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Abstract
This paper deals with the methodology of land use as an integrated part of a life cycle assessment (LCA). The conventional way of starting at the life cycle inventory (LCI) level with the search for suitable indicators, which are then followed along the cause-effect chain to the end points, is problematic in assessing land use, where knowledge of environmental mechanisms is limited. To avoid the obstacle of this lack of knowledge, the author has chosen a top-down approach, proceeding from the suggestion that an evaluation of the environmental values of land would be a good base for estimating the environmental burdens associated with land use. Parameters for this evaluation are suggested and default values for Sweden are calculated. Two safeguard subjects are involved: ecosystem productivity and biodiversity. The yield of biomasses is suggested as the main indicator of productivity, and the occurrence of redlisted species as that of diversity. Two levels of productivity and diversity are defined: a potential, reference level, and an actual level for each category. The divergence from the reference level is in both cases a measure of the degradation of the land or the effect of land use.

The feasibility of the methodology is shown with some examples. Advantages and disadvantages are discussed.

Keywords: LCA, Land Use, Biodiversity, Bioproductivity, Bioquality, Forestry, Agriculture.

1. Introduction

Generally Life Cycle Assessment (LCA) deals with the evaluation of the environmental loads that products and systems create during their whole life cycles. Often a division is made into Life Cycle Inventory (LCI) and Life Cycle Impact Assessment (LCIA). This report deals with the impact assessment of the Land Use part of an LCIA.

Land in an LCIA is any piece of the surface of the Earth, regardless of its nature; real land, lakes or oceans. However, in Land Use LCA, the impacts made during the use of land are normally focused on those belonging to the biosphere. The biosphere of a piece of land is not restricted to a thin surface. It extends down into the soil or the water as far as biotic life exists, and biotic life in the air above the surface is also included. Again, not all impacts on the bisophere are included – only those which are caused by managerial actions.

It can be noted here that the mere fact that an area is occupied for land use is not a reason to assign a load to the products from the area. An LCA does not take any position on the question of whether humanity has a right to use land, or for what purpose the land is used.

Only the losses of ecosystem value caused by the land use are recorded and evaluated here according to an anthropocentric yardstick.

Effects on biodiversity caused by the manner of managing forestry and agriculture are examples of land use impacts. The building of roads and other communications as well as the urbanisation of areas result in drastic reductions of the former natural biomass production as well as of biodiversity.

The methodology presented in this report has its origins in a CPM report from 1998 [1], but has been expanded stepwise and presented in proceedings from the meetings of COST Action E9 (Life Cycle Assessment for Forestry and Forest Products).

The assessment has been based on the EPS system and default method [2 and 3]. This method has five Safeguard Subjects: human health, ecosystem production capacity, abiotic stock resources, biodiversity, and cultural and recreational values. This study of damage to the biosphere deals mainly with the subjects of ecosystem production capacity and biodiversity.

Ecosystem production capacity, or bioproductivity as it will be called below, is also seen as a "life support function", because the lives of all "higher" organisms depend on the oxygen resulting from the assimilation process of the green vegetation and on green biomasses as nourishment. Biodiversity is likewise dependent on ecosystem production capacity, but it is also a kind of life support function, because most living species need other species to live on. Thus bioproductivity and biodiversity are the two themes that will be treated under land use. Cf: "The two themes of life support and biodiversity are seen as the most important contributions to the ecological value of an area ...." [Lindeijer 1998, 4]

A number of impact categories are mentioned in the EPS system, of which the following will be used in this land use method:

Table 1.1

Impact Category	Indicator	Unit	Weighting factor
Crop growth capacity	Crop	kg	0.15
Wood growth capacity	Wood	kg	0.04
Fish & Meat prod. capacity	F&M	kg	1
Soil acidification	Cation	mole H+	0.01

For this report the following additional default values are suggested:

Forage production capacity	Forage	kg	0.05
Fruit, berry & mushroom cap.	Berries	kg	0.2

If the bioproduction capacity figures are known for a piece of land, the bioproductivity value can be calculated using the weighting factors above. The weighting factors have the format ELU/indicator unit. The bioproductivity value that can be calculated represents the WTP value for avoidance of loss of this productivity. Conversely this load can be seen as the bioproductivity value of the land in question (per unit of area and time). In this report the dimension will be ELU/ha,y.

Resources, especially biotic resources, are often mentioned as part of the land use LCA, but are more seldom dealt with. See comments under section 3, Spatial and temporal aspects.

Biodiversity as a safeguard subject has only one impact category: species extinction. The assessment of species extinction is dealt with in section 6, WTP evaluation of species.

2. Top-down procedure suggested

Most LCIAs are made according to a bottom-up procedure, i.e. an attempt is made to assess the environmental consequences of known emissions and resource extractions. In the present work a top-down procedure has been adopted, in which an attempt is made to determine in what ways the product system interferes with certain environmental values. The starting point is thus the environmental values, and these are allocated according to various indicators along the cause-effect chain until scientific characterisation models can be used to link category indicators to LCI results. The allocation of values has some clear subjective elements in it, but some objective parts too.

The author's model will be presented in the next few sections, followed by some comments on the differences from other published models .

The sum of bioproductivity and biodiversity is here called Bioquality, having the dimension ELU/ha,y.

The hydrological state of an area of land influences its productivity as well as its biodiversity. For that reason it has been argued that the hydrological state should be incorporated as a third component of bioquality. However, as long as the effect of changes in the hydrological state can be evaluated as effects on productivity and biodiversity, no extra component needs to be added to bioquality.

Thus we have the relation Bioquality (Q) = Bioproductivity (P) + Biodiversity (S)

As has been explained above, the indicators of the impact categories biodiversity and bioproductivity, capable of expressing damage if such values are destroyed, can conversely be seen as expressing the environmental values of the land which is studied. Together with bioquality they can all be said to be functions of state, e.g. describing the state of an area. All three variables have different kinds of states; a potential state, describing a kind of natural state, and an actual state, which is found when one tries to evaluate the state.
They have the following relations:

$$Q = P + S$$
$$Qu = Pu + Su \quad \text{(potential values)}$$
$$Qa = Pa + Sa \quad \text{(actual values)}$$
$$\Delta P = Pu - Pa$$
$$\Delta S = Su - Sa$$
$$\Delta Q = \Delta P + \Delta S$$

where ΔQ is the difference between the potential bioquality and the actual bioquality, the difference being attributable to the use of the land. If $Qa < Qu$, ΔQ shows the fall in environmental value due to the land use. Thus the effect of land use is equal to ΔQ. ΔQ may be expressed in ELU/ha,y or other units, but when it is converted to the effect per functional unit, it will here be called E. If the land use is forestry, the unit might then be ELU/m^3 wood (or kg wood). If the land use is seed cultivation, the format might be ELU/kg seed (or tonne of seed):

$$\text{Effect } E = \Delta P + \Delta S = \Delta Q \text{ per functional unit} \qquad (1)$$

These terms are used in somewhat different ways than in conventional LCA. Normally top-down means starting with the general or holistic view, and then eventually coming down to detailed levels. If too many indicators are linked to a safeguard subject, the most relevant are found with a top-down procedure [9, SETAC WIA-2]. Conversely, bottom–up involves starting at the detail level and ending up at a total or general level. LCA is normally a bottom-up process, starting with detailed interventions, classification into impact categories, calculation of their effects at the end points, tabulating the impact categories in a common format, and – possibly – summing up to a total effect.

The land use issue is apparently different, because many recent authors have used a top-down procedure for land use (Lindeijer [5], Köllner [6], Baitz [7]). Thus a land quality value Q is defined and a reference value corresponding to an initial or natural state is presented, but not always calculated. Usually the difference between this initial value and an actual value indicates the result, the land use load, which should then be allocated to the product of the land use – the functional units.

It should be observed that in this model only natural biomass products are included in the calculation of Pu. The biomasses that form the yield of the land use (extracted wood, seed and other land use products) dictate the functional units and are the bases for the allocation of the land use load.

Following Equ.1, the environmental values Pu and Su of the land must first be estimated. Secondly, the actual states Pa and Sa have to be evaluated. In what follows the potential states will be dealt with first, and the actual states afterwards.

If environmental intervention between two points of time must be taken into account, the states at those two times have to be evaluated. The differences will show the effects of the intervention that caused the change during that time period.

The category indicators for the safeguard subjects "ecosystem productivity capacity" and "yearly extinction of species" both indicate the general willingness to pay for the loss of an indicator unit of the category, e.g. the loss of 1 kg of wood, or the yearly extinction of an endangered species. Using those estimations as parameters to describe the environmental value of the land gives two possible uses.

One is to map the environmental value of a landscape or a region. The environmental wealth of two landscapes can be compared. The values that would potentially be lost if any of the land was turned into communicational areas will become evident, making it possible to make choices that minimise the environmental load. A top-down approach is preferable as a means of doing this. With a bottom-up procedure, only differences in bioquality can be calculated, as linearities for all indicators used are out of the question.

The other possible use, which is directly applicable to land use, is to assess the effect of an intervention according to Equ.1.

This use assumes that the bioquality components of a land are readily available, preferably from databases of the GIS (Geographical Information System) type. This is not yet the case, but the author presents default bioquality values for Sweden below, illustrating how to make calculations on the general level

3. Spatial and temporal aspects - the format

Land use means a combination of environmental loads during a given period of time and over a specified area. Many authors prefer to calculate the load over two or more time units (years), apparently with a view to dividing the total load during the period by the total product output during the same time interval to arrive at the load per functional unit at the end. Lindeijer writes [4]:

$$\text{Land occupation} = A \cdot t \cdot Q$$

where A = Area (m^2), t = time (years, y), and Q = a quality difference (Qref – Qact).

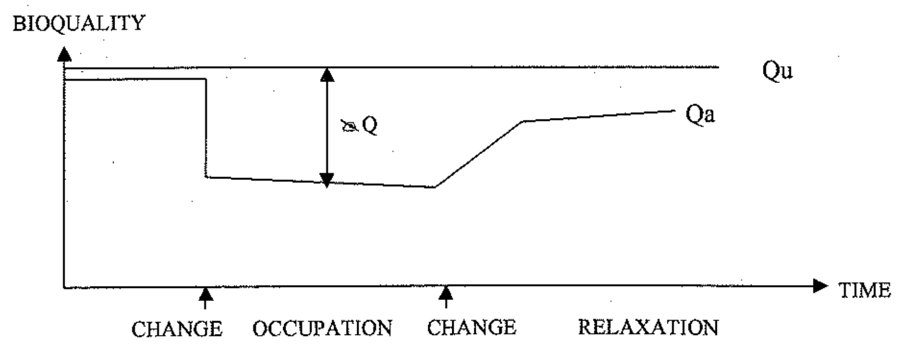
This method has an advantage if the calculation is made over a life cycle with different outputs during the cycle, e.g. for forestry. Under such circumstances the calculation can preferably be assigned to the time for the final cutting, when most of the silvicultural measures are taken that aim at assuring sustainability. In other cases, however, annual yields are taken, and then calculation per year is preferable.

But in an LCIA the load per functional unit is required. The final format is then different: EP/FU. EP = eco-points, FU = functional unit. See further below.

The following graph illustrates the shifts in bioquality with time.

Figure 1

Bioquality Phases and environmental load E



Authors describing land use often stress the that land use can be divided into three phases:

1. From a more or less natural state, the land is taken into use, meaning that the first phase is followed by a change or transformation to (real) land use. Changes can take place several times. This first change is often accompanied by a sudden loss of bioquality, primarily a fund resource, perhaps caused by a change to agricultural, arable land. Lindeijer [5] also uses the concept "net change" of land use, meaning the net change of a certain land use that is finished with a return to a near-natural state. This change is then taken as a measure of the net change.

2. During the next phase the land is used for a special purpose, an occupational phase. Bioquality changes may occur in this phase too, e.g. due to changing intensity of land use. However, this phase is mainly seen as a long period with a relatively stable bioquality level.

3. Different kinds of occupation may follow, but finally there may be a return to natural ecosystems, a relaxation, when no product is gained from the land.

The main phase of interest is of course the occupation. A forestry or agricultural use has usually existed for hundreds of years, which more or less nullifies the first transformation or change phase in the calculus. If one kind of forestry or agriculture is to be compared with another, the load per functional unit during an occupation phase is usually what is looked for.

However, if land in a near-natural state is converted to intensive agriculture, this change means that a biotic resource is seriously damaged. This corresponds to the first change or transformation in the graph above. In this model the load to be allocated to (say) the annual crops in the occupation phase starts with the load designated ΔQ. On top of that the value of the fund resource lost by the change should be allocated to the harvest that will follow until this "debt" is paid. After five to ten years the resource load is regarded as negligible compared to the ordinary annual load caused by the loss of the annual natural bioproductivity of the land.

This definition of the occupation load is somewhat different from that of other authors. Frischknecht 1996 [8] defines ecosystem degradation $ED = A \cdot t \cdot (fNPPref - fNPPact)$. fNPP = free Net Primary biomass Production. Lindeijer uses $\Delta Q = Qini - Qact$, where the initial value Qini might be something other than a potential, e.g. the level of the current relaxation potential, which is lower than the potential, see also SETAC WIA-2 [9].

Coming to the other end, the relaxation phase might be a more serious problem. The reversion of used land to forest may take hundreds of years, and still the original natural state may be difficult to reach. To which products should the loads be allocated? The land use that caused the low bioquality level is over, and little or no extraction of other products may take place. The solution may depend on the aim and scope of the study.

Thus, supposing that the occupation phase is at least a few years from a change point, the occupation load is easy to settle proceeding from ΔQ in a real graph of the type above. The potential bioquality is meant to mirror the bioproductivity and biodiversity that the land is

supposed to have in a kind of natural state. The total environmental load is calculated, and not only the difference from another land use, or from a level lower than the potential. Thus it is a biocentric view. However, the evaluation of this load will be made using an anthropocentric view – the only possible way?

According to Equ.1, the environmental value of the land before any intervention must first be estimated in order to designate the potential productivity Pu and biodiversity Su of the land. Secondly the actual states Pa and Sa have to be evaluated

4. The search for a reference state

Comparisons can hardly be made without reference values. This is especially the case when different kinds of land use need to be compared. When the aim is to evaluate the effects of land use, it is natural to use the imagined, natural and untouched state of the land as a reference. This has met with some serious objections relating to the dynamic nature of all lands and their ecosystems.

Ecosystem production capacity can simply be described as the annual yield of biomass on a piece of land. In a natural state this yield is very much dependent on climatic and edaphic (soil) conditions. As soil conditions depend on topographic variations, there are large local variations in production capacity. Thus larger areas must be involved if comparisons between averages for areas are to be meaningful. For smaller localities the specific variations must be known.

<u>Bioproductivity</u>

Most of the earth's land area was originally covered with woods or transitional stages to grasslands or steppes supplemented with other biomasses. Wood production capacity is normally estimated in terms of what is called site quality, having the format m^3sk/ha,y (solid cubic metres with bark), which in this method is recalculated to m^3ub/ha,y (solid cubic metres without bark. $1m^3sk = 0.84\ m^3ub$).The site quality is a fertility gauge that ought to give an indication of the potential productivity. Real growth is usually lower than the site quality – in Sweden usually 20–30 % lower depending on silvicultural measures that can be improved. However, in southern Sweden growth is significantly lower because of the high content of deciduous woods having a lower annual growth than softwoods. This fact has been allowed for by lowering the site quality values for the southernmost zones below. Even with ideal silvicultural management combined with a sustainable harvest of wood, the resulting productivity would be lower than that indicated by the site quality. When environmental considerations of biodiversity and cultural remains are included, still more concessions have to be made. In Sweden such calculations have led to the concept of "mean annual allowable cut" (MAAC) as part of the landscape planning process in forestry. It is suggested below that the potential wood productivity is defined as 70% of the site quality, thereby mirroring the mean annual allowable cut. About half of this remission is due to the conversion to m^3ub.

If the stand properties of a site are known, the site quality and the potential productivity as a mean annual allowable cut can be calculated. For general purposes regional values may be acceptable. The climate is the dominating factor for bioproductivity. Sweden can be divided into eight climatic zones, see for example [10, p. 49]. Within those regions the bioquality is roughly of the same character, but with local variations. The site quality of an area depends mainly on climatic and edaphic factors and is roughly homogeneous only within stands of the same kind. Thus there are wide variations even within the eight climatic zones mentioned. Regional or landscape averages are representative only when the study is confined to larger parts of a landscape, as the variations are then levelled out. If smaller areas are compared, local data are necessary, as the differences in variations will otherwise reduce the possibility of finding significant differences.

The EPS weighting factors for the category indicators of ecosystem productivity reported in table 1.1 in kg/ha,y (wood 0.04; fish & meat 1.0; forage = 0.05; and berries & mushrooms 0.2) are used to convert the productivity figures into ELU/ha,y

Swedish softwoods have a density of about 410 kg/m^3 (dry weight of moist volume). A forest with the site quality 6 m^{3}sk/ha,y has a softwood production capacity of $6 \cdot 410 = 2460$ kg/ha,y. This corresponds to a potential wood capacity of $2460 \cdot 0.70 = 1722$ kg/ha,y. The evaluation into ELU/ha,y gives $1722 \cdot 0.04 = 69$ ELU/ha,y

Other biomasses like bushes, grass, berries, mushrooms, meat and fish also contribute to biomass productivity. Average figures for berry and mushroom capacities in Sweden are 20 and 40 kg/ha,y resulting in $(20+40) \cdot 0.2 = 12$ ELU/ha,y. Hunting prey capacity is about 2 kg/ha,y resulting in an environmental capacity of 2 kg/ha,y. The total bioproductivity addition from non-wood biomasses is then 14 ELU/ha,y, which is thought to be relatively stable over Sweden. Nor do forestry land use operations affect this bioproductivity very much.

The average wood production capacity varies from 2 m^3 in the north of Sweden to 8 m^{3}sk/ha,y in southernmost Sweden, giving potential wood productivities between 1.7 and 5.6 m^{3}ub/ha,y and potential bioproductivity values from 28 to 94 ELU/ha,y.

Thus the average potential bioquality over the eight zones vary between 42 and 108 ELU/ha,y, see figure 2.

Biodiversity

Because of a lack of land use methodology, the default LCIA method of EPS 2000 stops at the average global level. However, in practice, most LCAs need local estimations, and at least a classification into different groups of species. Otherwise comparisons between different kinds of land use will be too uncertain for many applications. Reference levels are also suggested and calculated below.

5. Land structure issues

The biodiversity of a piece of land is often described as having three levels: the multitude of species, the multiplicity of genomes within the species, and the quality of the ecosystem. This paper – like most land use papers – deals mainly with the multitude of species. However, the importance of the ecosystem is also considered. This paper deals only with Swedish conditions, although the method is generally applicable.

The complex of questions to be answered to be able to make assessments includes primarily the following:

1. How should the environmental load be defined and the effects evaluated?
2. How does the structure of ecosystems affect the land use evaluation?
3. How should effects of individual interventions be evaluated?

The first question has been partly dealt with above. We shall now consider the land structure and its consequences for land use evaluation.

A country is built up of ground with mountainous areas, forests, and agricultural and urban landscapes. Lakes and seas or sea shores are additional elements of importance. Geological conditions usually differ substantially, as do climatic conditions. If we add to that that the prehistory of an area has always played an important part, and that history always goes on, meaning that all land is affected by dynamic forces – some called disturbances – it is clear that the biosphere takes a broad multitude of forms. This has led to the development of different kinds of biotopes, where some species have been successful in adapting to a relatively broad range of conditions (the generalists), and other have survived by specialising to more deviant biotopes and ecosystems.

This was the case even before Man started to overrun the earth thousands of years ago. The increasing occupation of land for hunting, farming, etc. has led to extinction and to decreased areas of natural land, but has also, in the long run, led to new forms of species that thrive in pasture lands and also in urban areas. Some old types of species have also found the old agricultural landscapes and urban environments to be safe for them.

The non-generalist species require some key elements to thrive, otherwise they will not survive in the environment. It follows that some habitats are key biotopes, where non-generalists will be found. There may be sufficient numbers of key biotopes to guarantee a sustainable future for some of the non-generalists. For others we have seen a decreasing number of habitats, which may eventually lead to extinction, at least regionally. Those species are then endangered, which means that they are noted in the national red list of threatened species.

The Swedish Red List 2000 comprises 4 120 threatened species out of a total of about 50 000 (multicellular) species [12].

In discussing the sustainability of biodiversity, the redlisted species are therefore the natural issue. If we can find and protect threatened species, at least in key biotopes (and reserves) the situation will improve, but sustainability will in many cases not be reached until measures are taken to increase the number of key habitats or key biotopes – a kind of restoration all over the forests. It must be stressed that the presence of redlisted species does not by itself guarantee sustainability. Those species present may be disappearing because there are too few of the key elements they depend on in the landscape to guarantee sustainability.

Which are the key elements that make up a key biotope? Some of them depend on edaphic (soil) or climatic factors. Such factors may or may not exist in the land in question: they are site-dependent. Other key elements can be created by good forestry management. Examples of such elements are standing and fallen dead wood, old trees and forests, dense and moist forests with a long and unbroken history without clearing, forest fires, etc. These are management-dependent biotopes that can be re-created within the framework of sustainable forestry.

In Sweden during the past decade forest inventories have been made to identify the key biotopes. "Signal species" – vascular plants or cryptogams – have been used to find habitats or sites with high natural values. These signal species or indicators have been chosen because they are possible to identify in the field and are either known to occur mainly in sites holding endangered species, or to be restricted to substrates or habitats rarely found in woodland landscapes, like aged habitats with long ecological continuity. The sites and habitats found may not always house redlisted species, but the conditions are those needed for species bound to those key elements. However, they are always sites with high natural values and therefore considered to be key biotopes.

The structure of the Swedish forests is thus that they differ widely as to their natural climatic and edaphic (soil) conditions, but also as to the manner of silvicultural management. From the biodiversity point of view the forests are composed of biotopes or stands with different ecosystems and natural values. On one hand we have large areas with low incidences of threatened species and minor areas with high natural values, but these may be scattered all over the place. The environmental loads on forestry and forestry products occur mainly when areas with high values are destroyed. Thus, if such areas have been identified and preserved from destruction, the land use loads will be reduced considerably.

Just as bad forestry leads to declining natural values in the forests, bioquality and biodiversity can be improved by planned actions leading to more key elements like dead wood, more deciduous trees of high age, protection of near-natural forests with long continuity of closeness, restoration of biotopes which are being overgrown, etc., etc. This does not mean that the aim should be to create near-natural environments in production forests. "There is a growing awareness that the co-evolution of biodiversity and human societies is one of the major paradigm shifts of conservation biology." (Sayer et al. 2000) [13].

Losses caused by ever-increasing human colonisation lead to fragmentation of forest areas. This is a major threat to biodiversity. When the remaining forest areas fall below 50%, and those areas are fragmented into several smaller areas, diversity begins to be threatened on the landscape scale. At a threshold level of only about 20% remaining forest, sustainability is seriously threatened, and recovery action is necessary to avoid local extinctions. The lack of forest areas with special care or reserves was analysed by Angelstam and Mikusinski 2001 [14]. Such actions fall within the province of landscape planning.

Within the landscape the ecosystems are usually divided into stands, where the ecosystems are fairly homogenous. The forester has to choose his management methods in such a way that they mimic the natural dynamics of the stand. Otherwise again, diversity will be lost.

Finally, at the smallest scale, single trees or patches of trees, snags, downed wood and living old trees have to be managed in order to reach sustainability. On the local scale, too, fragmented key elements are less effective for sustainability than assembled elements. As fragmentation is often the cause of extinction or endangerment of species, an indicator describing this phenomenon would be desirable. Angelstam & al. have formulated an index across the three spatial scales above in order to compare the actual biodiversity with the threshold values [15]. Yet for the future development of land use assessment, such an indicator – or set of indicators – ought to be part of the GIS to get the best effect. At the present state, a description of the degree of fragmentation in the landscape studied could be a start. The more fragmentation, the higher would be the ratio Su/Sa, which is discussed in some detail further below in section 9.

The footprint of humanity in forest environments is also indicated by four already mentioned forest properties: dead wood, thick trees, hardwood fillings, and old forest stands. These properties characterise the most important effects of silviculture on biodiversity. If the potential contents of each of these properties are set at 100, the actual states can be characterised by xx dead wood, yy thick trees, etc. A gradient through the landscape can be established, which also can serve to estimate the Su/Sa ratio [16].

There seems to be a new trend aiming at transferring the indicator focus from the occurrence of threatened species to the identification of key elements and how to create or restore different key biotope conditions so as to reach the requirements for sustainability. If the key biotope conditions are there, the threatened species connected to this biotope will also appear.

On the local scale the identification of key biotopes and other care-demanding biotopes has been going on in Sweden at least for the past few decades. So far about 1 to 3 % of the forest areas seem to be of such a quality that redlisted species are to be expected. Even if the national reserves are included, this is not considered enough to reach sustainability. In order to reach this goal, all the forests outside the care-demanding areas need to be managed in such a way that new key biotopes will be created. Something close to the potential biodiversity is probably possible only on a landscape level. Yet the restoration of management-dependent biotopes is feasible, and the same applies to those site-dependent biotopes for which key elements exist within the area studied.

From the land use assessment point of view these developments are promising and should end with good biodiversity evaluations down to the regional and landscape level. To reach the local level, local data will be necessary, not only for single species but also for stands. An evaluation of twelve of the most common forest stands would suffice to characterise a large part of our forests.

The final evaluation should also be directed to the efforts to meet these requirements, and should then look at these efforts as something positive in the evaluation, leading to a reduction in the environmental load.

6. WTP evaluation of species

Only the multiplicity of species will be dealt with here. The multiplicity of ecosystems and the degree of genetic variations within the species, which are also parts of biodiversity, will be neglected. However, species multiplicity is the easiest indicator to use for biodiversity. Positive relationships can probably also be found between species multiplicity and the other two parts, and further work can add to the bioquality levels reached so far.

In Life Cycle Assessments, where the final effects of interventions or causes are to be weighted and assessed, some kind of eco-points are commonly used. When it comes to impacts on biotic nature, there is no objective (biocentric) eco-scale. Only an anthropocentric view is possible. A well-known principle is to estimate the effects of the impacts by referring to Willingness To Pay (WTP) for avoiding the effect. A number of estimations have been made. The work by Steen has been used here.

Steen writes in EPS 2000 [2, p39]: "The most well known change in the safe-guard subject biodiversity, caused by human activities, is extinction of species. Each year a number of species is extinct. The value given to this change can be estimated from the cost of counteractive measures. Therefore it seems reasonable to use the yearly extinction of species as an indicator for the safe guard subject."

Steen estimated the contribution to species extinction via the probability of global extinction of redlisted species, defining the category indicator "NEX", (Normalised Extinction of species). It is normalised to the species extinct during 1990. Total Willingness To Pay (WTP) in OECD for 1 NEX was estimated from Swedish figures (178 MEUR = "NEXse"), giving a global figure of NEX = 1.1E+11 EUR/NEX. The ratio is the population ratio between Sweden's population and the global population. The figure was chosen as the weighting factor, and with an uncertainty factor of 3.

Because of a lack of land use methodology, the default LCIA method of EPS 2000 stops at the average global level. However, in practice, most LCAs need local estimations, and at least a classification into different groups of species. Otherwise comparisons between different kinds of land use will be too uncertain.

A simple way to relate the endpoint type category indicator NEX, which is not measurable, to the measurable indicator "number of redlisted species" is to divide the NEX weighting factor by the global number of redlisted species, giving all redlisted species the same weight. As is reported above, the Swedish WTP estimation (W_{NEXse}) has also formed the also for the global NEX weighting factor. In this case we can therefore allocate the Swedish weighted impact value to redlisted species by dividing 178 MEUR by 4 120 (tables 1 and 2), the number of redlisted species in Sweden, resulting in 43 204 EUR or ELU per species.

However, giving equal weightings to all species is not reasonable from an anthropocentric point of view.

An alternative is to ask whether Nature's ability to substitute a loss of species is the same for the main groups of species – vertebrates, invertebrates and plants. It is well known that insects and other invertebrates change their gene composition very easily, thereby linking their gene composition into new species. An indication of such possibilities is given by the number of generations per unit of time, say 100 years, for a species. Trees live long, but many plants often have a new generation each year, that is 100 generations per 100 years. Vertebrates have between five and maybe two hundred generations per century. An average for the larger vertebrates may be around ten. Invertebrates on the other hand may have a number of generations each year.

If a weighting is made according to the possibility to overcome the loss of species, the factors 10, 1 and 0.1 for vertebrates, plants and invertebrates respectively would then be closer to reality. The relative numbers of species in the three groups – vertebrates, invertebrates and plants – are 2, 80 and 18 percent respectively on the global scale. However, in Sweden there are 503 vertebrates, 33 500 invertebrates and 16 000 plants, in all 50 000 species, giving shares of 0.01, 0.67 and 0.32. This still leaves out bacteria and other single-cell organisms (7 700 species). This is an acceptable solution, as this group is different from the big plant group and, if included, would have a weighting factor of 0.01. Thus a weighted allocation of the NEXse value to only the three groups of species results from the following equation:

$$178\,000\,000 = a \cdot 0.01 \cdot 10 + a \cdot 0.67 \cdot 0.1 + a \cdot 0.32 \cdot 1 \qquad (1)$$

where a is a proportionality factor, found to be 365.5E+6. When this value is put into the right membrum, the W_{NEXse} values in ELU (=EUR) for the three groups can be calculated:

Table 6.0		
	Vertebrates	36E+6
	Invertebrates	25E+6
	Plants	117E+6
	The sum is	178E+6

The nature of the weighted impact values is that they represent the WTP for avoiding endangerment or extinction of species. These risks are estimated in the red lists published in many (European) countries. Thus the NEX values have to be broken down to the species level. The red list is built up as follows. Based on local population counts, the species that have too few representatives to survive in the long run or which show declining populations are added to a red list in one of the following categories: DD = Data Deficient, RE = Regionally Extinct, CR = CRitically endangered, EN = ENdangered, VU = VUlnerable, NT = Near Threatened.

Clearly the species belonging to the CR, EN and VU categories should be subjects of environmental care in the first hand. However, the other species on the red list also call for some measures. The higher the risk of extinction, the more intensive the measures. Thus the following weighting figures for the categories mentioned are suggested:

CR: 4, EN: 3, VU: 2 and the three other categories, below assembled under RED: 1.

In what follows the allocation of weighted impact values to redlisted species in Sweden will be calculated according to this principle.

As an example , let us consider the number of redlisted vertebrates in the new Swedish Red List [12, p. 35].

Category: CR	EN	VU	RED	Weighted impact values for vertebrates
12	20	52	(19+11+42)	36E+6

The following equation can be set up:

$$12 \cdot 4 \cdot x + 20 \cdot 3 \cdot x + 52 \cdot 2 \cdot x + (19 + 11 + 42) \cdot 1 \cdot x = 36E{+}6$$

The proportionality factor x is found to be 126 760. The CR part is then $12 \cdot 4 \cdot 126760 = 06.08E + 6$. For each of the 12 critically endangered species the characterisation value is (6.08E+6)/12 = 507 040 ELU. Fulfilling this calculation for CR, EN, VU and RED as well, and the same for the invertebrates and the plants, has given the following result (ELU/EUR) when the weighted impact values have been smoothed out:

Table 6.1 Weighted species indicator values in ELU including all species populations in Sweden

CATEGORY	**CR**	**EN**	**VU**	**RED**	**SUM**
Vertebrates					
No. of species		12	20	52	72
156					
ELU per species	**507 000**	**380 000**	**253 000**	**127 000**	
Invertebrates					
No. of species		123	281	618	1315
2337					
ELU per species	**26 000**	**19 000**	**13 000**	**7 000**	
Plants					
No. of species		144	269	434	780
1627					
ELU per species	**154 000**	**116 000**	**77 000**	**39 000**	

If the number of individuals in a population is known – or the number of habitats – the impact of the loss of one individual or one habitat can be calculated, assuming a linear relationship between these indicators and NEX. As an example: if we have 100 wolves in Sweden, category CR [12, p. 151], the loss of one of them corresponds to an environmental loss of 5070 ELU.

However, in most cases the population figures for the species are not known. At the same time the counting of redlisted species in a land use assessment is not a good solution for practical reasons. However, as was mentioned above, inventories of key biotopes have been made all over the country. Key biotopes are areas having such high natural values that redlisted species will probably be found there. These biotopes may be just one old tree, but they can also cover several hectares. Within landscapes they usually cover between one and three percent of the total area, but scattered in small patches. They can house between none and over 25 redlisted species, but the average seems to be between one and five redlisted species per biotope belonging to the invertebrate or plant groups. To be able to estimate the biodiversity value of key biotopes, a model estimating the biodiversity will be presented which hopefully will be close to an average key biotope value for the landscape in question. It will include only the plants and invertebrates, which are more bound to the substrate of the habitat than vertebrates which move over wide areas, although some of them may be related to a key biotope, e.g. the whitebacked woodpecker. The key elements sought for in finding the key biotopes are valid for plants, but many of the key biotopes are habitats for redlisted invertebrates also, thus they are included as well. Adding the weighted indicator values for invertebrates and plants above (Table 6.1) will give the following weighting indicator values per species:

CR: 180 000, EN: 135 000, VU: 90 000, RED: 46 000 (ELU)

The number of redlisted species in these groups are the sums of plants and invertebrates:

CR: 279 EN: 570 VU: 1 104 RED: 2169

Assuming an average of 6 000 key biotopes housing CR-listed species and the aforementioned relations between the groups, the following numbers of key biotopes for the four groups of redlisted species discussed above are found:

CR: 6 000, EN: 12 000, VU: 24 000 RED: 46 000

By dividing the weighted species indicator values by the corresponding key biotope numbers, we obtain the weighted key biotope category indicator values (ELU/category):

Table 6.2 Average weighted species indicator values for those belonging to a threatened key biotope category

CR: 30 EN: 11 VU: 4 RED: 1

The assumption of 6 000 key biotopes containing CR species is of course only an assumption – no figure is known. If the "right" value is twice as high, the key biotope values will be half as large.

These values can be used to assess the average values of key biotopes provided that the area under study is large enough to contain a decent number of key biotopes. Consider an average key biotope housing 5 redlisted species with the following plausible relative distribution of redlisted species per key biotope:

CR: 0.5 EN: 1 VU: 2 RED: 4

As these distribution numbers add up to 7.5 they should each be reduced by $5/7.5 = 2/3$. This average key biotope has a value of about 23 ELU: $(0.5 \cdot 30 + 1 \cdot 11 + 2 \cdot 2 + 4 \cdot 1)2/3$. For key biotopes with 5, 4, 3, 2, or 1 redlisted species, the weighting values will be:

Table 6.3 Key biotope weighted values when the number of redlisted species are:

5	4	3	2	1
23	**18**	**14**	**9**	**5 ELU/biotope**

So far the procedure has been to allocate the NEXse impact value to the category indicators, starting from the end point category biodiversity, as follows:

Impact category	Category indicator name	Category indicator unit	Note
Extinction of species	NEX	Dimensionless	Share of species extinct yearly and globally
Extinction of Swedish species	NEXse	Dimensionless	Share of species extinct yearly in Sweden
Extinction of redlisted species	NEXred	ELU/red,yr	% of total species population

The allocation has so far been made in five steps, but only the two last include weightings as indicated below:

Step 1	Step 2	Step 3	Step 4		Step 5	
			Vertebrates <			
			CR	154 000	*x %	Cf Table 6.1
Global level	Swedish level	Loss of < redlisted	Invertebrates < EN	116 000	*y %	
			VU	77 000	*z %	
			RED	39 000	*u %	
			Plants <			

The procedure is shown in detail only for the invertebrate group. Due to the practical difficulty of finding the extinct or threatened proportions of the total species populations, this procedure cannot yet be recommended. It would also be an advantage not to have to go down to the individuals of the species. The attempt to evaluate the key biotopes is better from that point of view – especially as those biotopes are being mapped in Sweden these days. However, the next chapter will show another solution to the assessment problem.

7. Geographical allocation of the weighted species indicator values

Another step towards more specific values is to aim at a finer geographical subdivision. The separation into eight climatic zones will be made. The Swedish National Atlas of Plants and Animals [11, p.103] subdivides the main groups of species among Sweden's 24 historical provinces (regions), and the share of the total number of redlisted species in Sweden is indicated. Combining these regional figures with the eight zones of the climatic map, the author has calculated the multiplicity of each of the three groups in each zone. The following table shows only the end result of the calculations.

Table 7.1 NEX shares of the redlisted species in Sweden over the eight zones, MELU.

Zone	Vertebrates	Invertebrates	Plants	All
1. Alpine Sweden	4.5	1.4	12.5	18.4
2. Subalpine Sweden	4.2	1.4	12.5	18.1
3. Central northern	4.2	1.6	13.7	19.5
4. Northern east coast	4.7	1.4	9.5	15.6
5. Southern east coast	5.6	5.4	16.0	27.0
6. Lake Vänern district	4.3	2.7	17.8	24.8
7. Central southern	2.6	6.0	14.3	22.9
8. West coast	6.1	5.1	20.2	31.4
SUM	**36**	**25**	**117**	**178** Cf Tab. 6.0

The first three columns show the partial NEXse figures for vertebrates, invertebrates and plants in MELU (or MEUR). The sums are given in the last column.

A rapid look at the columns reveals some well- known facts about biodiversity in Sweden, which is still more evident at the page 103 cited above. They show that the diversity is significantly higher in southern Sweden compared to the northern part, the part north of the Limes Norrlandicus. This difference is explained by the fewer species of invertebrates and vascular plants in the northern part, mainly owing to the lack of southern deciduous trees.

Only 849 species belong to the most endangered groups CR and EN of the 4 120 species redlisted [12]. A study of the distribution of the species in these two groups shows that 55% of the species are bound to the southernmost parts of Sweden (the west coast, Skåne, Blekinge Gotland and Öland), 34% are bound to middle Sweden, and only 11% to northern Sweden including the four northernmost climatic zones 1 to 4. These figures stress still more the fact that most of the endangered species are to be found in southernmost Sweden.

However, the values are for all kinds of land, woodlands and agricultural lands having the biggest share according to Table 2. Thus the next step will be to separate those redlisted species that are endangered by forestry and agriculture from the rest. Table 2 suggests that approximately 33% of the species are endangered by forestry and 32% by agriculture. As wetlands are mostly parts of the forest landscape, the percentage for forestry is set at 40%.

The first objective is to arrive at good default values for the biodiversity within each zone, and expressed in ELU (=EUR) units per hectare and year. First we need to know the forest area of each zone, expressed in hectares. By using forestry statistics and the partial NEX zone figures for all species from above, and considering the fact that forestry accounts for 40% of the redlisted species, we can calculate the following series of figures:

Table 7.2 Forest areas in 1000 hectare in the 8 climatic zones:

1	2	3	4	5	6	7	8
3 181	4 292	5 609	3 266	3 131	758	1 400	1 686

Knowing that 40% of the redlisted species are endangered by forestry, and that the NEXse is 178E+6, is enough to get mean actual values per hectare and year of forest in Sweden:

$$(178E+6) \cdot 0.4/23\,323\,000 = 3.05 \text{ ELU/ha,y}$$

Sweden has about 2.8 million hectares of arable land and about 450 000 ha of pasture land. The 32 % of species that are endangered by agriculture are probably to be found mainly in pasture lands. Thus an average potential value for those areas can be calculated as follows:

$$(178E+6) \cdot 0.32/450\,000 = 130 \text{ ELU/ha,y}$$

If arable land is also involved, the result is about 18 ELU/ha,year. These figures indicate that the numbers of habitats for these species are much lower than for those endangered by forestry.

The final conversion from NEX zone values to the average value per hectare for the 8 zones and the three groups of species is made below. First the NEXse zone values in Table 7.1 are multiplied by 0.4 to arrive at the values endangered by forestry. Secondly, these values are

divided by the forest area for the zone in question, see Table 7.2. The result is given in the following table:

Tab. 7.3 Mean environmental impact values on the forest biodiversity in ELU/ha,year

	Vert.	Invert	Plants	All		Thousand
Redlisted:	53	969	764	1786		hectares
Zone						
1	0.56	0.18	1.57	2.3		3 181
2	0.39	0.13	1.16	1.7		4 292
3	0.30	0.11	0.98	1.4		5 609
4	0.58	0.17	1.16	1.9		3 266
5	0.72	0.69	2.04	3.5		3 131
6	2.27	1.42	9.39	13.1		758
7	0.74	1.71	4.09	6.5		1 400
8	1.45	1.21	4.79	7.4		1 686
					SUM:	23 323
Average 1–4	0.5	0.2	1.2	1.9	ELU/ha	
Average 6–8	1.3	1.2	5.1	7.6	ELU/ha	

The picture given above of the biodiversity in eight climatic zones in Sweden has a low degree of precision, but should be qualitatively right. Zone No 6 has the highest values, not because of higher NEX values but because they are to be found in a smaller forest area, resulting in a higher value per hectare. The average for the total Swedish forests derived from this table and for all species is 4.7 ELU/ha, a value which is 50% higher than that calculated above (3.05), which mirrors the effect of the unevenness in the distribution of zone forest areas.

The above biodiversity figures are actual values. How to find the potential values?

The lack of key elements that are needed to reach sustainability and a potential bioquality level is hinted at below:

Lack of
- Old forests and overaged trees
- Wetlands
- Successional biotopes/stands regrown after a forest fire
- Moist and dense old spruce forests
- Deciduous forests and very old big trees
- Dead wood
- Forest grazing and other forms of old-fashioned silviculture

The biodiversity is not evenly distributed over the forest areas (or the agricultural areas). Many of the threatened species (plants) are to be found within key habitats, which make up only a few percent of the forest areas. Together with other "care demanding areas" which are also rich in biodiversity, although still lacking or low in redlisted species, they currently constitute only about 5% of the forest area. The rest, which is up to 95 % of the forest area, is "production forest", where general considerations to the environment are taken, or where in some cases special considerations apply, but which are principally a source of harvestable wood. Yet these areas may house about 50% of the red-listed species.

In a natural forest there is no great difference in biodiversity between key biotopes and the rest of the forest as we have today. Thus the biodiversity value for the rest (= 95%) of the forest is higher in a natural forest. In assessing the biodiversity of natural and actual forests it should be remembered that it is not only the number of species present that is higher in the natural forest. The number of individuals in the redlisted populations is also much higher, leading to an increased number of key biotopes, and a higher biodiversity in what we call production forests. Maybe the key biotopes in natural forests covered 15% of the area instead of 5 %. Probably the impact values outside were two to three times higher or more than in the actual forests today. Apparently the differences were greater in the forests that are more intensely used today than for those lying farther away from the product markets. In the next table the mean environmental impact values given above are smoothed out a little to avoid errors based on the differences in the forest areas and then doubled. With these changes, it is suggested that the values presented be used as potential biodiversity values at the regional or landscape level.

Table 7.4 Suggested regional potential biodiversity impact default values, ELU/ha,y

Zone	Vert	Invert	Plants	All
1. Alpine Sweden	1.0	0.5	2.5	4.0
2. Subalpine Sweden	1.0	0.5	2.5	4.0
3. Central Northern	1.0	0.5	2.5	4.0
4. Northern East Coast	1.0	0.5	2.5	4.0
5. Southern East Coast	1.6	1.4	4.0	7.0
6. Lake Vänern District	2.2	2.8	9.0	14.0
7. Central Southern	2.4	2.8	9.3	14.5
8. West Coast	2.6	2.8	9.6	15.0

In this paragraph the NEXse value has been allocated to the surface of Sweden according to the occurrence of redlisted species. This results in a somewhat different impact category list:

Impact category	**Category indicator name**	**Category indicator unit**	**Note**
Extinction of species	NEX	Dimensionless	Share of species extinct yearly and globally
Extinction of Swedish species	NEXse	Dimensionless	Share of species extinct yearly in Sweden
Occurrence of redlisted species	Existence of red listed species per area unit = biodiversity value	ELU/ha,yr	

The third and last line in this table illustrates the conversion of the WTP value for a given coincidence into an estimated biodiversity value for the area unit in question. This is probably the most apparent difference in this report from the ordinary bottom-up procedure. The consequence is an easy way to evaluate the impacts of land use interventions.

8. The general picture of Swedish potential forest bioquality

The need for reference levels for bioquality was discussed in section 4, "The search for a reference state". These figures are given below in more detail, together with the estimated biodiversity figures calculated above.

Bioproductivity: 70% of the site quality is multiplied by the wood density (410 kg/m^3 for softwood and 500 kg/m^3 for hardwood = birch. As an average, the birch content is below 10% in northern Sweden, 10–12 % in middle Sweden, and higher but variable in south Sweden). These productivities in kg/ha,y are then multiplied by 0.04 , the EPS weighting factor for wood, giving the wood potential productivity in ELU/ha,y.

Using generic figures for the other biomasses (20 kg/ha,y for berries, 40 kg/ha,y for mushrooms, both having a weighting factor of 0.2 ELU/kg), a biomass value of $(20 + 40) \cdot 0.2 = 12$ ELU/ha,y is calculated. A fish & meat value of $2.0 \cdot 1.0 = 2$ ELU/ha,y can be added in the same way [10]. These productivity values are thought to be equal all over Sweden, giving an additional productivity of 14 ELU/ha,y.

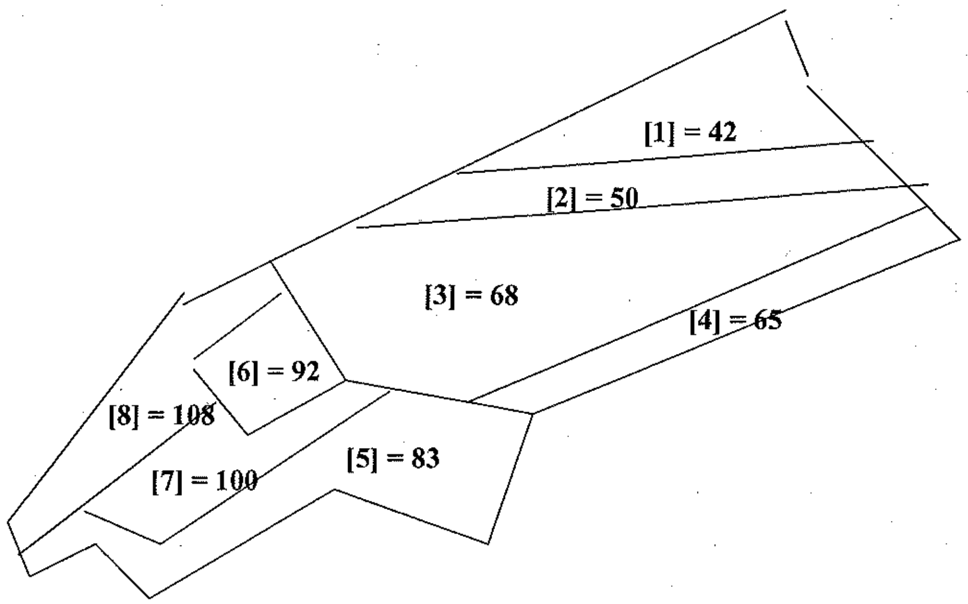
Biodiversity: The potential values are estimated above and inserted into the following complete table of potential productivity and diversity figures:

Table 8.1 Average potential bioquality values in eight Swedish climatic zones

Zone		1	2	3	4	5	6	7	8
Site quality,	m^{3}sk/ha,y	2.1	2.8	4.3	4.0	5.3	5.4	6.0	6.6
Pot. wood prod. value	ELU/ha,y	24	32	50	47	62	64	71	79
Other biomasses,	ELU/ha,y	14	14	14	14	14	14	14	14
Pot. diversity value,	ELU/ha,y	4	4	4	4	7	14	14.5	15
Pot. Bioquality,	**ELU/ha,y**	**42**	**50**	**68**	**65**	**83**	**92**	**100**	**108**

Other biomasses than wood have a high value, but aside from the fish & meat figures, which are based on the game yield, berries and mushroom figures mirror the annual productivities and not the annual harvests, which are only 6 to 16 % of the productivity figures. Worth commenting is also the relatively low contribution from biodiversity, which is only 10–16% of the total bioquality.

Average Bioquality within 8 Swedish climat zones, ELU/ha,yr



The figures given above can be used as default values when generic figures are acceptable or desirable. This is the case in product LCAs when the land use aspect is a relatively small part of the total environmental load.

9. Actual values and evaluation in product LCAs/case studies

So far the evaluation of the function of state called Potential Bioquality, with its components Bioproductivity and Biodiversity, has been treated. The potential levels have principally been taken from general estimations based on known data and without any use of cause-effect relations to interventions. The difference between Qu and Qa gives the environmental load on the land use. How to calculate the actual values?

Productivity: The aim of forestry land use is extraction of wood. Of course this means that the actual productivity goes down where thinning or final cut has taken place. Yet, as long as the extraction does not exceed the sustainability conditions over a life cycle or over the landscape, the productivity losses are accepted and taken into account in this method. This is the case if the annual yield at least on a long-term exceeds the extraction losses. Other biomasses than wood are less dependent on silvicultural measures, although the ecosystems are different in an old forest compared to those in juvenile forests. Thus the actual productivity is often not far from the potential, and can be greater – as this function has been defined here. The conclusion is that very often the bioproductivity is not damaged at all, or very little, e.g. when the forestry is acceptable according to FSC (Forest Stewardship Council) rules.

Biodiversity:

The actual biodiversity values on the regional and landscape level were given above (Table 7.3). Coming over to the local level the question is: How is the local diversity compared to that on the landscape level? Those vertebrates and invertebrates that easily move over the landscape, do they have habitats in the local area too? The redlisted species that cannot move easily, do they have habitats and the right key elements within the local area? Does the area show more or less diversity than other parts of the landscape?

Silvicultural measures may damage biodiversity very much if special care is not taken – especially at the final cut. But if the key biotopes within the forestry area are exempted from cutting, only the endangered species in the production forest part are jeopardised.
Suppose that all key biotopes survive the silvicultural measures that have been undertaken. What has happened to those in the production forest?

It is assumed here that half of the redlisted occurrences are within the key biotopes and the other half are outside in the production forest. Thus, if the potential biodiversity is 4 ELU/ha,y (as in zone 3), then the potential biodiversity in the (present) production forest is 2 ELU (50% of 4). However, in the potential case the key biotope area is supposed to be 15% of the area, but only 5% in the actual case. Thus the potential biodiversity outside the key biotope area is $2/0.85 = 2.3$ ELU/ha,y and $2/0.15 = 13.3$ ELU/ha,y within the key biotopes.

The same rule applied to the extensively used zone 3 forest of today will result in $1/0.95 = 1.05$ ELU/ha,y in the production area, and $1/0.05 = 20$ ELU/ha,y in the key biotope area. How does that change as a result of the silvicultural measures?

The now obsolete form of silviculture which aimed mainly at optimum wood yield meant that fewer measures were taken to pave the way for future improvements in biodiversity. The drop of the actual biodiversity in such a case was probably relatively high. Thus there was a clear risk of declining biodiversity afterwards when the key biotopes were not always known or cared about, and little was done actively to build up biodiversity after the intervention. .

Nowadays, when more stress is put on environmental issues and verification by certifications, silviculture is different. More biomasses are left in the final fellings, while measures listed above, aiming at restoring biodiversity and maybe even achieving improved actual biodiversity, may have been implemented. Thus even if the immediate drop could be about 50%, the measures taken would mean improved future gains in biodiversity. This should of course be mirrored in the land use assessment. It is suggested here that estimations should be made of the possible future effects of the measures taken. This could be expressed as a possible reduction of the ΔS. For instance, 50% of that reduction could be claimed in the assessment of the land use, see the case study below.

A calculation example

Land use load calculations are preferably made over a rotation cycle that may last 150 years in northern Sweden and 60–70 years in southern Sweden. Here is an example for a forest in zone No. 3. The landscape bioquality figures are given in Table 8.1 above. One hectare having bioquality figures equal to the average of the landscape is chosen. The rotation time is 90 years. It is a softwood forest with less than 5 % hardwood. The mixed wood density is estimated at 415 kg/m^3. As the site quality is 4.3 m^{3}sk/ha,y, the potential growth volume during the rotation period is $90 \cdot 4.3 \cdot 0.7 = 270$ m^3 under bark. This corresponds to a wood Pu

$= (270 \cdot 415 \cdot 0.04)/90 = 50$ ELU/ha,y. The final result proves to be only 250 m^3. At the final cut 20 m^3 are left as part of the nature conservation program. Thus the product yield over the rotation period is 230 m^3/ha or 230/90 = 2,56 m3/ha,y, and the wood Pa = $(250 \cdot 415 \cdot 0.04)/90 = 46$ ELU/ha,y. The productivity of other biomasses has been 14 ELU/ha,y over the whole period. Thus we have Pu = 50 + 14 = 64 ELU/ha,y. Based on the product yield, Pa = 46 + 14 = 60 ELU/ha,y, and thus ΔP = 4 ELU/ha,y or 4/ 2,56 = 1,56 ELU/m3

The potential biodiversity Su is 4 ELU/ha,y according to Table 8.1. That corresponds to Su/Sa = 2. Depending on the degree of degradation of the production forest, this ratio could even be 3, meaning that Sa is a third of 4ELU/ha,y in our example (= 1,33 ELU/ha,y).
Both ratios are used below to obtain an overview of the consequences for the assessment.

When Su/Sa = 4/2 = 2, the calculation is as follows:
Su = 4, Sa = 2, thus ΔS = 4 – 2 = 2 ELU/ha,y
ΔQ = ΔP + ΔS = 4 + 2 = 6 ELU/ha,y

When Su/Sa = 4/1,33 = 3, the calculation is as follows:
Su = 4, Sa = 1,33, thus ΔS = 4 – 1,33 = 2,77 ELU/ha,y
ΔQ = ΔP + ΔS = 4 + 2,77 = 6,8 ELU/ha,y

These figures are representative for the whole forest landscape as an average hectare was chosen. Concentrating on the final cut areas, the actual productivity as well as the biodiversity figures dropped simultaneously more than that. However, the productivity drop is already allowed for in the definition of the potential productivity, and the diversity variations within such a small part of the total area should not be considered.

The ΔQ value calculated is as an average for the whole rotation period. The actual yield was above found to be 230 m3 ub during 90 years or 230/90 = 2,56 m^3ub/ha,y. Thus the land use load E is

6/2,56 = 2,34 ELU/m^3. 6,8/2,34 = 2,66 ELU/m^3 respectively

Per kg wood the load E is:

2,34/415 = 0.0056 ELU/kg. 2,66/415 = 0.0061 ELU/kg.

As the environmental value of wood according to EPS is 0.04 ELU/kg, the land use load is only about 15 % of the wood's WTP value.

Although the actual biodiversity values (Sa) are found to be 2 and 1,33 respectively at the different Su/Sa ratios, the Sa value in the production part is lower. As the production part holds 50% of the Sa value the calculation gives;

Sa = 0,5*2/0,95 = 1,05 and Sa = 0,5*1,33/0,95 = 0,70

Apparently there must be good possibilities to improve the biodiversity by silvicultural measures.

It was pointed out above that if positive measures were taken to restore or improve the sustainability of the ecosystems, this should be considered in the assessment. Equ.1 can then be changed to:

$$E = \Delta Q - \Delta Q\mathrm{imp} \qquad (1b),$$

where ΔQimp is the reduction in the load ΔQ caused by silvicultural actionstaken to improve bioproductivity and/or bioquality.

Suppose that nursery-bred seedlings had been planted after the final cut. That would increase the wood yield in the future by about 10%. Suppose also that dead wood had been left as well as patches of old trees, and that part of the cut area had been burnt.

The future estimated increase in wood productivity corresponds to:

ΔP = $0.10 \cdot 46 = 4.6$ ELU/ha,y. ΔP = $0.10 \cdot 46 = 4.6$ ELU/ha,y.

The increase in Sa is estimated at 20%, thus:

ΔS = $0.20 \cdot .2 = 0.40$ ELU/ha.y ΔS = $0.20 \cdot 2.77 = 0.55$ ELU/ha.y
The sum ΔP+ΔS = 5 ELU/ha.y = 5.15 ELU/ha.y

50% of these sums (=ΔQimp) can be subtracted from ΔQ as an improvement compensation for the positive measures.

Thus E = $6 - 0.5 \cdot 5{,}0 = 3{,}5$ ELU/ha.y 6,8– $0.5 \cdot 5.15 = 4{,}2$ ELU/ha.y
or = 3,5/2,56 = 1,4 ELU/m^3. = 4,2/2,56 = 1.6 ELU/m^3

The result of the improvement measures is that the final impacts are reduced considerably, about 40% The comparison between two different levels of Su/Sa shows that the higher this ratio, the higher is the final load on the products – as could be expected.

The method of using the "footprint of humanity" to characterise the landscape gradient of Su/Sa, referred to in section 5, should be used to find the right Su/Sa ratio for the actual case.

A tonne of bleached softwood kraft pulp, needing 4.8 m^3 wood under bark, would then have a land use load between 3.8 and 7.8 ELU/t, depending on the Su/Sa ratio. Compared to earlier generic estimations made by the author, these figures are of the same order, but about 50% higher. On the other hand, this time the figures are much more accurate and sensitive to different local and external factors.

A continuation of the example

It is planned to build a road through the forest. A key biotope will be spoilt unless a diversion is made around the key biotope. What is the land use load for the two alternatives? On the road area the Sa will be zero.

Alt.1. A straight road, 100 m through the forest and over the key biotope. The road will affect a zone of 12 m width. The total area affected is then $100 \cdot 12 = 1200$ m^2. The key biotope has an area of 0.036 ha = 360 m^2. The road will cover two-thirds of the key biotope and the risk is that the whole biotope will be spoilt. According to the initial example Sa = 20 ELU/ha,y inside the key biotope and 1.05 outside. The road length outside is 80 m. Thus the biodiversity that will be lost outside is $(80 \cdot 12)/10000 \cdot 1.05 = 0.096 \cdot 1.05 = 0.10$ ELU/y, and inside the key biotope $20 \cdot 2/3 = 13$ ELU/y with a risk of losing the remaining 7 ELU/y – altogether 14 to 20 ELU/y.

Alt. 2 is to build the road entirely outside the key biotope, which would make its length 130 m. The load would be $(130 \cdot 12)/10000 \cdot 1.05 = 0.16$ ELU/y. Conclusion: It is very expensive to destroy key biotopes.

10. Comparisons of land uses

10.1 Comparisons of forestry (or agriculture) land uses of different intensities.

Going from extensive silviculture towards increasingly intensive silviculture generally means increased production per area and less biodiversity. Thus, if production increases more than the environmental load increases, the land use load per functional unit may decrease with higher intensity. Yet the total load of the products may still increase, because other inputs needed to reach higher intensity contribute to high environmental loads, e.g. fertilisers. On the other hand, intensive silviculture close to the sawmills or pulp mills contributes to lower transport energy consumption per functional unit.

Another factor to consider is that intensive culture means that smaller areas need be used to reach a given extraction volume. If the released area is kept as a reserve, the landscape bioquality might increase in spite of the partly intensive use. This might be especially true if the concentration of land use leads to less fragmentation of the forest.

If a comparison is made where two areas are involved, the potential bioqualities are usually different. If production of pine wood is compared in Sweden and in Brazil, both the potential bioproductivity and biodiversity will usually be much higher in Brazil. Thus the bioquality load will be very high in Brazil if large areas of the landscape are used for pine plantations. On the other hand, if over 50% of the forests are left aside, the degree of fragmentation will be low, and the biodiversity of the landscape high. Then the bioquality load will decrease very much, resulting in fairly low loads per m^3 wood if the wood yield is high.

10.2 Comparisons of different kinds of land uses.

The reference values are the same as for forestry. With other words, the potential bioquality of the area in question is calculated in the same way as for forestry. The actual values have to be estimated using local knowledge of the prevailing conditions.

11. Agricultural land use

An agricultural landscape is much more varieated than a forest landscape. Part is patches of forest and forest edges. Part of it may be settlement areas surrounded by park-like areas, often with old deciduous trees of high environmental value. Ditches and rivulets discharging into a lake are also common elements. But most of the area is arable and pasture land. Almost as many species are endangered by agriculture as by forestry. As was shown in section 7, the number of redlisted species is higher per hectare in the agricultural landscape. This is due to the smaller area for agriculture compared to forestry. Within the agricultural landscape few species occur on the arable land. Ditches and roadsides, together with pasture land and wetlands, used to contain high biodiversity values, especially in olden times.

The modern requirements for rationality therefore eliminate many of the key elements for biotopes in today's farmlands. One solution is to get together within the landscape to restore something of the old-fashioned way of farming, and keep it alive with public aid. Farming often results in different kinds of products. Thus there may be an allocation problem to calculate the environmental loads of the different products.

12. Reporting

In this report the results of land use evaluations are given in ELUs, the same unit as for all kinds of interventions in an LCA where the EPS system has been used. In this method land use assessment deals with the evaluation of the environmental loads on bioquality. Yet all calculations should be separated into the components of bioquality – bioproductivity, biodiversity and others if such are included. Otherwise information will be lost. Also, according to the ISO norms, for comparative assertions disclosed to the public, the interpretation must be conducted category indicator by category indicator.

13. Accuracy and precision issues

The fundament is the Willingness To Pay system as it has been presented in the EPS system. The strength is that all LCA impacts have been evaluated according to the same yardstick. The weakness is that WTP is an anthropocentric evaluation, not a biocentric one, which might have been the ideal. Thus Steen has suggested a standard deviation corresponding to a factor of 3 for the global NEX value. As the Swedish NEX value was the starting point, one might think that the factor could be less than 3. On the other hand, the NEX value is ten years old, and human willingness to pay for the biodiversity was probably lower then than today.

Once the NEX value is settled, the valuation of the over 4 000 redlisted species in Sweden is a question of allocation after some important weighting procedures. Thus the manner of allocation among the groups of species can be discussed, but the total sum is always the Swedish NEX sum. However, when the EPS NEX system is used, the relative positions among the species are fixed, resulting in acceptable precision when comparisons are made.

Another source of uncertainty in making the assessments lies in the estimation of the difference between the potential and actual biodiversity in the production forest. This difference should be based on the distribution of the redlisted species in the forest, and especially between the key biotopes and the production forest. For Sweden a 50/50 distribution seems to be an acceptable and simple suggestion, but no one knows. The hypothesis here is that there is a bigger difference between the production forest values and the potential values for this area than between the key biotope areas. For that reason the potential values for a modern Swedish production forest are said to be two or three times better than the actual values. It is well known, however, that the near-natural forests are close to their potential values. On the other hand, these forests cover less than 1% of the Swedish forest area. The conclusion is that this multiple between actual and potential values must be chosen with knowledge of the local situation.

14. Summary and conclusion

The land use method described above is based on the following principles:

- The method of valuation is the Willingness To Pay method used in EPS.

- A land area's environmental value is called its Bioquality, having the components Bioproductivity and Biodiversity. These components are seen as functions of state, which characterise the land use value and which can be damaged by actions in the use of land, as described in this report.
- Potential levels of bioproductivity and biodiversity, which serve as reference values, can be defined for each piece of land. The loads allocated by the assessment are equal to the distance between the reference state and the actual state in which the land is at the moment of the analysis. In comparing two actual states of the land, the difference in load is given by the difference in actual bioqualities for the areas being compared.
- The annual productivity of biomasses forms the base for the bioproductivity, and the occurrence of redlisted species is the platform for the biodiversity evaluation.
- Potential as well as actual bioproductivity and biodiversity default values have been estimated for eight Swedish climatic zones using available data.
- The inhomogeneity of forest lands, with patches of key biotopes in a network of production forests which are low in biodiversity, calls for different bioquality values for key biotopes and for the main "production forest land".
- On the regional level these default values prove the feasibility of the top-down method. The environmental impacts that the land use causes can be established for the functional unit, which in forestry can be 1 m^3 of solid wood or 1 kg of solid wood.

Advantages of this method:

- The same category indicators - productivity and diversity – can be used on any area to measure its potential environmental value. The corresponding actual values form the base for an estimation of the degree of degradation. The same yardstick is used as for other LCA interventions.
- In principle the top-down approach facilitates rapid general estimations by the use of default figures for regions and possibly landscapes. Full realisation requires the direct availability of data of the GIS type, which is still on the want list.
- Comparisons within as well as between land uses are possible and meaningful.
- The additive nature of the parameters used in this model makes it easy to include new parameters if necessary, and to fine-tune data when this is desirable.
- The final effects calculated (e.g. in ELU/ha,yr) have dimensions that facilitate the allocation of the functional unit given in amount per unit weight or volume per year, resulting in effect per amount per year on the functional unit (e.g. ELU/ton, y).

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TABLE A. Number of redlisted species

	CR	EN	VU	RED*	SUM
Vertebrates	12	20	52	72	156
Vascular plants	58	111	157	179	505
Algae	5	6	5	5	21
Mosses	15	20	69	134	238
Fungi	36	46	142	355	609
Lichens	30	56	61	107	254
Invertebrates	123	281	618	1 315	2 337
Total	**279**	**570**	**1 104**	**2 167**	**4 120**

*RED = DD+RE+NT (= not endangered, but on the red list)

TABLE B. Incidence of redlisted species in various landscape types

	Total, whereof	**vertebrates,**	**plants and**	**invertebrates**
Forests	2101 (33%)	59	945	1097
Agriculture	2013 (32)	78	733	1202
Urban biotopes	577 (9)	21	84	472
Mountains	166 (3)	19	118	29
Wetlands	620 (10)	63	226	331
Freshwater	258 (4)	53	52	148
Sea and sea shores	517 (9)	71	97	349
Total	**6250 (100%)**			

The data in these tables are all taken from The Red List 2000 [12]