stands, large stands, and homogeneity between stands in order to intensify production.</td>
<td>5' See 1' above. Increased proportion of deciduous trees in stands increases the spring light influx, potentially favouring the ground flora, epiphyte flora, and insect fauna compared with today's conditions. Still, however, shade-tolerant species and species on smaller dimensions are favoured more than the majority of the red-listed species which need larger dimensions and sun-exposed wood and trees. The same applies to a mixture of stands provided that the stand mosaic is fine-scaled. If increased mixture leads to more logging occasions species on newly dead fine wood may be favoured but perhaps with slight negative effects on some disturbance-sensitive species, e.g. species in litter. 6' Negative effects on ground flora, epiphyte flora, and insect fauna. The effects are less if the stands are composed of biologically rich tree species, e.g. oak. Reduced potential for considerations to biodiversity in forestry and for the potential to create a biologically rich matrix between value-cores. Strongly reduced range of ecological and structural variation, e.g. in terms of light influx, wood diversity, food plant diversity, soil heterogeneity. The reduced variation in turn leads to reduced ecosystem function.</td>
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6.2.2 Wetland draining in the forest landscape

Wetland habitats are characteristic for the Fennoscandian forest landscape and ranges ecologically from slightly moist and temporarily wet forest to bogs and fens that lack forest because of the hydrologic conditions. The Swedish flora and fauna is rich in species specific to wetland habitats, although rather few to one single habitat. The wetland species depend on specific combinations of soil conditions, air moisture, habitat structures etc. that are generated by the water. Wetlands also indirectly influence the habitats for species, for example by affecting tree species composition (e.g. alder, ash, and willow), the dynamics of forest stands, and the local climate. This implies that hydrology alone is not forming wetland habitats but natural processes in, for example, the tree dynamics are required as well.

The wetland biodiversity has declined due to drainage in both forest and the agricultural landscape. Hydrology often connects these two land use sectors. For example, large area of forest wetland was drained when lowland lakes and streams were regulated for agriculture in the 19th and early 20th centuries. The drainage is reflected in the red-list; for example, c. 40% of the red-listed birds, 15% of the beetles, and 13% of the bryophytes occur in wetland habitats (Gärdenfors 2010). The proportion of drained wetland varies between regions (Strömgren 2006) as well as between habitats. Wetlands close to agricultural areas, in lowland areas, and wetland habitats that are easily transformed to production forest, have in particular declined. On average, c. 15 % of the forested wetland area has been drained (Hänell 1990). Of the remaining wetland c. 80% of the objects are affected by draining (Gunnarsson & Löfroth 2009).

In the old agricultural landscape wetland comprised a resource and a variety of wetland habitats were mown for winter fodder, also in the forest landscape. Ceased traditional land use causes habitat changes that threaten biodiversity. This implies that hydrology alone is not forming wetland habitats but anthropogenic processes in related to traditional agriculture are required as well.

Forest draining has repeatedly been subject to governmental subsidiaries (Eliasson 2008), beginning already around 1840. As late as in the 1980s c. 15 000 hectares were drained, c. 30 000 hectares were subject to protective drainage after clearcutting, and c. 30 000 hectares subject to ditch restoration (Löfroth 1991).

Drainage is today regulated by 11 Kap. MB and the interpretation of this legislation is comprehensively treated in Naturvårdsverket (2009). In southern Sweden drainage is forbidden since 1994 and in northern Sweden permit is required. In 1994–1996 c. 400 hectares were drained annually based on dispensation in the south (Skogsstatistisk årsbok 2010). No national data are available thereafter. In Västerbotten in northern Sweden 5–10 permit applications were treated annually 1998–2002 (Naturvårdsverket 2007).

Protective temporary draining after clearcutting is allowed and since 1994 c. 50 000 hectares have been treated by protective draining (8 500 hectares
in 2009, Skogsstatistisk årsbok 2010). Although protective draining is not allowed for wetland it is unclear how large areas that ecologically constitute wetland habitats are affected.

Since 1850 c. 1.5 million hectares of wetland have been drained for forestry, of which c. 300 000 hectares are still considered too wet for production (Strömgren 2006). Also successfully drained wetland often return to wetland as the organic soils break down and the ground level thereby sink closer to the groundwater level. Also reduced efficiency of ditches contributes to the back-succession to wetland. This creates a need for restoring the drainage systems and cleaning of ditches in large areas of more or less wet drained areas. Ditch maintenance is allowed in order to maintain, but not increase drainage effects but as for protective draining ditch maintenance probably drains considerable areas that are ecologically wetland habitats (Naturvårdsverket 2007). In the province of Gästrikland c. 0.5–1 km of ditches are cleaned annually in the forest landscape, in Norrbotten coastal area c. 3 km, in Västmanland c. 10 km and in Kalmar c. 60 km. It is not likely that biodiversity considerations restrict the ditch maintenance significantly. For example, on the property of Sveaskog 16% (c. 50 000 hA) of the formerly drained wetland area are considered in the need of ditch maintenance (Hägglund 2009). Only c. 6 500 hA are considered to be less suitable for drainage for environmental reasons (here related to the FSC, Hägglund 2009).

**Important gaps of knowledge**

- The biodiversity values of formerly but unsuccessfully drained wetland are largely unknown.
- We don't know the extent of functional wetland draining due to protective draining (skydds dikning) and ditch maintenance, or the effects of such measures on biodiversity and ecosystem functions of the silvihydrological landscape.
### General land use change

<table>
<thead>
<tr>
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<tbody>
<tr>
<td>Climate change may indirectly result in reduced wetland draining if policy measures are developed that aim at reducing carbon release from oxidising organic wetland soils.</td>
<td>Drainage is regulated by 11 Kap. MB and the interpretation of this legislation is treated in (^4). In southern Sweden wetland drainage is forbidden since 1994 and in northern Sweden permit is required. In 1996 c. 400 hectares were drained based on dispensation in the south(^5). <strong>Protective draining</strong> after clearcutting is allowed and since 1994 c. 50 000 hectares have been treated by protective draining (8 500 hectares in 2009)(^6), with unknown ecological effects. <strong>Ditch maintenance</strong> is allowed and needed for large areas of formerly drained wetland of which a probably considerable proportion can be considered wetland ecologically(^6,7).</td>
<td>Alternative 1: Policy measures to stopping draining of undrained wetland</td>
<td>7’ Probably small because of the presently rather restricted drainage of ecologically intact wetland.</td>
</tr>
<tr>
<td>On the other hand, a warmer climate may increase the need for draining because of transportation problems on unfrozen wet and moist grounds(^1). Furthermore, if the view is taken that increased forest production helps reaching national and international climate goals (through long term substitution effects), then an increased forest production on formerly drained wetlands may be favoured by different policy measures (cf. (^8)).</td>
<td>Alternative 2: Policy measures to strongly reducing or stopping of new wetland draining as well as of protective draining and ditch maintenance in wetland and moist forest</td>
<td>8’ In general, we lack sufficient knowledge about biodiversity in already drained but still wet habitats, and about effects of ditch maintenance and protective draining. We may expect large positive effects on biodiversity connected to (a) moist forest, today frequently subject to protective draining. Moister forest habitats (on non-peat soils) are to large extent affected by drainage(^10) and forestry(^11). (b) Wetland habitats which are successional due to increasing soil moisture. (c) More or less stable wetland habitats in the hydrologic vicinity of drained areas. These were not affected by the original draining, but are often affected by restoration of the ditch systems. Ceased draining does not automatically equal ceased forestry and therefore, the effects will be largest on biodiversity least sensitive to forestry, e.g. the ground flora and fauna.</td>
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<td>Alternative 3: Policy measures to encouraging forestry on and further draining of formerly drained, but still or increasingly wet wetland</td>
<td>Alternative 3: Policy measures to encouraging forestry on and further draining of formerly drained, but still or increasingly wet wetland</td>
<td>9’ Negative effects on the groups of biodiversity in 8’ above. Decreased connectivity of wetland habitats since parts of larger wetland areas may be drained. negative (successional) effects on wetland in hydrological contact with drained areas. negative effects of forestry in the drained areas, which may also affect also undrained adjacent areas through edge-effects. Hydrological systems constitute structural networks in the forest landscape which may provide connectivity corridors and stepping stones. Drainage may therefore also increase fragmentation.</td>
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<td>Alternative 4: Policy measures to supporting draining of undrained wetland.</td>
<td>Alternative 4: Policy measures to supporting draining of undrained wetland.</td>
<td>10’ As 9’. In addition continued loss of wetland habitats as a function of the magnitude of draining. Large secondary effects of intensified forestry in and adjacent to wetland.</td>
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6.2.3 Stand continuity in forestry

Clearcutting has been the major silvicultural method in Scandinavia for the last decades, mainly being economically motivated. As an alternative, continuous cover forestry (CCF) has been suggested. The term CCF includes a number of silvicultural practices, of which “Naturkultur” is the one being debated for the longest time (Skogseko 3/2009). Other, largely overlapping terms are gap-cutting (luckhuggning), “skärmföryngring” (“överhållokn skärm”, i.e. regeneration under a canopy of older trees), stemwise selective cutting (“stamvis blädning”, aiming at optimising the diameter distribution), and volume-based selective cutting (“volymblädning”, aiming at optimising stand volume).

In a biodiversity context, CCF has been discussed from two, partly contradicting, point of views. On the one hand, CCF has been suggested as an alternative method in production forest that may have less negative effects on biodiversity than clearcutting. On the other hand, CCF has been suggested as an alternative to protection in continuity forest stands (SOU 2006:81). National surveys have produced different estimates of the area of continuity forest, i.e. forest not yet subject to clearcutting forestry. Of the 23.75 million hectares of productive forest land 900 000 hectares are legally protected, 600 000 in and 300 000 below the mountain zone. In addition, c. 975 000 hectares of un-protected high conservation value forest (high-value cores) on productive forest land are identified, 225 000 ha in the mountain zone and 750 000 ha below the mountain zone (Skogsstyrelsen 2008b). Most of the unprotected area of high-value cores can be assumed to be continuity forest. The Swedish national forest inventory estimates that 400 000 hectares are most likely to fit the definition of continuity forest and that the area may be as large as 1.8 million hectares (Cedergren 2008).

There is an ongoing debate on CCF both regarding its economy as an alternative to clearcutting forestry and regarding its ecological consequences as an alternative to protection (e.g. Skogseko 3/2009).

The legally protected area (900 000 hectares) together with the area of identified un-protected high-value cores (975 000 hectares) comprise 7.9% of the productive forest land, 55% in the mountain zone and 5.7% below the mountain zone. The ongoing forestry in the remaining high-value cores is by far the most important cause of the present loss of forest biodiversity (Fourth National Report).

In addition, deficit in a number of other ecological variables is also responsible for the biodiversity decline, e.g. sun exposure, volume and quality of dead wood, content of old-growth trees, and tree species composition. All these variables may vary quantitatively irrespective of whether clearcutting forestry or CCF is chosen. Several of the variables may also be affected by land use changes related to climate change.
A comparison of biological effects of clearcutting vs. CCF must, hence, be combined with analyses of other variables. Similarly, comparing logging and protection of continuity forest must consider the effects of logging method and considerations to biodiversity at logging.

**Important gaps of knowledge**

- The available field inventories are not sufficient to identifying the total area and location of the remaining value-cores for biodiversity. Some forest types can be assumed to particularly poorly known in this respect, e.g. pine forests, dry deciduous forests, and small value cores (key habitats).
- We have restricted knowledge of the effects of different methods for CCF on biodiversity, regarding the potential for preserving present biodiversity as well as for restoring biodiversity and ecosystem function.
- The potential of forming ecosystems in which harvest is based on shooting, as in coppice woodland, thereby resembling traditional land use, need to be evaluated in terms of biodiversity effects, economy, climate effects and other societal values.
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<tr>
<td>Increased use of forestry without clearcutting</td>
<td>The forestry's potential contribution to climate change mitigation has mainly focused on increasing the production in growing forest, thereby increasing the carbon storage per time and area unit. Although less in focus, one obvious alternative is to applying forestry practices that decreases the carbon release, both from growing and “grown-up” stands. Continuous cover forestry (CCF) may potentially decrease the loss of stored carbon at the logging phase compared to clearcutting, but still offering sufficient carbon uptake in the stands. It may also have advantages in a warmer climate because of hydro-logical problems on clearcuttings. CCF may also be used as an argument for forestry in continuity forest, even in high-value cores. Climate change may increase the interest for such forestry if the view is taken that increased forest production helps reaching national and international climate goals (through long term substitution effects).</td>
<td>None of the variants of CCF are common in Swedish forestry, but some are subject to ongoing studies, e.g. on the Sveaskog property. Legally, CCF often implies harvest of larger volumes than is allowed for thinning according to 10§SVL and therefore may require dispensation. The area of continuity forest in general as well as of un-protected high-value cores for biodiversity (c. 975 000 hectares) is continuously decreasing due to logging.</td>
<td>Alternative 1: A certain proportion of the clearcutting in production forest is replaced by CCF</td>
<td>11. Many species groups and ecosystem functions of forest ecosystems depend on other ecological variables than stand continuity. CCF per se is therefore not automatically favouring biodiversity connected to dead wood and old-growth trees, and may even be negative compared to clearcutting forestry for species depending on light influx and high temperatures. This includes species on light-generated substrates such as thick branches and hollow trees. Some pioneer tree species such as deciduous trees and pine, which are regenerating after clearcutting, may become rarer. CCF may favour species groups depending on (a) small-scale continuity of trees, e.g. mycorrhizal fungi, and species with low dispersal capacity utilising substrates of young and medium-aged trees; (b) small-scale continuity of thin wood (as a result of more frequent logging); (c) species favoured by shade continuity, e.g. drought-sensitive cryptogams on ground, trees, and wood, e.g. in moist forest; (d) species favoured by stand continuity, e.g. forest tits and branch-living lichens. In general, species typically occurring in moist forest and other fire refugia can be expected to benefit from stand continuity in CCF. In the traditional agricultural landscape, forests were used for a multiple of purposes, including grazing. This often resulted in semi-open forest with long continuity of trees. CCF will probably show little resemblance with such forest use, mainly because of focus on high stand density.</td>
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<td>Changed length of rotation period</td>
<td>Prolonged rotation periods may increase the stands’ carbon uptake period and delay its release in order to meet national and global goals for climate change mitigation. If delayed cutting is applied on old-growth forest, mainly the latter effect is achieved since carbon uptake per time unit is low in old natural forest. On the other hand, if intensified forest production is considered to meet climate goals through increased long-term substitution effects, then shorter rotation periods may be the result. Shortening of the rotation reduces conflict between short-term climate goals and long-term substitution by reducing the time lag between logging (release of carbon) and re-storage of carbon. Shorter rotation can also counteract the assumed increase of wind damage.</td>
<td>No prolongation of rotation periods occurs today. Shortening of rotation periods does probably not occur systematically, although high prices on thinning assortments (for biofuel, pulp, and timber) encourages earlier logging. Old-growth forest and other high-value cores are regularly cut since there is no fully functional national strategy for protecting all of the remaining c. 8% unlogged forest.</td>
<td>Alternative 2: Increased logging in the remaining continuity forest and high-value cores, using CCF.</td>
<td>12' Large negative effects on biodiversity and increased rate of biodiversity loss, due to reduced stand age, wood volumes, increased stand density etc. Recent reviews of literature show that considerations to biodiversity at logging are not sufficient for long-term maintenance of forest biodiversity and of more demanding species. Although a larger fraction of the original forest biodiversity may be preserved by CCF (mainly of the favoured species groups described in 11' above) compared to if the stands were logged by clearcutting, it is still only a fraction of the biodiversity in the high-value cores. So far no value-core habitats are identified in which threatened biodiversity would tolerate or benefit from CCF. Some types of traditionally grazed “farmer’s forest” may prove to be such a habitat under specific CCF-conditions.</td>
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<td>Alternative 1: Clearcutting in production forest is delayed by 1-2 decades</td>
<td>13' Probably only marginal effects on a restricted number of species groups that benefit from having a slightly longer period to re-establish viable populations after the last cutting, e.g. mycorhizal fungi. Reasonably prolonged rotation is however probably still much too short.</td>
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<td>Alternative 2: Clearcutting is shortened in production forest</td>
<td>14' Probably marginal for the same reasons as in 13' above. Very short rotation periods (10-25 years interval), based on production of biofuel rather than of timber and pulp, could potentially favour biodiversity connected to exposed old-growth trees of. e.g. Oak and Pine which is today threatened by shading in formerly semi-open stands.</td>
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<td>Alternative 3: Clearcutting or CCF is delayed 1-2 decades in old-growth forest and other high-value cores</td>
<td>15' If applied on all high-value cores, the detected present loss of forest biodiversity would more or less stop during the delay period. A delay is only a temporary rescue but would provide a future freedom of action compared to the irreversible effects of logging.</td>
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### General land use change

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<td>Changed policies for protection of value-cores for biodiversity</td>
<td>It has been suggested that permanently protected forest stands may be less useful in a changing climate and therefore should be replaced by more temporary forms of protection(^{20}). This may in practice imply reduced ambitions for long-term protection of the last value-cores for biodiversity.</td>
<td>Alternative 1: Reduced rate of and ambitions for the protection of value-cores for biodiversity, or reduced use of long-term protection in favour of temporary protection.</td>
<td>151(^{\prime}) Increased rate of loss of forest biodiversity, more or less directly proportional to the loss of value cores. Considerations to biodiversity at logging do not significantly bolster the detrimental effects of logging and thus do not reduce the negative trend for biodiversity(^{9}).</td>
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<td>On the other hand, the importance of the forests’ carbon stocks, and the negative effects of the release of those stocks through logging, have been emphasized in the international climate negotiations(^{21}), which may increase the attention to forest protection.</td>
<td>Alternative 2: Increased ambitions for and rate of protection of value cores to a level that implies a stop of the ongoing logging of value cores.</td>
<td>152(^{\prime}) Nearly completely stopped short-term loss of more demanding forest biodiversity. In a longer time perspective, losses will continue at a low rate due to fragmentation and suboptimal ecosystem function.</td>
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<td>At present, c. 8% of the productive forest in Sweden is legally protected, 55% in the mountain region and c. 6% outside the mountain region(^{9}). Another c. 4% (975 000 hectares) constitute known value cores for biodiversity, and pilot studies indicate that the area of unknown value cores is considerable(^{9}). The area of value cores is continuously decreasing due to logging, as is the area of continuity forest (ancient woodland)(^{9}).</td>
<td>Alternative 3: Increased rate of protection of value-cores for biodiversity as in alternative 2, and in addition protection of production forest in order to increase patch areas and connectivity.</td>
<td>153(^{\prime}) Short term effects as in 152(^{\prime}), above. Long-term loss of biodiversity will decrease compared to alternative 2.</td>
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**Increased use of coppice-based production systems, e.g. for biofuel**

Wood production based on shooting rather than on mortality/recruitment as in normal forestry, may potentially reduce three of conflicts related to normal forestry: (1) the time lag between cutting (release of carbon) and re-storage of carbon which creates a conflict between logging for potential substitution and short-term emission goals; (2) the problem with carbon release due to decay of dead stumps and root systems which further delays the re-storage function of growing stands following clearcutting\(^{22,23}\); (3) the negative effects on threatened biodiversity of logging for substitution in general and short-rotation forestry to speed up the re-storage in particular\(^{22}\).

Shooting-based forestry, i.e. coppicing of native deciduous trees, has been extremely widespread in the Eurasian nemoral and boreo-nemoral zones\(^{22,23}\), as well as in sub-alpine birch forest, but is today largely replaced by tall-tree production. The production form is used in Swedish energy forest based on *Salix* and *Populus* on former arable land, but has not been used based on native tree species on other soils. A recent review has suggested a number of plausible methods\(^{22}\).

Alternative 1: Introduction of coppicing mainly based on native deciduous trees, for example in edge zones, wetland and some types of arable land and forest areas. Production of wood biomass can be combined with harvest of grass, e.g. through grazing.

Alternative 1: The biodiversity in coppice ecosystems is largely unknown but the production form forms a number of microhabitats and a general habitat structure and disturbance regime that may be highly beneficial for threatened biodiversity, e.g. species connected to wood, bark, litter, and grass sward\(^{22}\).

**Notes:**

21 United Nations www, 22 Helldin et al. unpublished, 23 Emanuelsson 2009
6.2.4 Considerations to biodiversity in forestry

In Sweden a relatively high proportion of the productive forest area, >95 per cent in most regions, is transformed from natural forest to production forest based on clearcutting (Fourth National Report). This is motivated by a relatively high ambition regarding taking considerations to biodiversity in forestry ("the Swedish model"). Considerations are expected to be taken in all steps of the forestry production cycle. Considerations are aiming at preserving sufficient quantities and qualities of habitats for the forest biodiversity, e.g. dead wood, old-growth trees, deciduous trees, and some particularly important habitats, e.g. wetland habitats. Both the quantity and the quality of considerations are regulated in SVL 30§, but the forestry fails to observe these regulations on c. 25% of the logged area (Skogsstyrelsen 2008). Recent reviews of biological effects of considerations in forestry have concluded that the types of considerations used in the Swedish model are relevant and beneficial for some species groups but that the quantities are far below threshold values for sustainable conservation of forest biodiversity as a whole. (Johansson et al. 2009, Fourth National report).

Several of the different types of considerations may be affected by climate change-related modifications of land use. The effects may be both qualitative and quantitative and may affect both forest and a variety of wooded habitats in the agricultural landscape (Lennartsson et al. unpublished report to the Swedish EPA). Here, two climate-change related land-use changes are analysed, the removal of dead wood for pest control and the removal of logging residues for biofuel. Other removal of wood, e.g. for conservation purposes in the agricultural landscape, are not analysed, but see Lennartsson et al. (unpublished report to the Swedish EPA) and Götmark (2010) for concrete examples of methods, potentials, and problems.

LR-use and biodiversity is currently subject to a not yet published review by the Swedish Energy Agency. Another comprehensive review is Berglund (2006).
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<td>Changed routines for removal of dead wood, predominantly of Norway spruce</td>
<td>Norway spruce is assumed to be suffering from climate change, particularly in southern Sweden, due to a set of interlinked processes: (1) increased drought-damage, (2) more favourable conditions for bark-beetles, (3) increased storm frequency. The effects may act in cascade: more dead wood increases the populations of bark-beetles (e.g. <em>Ips</em> spp) at the same time as the trees are more sensitive, which further increase the bark-beetle outbreaks etc. Such cascades have been observed following the storms Gudrun and Per. Therefore intensified efforts to restricting the volumes of spruce wood can be expected</td>
<td>Storm-felled spruce trees damaged by bark-beetles are normally removed in production forest but not in protected areas. Up to 5 m³ spruce wood per hectare of recently fallen trees are allowed to be left in production forest, but larger quantities must be removed. Land-owners are already encouraged to intensifying removal of dead spruce wood and old spruce trees in the production forest. Also for nature reserves spruce wood removal is being discussed and decided, both for conservation purposes (for protecting the spruce in the reserve) and for stopping dispersal to adjacent production forest.</td>
<td>Alternative 1: No change of removal of fallen spruce wood. Leads probably to increased volumes of recently fallen trees because increased damage frequency is not fully matched by increased removal rates. Probably also increased volumes of older wood. Increased volumes of spruce wood in protected areas.</td>
<td>16° Positive effects on (a) generalist species with rather high dispersal capacity which survive in the production forest provided that continuity (but not necessarily large volumes) exists of different spruce substrates. (b) species which are favoured by episodic occurrences of large quantities of spruce and other wood in protected areas (and in the production forest following extreme storm damage). Examples may be poorly competitive species which benefits from released competition initially after a wood-producing pulse. These species may be generalists or vagabonding specialists. Further, species that normally occur in sparse populations but which during substrate pulses may build up large populations with high dispersal capacity.</td>
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<td>Alternative 2: Increased removal of spruce wood to stop increased growth of bark-beetle populations. Also some removal of spruce wood from protected areas.</td>
<td>17° Negative effect on the species in 16°, above, in production forest. Reduced potential for general considerations to biodiversity in forestry. In nature reserves there may be exceptional cases in which the spruce and its associated species are actually threatened by the bark-beetles. However, it is probably more common that episodic spruce death is positive to biodiversity since pulses of dead spruce wood may be important for a number of species; see 16° above.</td>
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<td>Use of logging residues as biofuel</td>
<td>Although the increased use of logging residues (LR) as biofuel is caused by high market prices, the production of biofuel in forestry is essential for substituition effects of forestry, in turn a major argument for the view that increased forest production for climate goals\textsuperscript{2,3}</td>
<td>LR are used for biofuels on more than half of the clearcuttings in Götaland and Svealand and c. 30% in Norrland\textsuperscript{5}. The trend is probably increasing. Harvest of LR is legally regulated as other forest activities.</td>
<td>Continued and further increased use of LR as biofuel.</td>
<td>18\textsuperscript{2} When logging old-growth or other biodiversity-rich forest, most of the biodiversity is lost already at logging, and the removal of LR thus affects the remaining species which survived the clearcutting. LR generally harbour few demanding and red-listed species\textsuperscript{10,11,12,13}, even though thin wood dimensions on clearcuttings are somewhat more species-rich than in forest\textsuperscript{14,15}. Therefore, removal of LR can be expected to mainly affect more generalist species on thin wood and stumps. There are however important exceptions for certain tree species in certain regions, e.g. Aspen in Uppland and Oak and other broad-leaved deciduous trees in eastern Småland\textsuperscript{14,15,16}. Biodiversity effects of systematic harvest of stumps are little known. Since stumps provide the main source of coarse dead wood on a clearcutting, stump harvest may affect the local abundance of wood beetles\textsuperscript{21}, and in the long run the overall abundance in landscapes dominated by forestry. For some rather generalist species on wood and litter, e.g. species of bryophytes, LR extraction has been shown to be negative because of reduced substrate abundance, reduced protection against dry-out on clearcuttings, and increased disturbance frequency\textsuperscript{17,19}. Considering that almost all of a forest’s biomass is removed at clearcutting, further substrate reduction through removal of LR may even break the last remaining substrate continuity in the production forest. The piles of extracted forest fuel are attractive for wood-living insects, especially if the cut area is cleaned of dead wood, and often function as ecological traps\textsuperscript{15,16}. When extracting LR on the clearcutting also other wood is frequently collected, including wood left at logging as biodiversity considerations, “consideration trees” that have fallen after the logging etc.\textsuperscript{18,19,20}. Use of LR affects biodiversity indirectly through increased need for fertilization. Effects of fertilization, e.g. through ash, are not analysed in this review.</td>
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6.2.5 Choice of crop on farmland

In the traditional agricultural landscape, i.e. before the large-scale use of artificial fertilisers, pesticides etc., arable land constituted a group of biologically rich habitats, with a flora and fauna much resembling ruderal habitats (Lennartsson 2010). A small-scale spatial heterogeneity of the farmland contributed to a rich biodiversity. Today’s arable land is to various degrees monocultures covering large fields with little spatial variation.

Also in the current, biologically poor farmlands there are, however, differences between crops in the effects on biodiversity. The effects are both direct and indirect. Direct effects are how the use of a crop affects, positively and negatively, biodiversity on the cultivated area. Indirect effects are how leakage of nutrients and pesticides affects biodiversity in other habitats, mainly waters and wetlands, and how the crop choice affects the production system on the farm as a whole. The latter may affect the management of other, biologically rich habitats such as semi-natural pasture.

Here, one climate-change related aspect of crop choice, changed use of ley, is analysed.
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<tr>
<td>Changed use of ley on arable land</td>
<td>The most direct link between climate change and grass production on arable land is an increasing interest in biogas production, partly based on policy measures to facilitate carbon-neutral fuels for transport. Ley has also been discussed as a tool for decreasing nutrient leakage from arable land, compared to crops that require more frequent cultivation. Although increased leakage can be expected in a warmer climate, the use of ley is however rather weakly linked to climate change since the leakage from arable land is already too large to reaching environmental objectives for nitrification or the Baltic. In other words, powerful measures for reducing nitrification are necessary irrespective of climate change. A warmer climate may, on the other hand, reduce the use of ley because of increased production potential. This includes potential for new crops, many of which require increased fertilization and annual soil cultivation.</td>
<td>There is a specific subsidiary within CAP for “environmental ley” (Miljövänlig vall), motivated both by decrease of nutrient leakage and positive effects on biodiversity. In forest-dominated regions the purpose of the subsidiary is to favour biodiversity and the survival of agriculture. In regions with intense agriculture the aim is to reduce nutrient leakage from 200 000 hectares of arable land. In 2009 c. 680 000 hectares had received the subsidiary in forest regions and c. 230 000 hectares in agricultural regions.</td>
<td>Alternative 1: Ley replaces other crops on an increasing proportion of the arable land</td>
<td>19' The main effect is that leys contribute positively to habitat variation in agricultural landscapes dominated by annual crops. Conversely, in (forest-dominated) landscapes already dominated by ley and similar non-intensive cultivation, further increase in ley proportion has mainly negative effects. Ley per se is somewhat more attractive for birds than high-intense annual crops. A similar advantage has been suggested for insects, especially of clover fields, but this advantage is doubtful considering the early harvest of leys. The proposed potential for grassland plants on long-duration leys is hardly relevant on normal soils and with shorter ley duration than several decades. If ley is compared with less intense annual crops, e.g. in ecological production, the latter are probably in most cases better habitats for birds, insects, and plants, due to rich weed flora, the use of fallow etc. Also biological effects of ecological farming varies between landscape types.</td>
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<td>Alternative 2: Increased use of (new) annual crops on the cost of ley</td>
<td>20' Positive effects on several species groups (see 19' above) in forest-dominated regions where cultivated arable land is rare. The effects of increased nutrient leakage on alluvial wetland is poorly known, mainly because no non-fertilised wetlands exist for comparison. Considering the already high nutrient levels on wetlands downstream arable land, a further increase in nutrient levels probably have marginal effects, especially compared to the effects on aquatic ecosystems.</td>
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6.2.6 Changed draining and use of wetland in the agricultural landscape

The Swedish arable land requires intense draining, both through underdrainage of the fields and through a general lowering and regulation of the water level in streams and lakes in the lowlands. The latter was initiated in the mid 1800s and continued until c 1950. The aim was both to gaining arable land from former wet meadows and to enabling draining of other arable land within the hydrologic system. The extent of lake and stream regulation is large in southern and central Sweden; for example, in the province of Uppland several lakes have disappeared and only two of the remaining lakes are unaffected by regulation (Brunberg & Blomqvist 1997). Lowering and regulation of water tables is often complemented with embankment of arable land.

As described for forest draining, many of the drained areas are now considered unproductive, both because of lowering of the ground level through breakdown of organic soils and because of larger quality demands on today’s arable land. Also old underdrainage systems gradually loses its function. Hence there is a need both for continuous maintenance and for restoration of drainage systems. The drainage systems are also considered sensitive to changes in, e.g., precipitation (SOU 2007:60d).

Parallel to the hydrologic loss of wetland in the agricultural landscape there has been an ecological loss of remaining wetland due to ceased traditional management, mainly mowing and grazing. The remaining alluvial wetlands are also affected negatively by water regulation.

In summary, draining and ceased management have caused strongly negative effects on biodiversity and ecosystem function in wetland habitats. As a result, there are CAP subsidiaries for the hydrological restoration and for resumed management of wetlands, respectively. The management is also subject to other CAP subsidiaries.
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<tr>
<td>Changed use of grass from wetland habitats</td>
<td>The most direct link between climate change and wetland use is an increasing interest in biogas production, partly based on policy measures to facilitate carbon-neutral fuels for transportation. The CAP subsidies for management of wetlands make such habitats economically interesting in spite of rather costly harvest and the fact that wetland grass gives c. 70% of the biogas produced by an equivalent amount of haylage from ley. Harvest of wetland grass is also necessary for obtaining a nutrient retention function of wetlands. This has in particular been discussed as a tool for reducing nutrient leakage from farmland. Although increased leakage can be expected in a warmer climate, the use of wetland is however rather weakly linked to climate change since the leakage from arable land is already too large to reaching environmental objectives for nitrification or for the Baltic. In other words, powerful measures for reducing nitrification are necessary irrespective of climate change.</td>
<td>There are specific subsidiaries within CAP for restoring and managing wetlands, respectively, motivated both by nutrient retention and by positive effects on biodiversity. There is a CAP goal to restore management and/or hydrology in 11 000 hectares until 2013, of which an area of c. 6 000 hectares was reached by 2009. Most of the restored wetlands are small: 65% &lt; 10 hectares and 5% &gt; 10 hectares. The use of wetland grass for biogas is still only of experimental extent. In the vicinity of the biogas plant in Örebro 2000 hectares of wetland need resumed management for conservation purposes, of which c. 600 hectares are estimated to be suitable for harvest of biogas grass. The experiments connected to this plant show some practical problems related to quality demands and the need to mulch the grass before delivery.</td>
<td>Increased use of wetland grass for biogas production</td>
<td>21° The positive effects of resumed management of former wet meadows are well documented, for example for birds and carabid beetles. At present, no disadvantages are known of harvesting “biogas hay”, compared to the commonly performed mowing and grazing for conservation purposes. The problem of too early disturbance for some wetland birds need to be considered in relation to the biogas plants’ quality demands. Harvest may favour restoration of larger wetlands which would be positive for biodiversity compared with today’s restoration. The above effects regards semi-natural wetland habitats, i.e. habitats formed by anthropogenic disturbances (harvest) but in which the vegetation is built up by spontaneous colonisation and competition among native species. If wetland use is based on planted crops, e.g. energy crops, several of the positive effects on biodiversity will become considerably smaller because biodiversity is largely depending on native species.</td>
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<td>Changed draining in the agricultural landscape</td>
<td>Reduced draining of low-lying arable land may be the result of climate change if hydrological conditions make draining too expensive in relation to the productivity of the land. Also deliberate restoration of wetlands on former arable land may increase as an additive effect of such cost-benefit thresholds and policy measures for wetland restoration. The present policy measures for wetland restoration are motivated by both biodiversity and nutrient retention. Changed hydrological conditions may on the other hand also increase draining efforts, e.g. in terms of underdrainage, embankment, and general restoration of whole drainage systems.</td>
<td>Due to the CAP subsidiaries for restoration of wetlands (above) some arable land has been transformed to wetland, either as a direct measure or as an indirect consequence of restoration of adjacent wetlands.</td>
<td>Alternative 1: Increased transformation of wet arable land to wetland habitats.</td>
<td>22° This measure would affect two main types of wetland habitats, (a) new wetland on former arable land; (b) existing, although hydrologically and ecologically changed wetland that has been drained in order to drain adjacent arable land. Biodiversity effects of new wetland on former farmland (a) are poorly known, in particular regarding long-term development of wetland flora and fauna. Some knowledge of wildlife waters may be relevant for this type of wetland. Pioneer and opportunistic species of, e.g. beetles, dragonflies and birds are known to rapidly colonise new wetland, the latter group also using the habitat on migration. Restoration of existing but deteriorated wetland (b) is better known. The effects of restoration differ depending on, e.g. degree of isolation, habitat change, and wetland type.</td>
</tr>
<tr>
<td>Alternative 2: Increased draining of arable land and wetlands.</td>
<td></td>
<td></td>
<td>23° Negative effects on biodiversity in existing wetlands that are affected by the farmland draining. Examples are alluvial wetlands (due to improved regulation of streams and lakes, improved embankment etc.), and wetland downstream arable land that need to be drained in order to drain arable land.</td>
<td></td>
</tr>
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7. Knowledge gaps and uncertainties

Throughout the report, we have pointed at knowledge gaps and uncertainties. These gaps and uncertainties primarily concern effects on biodiversity, emission factors for greenhouse gases, and environmental systems analyses using such site specific field data as well as life cycle data on resources and emissions. Systems analyses would e.g. result in better knowledge of the actual size of the substitution effect and the impact of different management practices. Systems analysis could also be performed using backcasting, i.e. defining the desirable future with regard to the various values land management should safeguard, and work out strategies to reach that state. To support either kind of systems analysis, there is a need for methods to assess the value of ecosystem services that have no obvious economic value (to date) as well as for methods to better assess and understand the preferences of people (preferences that may influence the value of e.g. cultural ecosystem services). Filling these gaps and resolving these uncertainties would considerably improve the chances for researchers to support policymakers at various levels in the choice of land use measures, whereas the decision concerns the local, regional or national level.
8. Prioritized research fields

In this report we have, not surprisingly, identified a number of knowledge gaps which all necessitate further research for better understanding of environmental consequences of land use and land management. Below some prioritized research areas are elucidated for forestry, agriculture and both. They are NOT listed in order of priority.

**Forestry**

1. One main question concerns management of drained afforested peat land. There still is, despite previous research, a need for better knowledge on emission factors for greenhouse gases and their relation to site factors. More knowledge is also needed on the methylation of mercury in forest wetlands. We further want to prioritize systems perspectives approaches with a life cycle assessment perspective on the use of peat land for forestry, where the substitution impact is balanced towards emission from drainage and other measures. It is important to stress the impact of various site factors, such as fertility, hydrology and forest properties, on the net emissions of greenhouse gases.

2. Another important issue is the cutting strategies, i.e. selective cutting versus clearcutting. Selective cutting generally is beneficial in terms of biodiversity and run-off water quality, however potentially at the expense of less climate mitigation. Here too we want to prioritize a systems perspectives approach with a life cycle assessment perspective where e.g. the substitution impact is balanced towards emissions during management. Selective cutting is a broad concept and may be implemented in various ways. It is important to suggest methods that optimize climate mitigation, biodiversity and water quality.

3. Characteristics of different deciduous species and production methods with regard to substitution potential as well as emissions during establishment, management, harvest and transport.

4. Effects of N fertilization on biodiversity

5. Effects of harvesting residues after forest cutting on biodiversity, as well as on decomposition of old organic matter (the C-pool in the mineral soil).

**Agriculture**

1. In optimising land use for different regions and under changing economic and biophysical conditions, research is required that brings global risks and opportunities to a local context under different future scenarios. By developing and applying scientific methods for integrated assessment of different future land use options, it will be possible to provide a tool to balance the conflicting interests of producing biomass for food, energy and other benefits including ecosystem services, and at the same time reach environmental targets and maintain the long-term productivity of the land.
2. Research on reducing fossil fuel consumption and optimizing co-production of biomass for energy and food.

3. Research on how marginal land can be used for production of commodities.

4. Development of strategic crop rotation for reduced negative environmental impact (pesticides, fertilizers and greenhouse-gas emissions)

5. Methods for integrating animal and crop production in order to reduce nutrient losses, to enable the use of ecosystem services from livestock management, and to increase biodiversity.

6. How to design the agricultural ecosystems to optimize ecosystem services in order to maximize yields – how to increase the abundance of insects and animals that mitigate pest and crop deceases.

**General**

1. There is a need for tools to facilitate decision-making in land management aiming at maximizing climate change mitigation and minimizing hazardous impacts on e.g. biodiversity and water quality, at local as well as aggregated (regional and national) scale. Such tools must necessarily be able to assist in making trade-offs between different environmental aspects, but also between different dimensions of sustainable development (social, economic and environmental). Balancing different dimensions, in turn, requires a way of valuing these different dimensions in some kind of common unit. This is a particular challenge with regard to values that are complex and lack an obvious economic value, like e.g. biodiversity and cultural ecosystem values.

2. The effects of climate change, in terms of altered precipitation regime and changes in temperature, on the mobility and dispersal patterns of organisms is poorly investigated. Effects of climate change on the response of organisms to management and in relation to the landscape is also poorly investigated. In general is the relation between the landscape and mobility, dispersal and survival of organisms poorly understood.
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Land management meeting several environmental objectives

Minimizing impacts on greenhouse gas emissions, biodiversity and water

Knowledge compilation and systems perspectives

MATS OLSSON, PETRA ANDERSSON, TOMMY LENNARTSSON, LISETTE LENOIR, LENNART MATTSSON & ULRIKA PALME

Which mitigation options in land-use management do meet the goals for greenhouse gas emissions, biodiversity and water security?

This report makes a systems analysis of land-use and its implications for greenhouse gas emissions, biodiversity and water.

The main conclusions are that:

• Most land management strategies can meet the goals for biodiversity and water in an adequate way, except intensive forestry, although trade-offs between different environmental values will be necessary.

• It is important that there is an understanding of how the prerequisite for impacts of land-use changes in a changing climate.

The authors assume sole responsibility for the contents of this report, which therefore cannot be cited as representing the views of the Swedish EPA.