Physical impacts of land use in product life cycle assessment

Final report of the EURENVIRON-LCAGAPS sub-project on land use

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Introduction

This is the final report from the sub-project “Quantitative environmental assessment of land use in relation to the product life cycle” of the EUREKA project EU-1296 entitled “Development and application of major missing elements in the existing detailed Life Cycle Assessment methodology (LCAGAPS),” which was funded by the Danish EUREKA-secretariat at the Danish Agency for Industry and Trade. Through the Danish funding it was possible to involve a Dutch expert in the field, Erwin Lindeijer, to participate in the work.

The original concepts upon which this report is based were presented to the international scientific community in 1996 (Weidema & Mortensen 1996, Blonk *et al.* 1996), and within the field of biodiversity assessment some key ideas were developed in the report by Schmidt (1997). Several of the scientific topics related to environmental assessment of land use have been in rapid development during the scheduled period of the LCAGAPS project, especially in the fields of assessment of biodiversity and biogeochemical substance cycles. The finalisation of the project was postponed to take advantage of this concurrent and still ongoing development, and in the following years we focused on contributing to the conceptual development, especially in the SETAC working group on impact assessment (as documented *e.g.* in Lindeijer *et al.* 1998). In view of the rapid advancement in modelling and data availability, we have placed emphasis on assessment indicators that can function at the current level of available information, while being amenable for refinement as more data become available. For the same reason, not all aspects of the topic have been treated in equal detail. The final results of the project are presented with the present report.

Executive summary

Land uses, such as agricultural production, mineral extraction, and human settlements and infrastructures, have a number of physical impacts on flora, fauna, soil, and soil surface, which are often neglected in product life cycle assessment (LCA) because of lack of adequate impact indicators.

In this report we discuss the quantified assessment of the physical impacts of land use in terms of indicators for biogeochemical substance and energy cycles, ecosystem productivity, biodiversity, cultural value and migration and dispersal. The indicators are placed in a comprehensive framework (Chapter 1) describing the impact chains from the starting points (the physical impacts) to the endpoints (areas of protection).

We distinguish between occupation impacts (from land occupation) and permanent ecosystem impacts that involve a permanent change in the relaxation potential, i.e. the steady state that can be reached if the land is left to relax at the end of human activity. This report deals mainly with occupation impacts.

The occupation impact (I_{occ}) from any human activity can be calculated from the formula: $I_{occ} = A * t_i * (Q_{pot} - Q_{act})/s_i$, where A is the area occupied, t_i is the period of occupation (also including the relaxation period), Q_{pot} is the indicator value (e.g. for ecosystem productivity and biodiversity) for the relaxation potential, Q_{act} the indicator value during the human activity, and s_i is a slope factor to reflect that during relaxation, the indicator value Q_{act} will gradually approach Q_{pot}.

The indicators for ecosystem productivity and biodiversity are developed in relative detail (in Chapter 2) and we have placed emphasis on developing the indicators in such a way that they can be used at the current level of available information, while being amenable for refinement as more data become available.

As indicator for ecosystem productivity, we select Net Primary Productivity (NPP) as a reasonable mid-point indicator for the impact on biotic resources, the potential for agriculture, and most of the life-support functions of natural systems.

For biodiversity, we develop an indicator that includes species richness, inherent ecosystem scarcity (expressed as the inverse of the potential ecosystem area that could be occupied by the ecosystem if left undisturbed by human activities), and ecosystem vulnerability (indicating the relative number of species affected by a change in the ecosystem area, as expressed by the species-area relationship).

In Chapter 3 and 4, we present tentative data and data references for the different indicators and their practical application, including normalisation references for the ecosystem productivity and biodiversity indicators. In Chapter 5 these indicators are applied on a few product examples.

1. Concepts and definitions

1.1 Land use and physical impacts

The term “land use” is traditionally used to denote a classification of those human activities, which occupy land area. In the field of product life cycle assessment (LCA), the term “land use” or “land use impacts” has been used to denote the environmental impacts related to physical occupation and transformation of land areas. LCA operate with a number of other environmental impact categories, such as “climate change (global warming)”, “stratospheric ozone depletion”, “human toxicity”, “eco-toxicity”, “photo-oxidant formation”, “acidification” and “nutrification” (see Udo de Haes *et al.* 1999).

To avoid double-counting, it is important to make a clear distinction between the different impact categories and between the impacts and the human activities that cause the impacts. Human activities (including different land uses in the traditional sense) may have several physical and chemical exchanges with the environment:

- Substances emitted to air
- Substances emitted to water bodies
- Substances that are removed from the soil, through wind erosion, run-off from the surface, with crops, or directly by physical removal
- Substances that are left in the soil or on the soil surface
- Physical impacts on humans (accidents) and animals in human care
- Physical changes to the original flora, fauna and soil, including soil compaction and other changes in water infiltration and evapotranspiration
- Physical changes to the surface, including changes in reflection of solar radiation (albedo)

Some of these exchanges may affect several of the above mentioned impact categories.

Among these exchanges, it has been suggested (Lindeijer *et al.* 1998) that it is the physical changes (except those on humans) that should be covered by the term “land use”. In line with this, other terms have been suggested that focus more on the physical change: “Physical impacts of land use” or focusing more narrowly on the flora/fauna aspect: “Physical habitat depletion” (Cowell 1998).

In this document, we use the term “land use” in its traditional sense, as a classification of human activities that occupy land area, and we use the term “physical impacts of land use” to denote the physical changes mentioned above, which are the subject of this report.

The relations to other impact categories, not included in this paper, are described in more detail in section 1.2.

1.2 What is not included here (relationship to other impact categories)

In this report, we do not consider the following impacts from land use:

- Impacts *from* the area under human use to the surrounding area, caused by substance emissions (to air, water, soil or soil surface, including emissions from decomposition of organic matter and emission of soil through erosion), since such impacts are typically modelled as separate impact categories in LCA, even when these substances are emitted as a result of the land use, and even when the impact chain of these substance emissions affect the same mid-points as the physical impacts (however, when such an overlap in impact chains occur, the indicators discussed in chapter 2 may be relevant as indicators for the impacts from substance emissions). Note that the concepts developed in section 1.4 and 1.5 may nevertheless also be applied to the issue of background emissions, i.e. the emissions from the area in its potential steady state after all human use has ended.
- Impacts *to* the area under human use from emissions from the surrounding area and from impurities in substances applied as part of the land use, since in LCA such substance impacts are typically modelled as separate impact categories to the area of protection "Made-made structures and ecosystems" (see section 1.3.2).
- Impacts *to* the area under human use from the intended addition of substances (i.e. not covered in the above point) or from the removal of substances (including soil and nutrients), directly, through decomposition or erosion, or with manure or crops, since such impacts are modelled separately as an impact to the area of protection "Resources" (see section 1.3.2), either in parallel to other deposits of materials or as an impact to the potential for agriculture. Note here that midpoints in these impact chains may be "Regulation of nutrient concentration" and "Topsoil preservation" under the protection area "Life support systems" (see also section 1.3.3).
- Impacts from the use of water, since this is typically modelled separately, when relevant (but may then affect the same mid-points as the physical impacts, which implies that the indicators discussed in chapter 2 may also be relevant as indicators for the impacts of water use).
- Physical impacts to aquatic ecosystems, since the indicators for these impacts are essentially different from those appropriate for terrestrial land use (Dankers & Leopold 1998). However, some of the theoretical concepts developed in this paper may also provide a basis for developing indicators for aquatic impacts.
- Physical impacts on humans (accidents) and animals in human care, even when these impacts occur as a result of the land use, since this is typically covered by separate modelling in LCA (if not intentionally omitted).

1.3 Impact chain

An impact chain describes the relation between a given human activity and its environmental impact.

1.3.1 The starting point: Boundary between inventory analysis and impact assessment

In LCA, the starting point of the impact chain is the inventory, the result of the product life cycle inventory analysis. The inventory is a more or less aggregated list of the exchanges of a product system. A product system is the totality of human activities related to the life cycle of a specific product. It is therefore essential to define clearly the borderline between the product system (the human activities) and its environment (upon which the product system impacts). The same item may be an impact in one study, while being a part of the product system in another study.

There are three ways in which an impact may become an inventory issue rather than an impact assessment issue:

1) As an object of study: The object of the LCA, its so-called functional unit, may be related to the physical impacts of land use, as when investigating different ways of landscape maintenance, *e.g.* mechanical versus biological maintenance activities. In this case, all product systems under study (the human activities related to the different forms of maintenance) are deliberately normalised with regard to the physical impacts of land use. These impacts are thus equal in all systems and not relevant as an impact assessment issue. The inventory will thus be limited to all other environmental effects, besides the physical impacts of land use.
2) As an "impact for treatment": In the early days of LCA, "Waste" was listed as an inventory item, because the waste was largely leaving the human interest-sphere in an untreated form. Today, we are accustomed to include "Waste treatment" as a human activity within the product system, so that it is the environmental exchanges from waste treatment that are the starting point for the impact assessment. The same is now relevant for other impact categories, if it can be reasonably justified that remediation actually will take place. This is *e.g.* relevant for some land use activities (the mandatory remediation after many mining and quarrying operations, the demand to replant forests elsewhere when harvesting, etc.), which are included as part of the inventory analysis. Thus, only impacts that are not (expected to be) remedied are to be included in the impact assessment.
3) As a by-product: A physical impact of land use should be regarded as a by-product when it actually leads to the displacement of dedicated landscape maintenance activities. This is the case of some "positive" impacts, *i.e.* impacts with a positive value to humans, such as physical changes that increase the biodiversity of an area, *e.g.* when grazing (with meat or milk as the main product) as a side-effect keeps an introduced obnoxious weed under control and thereby obliterates the need for mechanical control of this weed (i.e. the by-product is weed control). The resulting effect on the environment (the physical impact) is neutral (the by-product displaces equal amounts of dedicated maintenance) and there is therefore no physical impact to be assessed.

It has been suggested (Heijungs & Guinée 1997) that the human competition over land area (or any other scarce resource for that matter) should be regarded as an impact in LCA (Note that this is different from competition between humans and nature, which is the main topic of the physical impacts of land use as understood in the present report). However, the consequence of human competition over land is simply that another land use (the use most sensitive to competition) will either be given up, be transferred to other land areas, or be displaced by other human activities fulfilling the same need. Thus, it is a special case of point 2 above: The "impact" competition is "remedied" by other human activities, which are therefore included in the studied product system. This is parallel to the way other scarce resources can be treated in LCA: by including the future consequences of the current use in the inventory analysis (see Weidema 2000). Thus, there is no need to include "competition" as a special impact category in LCA.

In conclusion, those physical changes, which are not part of the object of study, not remedied, and not displacing other activities, are listed in the inventory, as the starting point for the impact chain.

From this starting point, we could now follow the impact chains of the different physical changes (of flora, fauna, soil, and soil surface) one by one. However, it may be more enlightening first to know where we are headed:

1.3.2 The end points: Areas of protection

Four areas of protection (valuable in themselves or to humans) have been identified by Udo de Haes *et al.* (1999): Human health, man-made environment, natural environment, and natural resources. Natural resources as a protection area reflects the concern of availability to future generations. Natural resources may be any part of the natural environment, but the protection area is only affected if availability to future generations is affected, i.e. through irreversible depletion. In contrast, natural environment as a protection area is defined in terms of its current value (to humans or in itself), and may be affected both by reversible and irreversible depletion.

Since the natural environment (understood in every-day language terms) physically includes natural resources, it may be useful to rename the protection area "Natural environment" into *e.g.* "Natural Ecosystem", "Ecosystem health", "Ecosystem functions" or "Life-support functions", leaving the irreversibly depletable aspects "Biotic resources and biodiversity" under "Resources."

Similarly, the protection area "Man-made environment" includes aspects which are of concern to future generations (unique cultural assets), and aspects that are only of current value (*e.g.* ordinary buildings and current crop yields), which makes it useful to rename (redefine?) the protection area "Man-made environment" into "Man-made structures and ecosystems", leaving the unique cultural aspects under "Resources", which thus becomes not only "*Natural* resources."

Thus, we may outline the following areas of protection (see also figure 1):

- Resources, consisting of:
  - Deposits of materials and energy carriers,
  - Biotic resources,
  - Biodiversity,
  - Land area with potential for agriculture,
  - Unique types of landscapes,
  - Unique cultural assets (unique cultures, historical or archaeological sites or structures and other non-reproducible cultural media).
- Health/welfare of humans (and animals in human care – forgotten or intentionally omitted in Udo de Haes 1999?).
- Man-made structures and ecosystems.
- Life-support functions of the natural systems, which includes[1]:
  - Temperature regulation of air, water and land surface (through the "greenhouse effect", movements of air and water currents and interaction with the hydrological cycle),
  - Regulation of fresh water availability (through precipitation, runoff, evapotranspiration, and soil storage),
  - Regulation of nutrient concentrations (via weathering, photosynthesis, nitrogen fixation, movement by and through organisms, decomposition, and sedimentation)
  - Topsoil formation and preservation,
  - Removal of unwanted substances (by filtering, immobilisation, and biological decomposition),
  - UV-protection (by the stratospheric ozone layer).

Several ways of sub-dividing and describing ecosystem life support functions have been suggested, *e.g.* de Groot (1987, 1992) and Daily (1997). The short list suggested here summarizes these typologies according to functional characteristics.

The impact chain is a description of how the starting points (physical changes to flora, fauna, soil, and soil surface) are connected to the above end points. However, as shown by figure 1, the endpoints are not independent, i.e. an area of protection can be an endpoint for the valuation, while at the same time being a midpoint in the impact chain of another endpoint.

[1] Barbier *et al.* (1994) describe the Life Support System (LSS) as the continuous interaction between organisms, populations, life communities and their physio-chemical environment. IUCN/UNEP/WWF (1991) define it as the ecological process that maintains the productive, adaptive and renewal capacity of land, water and/or the whole biosphere. One could say that the LSS is an anthropocentric perception of ecosystems and the biosphere, wherein above-mentioned interactions and ecological processes are considered essential for human existence and its ways of life (van Wetten *et al.* 1996). We find it easier to operationalise nature value in functional terms than in terms of existence, although it may be argued that this somewhat anthropocentric description leaves out the inherent value of ecosystems. However, the existence of ecosystems is conditional upon the existence of the life support functions and vice versa, so in practice the anthropocentric position will also cover a more biocentric perspective (Turner 1991).

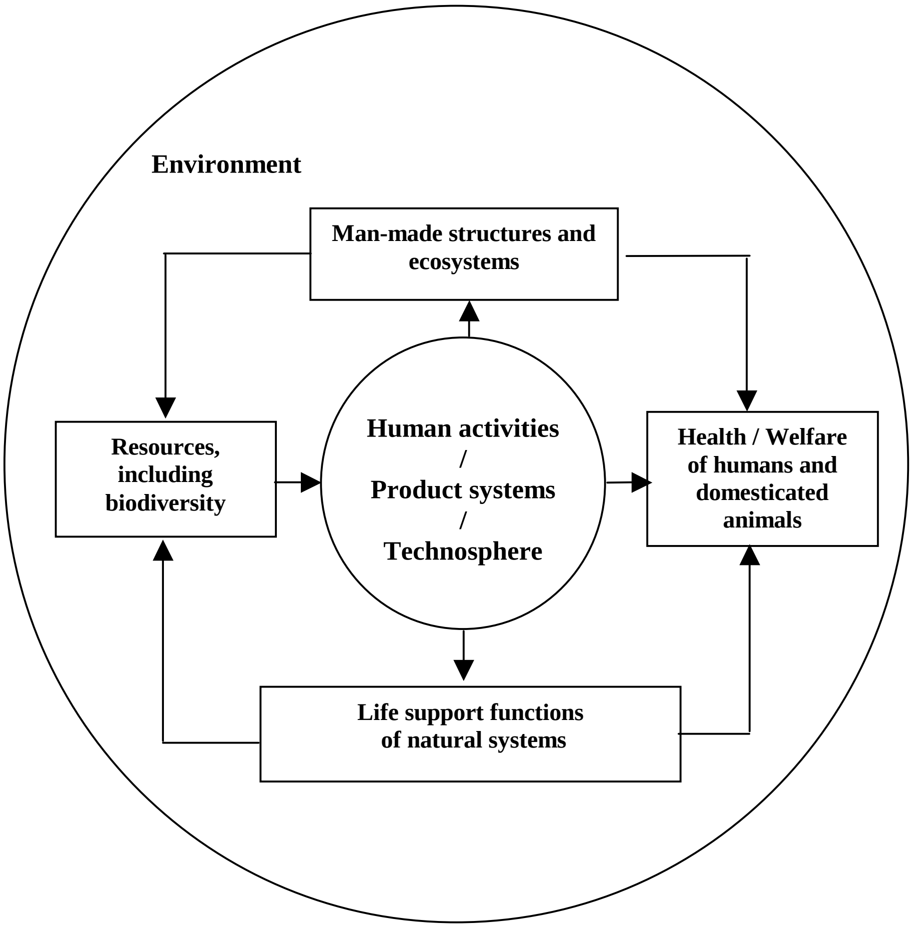

Figure 1. Areas of protection and their relations.

1.3.3 Mid-points of the impact chain

The impacts resulting from *physical changes to flora* or *fauna* are closely interrelated and can hardly be viewed in isolation. A change in flora will affect the fauna, *e.g.* in the form of pollinators, pests and herbivores, which may be specifically dependent on the affected flora. In the same way, a change in fauna will affect the flora. The degree of interrelationship will depend on the specificity of the dependencies, and the role of the altered flora or fauna within the specific ecosystem. It may be possible to distinguish key species within an ecosystem, for which a change may imply larger consequences for the ecosystem than a change in other species. In general, our knowledge of ecosystems and species interrelationships does not have a degree of detail that allows us to quantify the consequences of a change in a specific species. Thus, at the present state of knowledge we may have to express the impact from changes in flora and fauna in a general term such as *"altered species composition and population volumes"* (arrow 1 in figure 2). It should be noted that physical changes to flora and fauna may take place through a number of very different vectors, spanning from the physical introduction of new species, over activities like hunting, to the intentional change of natural ecosystems into man-made ecosystems for production, recreation, martial activities or habitation.

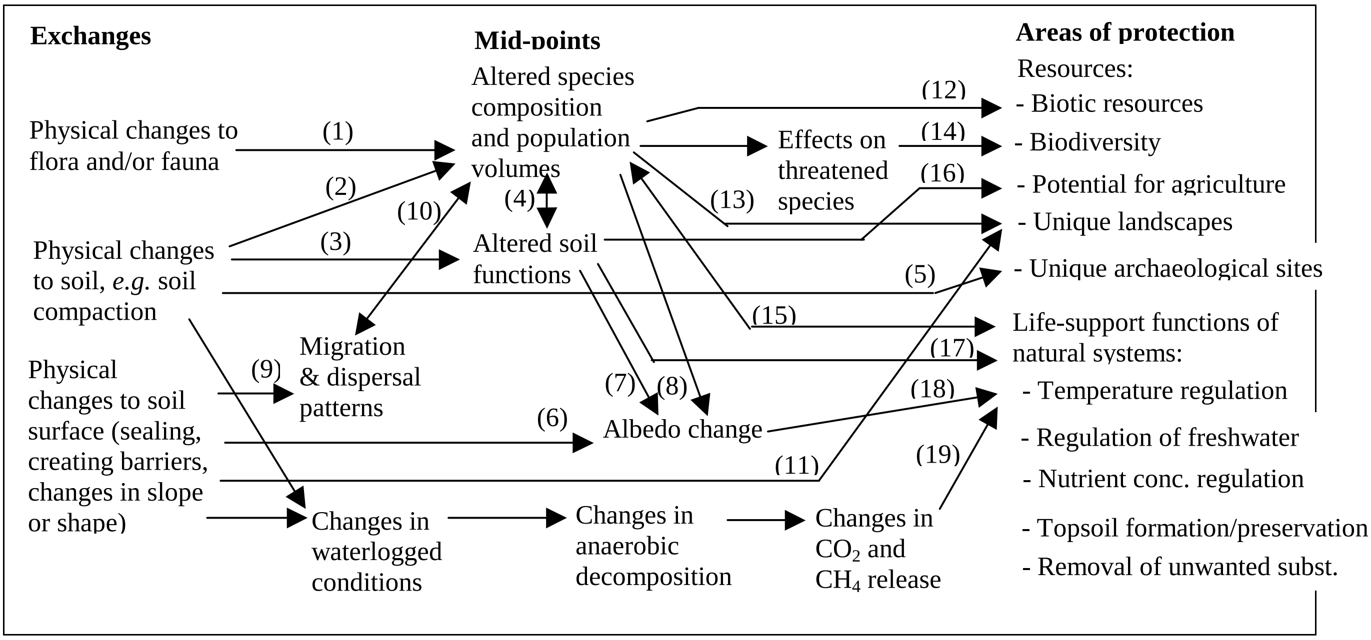

Figure 2. Impact chains for physical impacts of land use

A *physical change to the soil* may also lead to *"altered species composition and population volumes"*, either directly (arrow 2 in figure 2) or as a consequence of *altered soil functions*, especially related to water infiltration and water holding capacity (arrow 3). Altered soil functions may itself be caused by altered species composition, thus forming a feedback loop (arrow 4). Physical changes to the soil may also have a *direct impact on archaeological sites* (arrow 5), which is a sub-category under unique cultural assets of the area of protection "Resources," and on *temperature regulation* via changes in waterlogged conditions (arrow 19).

Physical changes to the soil surface may have an impact (arrow 6) on the *albedo* (which may also be affected by the above mentioned altered soil functions and altered species composition, arrows 7 and 8) and on migration and dispersal patterns of flora and fauna (arrow 9), thus interacting with the *species composition* of ecosystems (arrow 10). Physical changes to the soil surface may also directly impact on *unique types of landscapes* (arrow 11), which is a category under the area of protection "Resources."

The mid-point *altered species composition and population volumes* may be related directly (arrow 12 & 13) to *biotic resources* and *unique types of landscape* of the area of protection "Resources". The relationship to *biodiversity* (arrow 14) is also fairly straightforward, through the mid-point *"effects on threatened species"*. An altered species composition, especially of the vegetation, may also affect practically all the categories under the area of protection *"Life-support functions of natural systems"* and vice versa (arrow 15).

The mid-point *altered soil functions* can be related directly to the *potential for agriculture* (arrow 16) under the area of protection "Resources" as well as to several categories under *"Life-support functions of natural systems"* (arrow 17).

The mid-point *albedo* relates directly to the *temperature regulation* under the area of protection “Life-support functions of natural systems” (arrow 18).

It should be noted that in addition to the impacts chains of figure 2, the different *life-support functions are interrelated both within themselves* and back to the midpoints *migration & dispersal patterns, altered species composition and population volumes* and *altered soil functions*, thus creating a complicated network of relationships, for which it may appear difficult to find any simple indicator. This issue is dealt with in chapter 2.

Furthermore, it should be noted that the areas of protection mentioned in section 1.3.2 are not in themselves independent, which implies the possibility for further modeling of the impacts, *e.g.* towards the area of protection “Health/welfare”, as shown in figure 1. This issue is not dealt with in this paper.

1.4 Occupation and transformation of land

The physical impacts of land use are related to either land transformation or land occupation.

Land transformation is the process of changing the flora, fauna, soil or soil surface from its original state to an altered state. The altered state (level B to C in figure 3) may be temporary, so that after the human activity (ending at t_2), the flora, fauna, soil or soil surface is undergoing a *relaxation period* (with or without human intervention), finally arriving at a new steady state (level D in figure 3, which may be lower, equal to, or higher than the original level A). The transformation may be instantaneous (as at t_1) or gradual (as during the human activity from t_1 to t_2 in figure 3).

Land occupation is the maintenance of the flora, fauna, soil or soil surface in a state different from that steady state which can be reached after the relaxation period. This includes the occupation between t_1 and t_2, which postpones the beginning of the relaxation period, and the occupation during the relaxation period (t_2 to t_3), where the flora, fauna, soil or soil surface is also different from the potential steady state (D).

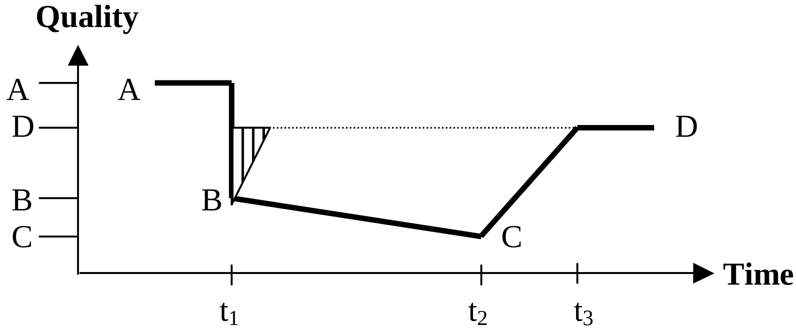

Figure 3. State of the flora, fauna, soil or soil surface before, during and after a human activity taking place in the time interval $t_1 - t_2$.

Occupation impacts refer to the impacts from land occupation. In figure 3, the occupation impacts can be illustrated by the area between the fully drawn curve and a reference level to which this altered state is measured (typically level D, see the discussion in section 1.5). In figure 3, the temporary state (level B to C) is drawn at a lower level than level A and deteriorating, although it may as well be at a higher level, and constant, increasing or fluctuating. Occupation impacts are expressed in units of quality * area * time.

The difference between levels A and D is the net land transformation or *permanent ecosystem impact*, which can be expressed in units of quality * area. This involves a change in the *relaxation potential*, i.e. the steady state that can be reached after relaxation. Thus, it represents the permanent or irreversible changes in the quality of an area.

Permanent ecosystem impacts may be caused either instantaneously by a land transformation, as illustrated in figure 3 (where the impact occurs at t_1) or gradually (as shown in figure 6 in section 1.5, where the change in the relaxation potential takes place gradually over the entire period t_1 to t_2).

A human activity that effectively has no duration, such as the clear-cutting of a forest, and thus involves only a transformation at time t_1, may or may not have permanent ecosystem impacts, but it will always have an occupation impact, since relaxation is never instantaneous. The occupation impact of such an activity is illustrated by the triangle with vertical lines in figure 3 (given by the relaxation time that it takes before level D is reached – note that the slope of the curve during relaxation may be different for an activity that ends at t_1 and an activity that ends at t_2). The permanent ecosystem impact is the difference between level A and level D.

A human activity with a certain duration, and which changes the current state, but does not affect the level of the final steady state, has no permanent ecosystem impact but only an occupation impact, which can be measured as the additional area below the reference level due to the activity. In figure 3, the activity taking place *between* t_1 and t_2 (i.e. not including the initial land transformation) is causing the additional white area between the fully drawn curve and the reference level D, i.e. the full area between these two curves minus the area ascribed to the previous activities (in this case the triangle with vertical lines ascribed to the land transformation at t_1).

A human activity with a certain duration, but which affects neither the current state, nor the level of the potential final state (as illustrated in figure 4 by the rectangle between t_1 and t_2 and level B and D, where the activity maintains the current state B and does not affect the relaxation potential D), does not have any permanent ecosystem impact, nor any impact from relaxation, but can be simply measured as $(D-B)*(t_2-t_1)$. It can be seen as a simple postponement of the start of relaxation from time t_1 to time t_2. Despite the postponement, the need for relaxation is still caused by the initial land transformation at time t_1, and shall therefore be ascribed to this land transformation.

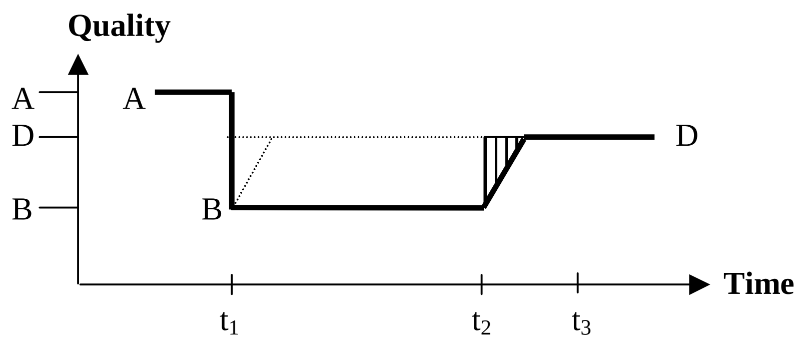

Figure 4. State of the flora, fauna, soil or soil surface before, during and after a human activity in the time interval $t_1 – t_2$, indicating how relaxation is postponed by the duration of the activity.

However, it should be noted that land occupation can only be seen as a postponement of relaxation in the occupied area (or a similar area) when the general trend is an increase in the relaxation of ecosystem area. When the general trend is an increase in ecosystem area under human use, a continued occupation implies that the area is not released for other human uses, which will therefore instead bring a similar area into human use elsewhere, i.e. implying a transformation there.

1.5 Choice of reference state for the measurement of occupation impacts

The occupation impact caused by a human activity is measured in relation to a reference state, i.e. it is expressed as the difference between the actual state and the reference state. The choice of reference state is not arbitrary, but relates to the distinction between occupation impacts and permanent ecosystem impacts as defined in section 1.4. The occupation impact should be defined in such a way as to avoid any overlap with the permanent ecosystem impact and give a full expression of the impacts not captured in the permanent ecosystem impact. This can only be done by using the final steady state (the *relaxation potential*, level D in figure 3 and 4) as the reference state, since the permanent ecosystem impact is defined as the difference between the original and the final steady state (levels A and D in figure 3 and 4). Note that the final steady state is not necessarily fixed as shown in figure 3 and 4, but may itself change as a result of the human activity in question, either at the start of such an activity (as shown in figure 5) or as a gradual degradation of the relaxation potential (as shown in figure 6).

Thus, the occupation impact is measured as the difference between the actual level (the fully drawn curve in the figures) and the level of the *current relaxation potential*, i.e. the final steady state if the land occupation was to end immediately after the current land occupation (level D' for the activity with level B in figure 5, level D for the activity shown by the rectangle with horizontal lines in figure 5, and level D'' for the activity shown by the rectangle in figure 6.

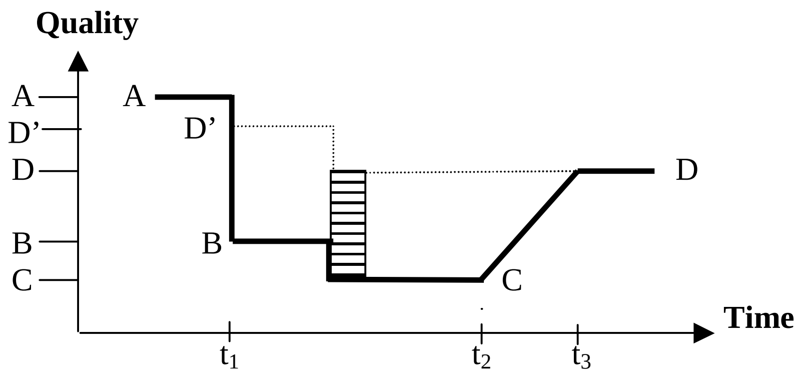

Figure 5. State of the flora, fauna, soil or soil surface before, during and after a human activity, with indication of two different reference states depending on the nature of the human activity.

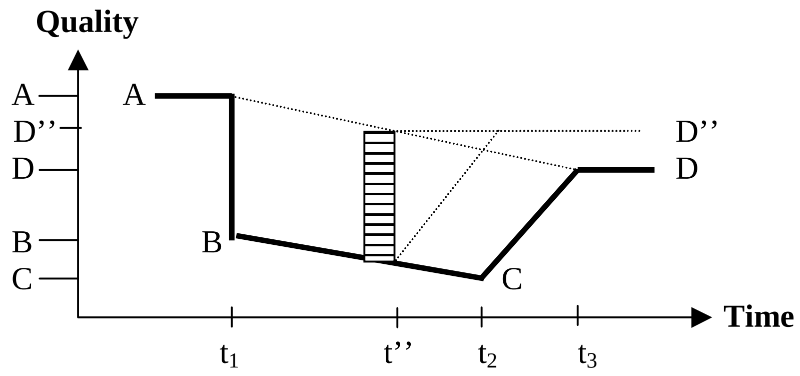

Figure 6. State of the flora, fauna, soil or soil surface before, during and after a human activity, with indication of gradual (and continuous) degradation of the relaxation potential (= reference state), and the reference state (D’’) if human activity was terminated at t’’ .

It has been suggested (Blonk & Lindeijer 1995) that the original state before any human intervention (level A in the figures) could be used as reference state. This would imply that all permanent ecosystem impacts since the initial human use would be allocated over the subsequent activities in relation to their duration, while disregarding the actual relaxation potential. While the original state is relevant as reference level for an initial impact on the potential relaxation level, it does not have any relation to the impacts of any subsequent activities, and its use would therefore lead to sub-optimisation of the current land use. Figure 7 illustrates the independence of the current impact from any initial state, when the permanent ecosystem impact is measured with reference to the relaxation potential at the end of the preceding activity (i.e. D), and when the occupation impact is measured in relation to the current relaxation potential (D’). In figure 7, the triangle with vertical lines illustrates the relaxation period from the preceding activity (i.e. *before* t_1, *not* the relaxation from the land transformation *at* t_1 shown in figure 3), which must be subtracted from the area (t_1D’, t_1B, t_2C, t_3D’) to arrive at the occupation impact of the activity taking place in the time t_1 to t_2.

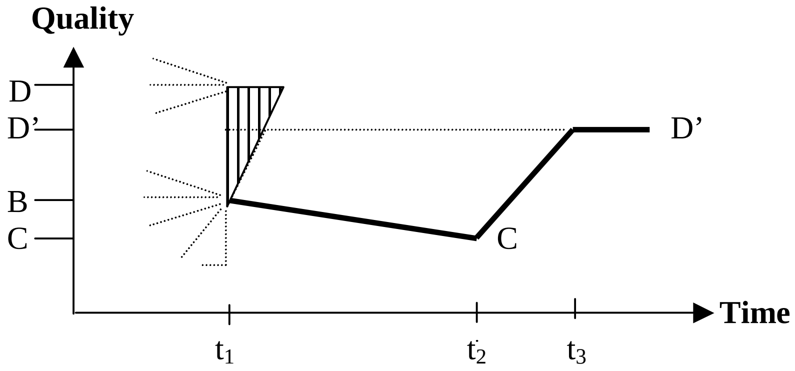

Figure 7. State of the flora, fauna, soil or soil surface before, during and after a human activity taking place in the time interval $t_1 – t_2$.

Likewise, it has been suggested (Baitz *et al.* 1998) that the state immediately before the studied activity could be used as reference state (*e.g.* state B in figure 5 and 8). This would imply that until all human activity on the area ends, the first human activity on the area (the one with level B in figure 5 and 8) would continue to be ascribed occupation impacts (the area between the two dotted lines in figure 8), as if it had not terminated, or these impacts would not be ascribed to any activity, while any subsequent activity would be ascribed occupation impacts relative to the preceding activity only (the area below the lowest dotted line in figure 8). This would also eliminate the distinction between permanent ecosystem impacts and occupation impacts. Occupation impacts would only be attributed to land use that involves transformation. Continuation of land use as it is would not be ascribed any impact. This implies ignoring the impacts due to this land occupation and prevention of relaxation. This is clearly not reflecting any causal relationship between the human activities and their impacts, and thus cannot be recommended.

For the same reasons, the state immediately after the studied activity (i.e. before relaxation) would not be relevant as a reference state, nor would the average current state (as suggested by Köllner 2000a, and mainly determined by agriculture and other human activities). This latter proposal would imply a different interpretation of occupation, namely as postponing the return to average human activities.

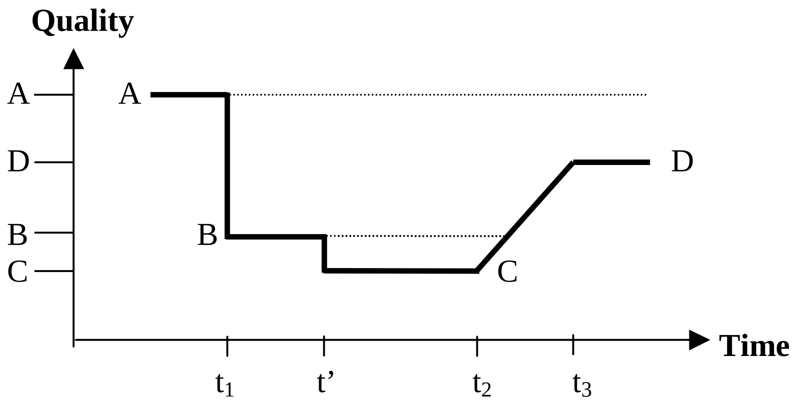

Figure 8. State of the flora, fauna, soil or soil surface before, during and after two human activities (in the periods t_1 to t' and t' to t_2, and its (mis)allocation over the two activities when the state immediately before the studied activity is used as reference state. Note that in this figure, the reference state is chosen as A for the activity t_1-t'. In all previous figures (3 to 7) the reference state = dotted line = current relaxation potential.

It can be concluded that:

- if the reference state is chosen at a level above the current relaxation potential, the impact measured would include a part of the permanent ecosystem impact, i.e. it would not be a representation of the occupation impact alone.
- if the reference state is chosen at a level below the current relaxation potential, the impact measured would exclude part of the occupation impact of the current land use.

1.6 Changes in the rate of relaxation

The rate of relaxation (the slope of the curve during relaxation) may be affected by the human activity before relaxation (as illustrated in figure 3, where the slope during relaxation is different for the activity that ends at t_1 and the activity that ends at t_2). A human activity that changes the rate of relaxation will be ascribed the consequences of this change through the occupation impact as described in section 1.4.

The rate of relaxation may also be affected positively by human intervention in the relaxation phase itself (*e.g.* by fertilisation or forced introduction of species). If such intervention can be foreseen, it may be included as a process in the analysed product system, and the altered rate of relaxation used when calculating the impacts.

2. Indicators for measuring nature value

In order to make the impact assessment operational, it is necessary to find one or more indicators that adequately reflect and quantify the "Quality" on the Y-axes of figures 3 to 8. It follows from these figures that the same indicators should be relevant for both permanent ecosystem impacts (measured in units of quality * area) and occupation impacts (measured in units of quality * area * time).

An adequate set of indicators must reflect the key aspects of the impact chains as described in section 1.3.3. Indicators may be defined at several levels of the impact chain (exchange, midpoint or endpoint).

Taking a starting point in the areas of protection in figure 2, the following types of indicators may be distinguished:

- Indicators for the biogeochemical *substance and energy cycles*, being part of the impact chains for the life-support functions.
- Indicators for the actual or potential *productivity* of the ecosystems, relating to the availability of biotic resources, the potential for agriculture, and most of the life-support functions.
- Indicators for the *biodiversity* of the ecosystems, relating directly to the endpoint "Biodiversity" under "Resources", and also being indicators for species composition as a mid-point to other areas of protection.
- Indicators for the *cultural value* of the affected sites, in terms of uniqueness of landscapes and archaeological remains.
- Indicators for *migration and dispersal*, as one of the midpoints towards altered species composition.

These five types of indicators are dealt with in more detail in the following sections.

2.1 Substance and energy cycles

Altered species composition and population volumes will affect the biogeochemical cycles of substances and energy in many ways.

The *carbon cycle* is of particular interest to the global temperature regulation, as the atmospheric concentration of CO_2 and CH_4 play a key role in this regulation. An increased atmospheric concentration of these so-called greenhouse gases will decrease the heat loss from the earth through infrared radiation, thus leading to global warming.

By photosynthesis, CO_2 is taken up by plants, later to be released through respiration and decomposition of organic matter. Although this implies a neutral long-term net effect on the atmospheric concentration of CO_2, the temporary storage in organic matter is of a size that cannot be ignored as sink and source of atmospheric CO_2 when assessing the impact of land use on the global temperature regulation. Net primary productivity (NPP, see section 2.2) may be used as an intermediate indicator for the change in fixation and release of CO_2 as a result of altered species composition and population volumes. When converted to CO_2, the impact on global warming can be calculated from the same models that are used to assess the impact of industrial sources of CO_2. However, the dynamics of the modelling and the time horizon of the assessment are of particular importance for CO_2 from area sources.

Decomposition of organic matter under waterlogged conditions in wetland soils result in release of CH_4. Physical changes that affect the size of the waterlogged area or the duration of the waterlogged condition will obviously have a direct impact on the CH_4 release. For the waterlogged soils, NPP (see section 2.2) may be used as an intermediate indicator for CH_4 release, since decomposition is here proportionally related to NPP. Also for this issue, the ordinary models for global warming can be used to assess the impact.

The fixation of *nitrogen* through soil-living organisms, especially those linked to leguminous plants, is an essential part of the natural supply of nitrogen for plant growth. However, the importance of natural fixation is currently limited, since the global supply of industrial nitrogen-sources is now equivalent in size to the natural fixation, which has had the consequence that it is rather the amount of unwanted nitrogenous substances that is of concern, causing eutrofication (nutrification) both in aquatic and terrestrial environments, and with serious implications for the biodiversity of ecosystems adapted to nitrogen-poor conditions. Thus, compared to other mechanisms, the importance of vegetation-changes on the nitrogen cycle are limited.

The concentration of aerosols and *dust* in the atmosphere is influenced by vegetation, since vegetation reduces wind speeds and surface susceptibility to wind erosion, and at the same time filters the air, thus removing dust from the atmosphere. For most plant micro-nutrients, redistribution through wind is the main natural process for transfer between ecosystems. In the absence of more detailed models and data, the influence of vegetation on atmospheric dust may be estimated by indicators related to vegetation cover, leaf area index, above-ground biomass, or primary productivity (see section 2.2). The same models that are used to assess the impact of industrial sources of dust can be used to assess the impact from a vegetation-induced change in the atmospheric dust concentration.

The *hydrological cycle* is influenced in many ways by the amount and structure of the vegetation as well as by surface characteristics. Vegetation plays a role for water interception, reduction of run-off, and modification of evaporation (increasing evaporation when water is freely available and decreasing evaporation when water is scarce). By changing the albedo and surface roughness, vegetation may also influence the distribution of precipitation. If albedo increases, evaporation decreases and precipitation markedly decreases. This effect may be self-amplifying, since reductions in vegetation typically increase the albedo and decrease surface roughness. The surface characteristics are important for direct evaporation as well as for the partitioning of precipitation between run-off and infiltration. Many empirically based models exist for parts of the hydrological cycle on local scales (and dependent on the locally available input data), but models based on proven, generalised relationships on regional and global level are still under development. For a survey of current research, see http://www.gewex.com/. It appears that globally applicable indicators are not yet available for the impact of vegetation and soil properties on the hydrological cycle. Until more universally applicable models become available, the influence of vegetation on water inception, reduction of run-off, and stabilization of evaporation, may be assumed linearly related to the amount of vegetation, *e.g.* expressed in terms of above-ground biomass or primary productivity, which may therefore be used as a preliminary rough indicator (see section 2.2).

The influence of the vegetation on the *energy balance* takes place both indirectly, via the hydrological balance just described, and directly through the albedo. As for the issues above, the influence of vegetation on the energy balance may - in the absence of more detailed models and data - be assumed linearly related to above-ground biomass or primary productivity (se section 2.2).

2.2 Ecosystem productivity

In terrestrial ecosystems, the availability of *biotic resources* is - on a very general level - related to their productivity potential. While it may be possible in specific situations to model the direct influence of altered species composition and population volumes on specific species that supply goods and services for human use, the typical level of information will only allow the use of general indicators for productivity. The productivity potential is an even more directly appropriate indicator for the *potential for agriculture*. And as already indicated in the preceding section, ecosystem productivity can be used as a more or less satisfying indicator for many biogeochemical substance and energy cycles that are part of the impact chains for many *life-support functions*. In addition to the mechanisms described in the preceding section, primary productivity is linked to further important mid- or end-points of life-support functions, namely:

- Decomposition, which plays an important role for nutrient availability and topsoil formation, is determined mainly by turnover of organic matter, as measured by net primary productivity.
- Dislocation of substances by and through organisms, closely related to the above, but also including the transport of nutrients between ecosystems by and in migrating species. Again, in absence of more specific indicators, the net primary productivity appears to be a reasonable proxy indicator.
- Topsoil formation and preservation. Along with weathering, organic matter plays a key role in topsoil formation, and vegetation cover plays a key role in topsoil preservation. Thus, net primary productivity is a reasonable indicator for both mechanisms.

In conclusion, change in *net primary productivity (NPP)*, which is defined as the net carbon uptake of the ecosystem (fixation through photosynthesis minus losses through respiration) over time, appears to be a reasonable mid-point indicator for the impact of altered species composition and population volumes on biotic resources, potential for agriculture, and life-support functions of natural systems. This may be substituted and/or amended by other indicators, such as above-ground biomass, when relevant data and models become available for the described processes.

As an indicator for impacts from land use on life-support, Blonk & Lindeijer (1995) and Lindeijer *et al.* (1998) have suggested *free net primary productivity (fNPP)*, i.e. NPP minus the amount of carbon sequestered for human use. However, they view this as an indicator for "nature development space", i.e. the amount of biomass nature can apply freely for its own development. In this sense, it may be a better indicator for impacts on biodiversity (see section 2.3) than for impacts on life-support, since most life-support functions listed in section 1.3 are related to the full NPP, i.e. the overall turnover of biomass, disregarding whether human use is a part of this turnover. The impact of ecosystems on the carbon cycle, nitrogen fixation, the concentration of atmospheric dust, the hydrological cycle, and the energy balance are all closer related to NPP than to fNPP, since it is the productivity and functioning of the ecosystem which is essential, not the route of utilisation of the products (by humans or within the ecosystem itself). A possible exception to this is decomposition and the formation of topsoil, which obviously depend on the organic matter left in the ecosystem. However, the human removal of organic matter and nutrients is separately modelled as a substance flow impact of the specific land use (see section 1.2) which may then subtract from the positive effects related to NPP.

The use of NPP as an indicator for "nature value" may appear counter-intuitive when confronted with the fact that man-made ecosystems may have a higher NPP than natural ecosystems on the same latitude, partly due to the higher proportion of young individuals with a high productivity, partly due to fertilisation, irrigation and other management practices. This implies that managed ecosystems can have a higher "nature value" than natural systems. This is, however, not surprising when considering that the "nature value" we seek to capture with the indicator NPP is mainly related to the gross influence of vegetation on climate and substance flows, which is clearly related to vegetation turnover, which is exactly promoted in managed ecosystems. Nevertheless, it is possible to criticize this choice of indicator for not adequately including the effects of harvesting. If a managed ecosystem is repeatedly harvested by removing the majority of above-ground biomass, the NPP may be kept high although important life-support functions may be endangered in the periods immediately after harvest and until an adequate plant cover is re-established. This is especially true for those impact chains that involve mid-points or mechanisms related more to vegetation cover or structure than to biomass turnover, such as the influence of vegetation on wind speed, interception of precipitation and dust, evapotranspiration, and albedo. When possible, it may therefore be appropriate to combine NPP with other indicators that better reflect vegetation cover and structure. Another option may be to make the NPP measure more dynamic and assign more importance to particularly low levels of NPP even when these low levels appear only at shorter intervals that are not reflected adequately in the annual average NPP.

2.3 Biodiversity

Biodiversity may be seen both as an indicator for the species composition of ecosystems, which is a mid-point for many impact chains (see figure 2), and as an endpoint in itself under the area of protection "Resources". Biodiversity is a term covering genetic diversity, species diversity, and ecosystem diversity. The three levels are interrelated in the sense that the preservation of diversity at the higher levels requires genetic diversity, and the maintenance of genetic diversity often requires in vivo conditions of species, which again requires the existence of the ecosystems. Since the area of protection “life-support functions” includes the key, irreversible aspects of ecosystem maintenance, it is reasonable to regard genetic diversity as the remaining irreversible aspect, i.e. the resource aspect, of biodiversity. In practice, nevertheless, data are most easily available for species diversity, and especially for vascular plant species. Thus, vascular plant diversity is an obvious proxy upon which to base any practical biodiversity indicator. Vascular plants also play a key role in ecosystem functions, and vascular plant diversity appears to be reasonably well correlated with terrestrial species diversity in general (Rosenzweig 1995, Barthlott et al. 1996). This proxy may either be further validated by specific investigations, or may later be modified to include also other species.

In the context of LCA, a biodiversity indicator based exclusively on vascular plant species richness[2] was developed by Lindeijer *et al.* (1998) and Köllner (2000a & b), the latter also applied in a modified draft form in the Eco-Indicator 99 (Goedkoop & Spriensma 1999). These indicators express loss of vascular plant species richness in relative terms, through dividing by a local reference state, in accordance with the 1992 Rio de Janeiro Convention on Biological Diversity, which discourages absolute scores for species diversity, since a reduction of say 5 species out of a total of 20 species in the Northern countries is considered worse than a reduction of 5 species out of a total of 200 species in the tropics. Köllner chose average actual or historical levels of species diversity as reference states, while Lindeijer *et al.* use the maximum actual species diversity. The latter is in better accordance with the rationale presented in section 1.5.

Also in the context of LCA, a biodiversity indicator based on the absolute number of rare species was proposed by Müller-Wenk (1998). The basis for this proposal is an assumption regarding the relationship between the number of rare species and the area of intensively used land, combined with data on rare species from Switzerland and Germany. The approach is noteworthy as an attempt to quantify the marginal impact of land use on biodiversity. Nevertheless, the current applicability on a global level is limited, due to lack of data.

In the early work from the LCAGAPS sub-project on land use (Schmidt 1997), we suggested to base the biodiversity assessment on the concept of relative scarcity on each ecosystem level (biome, biotope, habitat, species). In vivo species conservation is completely dependent on the conservation of the ecosystems. Thus, ecosystem scarcity is regarded as an essential part of a biodiversity indicator. To use factors for each ecosystem level is in line with the concepts of the 1992 Rio de Janeiro Convention on Biological Diversity. It was also suggested that the scarcity indicator could be supplemented by a vulnerability factor to take into account that two equally scarce ecosystems may not be equally vulnerable.

Also Cowell (1998) proposed a semi-quantitative indicator system for global biodiversity impacts of land use in the context of LCA, with four factors expressing relative ecosystem scarcity (through the relative area of such ecosystems), relative number of rare species (relative to a postulated maximum), relative species richness (relative to a maximum differentiated in relation to latitude), and relative number of individuals (expressed by the proxy indicator NPP relative to a maximum NPP per ecosystem type).

[2] We use “species richness” to signify “number of species per area”. In technical literature, both this term and the term “species density” are used in this meaning, but unfortunately both terms are also found to be applied with different meanings.

As part of the LCAGAPS sub-project on land use, the results of which is presented with the present report, we have developed the above concepts of scarcity and vulnerability further. As a basis, we use the most simple measure available for biodiversity:

- Species richness (SR) of the ecosystem.

This basic measure is then modified by two factors at ecosystem level:

- Inherent ecosystem scarcity (ES), expressed as the inverse of the potential area ($1/A_{pot}$) that could be occupied by the ecosystem if left undisturbed by human activities.
- Ecosystem vulnerability (EV), indicating the relative number of species affected by a change in the ecosystem area, as expressed by the species-area relationship.

When combining several factors expressing different aspects of biodiversity, these must be related to one another in order to arrive at a consistent indicator system that can be applied on various levels of detail once data is available. Therefore, each factor should have a conceptual link with the others. We link the three factors by multiplication, which forces us to relate each factor to one another, and to determine the weights of each factor. In a multiplication, the inherent weight of each factor is determined by the range of its possible values. To compare these inherent ranges, we therefore normalise each factor so that the lowest scoring ecosystem is unity, and analyse the resulting quality indicator for bias.

The development of this biodiversity indicator is described in more detail in the following sub-sections.

2.3.1 Species richness (SR)

The most simple measure for biodiversity is species richness (SR), i.e. number of species per area. Data on the largest groups (about 160'000 micro-organism species, 130'000 soil macrofauna species, 31'000 lower plant species and 15'000 nematode species) is largely lacking (Groombridge, 1992). Higher – vascular - plants (about 263'000), fishes (nearly 22'000 described) and vertebrates (43'000 species) offer the best chance. Of these, vascular plant species richness is proposed as reasonable indicator for species richness in general, since there are no other species group for which global data are available to a reasonable extent, and since vascular plant species are generally regarded as predictive for the diversity of other species (Rosenzweig 1995, Barthlott et al. 1996). Even when vascular plant species are accepted as a reasonable indicator for other species, a refinement could be to correct this proxy for actual values whenever available.

To normalise SR, so that the value of the lowest scoring ecosystem is unity, we must divide by the minimum vascular plant species richness (SR_{min}), arriving at:

$$nSR = SR/SR_{min} \qquad (1)$$

At the biome level, SR_{min} is set to 100 species per 10000 km^2 (found in deserts, tundra and on ice caps - see table 5). The setting of the minimum species richness determines the inherent weight of this factor, since this determines the range of its possible values (1 to >90, see table 1 in section 2.3.4).

It may be argued that the biodiversity indicator should rather reflect the number of indigenous species (as opposed to non-indigenous/neophytes, intentionally or accidentally introduced by man) and endemic species (species not found elsewhere) than the overall species number (Barthlott *et al.* 1999, Williams *et al.* 1994, Kier & Barthlott in press). With large numbers of non-indigenous species, the overall species richness may well be high, while indigenous and endemic species are still endangered, e.g. by invasive alien species. When improved data become available for indigenous and endemic species at a global level, these concerns may be included in the species richness (SR) factor.

Also from a conservationist viewpoint, those species would be crucial that are endangered or rare. The number of rare species would then be taken as an indicator for the ecosystem value. A potential refinement in line with this would be to determine the rareness of species (and possibly indirectly ecosystems) based on their mutual genetic distance (phylogenetic diversity). This would directly address the genetic part of the endpoint “biodiversity.”

Another possible refinement is to amend with diversity indicators determined by ecologists, such as key species (performing key functions in the ecosystem), or other indicators (see examples for Northern forest ecosystems in Hansson 2000). However, a requirement for a refinement would be that the resulting indicator should be unbiased across ecosystems.

As a central measure for biodiversity, the choice of species richness is in line with common sense understanding of biodiversity (“more species is better”), but may be seen as conflicting with the 1992 Rio de Janeiro Convention on Biological Diversity or national policies on biodiversity, because also species-poor ecosystems, such as heathlands and moors, are considered valuable and contributing to ecosystem diversity in themselves. The adjustment factors described in sections 2.3.2 and 2.3.3, aim to some extent to overcome this conflict.

2.3.2 Inherent ecosystem scarcity (ES)

The smaller the area in which an ecosystem is viable (due to its requirement for a particular and possibly rare geophysical condition), the more scarce it (and its species) can be said to be, even in a natural situation without human impacts. Thus, the same area should be assigned a higher value in an ecosystem with a small potential area than in an ecosystem with a larger potential area. This implies that the *Inherent Ecosystem Scarcity* (ES) can be expressed by a reverse relationship with the potential area of the ecosystem (A_{pot}), resulting in the following formula:

$$ES = 1 / A_{pot} \qquad (2)$$

Currently, data for A_{pot} are globally available at the biome level (see section 3.3). When data become available at lower ecosystem levels, the ES factor can be redefined to include ecosystem scarcity at these lower levels (following the suggestion of Schmidt 1997), to take into account that these factors may well be different between biotopes and between habitats. Prentice (pers. comm.) indicates that data are available for the development of such factors at least at the biotope level.

To normalise ES, so that the value of the lowest scoring ecosystem is unity, we must multiply by the area of largest potential ecosystem area ($A_{pot,max}$), arriving at:

$$nES = A_{pot,max} / A_{pot} \qquad (3)$$

At the biome level, $A_{pot,max}$ is boreal forests with the area $25*10^6$ km^2 (see table 5 in section 3.3). The setting of the largest and the smallest potential ecosystem area determine the inherent weight of nES, since this determines the range of its possible values (1 to 5.4, see table 1 in section 2.3.4).

2.3.3 Ecosystem vulnerability (EV)

The larger the current occupation of a potential ecosystem area, the more stressed and vulnerable the remaining unoccupied ecosystem area will be. Figure 9 illustrates two otherwise equally valuable ecosystems b1 and b2, of the same potential climax area. Of b1 only 10% is left and of b2 35% is left. In ecosystem b1, a continued occupation has a larger impact than in ecosystem b2, since the relative nature value would increase more with an incremental area increase in ecosystem b1.

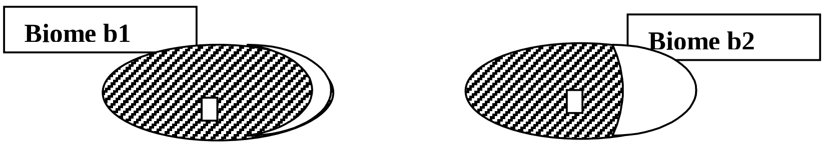

Figure 9. Illustration of the importance of the remaining unoccupied ecosystem area.

Using species richness as a measure of ecosystem value, this is illustrated by the theory of island biogeography (see Myers & Simon 1994, p.75 & references on p.217) saying that when only 10% of a habitat is left, a maximum of 50% of the original species richness can be supported. Depending on whether the still existing ecosystem loses part of its intactness, this maximum supporting capacity can easily go down to 30%. The basis for this rule is the species-area curve, determined by the formula $S = cA^z$, where S is the number of species, A is the area and c and z are fitting parameters (see figure 10). The parameter z determines the steepness of the curve, with values between z = 0.35 (causing a less steep slope thus earlier loss of species upon area reduction) for isolated areas such as islands and z = 0.15 in larger areas such as continents (Connor & McCoy 1979). For z = 0.15 the 50% reduction happens at 2% of the original habitat and for z = 0.35 it occurs at 14% of the original habitat (Reid & Miller 1989). This illustrates the importance of the remaining undisturbed area of the total potential ecosystem area, and the uncertainties relating to this.

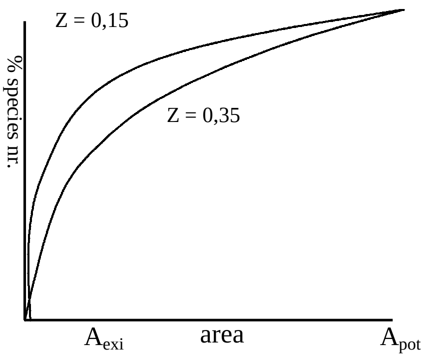

Figure 10. The species-area curve

Assuming that species numbers are a good proxy for biodiversity, ecosystem vulnerability should indicate the relative number of species affected by a change in the ecosystem area, as expressed by the species-area curve (see figure 10). When less is left of the ecosystem, the same area impact gives a larger change in species numbers. It is this tendency that makes the difference between ecosystem b1 and b2, expressed by the steepness of the slope, thus by the relative value of the derivative of the curve at A_{exi} and A_{pot}. The derivative is expressed by $S' = zcA^{z-1}$. This implies that the *Ecosystem Vulnerability* (EV) factor can be expressed by the derivative at A_{exi}. Normalising relative to the potentially protected or undisturbed ecosystem A_{pot} gives us:

$$nEV = (A_{exi}/A_{pot})^{z-1} \tag{4}$$

Its relative values for the two extreme[3] values for z, and factors for 10% resp. 50% of the potential ecosystem left ($nEV_{10\%}$ and $nEV_{50\%}$) are:

z = 0.15 ('mainland'): $nEV_{10\%/0.15} = 0.1^{-0.85} = 7.1$ and $nEV_{50\%/0.15} = 0.5^{-0.85} = 1.8$

z = 0.35 ('island'): $nEV_{10\%/0.35} = 0.1^{-0.65} = 4.5$ and $nEV_{50\%/0.35} = 0.5^{-0.65} = 1.6$

The size of the scores conveys what we would miss out if we disregarded this factor, i.e. if we assumed that z = 1 (S = cA) so that the species number would be assumed to decrease constantly upon reducing the area, no matter how much of the ecosystem is left. A higher value of z means that the extra impact per extra m^2 occupied is lower, because the first occupation causes relative more impact already (so the impact is divided more evenly over the whole curve). As we consider vascular plant species diversity in a wide range of situations, where the surrounding area can be expected to support some exchange of species[4], we suggest to apply the extreme 'mainland' value of z = 0.15.

A maximum score needs to be determined for nEV, because else the scale reaches to infinity as the area of ecosystem left goes towards zero (resulting in an infinite weight for nEV). As for the other factors, it seems unavoidable that the setting of such a maximum score will imply some degree of normative judgement. Here, we argue that percentages of actual to potential area below the present average protected area (6% of the global ecosystem according to Green & Paine 1997) will not be relevant. Thus, nEV_{max} will be $EV_{6\%/0.15} = 10$.

Currently, data for A_{exi} are globally available at the biome level (see section 3.3). When data become available at lower ecosystem levels, the EV factor can be redefined to include ecosystem vulnerability at these lower levels, to take into account that it may well be different between biotopes and between habitats.

[3] For birds on islands, the z-value may even increase to over 0,5. For so-called interprovincial species-area curves (for assessments on multiple intermediate ecosystem levels) z actually ranges between 0,6 to over 0,9 for tropical rainforest provinces (Rosenzweig 1995, p.276).
[4] Isolated ecosystems may be more vulnerable to changes. This aspect is covered by the ES factor, described in section 2.3.2.

2.3.4 Combining the biodiversity factors

Now, the three factors nSR, nES, and nEV are combined through multiplication to arrive at the biodiversity indicator adequately reflecting and quantifying the Quality (Q) on the Y-axes of figures 3 to 8. This forces us to relate each factor to one another, and to determine the weights (a, b, and c) of each factor:

$$Q_{biodiversity} = nSR^a * nES^b * nEV^c \tag{5}$$

If we set a = b = c = 1, the weight of each factor is determined by the inherent range in its values. The preliminary maximum ranges and main choices therein are summarized in table 1.

Table 1. Overview of factor ranges (see also table 5 in section 3.3)

Factor	Preliminary range	Main choices
Species Richness	1 – 90	Including all species or only indigenous, endemic or rare species – here all species. The expression for species richness: SR, the Fischer log form, or the Arrhenius exponent form – here the plain SR. The minimum value of SR: 100/10000km^2.
Ecosystem Scarcity	1 – 5.4	The level at which ecosystems are assessed: biome, biotope, habitat – here biome only. The ecosystem classification system used – here Biome 2.
Ecosystem Vulnerability	1 – 10	The maximum value of EV, determined by z and the % of ecosystem currently protected – here 0.15 and 6%, respectively.

A more detailed classification, for instance based on vegetation types (such as the Biome 3 model), may reduce the range in some of the factors considerably. Nevertheless, the area occupied will still dominate the whole impact assessment, as its range can in practice be a factor 10'000 for cases comparing renewable and non-renewable materials. This dominance of the area occupied is inherent to the focus on occupation. For permanent ecosystem change, area becomes less important and the scarcity aspect more important.

The total weight of all proposed factors is equal to their multiplied range. No arguments have been found to warrant the setting the weights a, b or c to any other value than 1, which means that we can simplify formula 5 to:

$$Q_{biodiversity} = nSR * nES * nEV \tag{6}$$

or, for the purposes of relating to the basic data:

$$Q_{biodiversity} = (SR / SR_{min}) * (A_{pot,max} / A_{pot}) * (A_{exi} / A_{pot})^{-0.85} \tag{7}$$

Note that $Q_{biodiversity}$, and especially the factor nEV, may change over time, which means that the scale of the Y-axis in figures 3 to 8 is not stable over longer time periods - not as a result of the currently studied land use, but due to changes in the human land use in general (A_{exi}), or even due to changes in natural conditions (A_{pot}). This implies that the figures 3 to 8 must be understood to illustrate marginal changes, i.e. changes that are small compared to the overall size of the ecosystems. In parallel, the data in section 3 are applicable only for marginal changes.

2.4 Cultural value

In contrast to the impacts and indicators discussed above, unique landscapes and unique archaeological sites under the area of protection “Resources” do not have a value outside the context of human culture. While unique landscapes may indeed be natural, they may as well be man-made, and certainly archaeological sites cannot be measured in terms of “nature value.” This implies that the concepts outlined in sections 1.4 to 1.6 and illustrated in figures 3 to 8 are not appropriate for these culturally dependent areas of protection.

The impacts on unique landscapes and unique archaeological sites are not related to occupation. Indeed, the conservation of landscapes and archaeological sites may actually require human occupation, since the relaxation into a natural state may lead to their disruption or disappearance. Nor are the impacts related to any relaxation level, since unique landscapes and unique archaeological sites per definition cannot be recovered.

The impacts on unique landscapes and unique archaeological sites are fundamentally related to transformation itself, whether this transformation is one from a natural state into human use, from one human use to another, or a relaxation from human use. Thus, for these impacts, the reference level (cf. the discussion in section 1.5) is indeed the state immediately before the studied activity. Any change from this previous state may imply an impact.

Exactly because of their uniqueness, the value of unique landscapes and unique archaeological sites cannot be determined in terms of a general indicator, but must be treated on a case-by-case basis. However, an indicator may be developed for disruption of unknown archaeological sites (i.e. possibly but not necessarily unique), since this can be related to increases in ploughing depth, introduction of deep-rooted plants, and other activities that disturb or remove soil layers that were previously undisturbed. Thus, an indicator may be based on the thickness of soil layer disturbed. This indicator should be multiplied by the area, and weighted by a factor determined by archaeologists and historians, expressing the probability of occurrence of archaeological remains in different area types (and soil depths). Predictive models for this purpose are in development (see *e.g.* Dalla Bona 1994). As for ecosystems, the possibility for a meaningful classification of area types depends also on the ability of the life cycle inventory to identify the location of specific activities within such classes (see section 4.1).

2.5 Migration and dispersal

The subdivision of habitats or creation of barriers by cutting corridors, typically by draining or by construction of larger roads or barrages in streams, will impact biotic migration and dispersal patterns.

In some species, migration is an essential part of the life cycle of individuals, as known especially from birds and larger sea-living animals. But also smaller movements may be of importance, such as animals in the mating season roaming in search of a partner. Dispersal refers to the broader issue of gene flow at population level. However, dispersal capabilities of single organisms are increasingly considered to be crucial for the survival probability of threatened species.

For both migration and dispersal, indicators related to landscape structure are of interest. One such indicator is connectivity, which is the degree to which a landscape facilitates or impedes movement of organisms among resource patches. However, measurement of connectivity is not uncontroversial (Tischendorf & Fahrig 2000a & b). Although the role of fragmentation increases in landscapes with low habitat coverage, it appears that the amount and quality of habitats in general play a larger role for landscape connectivity than landscape fragmentation. This appears to be true, even for the larger scale of wetland areas as stopover sites for migratory birds, where the quality of wetland sites (i.e. food availability) was found to be substantially more important than the spacing of sites along the migratory pathway (Farmer and Wiens in press). This suggests that the general indicators for productivity and biodiversity may to some extent cover the concern for connectivity.

We therefore refrain from development of additional indicators for migration and dispersal.

A possibly more serious problem is the reverse: namely the increase in dispersal of invasive species, alien to the local ecosystems. This may happen through creation of new corridors or dispersal vectors, or through intentional introductions. This may be classified as a physical change to flora and fauna, with direct impacts on altered species composition (arrow 1 in figure 2). As mentioned in section 2.3, this may be reflected in a refined biodiversity indicator, when models are developed that allow data on indigenous and endemic species to be distinguished from the overall species number.

3. Data and maps

For the practical application of the different indicators discussed in chapter 2, data availability is a crucial issue. We have focussed on presenting data for a few of the most central indicators, while for other indicators we limit ourselves to describe the availability of data in general terms, either because these indicators have been treated extensively elsewhere, or because a treatment here would be meaningless in the light of the rapid developments in the particular fields.

3.1 Data for substance and energy cycles

Carbon dioxide sequestration can be calculated directly from NPP values (which are either reported as carbon or easy to convert from biomass to carbon). For land transformations, the carbon in live vegetation is of interest. Data can be found in Olson *et al.* (1985), or see table 2.

According to the IPCC methodology, methane emission from rice fields is estimated at an average of $20 g/yr/m^2$. The ratio of methane emission to net primary productivity can be estimated to an average 3% on a carbon-to-carbon basis, based on field measurements in rice fields (Huang *et al.* 1996). See also Matthews (1993).

Atmospheric dust from unvegetated areas is mainly related to recently disturbed soils. Data can be found in Gillette (1978) and Tegen & Fung (1994).

Albedo for vegetated areas are typically 10-20%, while for unvegetated areas, the albedo is 20-45% depending on clay and organic matter content (see *e.g.* Matthews 1983).

3.2 Data for ecosystem productivity

Data for net primary productivity are given in table 2. Dobben *et al.* (1998) provide a world map of net primary productivity per physiotype. Table 3 provides a rough summary of this map relating NPP to latitude and altitude. Figure 11 displays a similar map, based entirely on satellite imagery by the Laboratory for Global Remote Sensing Studies (http://www.geog.umd.edu/glopem/), see also Prince & Goward (1995).

Table 2. Annual net primary productivity (NPP), and carbon content in plants and soil (0-1 metre) in broadly categorized terrestrial ecosystems (from Amthor et al. 1998)

Ecosystem	NPP gC/m2/year	Plant C g/m2	Soil C g/m2
Forest, tropical	925	16500	8300
Forest, temperate and plantation	670	12270	12000
Forest, Boreal	355	2445	15000
Woodland, temperate	700	8000	12000
Chaparral	360	3200	12000
Savanna, tropical	790	2930	11700
Grassland, temperate	350	720	23600
Tundra, arctic and alpine	105	630	12750
Desert and semidesert scrub	67	330	8000
Desert, extreme	11	35	2500
Lake and stream	200	10	-
Wetland	1180	4300	72000
Cultivated and permanent crop	425	200	7900
Human area	100	500	5000
AVERAGE	391	3220	13640

Table 3. Estimates of NPP (gC/m2/year) for dry physiotopes (table from Dobben et al. 1998, recalculated assuming 45% C in biomass).

Altitude: / Latitude:	0 - 1000	1000-3000	>3000
80	<=50	0	0
60	360	140	0
40	540	360	90
30	<=50	<=50	<=50
20	720	360	140
0	990	540	180

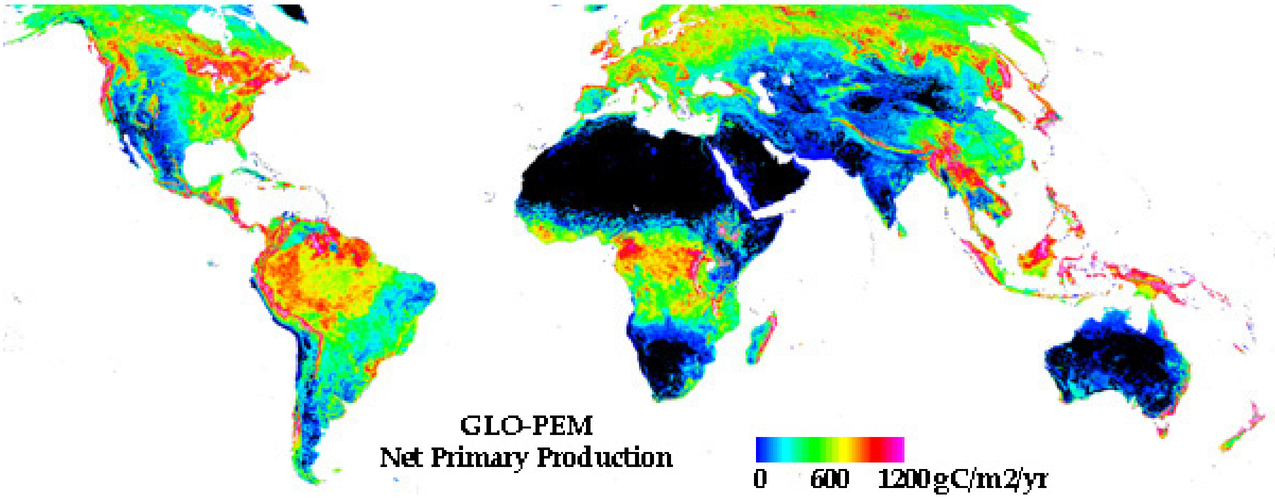

Figure 11. World map of Net Primary Productivity based entirely on satellite imagery. From the Laboratory for Global Remote Sensing Studies (http://www.geog.umd.edu/glopem/).

As a normalisation reference, to enable interpretation across different impact categories, the following tentative value can be applied: 9800 E12 g C / year or *1750 kg C / capita / year* (when applying the world population of $5.6*10^9$ persons for 1994, which has been used as reference year).

This has been calculated in the following way, see also table 4:

1. Human settlements constitute 2 E12 m^2 with an NPP of 100 g C/m^2/year (Amthor *et al.* 1998), and are mainly situated in the more productive natural ecosystems for which an average NPP of 700 gC/m^2/year is applied.
2. Arable and permanent crops constitute 15 E12 m^2 (FAOSTAT) with an NPP of 425 gC/m^2/year (Amthor *et al.* 1998), and are mainly situated in the more productive natural ecosystems for which an average NPP of 700 g C/m^2/year is applied. The uncertainty on both these NPP figures are important. Depending on agricultural intensity, the actual NPP may range from 100 to 700 g C/m^2/year, and the NPP of the natural ecosystems occupied may vary from 350 to 950 g C/m^2/year.

3/4. Permanent pasture occupy both high productivity ecosystems, such as tropical and temperate forests and savanna (16 E12 m^2 with an average NPP 700 g C/m^2/year), and low productivity ecosystems such as natural grasslands and boreal forests (19 E12 m^2 with an average NPP of 350 g C/m^2/year). In both situations we have assumed an average reduction in NPP of 15%, mainly based on Milchunas *et al.* (1994). The areas occupied are estimated from the difference between potential and existing natural areas according to table 5 (taking into account the above areas occupied by human settlement and arable and permanent crops), and the sum has been validated against the total area of permanent pasture from FAOSTAT.

5. Human induced desertification has been included with a 25% reduction of the potential NPP of 67 of semi-desert scrub in an area of 15 E12 m^2 (Vitusek et al. 1986). It is questionable to what extent desertification should be included as human induced, but the NPP value is in any case not significant compared to the other sources of impact (and uncertainty) on global ecosystem productivity. For the same reason, we have not sought a more accurate and updated estimate for this impact.

6. The above occupation impacts all relate to the period of the current human activities in themselves, seen as the annual postponement of potential relaxation. In addition, there are occupation impacts during relaxation of the areas that are transformed. The annual transformation of e.g. tropical forests into agricultural areas or human settlement has an occupation impact during its potential relaxation, which can be calculated as the average reduction in NPP multiplied with the relaxation time and the area transformed. With respect to its importance for ecosystem productivity relaxation requirement, the dominant transformation is deforestation. Gross deforestation currently amounts to an average 0.135 E12 m^2 /year (FAO 2001). Applying an average relaxation period of 90 years (see section 4.5) and an average depression in NPP of 137.5 g C/m^2/year during this period, we obtain an occupation impact of 1670 E12 g C from relaxation from the current annual deforestation.

Table 4. Calculation of a global normalisation reference for occupation impact on ecosystem productivity. See the text for explanations.

Land use	NPP of relaxation potential	NPP during human activity	Reduction in NPP	Area affected	Annual occupation impact on ecosystem productivity
	g C/m^2/year	g C/m^2/year	g C/m^2/year	m^2	g C / year
1. Human settlement	700	100	600	2 E12	1200 E12
2. Arable and permanent crops	700	425	275	15 E12	4125 E12
3. Pasture in high productivity areas	700	600	100	16 E12	1600 E12
4. Pasture in low productivity areas	350	300	50	19 E12	950 E12
5. Human induced desertification	67	50	17	15 E12	255 E12
6. Relaxation from deforestation	700	425	275/2	0.135 E12	1670 E12
Global total					**9800 E12**

3.3 Data for biodiversity indicators

There are several classification systems possible for ecosystems on the biome level. For the potential biome area (the dominating of the factors), a simple policy-oriented system has been chosen: the Biome 2 model based on Prentice *et al.* (1992), which has been used to model impacts of global warming for policy goals. There are other models, such as the Box model (Box 1995) or the Simple Biosphere model (Sellers *et al.* 1996), for which area data was not available to us.

For the existing area left of each potential biome area, the classification scheme of the USGS[5] (the Biosphere Atmosphere Transfer Scheme) - fitting Biome 2 and the species richness data best - was selected, and compared with that of the Biome 2 model. Both the potential and existing biome areas, according to the selected data sources, are given in Annex 1. Especially

[5] The USGS has a website (http://edcwww.cr.usgs.gov/eros-home.html) which contains a wealth of global GIS data on present land use/land cover. The data has been (or rather is being) interpreted in terms of different ecosystem (biome) classification systems. See annex 2 for a legenda on 5 of them. The classification of the Biome I model has been added to annex 2. We thank Christian Bauer from Aachen University of Technology, Germany for initial data treatment of the USGS data, as large mainframes are required to manipulate the extensive GIS data files.

the difference in agriculture land cover between BATS and Biome 2 is disturbing. The data from BATS seem erroneous, as FAO gives a global agriculture area of $49*10^6$ km^2 for 1998 and of $46*10^6$ km^2 for 1970.

Thus, in table 5, we have used the data from the Biome 2 model to calculate $Q_{biodiversity}$. Rough values for the species richness (SR) per biome are taken from Barthlott *et al.* (1996), a first version of the map in figure 12.

These initially collected data may be somewhat erroneous. The two sources for existing land use (BATS and Biome 2) including different % of contributions from each type illustrates the uncertainty range of these data. Clustering of biomes may also be under discussion. This clustering may determine the values for $A_{pot,max}$ and SR_{min}. We for instance chose boreal forest for $A_{pot,max}$ because the clustering of all grasslands is very different for different classifications and may include very different climate zones, whereas boreal forest seems a more coherent biome class.

Thus, uncertainty may be considered high and may be above 50%. However, once a consistent classification scheme is chosen, mainly model uncertainties remain. Depending on the credibility of the model, this may be considered a larger or a minor problem. The data uncertainty on the global level may then be reduced to below 50%. For the present report, these data do give an impression of what the range in the various factors may be (see the overview in table 1).

Data collection on a more detailed level has not yet been attempted. An attempt to use Biome 3 may be worthwhile. However, whether all required data would still be available in a consistent manner on this level of resolution is unclear. The ecosystem data will be in terms of vegetation types, and biodiversity data should be available on that level too. Also, more regional scarcity prioritisation schemes may be valid, but in a global approach such as LCA, may lead to serious credibility problems and subsequent data use limitations.

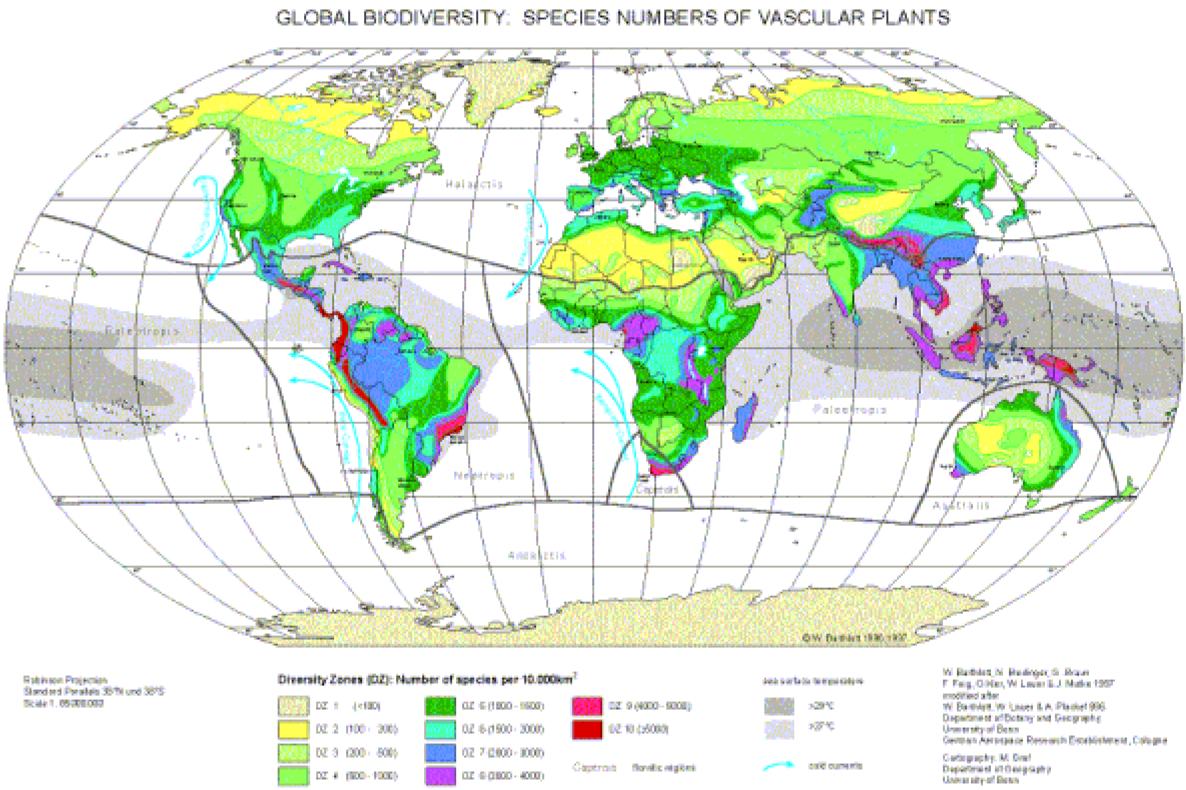

Figure 12. Richness of vascular plant species (reproduced from Barthlott et al. 1996; http://www.botanik.uni-bonn.de/system/phytodiv.htm)

Table 5. Calculating $Q_{biodiversity}$ for remaining area of natural biomes (data from annex 1)

Potential biome	**Species richness** *SR* [no./10^9 m^2]	**Potential area** A_{pot} [10^{12} m^2]	**Land cover area** A_{exi} [10^{12} m^2]	**Ecosystem vulnerability** $nEV = (A_{exi} / A_{pot})^{-0.85}$	$Q_{pot, biodiversity}$ $(SR / SR_{min}) * (A_{pot,max} / A_{pot}) * nEV$
Grassland/Steppe	200-1500	17.2	9.0	1.7	5-38
Savanna	200-3000	11.5	5.4	1.9	8-130
Boreal forest	200-1000	**25.3**	20.7	1.2	212
Cool conifer forest	500-1000	4.7	2.4	1.8	48-97
Temp. deciduous forest	1000-1500	6.2	1.0	4.7	190-290
Tropical forest	1500-9000	5.7	3.8	1.4	93-560
Hot desert	**100**-200	14.2	11.6	1.2	2-4
Tundra	**100**-500	12.1	13.3	~ 1	2-10
Ice	<**100**-200	6.7	6.7	1.0	1-2
Scrubland	500-4000	8.1	2.2	3.0	47-370
Temp. mixed forest	500-3000	7.7	1.7	3.6	59-350
Warm mixed forest		4.8	0.9	4.1	110-650
Mixed forest total		12.5	2.6	3.8	38-230
Tropical woodland	1000-3000	6.8	4.3	1.5	56-170

As a normalisation reference, to enable interpretation across different impact categories, the following tentative value can be applied: 21 E15 $Q_{biodiversity}$-weighted m^2*years/year or 3.8 * 10^6 *$Q_{biodiversity}$-weighted m^2*years/capita/year* (when applying the world population of $5.6*10^9$ persons for 1994, which has been used as reference year).

This has been calculated in the following way, see also table 6:

1-2. Human settlements as well as arable and permanent crops are mainly situated in the more productive natural ecosystems for which the range of $Q_{biodiversity}$-values for tropical forest from table 5 has been applied. This range includes the values for temperate forests. The area is 17 E12 m^2 (FAOSTAT) and it is assumed that practically all natural forest species are negatively affected.

3-5. Permanent pasture occupy both high productivity ecosystems, such as tropical and temperate forests, with $Q_{biodiversity}$-values between 93 and 560 (see table 5), and low productivity ecosystems such as natural grasslands and boreal forests (with $Q_{biodiversity}$-values between 2 and 38. In both situations we have assumed that 1/3 of the natural species are negatively affected. Due to the low $Q_{biodiversity}$-value for the latter ecosystems, they do not contribute significantly to the overall normalisation value. The same is true for desertification.

6. For the biodiversity indicator, the occupation impacts during relaxation of transformed areas are very important due to the long relaxation periods that must be assumed (see section 5.4). In these calculations, the gross deforested area of natural forests of 0.15 E12 m^2 /year (FAO 2001) is multiplied by an average relaxation time of 540 years and an average depression of $Q_{biodiversity}$ of 165 during this period. The resulting occupation impact from relaxation of this current annual deforestation is 13000 $Q_{biodiversity}$-weighted m^2*years or nearly 2/3 of all occupation impacts on biodiversity.

It is important to note that this normalisation reference relates to marginal changes, i.e. changes that are small compared to the overall size of the ecosystems. If the normalisation reference given here was used in estimations of the global impact of all human activities on biodiversity, this would imply a double-counting, since this global pressure is already factored into the $Q_{biodiversity}$-factor once, through the factor $nEV = (A_{exi} / A_{pot})^{-0.85}$. Reflecting the current pressure on ecosystems, this factor it is between 1 and 5 (see table 5), while in a situation without any human activities it would be unity for all ecosystems and the global value of $Q_{biodiversity}$ would be equally lower.

Table 6. Calculation of a global normalisation reference for occupation impact on biodiversity. See the text for explanations.

Land use	$Q_{biodiversity}$ of relaxation potential	$Q_{biodiversity}$ during human activity	Average reduction in $Q_{biodiversity}$	Area affected [m^2]	Annual occupation impact on biodiversity [$Q_{biodiversity}$-weighted m^2*years]
1. Human settlement	93-560	0	330	2 E12	660 E12
2. Arable and permanent crops	93-560	0	330	15 E12	5000 E12
3. Pasture in high productivity areas	93-560	62-373	110	16 E12	1800 E12
4. Pasture in low productivity areas	2-38	1-25	7	19 E12	130 E12
5. Human induced desertification	2-4	1-3	1	15 E12	15 E12
6. Relaxation from deforestation	93-560	0	330/2	0.15 E12	13000 E12
Global total					**~21000 E12**

4. Implications for inventory

In the context of LCA, the concepts, indicators, and data described in the previous chapters would not be meaningful if they could not be related to the product life cycle inventory data, i.e. the available data on land use for specific human activities. Therefore, this chapter looks at the possibilities to record:

- The location of a specific human activity
- The size of the area affected
- The current relaxation potential for the affected area
- Relaxation periods

4.1 Typical locations of specific human activities

In LCA, the human activities that are involved in the production of a specific product can often be determined only in terms of their general nature, such as “tin extraction” or “soy bean production.” In some instances, the geographical location can be determined, at least at the level of country. For the indicators described in chapter 2 and 3, the location needs to be described in terms of ecosystem characteristics.

A default solution to this problem would be simply to apply the average country-wise location of each activity type, *e.g.* all tin or soy bean producing countries would be assumed to be affected in proportion to their share of the global production, and within each country, all ecosystems would be assumed to be affected in proportion to their share in the total area.

An example of this approach for resource extraction can be found in Lindeijer *et al.* (1998), see table 6.

Table 6. NPP of the average affected area for resource extractions (from Lindeijer et al. 1998, recalculated assuming 45% C in biomass).

Resource extraction	Average NPP-value $gC/m^2/year$
Energy	
Coal mining	360
Oil extraction	90
Gas extraction	315
Wood	
Forestry of hard wood	990
Forestry of pine	360
Metals	
Bauxite mining	450
Cadmium ore mining	450
Chromium ore mining	400
Cobalt ore mining	450
Copper ore mining	230
Iron ore mining	500
Ilmenite mining	450
Lead ore mining	360
Lithium ore mining	270
Manganese ore mining	500
Mercury mining	270
Nickel ore mining	360
Platinum group ore mining	320
Silver ore mining	590
Tin ore mining	630
Uranium ore mining	360
Zinc ore mining	400

This approach will obviously even out the data, compared to a more specific identification of sites.

A preferable approach would be to identify typical locations of human activities in terms of ecosystem characteristics. For example, agriculture will typically take place in relatively flat areas with fertile soils, and each crop type has certain demands to soil and climate conditions, which point to a certain delimitation of possible locations, even within a country. The same is true for forestry. The location of mines can be predicted fairly precisely from geological information (besides the fact that the location of mining sites is relatively stable and well-known). Thus, a relatively precise identification of the correct ecosystem can be obtained by super-imposing the potential locations on maps of the different ecosystem indicators.

4.2 Size of area affected

For agriculture and forestry, the area affected for a specific production can be readily calculated from data on yields per hectare (FAOSTAT http://apps.fao.org/). For mining and other activities, data are available in several LCA databases, *e.g.* the ETH data (Frischknecht *et al.* 1996) and the IVAM database (http://www.ivambv.uva.nl/uk/producten/product5.htm).

4.3 Current relaxation potentials

For biodiversity, the extinction of species at the global level will imply a reduction in the biodiversity relaxation potential, in proportion to the reduction in global species density for that ecosystem caused by the land use. As the vulnerability of the particular ecosystem has already been factored into the $Q_{biodiversity}$ in section 2.3, it is reasonable to assume for all ecosystems a fixed percentage of $Q_{biodiversity}$ being affected by a similar size of area occupied. To calculate this percentage, the method suggested by Müller-Wenk (1998) may be appropriate. For German conditions, Müller-Wenk calculates the marginal effect of land use on biodiversity to be 0.00005% per incremental km^2*year occupied, or $5 * 10^{-13}$ per m^2*year. To fit the formula of $Q_{biodiversity}$ from section 2.3, the percentage should be established for the least valuable/threatened ecosystem, but in the absence of such data the percentage given for German conditions could be used as a “better-than-nothing” value. This would imply, for example, that an occupation of 70 km^2 of an area with a $Q_{biodiversity}$ of 160 (typical of rainforest) for a period of 180 years, would be attributed a permanent reduction in $Q_{biodiversity}$ of $0.0000005/km^2/year * 70km^2 * 160 * 180years = 1$.

For net primary production, the current relaxation potential can generally be assumed to be the climax ecosystem, as described by the indicators and data in chapter 2 and 3. The exception is instances where prior or current land use has altered the physical conditions to an extent that a full recovery to the climax ecosystem is prevented. This may be caused by nutrient depletion or enrichment, soil compaction, pollutants, or dominance by invasive species. We are not aware of any studies that have determined the relaxation potential in such human-influenced ecosystems, and must therefore recommend a case-by-case approach.

4.4 Current indicator level for different human activities

Land uses will differ in their degree of influence on the indicators described in chapter 2 and 3. The complete removal of all vegetation and sealing of the surface (*e.g.* for buildings and roads) is one extreme, to which open mining sites also belong. The other extreme is the hunting of one particular species, which may or may not affect overall ecosystem functions, depending on the constraints on the particular hunting and the role of that species within the ecosystem. In between these extremes we find different silvi- and agricultural land uses as well as residential areas and recreative and military use of land, which may in some instances increase the “nature quality” of an area, in other cases reduce it, but generally not to zero.

For the indicators related to substance cycles and ecosystem productivity, the identification of the indicator level of the current land use is straightforward. Heavy grazing has small negative effect on NPP (Milchunas *et al.* 1994), which may be estimated to 15% of the NPP of the corresponding natural grasslands. For arable lands, especially in developing countries, the NPP may be as low as 10-20% of the NPP of the native vegetation, and only in the most developed countries can NPP of arable land exceed that of the native vegetation (Esser, 1994). The net primary productivity of crops may be estimated from crop yields by applying the crop-specific coefficients of dry matter content, harvest index, root production, and carbon and nitrogen content provided by Goudriaan (1997). For silviculture, management practices may influence net primary productivity, typically in an upward direction, but in general it can be assumed that there is no difference relative to the NPP of natural forests. For residential areas, a default value of 100 g C per m^2 per year can be used, while for recreative and military use of land, the estimates may be based on a qualitative assessment of the degree of deviation from the current relaxation potential.

For the biodiversity indicator, it is less straightforward to identify the indicator level for current land use. While the overall species richness may increase, be stable, or decrease under human land use, most land uses will favour some species at the expense of others. The use of species richness as such would therefore not correctly reflect the threat to biodiversity. Rather, the current indicator level should reflect the relative number of species not negatively affected by the current land use. Approximately 1/3 of naturally occurring rangeland species may be negatively affected by increased grazing (Landsberg *et al.* 1997). Most arable farming will imply a complete replacement of the naturally occurring ecosystem, and will therefore receive a zero score, equivalent to mining activities and residential use[6]. For silvicultural and recreative land uses, the relative number of species not negatively affected will depend heavily on management practices, and estimates need therefore to be based on an assessment of the specific management practice (see *e.g.* Hansen *et al.* 1991). For biodiversity indicators of managed forests, see also Bachmann *et al.* (1996) and Hansson (2000).

4.5 Relaxation periods

Tentative data for the relaxation period required to reach maximum potential biomass after a complete system removal, and under the assumption of no degradation in the relaxation potential, is given by Dobben *et al.* (1998), see table 7. The relaxation period to reach maximum potential biodiversity will be much longer. A tentative value can be obtained by multiplying the values in table 7 with a factor 6.

[6] This does not imply a neglect of agricultural or horticultural diversity, but rather that the maintenance of this aspect of biodiversity needs to be captured by a separate indicator, additional to the indicator for biodiversity of natural systems described in this paper.

Table 7. Relaxation periods (in years) to reach potential biomass in physiotopes by altitude and latitude (from Dobben et al. 1998).

Altitude: / Latitude:	0 - 1000	1000-3000	>3000
80	150	200	220
60	90	110	120
40	70	90	100
30	150	175	185
20	60	70	90
0	50	70	100

5. Examples

As examples of the application of the methodology outlined in chapters 1 to 4, we have chosen a few typical products: soybean, rapeseed, sand, beef and road transport. The examples deal exclusively with the indicators for ecosystem productivity and biodiversity. Furthermore, we have not included possible changes in the areas surrounding the investigated human activities.

Soybean production for European market

The current trend in soybean production is an increase in cultivated area, especially in Brazil, where new land is taken into cultivation. Thus, the marginal soybean production implies a land transformation from the natural biome (tropical forest) in Brazil (latitude around 20°) to agriculture. For 1000 kg soybeans, the *occupation impact* can be calculated as follows (see also table 8):

1. The area (A) transformed and occupied during the activity is 4250 m^2 calculated from a yield of 2350 kg/ha/yr according to FAOSTAT.
2. The duration of the activity (t_{act}) is 1 year, and the relaxation time (t_{relax}) for latitude 20° is 60 years for ecosystem productivity and 360 years for biodiversity (see section 4.5).
3. $Q_{pot,\ productivity}$ in the relaxation potential (assuming no permanent ecosystem impact) is 720 g C/m^2/year (from table 3). $Q_{pot,\ biodiversity}$ in the relaxation potential is 155, calculated from formula 7: $(SR / SR_{min}) * (A_{pot,max} / A_{pot}) * (A_{exi} / A_{pot})^{-0.85}$, with an intermediate value for SR = 2500 species/10000 km^2; SR_{min} = 100 species/10000 km^2 and data for tropical forest from table 5: $A_{pot,max} = 25.3 * 10^6$; $A_{pot} = 5.7 * 10^6$; $A_{exi} = 3.8 * 10^6$.
4. $Q_{act,\ productivity}$ during soybean production is 420 g C/m^2/year (assuming a grain to NPP ratio of 0.25 and 45% C in biomass). $Q_{act,\ biodiversity}$ in soybean production is zero, as there are none of the original species left on the area.

5/6. During the production, the above quality is maintained constant, while during relaxation the quality increases gradually from the actual quality level of the production to the level of the relaxation potential. Thus, during production, the reduction in quality is simply the difference between the potential and the actual levels, while during relaxation this difference is gradually diminished. Assuming an even curve between the two levels (whether straight or sigmoid), the quality will on average be half of the difference between the potential level and the level during production. This is reflected in the slope factor s = 2 during relaxation.

7. The occupation impact can then be calculated from the formula:

$$I_{occc} = A * t_i * (Q_{pot} - Q_{act})/s_i \qquad (8)$$

where the index i signifies either “activity” for the period during soybean growing or “relaxation” for the relaxation period, and finally the separately calculated values for these two periods can be added, resulting in the values $39.5 * 10^3$ kg C for ecosystem productivity and $119 * 10^6$ $Q_{biodiversity}$-weighted m^2*years for biodiversity.

These values can be normalised and expressed in person-equivalents by dividing by the normalisation references from section 3.2 and 3.3: 1750 kg C / capita / year and 3.8 * 10^6 $Q_{biodiversity}$-weighted m^2*years/capita/year, respectively, resulting in the following normalised values per 1000 kg soy-beans: 22.6 person-equivalents for ecosystem productivity and 52.4 person-equivalents for biodiversity. The fairly large normalised values reflect that soybean production involves land transformation that constitutes an important part of the global impact.

Table 8. Calculation of the occupation impact from growing 1000 kg soy beans

		Impact on ecosystem productivity during activity	Impact on ecosystem productivity during relaxation	Impact on biodiversity during activity	Impact on biodiversity during relaxation
1. Area	A	4250 m^2	4250 m^2	4250 m^2	4250 m^2
2. Time	t_{act} and t_{relax}	1 year	60 years	1 year	360 years
3. Potential quality	Q_{pot}	720 g C/m^2/year	720 g C/m^2/year	155	155
4. Actual quality	Q_{act}	420 g C/m^2/year	420 g C/m^2/year	0	0
5. Slope factor	*s*	1	2	1	2
6. Quality reduction	$Q_{red} = (Q_{pot} - Q_{act})/s$	300 g C/m^2/year	150 g C/m^2/year	155	77.5
7. Occupation impact	$I_{occ} = A * t_i * Q_{red}$	39.5 * 10^3 kg C		119 * 10^6 $Q_{biodiversity}$-weighted m^2*years	

The *permanent ecosystem impact* on biodiversity can be calculated by multiplying the 119 * 10^6 $Q_{biodiversity}$-weighted m^2*years with the factor 5 * 10^{-13}/m^2/year from section 4.3, resulting in a $Q_{biodiversity}$-value of 6 * 10^{-5}.

Rape seed production for European market

Rape seed is currently a crop grown on set aside areas in Europe, where the trend is that agricultural land is taken out of cultivation. Rape seed production can therefore be seen as a simple postponement of relaxation in the natural biome temperate deciduous forest. For 1000 kg rapeseed, the impact can be calculated as follows (see also table 9):

1. The area (A) occupied during the activity is 4000 m^2 calculated from a yield of 2500 kg/ha/yr according to FAOSTAT.
2. The duration of the activity (t_{act}) is 1 year. As there is no transformation, there is no relaxation time allocated to this activity.
3. $Q_{pot,\ productivity}$ in the relaxation potential at latitude 40° is 540 g C/m^2/year (from table 3). $Q_{pot,\ biodiversity}$ in the relaxation potential is 192, calculated from formula 7, with SR = 1000 species/10000 km^2 and data for temperate deciduous forest from table 5: A_{pot} = 6.2 * 10^6; A_{exi} = 1.0 * 10^6.

4. $Q_{act,\ productivity}$ during rapeseed production is 450 g C/m^2/year (assuming a grain to NPP ratio of 0.25 and 45% C in biomass). $Q_{act,\ biodiversity}$ in rapeseed production is zero, as there are none of the original species left on the area.
5. During the production, the above quality is maintained constant, as reflected in the slope factor s = 1.

6/7. The occupation impact can then be calculated from formula (8): $I_{occ} = A * t * (Q_{pot} - Q_{act})/s$, resulting in the values 360 kg C for ecosystem productivity and 0.77 * 10^6 $Q_{biodiversity}$-weighted m^2*years for biodiversity.

These values can be normalised and expressed in person-equivalents by dividing by the normalisation references from section 3.2 and 3.3: 1750 kg C / capita / year and 3.8 * 10^6 $Q_{biodiversity}$-weighted m^2*years/capita/year, respectively, resulting in the following normalised values per 1000 kg rape-seed: 0.2 person-equivalents for ecosystem productivity and 0.2 person-equivalents for biodiversity. The fairly large normalised values reflect the importance of agricultural production in the global impacts from land use.

Table 9. Calculation of the occupation impact from growing 1000 kg rapeseed

		Impact on ecosystem productivity during activity	Impact on biodiversity during activity
1. Area	A	4000 m^2	4000 m^2
2. Time	t_{act} and t_{relax}	1 year	1 year
3. Potential quality	Q_{pot}	540 g C/m^2/year	192
4. Actual quality	Q_{act}	450 g C/m^2/year	0
5. Slope factor	s	1	1
6. Quality reduction	$Q_{red} = (Q_{pot} - Q_{act})/s$	90 g C/m^2/year	192
7. Occupation impact	$I_{occ} = A * t_i * Q_{red}$	360 kg C	0.77 * 10^6 $Q_{biodiversity}$-weighted m^2*years

Rape is not a particularly deep-rooted plant and its cultivation does not imply an increase in ploughing depth. Thus, cultural values are not threatened.

The soybean and rapeseed examples demonstrate the important difference between transformation and occupation. Also, it is interesting to note the high weight assigned to biodiversity in temperate deciduous forest ($Q_{pot,\ biodiversity}$), due to how little is left of the potential area, compared to tropical forest (as expressed in the factor nEV = $(A_{exi} / A_{pot})^{-0.85}$). This is enough to outweigh the larger number of species in tropical forest (here assumed a factor 2.5, but even if assuming a factor 5, the two ecosystems would receive $Q_{pot,\ biodiversity}$-scores in the same range).

Sand extraction for European market
Sand extraction in Europe is located in areas formerly used for agriculture. In some instances, the site is not restored after the termination of the extraction process but instead left as *e.g.* a lake, implying a permanent ecosystem change. In other instances, the ground is restored after the extraction and the extraction can therefore be seen as a simple postponement of relaxation in the natural biome temperate deciduous forest. In the latter case, the occupation impact per 1000 kg sand can be calculated as:

- Area*time occupied: 0.18 m^2*years (from Lindeijer *et al.* 1998).
- $Q_{pot, productivity}$ in the relaxation potential: 540 g C/m^2/year (as for rape, see this).
- $Q_{pot, biodiversity}$ in the relaxation potential: 192 (as for rape, see this).
- $Q_{act, productivity}$ during sand extraction: zero.
- $Q_{act, biodiversity}$ during sand extraction: zero
- Occupation impact: 540 gC/m2/year * 0.18 m^2 = 0.097 kg C for ecosystem productivity and 192 * 0.18 m^2*years = 35 $Q_{biodiversity}$-weighted m^2*years for biodiversity.

These values can be normalised and expressed in person-equivalents by dividing by the normalisation references from section 3.2 and 3.3: 1750 kg C / capita / year and 3.8 * 10^6 $Q_{biodiversity}$-weighted m^2*years/capita/year, respectively, resulting in the following normalised values per 1000 kg sand: 0.055 milli-person-equivalents for ecosystem productivity and 0.009 milli-person-equivalents for biodiversity.

As sand extraction involves the complete removal of soil to a great depth, there may be a significant impact on cultural values (see section 2.4), unless particular precautions are taken.

Beef production in Europe
The marginal beef production is taking place on permanent grasslands, which are currently being taken out of cultivation. Thus, beef production can be seen as a simple postponement of relaxation in the natural biome. If the natural biome is assumed to be temperate deciduous forest, the calculation will be parallel to the above. However, an assessment at the biome level will not reflect that permanent grasslands may be placed in biotopes quite different from the average biome area, and thus stresses the importance of improving the availability of indicators at lower ecosystem levels than biome, as suggested in chapter 2.

Road transport in Europe
Road transport is mainly taking place between high-density population centres placed in fertile areas. Although areas for roads are therefore often taken out of agricultural use, and the trend is an increase in area for infrastructure, the general trend for agricultural land is a reduction of cultivated area, also beyond that which is transferred to other human use. Thus, roads can be seen as a simple postponement of relaxation in the natural biome temperate deciduous forest, and the calculation will be parallel to the above, with area*time being 0.004-0.021 m^2*year per t*km transport (Lindeijer *et al.* 1998). It should be noted that roads often have considerable effects on the neighbouring areas.

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Annex 1

Existing A_{exi} and potential A_{pot} ecosystem areas on a simple biome level

Biosphere Atmosphere Transfer Scheme (USGS 2000)		**Biome 2.0 model** (Leemans *et al.*, 1998); data from 1970		
Existing biome / land use	*Area [km²] (A_{exi})*	*Related potential biome*	*Area [km²] (A_{pot})*	*Land cover area (A_{exi})*
1 Crops, Mixed Farming 10 Irrigated crops Agriculture total	14339934 3362275 17702209	1 Agriculture 4 Extensive grassland Agriculture total	-	31513480 12311561 43825041
2 Short Grass, 7 Tall Grass Grassland total	10619309 9417805 20037114	11 Grassland/Steppe, 14 Savanna Grassland total	17170062 11534288 28704351	9016363 5390550 14406913
3 Evergreen Needleleaf Trees	6088235	6 Boreal forest	**25332520**	20721432
4 Deciduous Needleleaf Trees	2990132	7 Cool conifer forest	4722752	2441293
5 Deciduous Broadleaf Trees	5856440	9 Temp. Deciduous forest	6198476	1013739
6 Evergreen Broadleaf Trees	12819330	16 Tropical forest	5738681	3776888
(7 Tall Grass)	(9417805)	-	-	
8 Desert 11 Semidesert Desert total	15979128 12344809 28323937	12 Hot Desert	14181394	11567131
9 Tundra	7542839	4 Tundra, 5 Wooded tundra Tundra total	9029500 3076248 12105748	9537464 3774699 13312163
(10 Irrigated Crops)	(3362275)	-	-	
(11 Semidesert)	(12344809)	-	-	
12 Ice Caps and Glaciers	16573114	3 Ice	6660460	6684545
(13 Bogs and marches)	(1385551)	-		
14 Inland Water, 15 Ocean	371478604	-	-	
13 Bogs and Marshes 16 Evergreen Scrubs 17 Deciduous Scrubs Scrubland total	1385551 1090829 3788346 6264726	13 Scrubland	8120857	2239859
18 Mixed Forest	3673596	8 Temp. mixed forest, 10 Warm mixed forest Mixed forest total	7689525 4759974 12449499	1714379 895506 2609885
19 Interrupted Forest	17746431	15 Tropical woodland	7622253	4254199
20 Water and Land Mixtures	-	-	-	
Total land area (excluding inland water & oceans)	145618000		131000000	131000000

Annex 2

Land classification schemes with their number codes: five from USGS and one (bold) from RIVM

USGS Land Use/Land Cover System Legend (Modified Level 2)	IGBP Land Cover Legend	Simple Biosphere Model Legend	Simple Biosphere 2 Model Legend	Biosphere Atmosphere Transfer Scheme Legend	Biome 1 model (Potential vegetation)	**Biome 2 model** (Potential vegetation)
1 Urban and Built-Up Land	1 Evergreen Needleleaf Forest	1 Evergreen Broadleaf Trees	1 Broadleaf Evergreen Trees	1 Crops, Mixed Farming	1 Ice/Polar Desert	1 -
2 Dryland Cropland and Pasture	2 Evergreen Broadleaf Forest	2 Broadleaf Deciduous Trees	2 Broadleaf Deciduous Trees	2 Short Grass	2 Semidesert	2 -
3 Irrigated Cropland and Pasture	3 Deciduous Needleleaf Forest	3 Deciduous and Evergreen Trees	3 Broadleaf and Needleleaf Trees	3 Evergreen Needleleaf Trees	3 Tundra	3 Ice
4 Mixed Dryland/Irrigated Cropland and Pasture	4 Deciduous Broadleaf Forest	4 Evergreen Needleleaf Trees	4 Needleleaf Evergreen Trees	4 Deciduous Needleleaf Tree	4 Taiga	4 Tundra
5 Cropland/Grassland Mosaic	5 Mixed Forest	5 Deciduous Needleleaf Trees	5 Needleleaf Deciduous Trees	5 Deciduous Broadleaf Trees	5 Cold Deciduous Forest	5 Wooded tundra
6 Cropland/Woodland Mosaic	6 Closed Shrublands	6 Ground Cover with Trees and Shrubs	6 Short Vegetation/C4 Grassland	6 Evergreen Broadleaf Trees	6 Cool Grass/Shrub	6 Boreal forest
7 Grassland	7 Open Shrublands	7 Groundcover Only	7 Shrubs with Bare Soil	7 Tall Grass	7 Cool Conifer Forest	7 Cool conifer forest
8 Shrubland	8 Woody Savannas	8 Broadleaf Shrubs with Perennial Ground Cover	8 Dwarf Trees and Shrubs	8 Desert	8 Cold Mixed Forest	8 Temp. mixed forest
9 Mixed Shrubland/Grassland	9 Savannas	9 Broadleaf Shrubs with Bare Soil	9 Agriculture or C3 Grassland	9 Tundra	9 Cool Mixed Forest	9 Temp. deciduous forest
10 Savanna	10 Grasslands	10 Groundcover with Dwarf Trees and Shrubs	10 Water, Wetlands, Ice/Snow	10 Irrigated Crops	10 Temperate Decidous Forest	10 Warm mixed forest
11 Deciduous Broadleaf Forest	11 Permanent Wetlands	11 Bare Soil		11 Semidesert	11 Evergreen/Warm mixed Forest	11 Grassland/Steppe
12 Deciduous Needleleaf Forest	12 Croplands	12 Agriculture or C3 Grassland		12 Ice Caps and Glaciers	12 Warm Grass/Shrub	12 Hot desert
13 Evergreen Broadleaf Forest	13 Urban and Built-Up	(missing in HTML-file)		13 Bogs and Marshes	13 Hot Desert	13 Scrubland
14 Evergreen Needleleaf Forest	14 Cropland/Natural Vegetation Mosaic	(missing in HTML-file)		14 Inland Water	14 Xerophytic Woods/Shrub	14 Savanna
15 Mixed Forest	15 Snow and Ice	(missing in HTML-file)		15 Ocean	15 Tropical Rain Forest	15 Tropical woodland
16 Water Bodies	16 Barren or Sparsely Vegetated	(missing in HTML-file)		16 Evergreen Shrubs	16 Tropical Seasonal Forest	16 Tropical forest
17 Herbaceous Wetland	17 Water Bodies	17 Persistent Wetland		17 Deciduous Shrubs	17 Tropical Dry Forest/Savanna	
18 Wooded Wetland		18 Dry Coastal Complexes		18 Mixed Forest		
19 Barren or Sparsely Vegetated		19 Water		19 Interrupted Forest		
20 Herbaceous Tundra		20 Ice Cap and Glacier		20 Water and Land Mixtures		
21 Wooded Tundra						
22 Mixed Tundra						
23 Bare Ground Tundra						
24 Snow or Ice						