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Biodiversity and life support indicators for land use impacts in LCA

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Dit rapport beschrijft de tweede fase van een onderzoek naar een operationele methode om het thema 'aantasting van *ecosystemen en landschap' in onder te brengen in* milieugerichte levenscyclusanalyse (LCA).

Het rapport beschrijft de huidige methoden en geeft aan welke tekortkomingen deze hebben. Vervolgens wordt een nieuwe *methode beschreven die voldoet aan de criteria:* praktisch haalbaar, mondiaal toepasbaar, wetenschappelijk gefundeerd en inpasbaar in LCA.
De methode is nog niet bruikbaar voor het subjectieve thema aantasting van landschap, voor ingrepen in aquatische *ecosystemen en voor* versnippering.

In de methode worden twee ingrepen onderscheiden: veranderen van soort landgebruik (m^2) en continu bezet *houden van land* ($m^2 \bullet j$). *Voor beide ingrepen worden twee* indicatoren benoemd: een indicator voor de invloed op het life support systeem (fNPP) en een indicator voor de invloed op de biodiversiteit (α). Dit resulteert in vier indicatoren. Verdere weging vereist subjectieve keuzes en deze vallen buiten de *reikwijdte van dit project.*

De methode is beproefd in een zevental cases en bleek uitvoerbaar. Een beperking is de beschikbaarheid van voldoende betrouwbare gegevens. Nadere beproevingen met *betrouwbaarder gegevens zullen naar verwachting volgen in* fase 3.

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Biodiversity and life support indicators for land use impacts in LCA

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Content

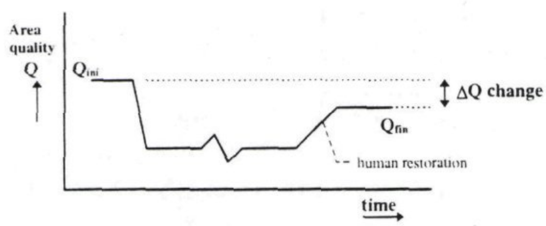
Managementsamenvatting

In opdracht van Rijkswaterstaat Dienst Weg- en Waterbouwkunde (RWS DWW) is een methode operationeel gemaakt om de effecten van landgebruik op ecosystemen in produkt-levenscyclusanalyses (LCA) mee te kunnen nemen. IVAM Environmental Research was hierbij projectleider en bewaakte het methodologisch kader, terwijl het Instituut voor Bosbouw en Natuuronderzoek (IBN-DLO) de indicatoren nader uitwerkte en gegevens verzamelde.

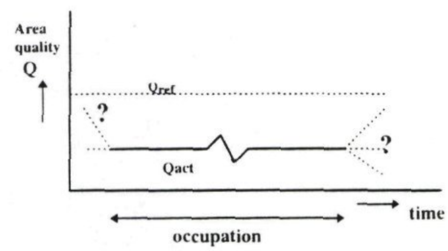
De belangrijkste aanleiding voor RWS DWW om dit onderzoek uit te zetten is dat bij beleidsbeslissingen omtrent bouwgrondstoffen en infrastructuur steeds meer aandacht is voor LCA als beleidsondersteunend instrument, ook bij RWS DWW. Omdat in LCA's effecten van landgebruik tot op heden nog niet worden meegenomen, zijn sub-optimale beleids-aanbevelingen te verwachten. RWS wil bijdragen aan een verandering van deze situatie. Eerder is al een voorstudie uitgevoerd voor RWS DWW door IVAM ER. Deze studie bouwt daarop voort en heeft als doel het ontwikkelen van een operationele methode voor effecten van landgebruik op ecosystemen in LCA.

Onder landgebruik wordt hier begrepen zowel het *veranderen* van het soort landgebruik (eenheid m^2), als het continu *bezet houden* van land voor eenzelfde menselijke activiteit (inclusief tijdsdimensie, m^2xjaar). Zie onderstaande figuren voor een weergave van de verschillende percepties van landgebruik:

Verandering van landgebruik (change):



Bezet houden van land (occupation):



Met deze methode kunnen de effecten van dit landgebruik op de zogenaamde 'life support' functie en op de locale biodiversiteit (en daarmee indirect op het potentieel risico van mondiaal biodiversiteitsverlies) op een systematische en reproduceerbare manier worden weergegeven.

De life support functie betreft de rol die (delen van) een ecosysteem speelt (spelen) in het onderhouden van levensprocessen, zoals het sluiten van stofstromen en een goede bodemstructuur. De lokale biodiversiteit geeft de intrinsieke (statische) natuurwaarde weer van een gebied. De twee thema's biodiversiteit en life support worden gezien als de belangrijkste bijdragen aan de ecologische waarde van een gebied.

Door deze uitsplitsing in vier elementen ontstaan in principe vier scores voor de effecten van landgebruik op ecosystemen. Een verdere aggregatie tot 1 score vergt een subjectieve weegstap, die hier niet is uitgevoerd.

Gekozen indicatoren

Als maat voor de bijdrage aan life support is de vrije netto primaire biomassa productie (fNPP) gekozen. Dit is de hoeveelheid biomassa die de natuur vrijelijk kan benutten voor de eigen ontwikkeling, ook als de mens de rest van de geproduceerde biomassa voor eigen consumptie gebruikt (zoals bij bos- en landbouw).

De bijdrage aan biodiversiteit wordt in deze studie gemeten aan de terrestrische soortendiversiteit van (vasculaire) planten, uitgedrukt in de parameter α. De doorslaggevende reden om deze maat te gebruiken voor biodiversiteit is dat hiervoor mondiaal voldoende wetenschappelijke gegevens beschikbaar zijn.

Beide maten worden afgezet tegen een referentiewaarde en (bij verandering van landgebruik) tegen de situatie direct vóór de verandering. Dit leidt tot twee formules per indicator, één voor het bezet houden van land en één voor een verandering in landgebruik:

ECOSYSTEM CHANGE (EC):

$EC = A \cdot (fNPP_{ini}-fNPP_{fin})$ als een maat voor life support en
$EC = A \cdot (\alpha_{ini}-\alpha_{fin})/\alpha_{ref}$ voor biodiversiteitseffecten door verandering van landgebruik

ECOSYSTEM OCCUPATION (EO):

$EO = A \cdot t \cdot (fNPP_{ref}-fNPP_{act})$ als een maat voor life support en
$EO = A \cdot t \cdot (\alpha_{ref}-\alpha_{act})/\alpha_{ref}$ voor biodiversiteitseffecten door bezet houden van land

De uitkomsten van de formules zijn de eigenlijke indicatoren voor effecten van landgebruik op ecosystemen.

Geproduceerde data
Voor beide indicatoren zijn gegevens verzameld voor referentie gebieden op mondiale schaal, weergegeven op wereldkaarten. De referentiewaarden zijn gebaseerd op de meest recente wetenschappelijke metingen.

Verder is voor een aantal concrete situaties van landgebruik een inschatting gemaakt van de biodiversiteits- en vrije biomassawaarden. Voor deze situaties zijn dus reeds default indicatorscores opgesteld voor zowel verandering als het bezet houden van het land. Het gaat om de cases aluminiumwinning in Zuidamerikaans tropisch bos, zandwinning, industriële productie, wegtransport en afval storten in Europees landbouwgrond en bosbouw en waterkracht in Scandinavische heuvels.

Tenslotte is voor een groot aantal metalen, fossiele brandstoffen en houtsoorten de gemiddelde referentiewaarde in de winningsgebieden bepaald. Hiermee kan het bezet houden van land tijdens de winning beschreven worden.

Beperkingen
De belangrijkste beperking van de gegevens die nu gebruikt zijn om de methode te operationaliseren is dat ze te globaal zijn om erg specifieke situaties te beschrijven. Dat betekent dat landgebruik in gebieden met uitzonderlijke locale natuurwaarde te gunstig beoordeeld zal worden. Voor dergelijke situaties is een meer gedetailleerde analyse, zoals soms toegepast in MER procedures, beter op zijn plaats. Meer gedetailleerde gegevens over soortendiversiteit en biomassa-productie (voor, tijdens en na de activiteit) kunnen wel in de huidige systematiek worden geïntegreerd.

Voor de situatie op zee zijn er geen vergelijkbare gegevens beschikbaar.Voor zeebodems zonder extreem hoge natuurwaarde is het effect van kortstondige ingrepen vermoedelijk verwaarloosbaar door de dynamiek van die ecosystemen.

In huidige LCA's kan met beschikbare gegevens over landgebruik door menselijke activiteiten en de hier gegenereerde data alleen het effect van het bezet houden van land worden beoordeeld. Voor de effecten van verandering van landgebruik zijn meer specifieke gegevens nodig. Een voorbeeld is welke eindsituatie wereldwijd gemiddeld wordt bereikt na beëindiging van koperwinning, op welk gemiddeld oppervlak en bij welke gemiddelde productiecapaciteit.

De effecten van verdroging kunnen waarschijnlijk ook met deze methode worden weergegeven, hoewel de milieu-ingreep dan niet zozeer landgebruik als wel grondwater-onttrekking is. Hiervoor zijn echter nog geen data verzameld.

Landschap betreft in eerste instantie niet de waarde van ecosystemen, maar eerder de subjectieve menselijke beleving van een omgeving. Landschappelijke aantasting kan dan ook niet door bovengenoemde indicatoren worden beschreven. Dit thema verdient een aparte operationalisatie, die in het kader van deze studie niet mogelijk was.

Ook de effecten door versnippering kunnen niet met deze indicatoren worden weergegeven. Een meer op fauna gerichte indicator zou hier op zijn plaats zijn, naast een factor die de omtrek van de ingreep beschrijft.

Deze lijst van beperkingen is begrijpelijk gezien het innovatieve karakter van deze studie. Deze studie is de eerste uitwerking van een karakterisatiemethode voor effecten van landgebruik in Nederland. De uitwerking is echter zodanig dat de methode werkelijk gebruikt kan worden, en na deze eerste stap verder ontwikkeld kan worden.

Conclusies en aanbevelingen

Gebleken is dat de methodiek toepasbaar is op cases en ingevoerd kan worden in bestaande databases. Voor een algemene toepasbaarheid dient eerst meer data over landgebruik bij economische processen verzameld te worden, en ingevoerd te worden in alle veel gebruikte LCA databases. Ook dient er ten behoeve van een integrale afweging meer bekend te zijn over het relatieve belang van effecten van landgebruik ten opzichte van andere effecten op de kwaliteit van ecosystemen. Er is gewerkt aan een internationale acceptatie van deze methode, maar een officiële instantie die hier uitspraken over kan doen is pas over enkele jaren te verwachten.

Voor infrastructurele werken en andere belangrijke veranderingen in landgebruik is het nodig om lokaal gegevens te verzamelen over biodiversiteit en biomassa-productie voor, tijdens en na de activiteit. Deze kunnen dan worden ingepast in een complete LCA, gebruik makend van dezelfde referentiewaarden.

Leeswijzer

De hoofdtekst bestaat uit 5 hoofdstukken. Hoofdstuk 1 beschrijft de onderzoeksomgeving (aanleiding, doel, inhoudelijke randvoorwaarden en begeleidingscommissie). Hoofdstuk 2 gaat in op de basis voor de eigenlijke methodiek, het landgebruik zelf. Hoofdstuk 3 is de methodische kern van het verhaal en verantwoordt de voorgestelde indicatoren. Hierin wordt verwezen naar annex 1 voor de verdere onderbouwing en dataverzameling voor de indicatoren. Voor de situatie in zee wordt verwezen naar annex 2. De relatie met gerelateerde milieu-effecten zoals landschappelijke aantasting en verdroging wordt ook in hoofdstuk 3 beschreven. In hoofdstuk 4 worden de methodische data toegepast op een aantal cases. Naast de conclusies per hoofdstuk wordt in hoofdstuk 5 de geoperationaliseerde methodiek nog een geëvalueerd, worden de conclusies nog eens samengevat en worden onderwerpen voor nader onderzoek opgesomd.

Bij de hoofdtekst horen 4 appendices. In appendix 1 staat een overzicht van alle bekende methodes om landgebruik te kwantificeren in het kader van LCA. Appendix 2 beschrijft de resultaten van een workshop die IVAM ER gehouden heeft op een LCA congres in Bordeaux, inclusief een vragenlijst die naar de deelnemers is verstuurd. Appendix 3 geeft de landgebruiksgegevens weer die gebruikt zijn voor de cases. Appendix 4 tenslotte bevat tabellen met gemiddelde referentiedata voor allerlei soorten grondstofwinning, om de toepasbaarheid van de data te vergemakkelijken.

Tenslotte verdienen appendices 4 en 5 bij annex 1 nog een aparte vermelding. Dit zijn de twee wereldkaarten waarop IBN-DLO de referentiewaarden voor biodiversiteit en life support op hebben aangegeven. Deze kaarten zijn onmisbaar voor een eerste globale uitwerking van deze methodiek in bestaande LCA's.

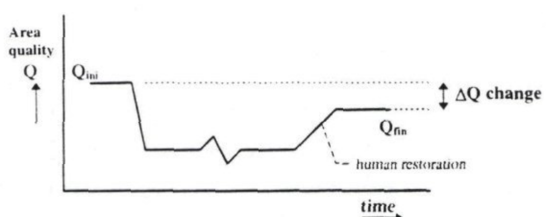
Summary

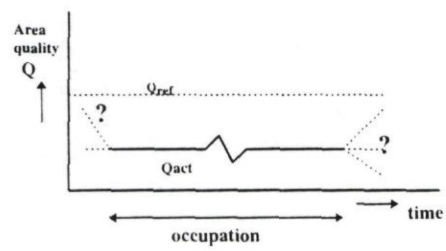
A method has been operationalised to include the main impacts of land use on ecosystems in LCAs, under commission of the Dutch Ministry of Transport, Public Works and Water Management (RWS, division DWW). IVAM E R, the environmental consultancy of the University of Amsterdam, was project leader and guarded the methodological framework, whereas the Dutch Institute for Forestry and Nature Research (IBN-DLO) operationalised the impact indicators in detail, gathering the necessary data.

The most important reason for RWS DWW to commission this research is that in policy decisions about aggregates and infrastructure more and more attention is given to LCAs as decision support tool, also within RWS DWW. However, as in LCAs impacts due to land use have not been incorporated up till now, sub-optimal policy recommendations are to be expected. RWS DWW wishes to improve this situation. An earlier pilot study on this subject has been commissioned by RWS DWW to IVAM ER. This study continues where that study ended and has as aim to develop an operational method to include land use impacts on ecosystems in LCA.

Terminology land use

Land use is being divided into net *changing* the type/quality of land use (units m^2) and *occupying* land for a certain activity (units m^2 times year). These two aspects of land use can be illustrated with the following two figures. The horizontal axis depicts the course of time and the vertical axis depicts the quality (change) relative to a reference state.





These two aspects of land use are by now internationally accepted among LCA experts, according to a workshop and a survey based on a presentation on this subject at a SETAC conference in April 1998. Two proposals have been given to combine the two types of land use. One is to make an integral, neglecting the situation after the activity (thereby assuming no net change). The other is to assume a hypothetical regeneration time to the situation before, resulting also in an imaginary situation of no net change. The last option is used in the ETH database. We have not chosen to perform such solutions, in order to allow all possible perceptions to be considered.

Determining the impact scores

With this method the impacts on the so-called 'life support' function and on the local biodiversity (therewith indirectly indicating the potential risk of global loss of biodiversity) can be expressed in a systematic and reproducible manner. The two themes life support and biodiversity are seen as the most important contributions to the ecological value of an area, although not including scarcity is seen as a limitation.

We have chosen to select only two indicators for ecosystem quality using a top-down approach. This was done with a view of the difficult task of valuating the many disaggregated indicator scores that would result if all land use type specific indicators would be operationalised without prior view of their relative importance. The cause-effect chain for land use and related issue has been drawn, showing the different possible levels of indicators.

The life support function concerns the role that (parts of) an ecosystem plays in maintaining life processes, such as the closing of substance cycles and a properly functioning soil structure. Life support can therefore be considered as a an expression of the dynamic nature value of an area. As a measure for the contribution to life support the free net primary production (fNPP) is chosen. This is the amount of biomass which nature can freely apply for its own development, when human consumption of biomass is subtracted, as in forestry and agriculture. This fNPP is compared to the biomass production in a natural ecosystem from the same region (NPPref) and, for land use changes, to the (free) biomass production before the change.

Local biodiversity expresses the intrinsic nature value of a region. The contribution to biodiversity is here measured using the species diversity of vascular plants (S), expressed in the parameter α. The relation between S and α is as follows:

$$S_{map} - S_{ref\ cell} = \alpha * LOG(A_{map}/A_{ref\ cell})$$

where $_{map}$ stands for the situation in the mapping area in which data are collected, and $_{ref\ cell}$ is a reference mapping cell of 1 m^2 stated to contain 10 species ($S_{ref\ cell} = 10$, $A_{ref\ cell} = 1$). This relation is used to compare data from various data collection schemes. For land use impacts themselves, the relation between α and the area of land use A is considered to be linear, using the so-called hot spot theory.

The most important reason for choosing species diversity as indicator for biodiversity is that sufficient scientific data are available globally. Also this measure is compared to a reference state and, for land use change, to the situation just before the change. As with life support, this leads to two formulas for expressing biodiversity impacts due to land use. The numerical values of the formulas are the actual indicators, or in LCA terms the equivalency factors, for impacts of land use on ecosystems.

By this separation into four elements in principle four scores result. A further aggregation to one score requires a subjective weighting step, which is not performed here.

Generated data

For both indicators data are generated for reference areas on a global scale, presented on world maps. The reference values are based on the most recent scientific measurements. The classification of areas with the same reference state is based on important abiotic factors such as altitude, latitude and amount of rainfall. This results in a so-called physiotope classification. This is a more flexible and better defined classification than the classification of 'naturalness' also used for land use typology in LCA. The reference state that results from these data concerns the most undisturbed situation available in the *present* region.

Furthermore, biodiversity and free net biomass values have been estimated for a number of specific land use situations. For these situations default values have been generated for occupation as well as for change of land use. The cases considered were bauxite mining in South-American tropical forests, sand extraction, industrial production, road traffic and landfill in European agriculture grounds, and forestry and hydropower in Scandinavian hills.

For biodiversity the scores per case are based on diversity reduction factors derived from studying Dutch GIS (geographic information system) data. The fNPP values are based on NPP measurements in comparable situations and calculation of the extraction of biomass for human consumption. These data are linked to the available data on the intervention, land use.

Finally, for a large amount of metals, fossil fuels and types of wood the average reference values for occupation have been determined for the exploitation areas. Occupation impacts for most resources have become available with this approach.

All the occupation data have been put in an updated version of the IVAM ER database, making the method operational in at least one public database.

Restrictions
The most important restriction to the data gathered to operationalise the present approach is that they are too generic to describe very specific situations. This implies that land use in regions with extraordinary local nature value will not be judged sufficiently severe. For such situation a more detailed analysis, as sometimes applied in Environmental Impact Assessments, is more appropriate. Collected detailed (site-specific) data on species diversity and biomass productivity before, during and after the activity can be integrated in the present method, however.

For land use at sea no comparable information is available to select only species diversity as an indicator for biodiversity. The age distribution and the species composition should be included in an expression of the regeneration time relative to a reference in the same ecotope. For seabeds without specific nature value the impact of short-term land use interventions are probably negligible due to the high dynamics of those ecosystems, especially for life support.

In present LCAs only land occupation impacts can be judged with the now available data on land use and the here generated data. For impacts of land use change more specific data are required. An example of such data is which average final situation is reached after closing copper mines, using what average area and with which overall production capacity.

The impacts of desiccation can probably also be expressed with these indicators, although the intervention is then groundwater extraction rather than land use. No data are available for this, however.

Landscape degradation can not be described by above indicators. These impacts do not relate to ecosystem quality in first instance, but rather to the subjective human perception of the surroundings in terms of historic, geological and aesthetical values. This theme deserves a separate operationalisation, which was not possible within the framework of this study.

The impacts of fragmentation can not be expressed using these indicators either. An indicator for fauna would be more appropriate here, linked to a factor based on the circumference of the land use activity.

The list of restrictions is long. However, this is only a first step towards full operationalisation of the approach. Many options for further research have been given. Nevertheless, this approach can already be used, and is for instance used in the LCA software tool Eco-Quantum.

Conclusions
Above operationalisation has been presented to international LCA experts and users via contacts within SETAC throughout the research. It seems that the general framework for dealing with land use and the indicator for biodiversity is accepted. There were more doubts on the indicator for life support, as this resulted in contra-intuitive results for interventions in biomass-poor ecosystems, and because the concept of life support is still insufficiently defined. Standing biomass has been proposed as an alternative indicator for life support.

The method has proven to be applicable on cases and (at least for occupation) can be included in present databases. For a more general use of the method, first more data needs to be gathered about land use connected to the various economic processes and inserted in all popular databases. For an overall assessment some indication of the relative importance of land use impacts compared to other impacts on the quality of ecosystems is needed. Some effort has been put into the international acceptance of this approach, but an official organisation to state such acceptance for LCAs in general can be expected only after some years to go.

For infrastructural activities and other important land use changes local information must be gathered on biodiversity and biomass productivity before, during and after the activity. These data can then be incorporated in a complete life cycle analysis, using the same reference values.

Readers guide

The main text of this report consists of 5 chapters. Chapter 1 describes the goal and the scope of the research and the guiding committee. Chapter 2 describes the framework of land use itself. Chapter 3 is the main chapter, discussing the various options and choices made to arrive at the final expressions for land use impacts. In this chapter references are made to annex 1 for the further argumentation and data collection for the indicators. For the situation at sea it is referred to annex 2. The relation with other impacts such as landscape deterioration and desiccation is also shown in chapter 3. In chapter 4 the data are applied to a number of cases, including the problem of allocation. Chapter 5 contains an evaluation of the approach, a summary of the conclusions drawn in the report and a list of possible future research topics.

The main text has 4 appendices, with an overview of existing land use approaches (1), the results of the SETAC workshop held at Bordeaux including the survey text (2), the land use data used in the cases (3) and tables with mean reference data for all kinds of global resource extractions, to facilitate use of the collected information.

Appendices 4 and 5 to annex 1 deserve separate mentioning. These are the two world maps with reference data for biodiversity and life support which IBN-DLO have produced. These maps are indispensable for a first global operationalisation of this approach in current LCA studies.

1. Goal and scope of the study

Introduction
The Ministry of Transport, Public Works and Water Management (RWS, division DWW) has commissioned a research on the operationalisation of land use impacts in LCA. It is the follow-up of a theoretical survey study on this issue [Blonk & Lindeijer, 1995] and includes all known previous studies on the subject. RWS DWW is involved with LCAs for construction and for aggregate extraction policy. In such cases land use aspects tend to be major aspects in an environmental assessment. Therefore, within RWS DWW some discontent raised on the fact that land use issues were not adequately included in LCA studies. Within DWW the discussions on land use impacts are well-known but up till now no approach was able to include this issue consistently in an LCA. The was the starting point for the consecutive research projects on this subject.

In European environmental policy, land use impacts are considered of major importance. Nearly all of European area has been disturbed or (sometimes drastically) changed by human interference with the natural environment. Nowadays, many of human activities are regulated through spatial planning. Nevertheless, some 70% of ecosystem degradation in Europe is considered to be due to changes in land use, either destruction or disturbance (see [RIVM, 1992]). Over half of European land area is permanently in use for human activities ([Eurostat et al., 1995]). Assessing contributions to this general effect of large scale land occupation and land use changes due to human activities is what is aimed at when trying to operationalise (quantify) land use effects in LCA.

Content
The aim of this research is to evaluate and (further) develop indicators for potential land use impacts in life cycle assessment (LCA). In LCA terms this implies the operationalisation of the characterisation of these potential land use impacts in terms of category indicators to be applied in LCIA (Life Cycle Impact Assessment). In general terms this means that physical measures are chosen and data is collected to express the potential impacts of very different types of land use (all over the world) due to a product or service life from cradle to grave.

What are the requirements set by this goal? There are several requirements to be met ideally, of which the major ones are mentioned here.

- One requirement is about the *applicability on different scale levels*. The most generic level of aggregation is that of possible impacts due to land use in an area as big as a country or larger (without knowing where exactly in that country the land use occurs). But the framework should also allow for dealing with more detailed information on certain parts of the life cycle, and more generic information on the rest of the life cycle. Thus, the framework should allow for consistent application at different levels of detail.
- Another requirement has to do with the pragmatic ambition of this study. It is envisioned to produce equivalency factors (indicator scores per land use impact) for at least the most generic level of detail and a number of land use cases. The ambition is to produce a *readily applicable set of equivalency factors* for all main categories of land use.
- An important requirement is that the major kinds of land use impacts encountered in LCA are indeed covered by the indicators which are to be selected. According to the present draft of ISO 14042 [ISO, 1998], these indicators should have a clear link to land use impact endpoints. Endpoints are higher order objects affected by the intervention in nature. To clarify this link, the presumed cause-effect chain related to land use should be explicitated and *relevant end-points and indicators* should be defined. These indicators should preferably be based on scientific knowledge.
- A crucial constraint to the indicator due to the requirement of *linearity with respect to the functional unit* (a general requirement of traditional LCA) is that a linear extrapolation to larger or smaller units of human activity (functional units) should be adequate: the

indicator should relate to the type of land use, but its value should not be dependent on the amount of land used. This implies that the indicator can not be used to describe or evaluate situations where an extremely large impact can be expected due to the amount of land use assessed. This should secure that the assumption, that doubling the amount of land used causes a doubling of the impacts, is valid. LCA should thus not be used to describe the loss of the last 10% of the tropical forest or the last habitat for the rhinoceros, nor for the first road cutting through a primary forest. Generally speaking, LCA should not be used to replace Environmental Impact Assessments or other specific assessment approaches where those are more appropriate and sufficient.

There are also some limitations given by the extent of this research. Aesthetic aspects, largely captured under the heading *landscape degradation* could only be considered marginally, because a totally different end-point is at stake than the one focused on in this study and no serious suggestion for operationalisation was available to start with.

Also, impacts with no linear relation to the area of land used were not included here. Thus, impacts outside of the range of the actual land use area (*desiccation* impacts of an activity, or impacts due to *fragmentation*) are not taken into account. Nevertheless, some suggestions are given on how to operationalise these impacts.

Process

This research has been conducted for the Road and Hydraulic Engineering Division (a division of the Ministry of Transport, Public Works and Water Management) by IVAM Environmental Research University of Amsterdam (IVAM ER) in co-operation with the Dutch Institute for Forestry and Nature Research (IBN-DLO), locations Wageningen (for biodiversity and biomass) and Texel (for marine aquatic systems case). The commissioner (project leader J. Broers) has appointed a guiding committee, consisting of the following persons:

D. Bal	Ministry of Nature Management, National Reference Centre for Nature	IKC Natuurbeheer	Wageningen
H. van Bohemen	Road and Hydraulic Engineering Division[1])	RWS DWW	Delft
L. Breedveld	Institute for Inland Water Management and Waste Water Management[1])	RWS RIZA	Lelystad
B. ten Brink	National Institute of Public Health and the Environment	RIVM	Bilthoven
A. Eijs	Ministry of the Environment	VROM SVS	The Hague
R. Heijungs	University of Leiden, Centre of Environmental Science	RUL CML	Leyden
J. Klein	Winand Staring Centre for Integrated Land, Soil and Water Research	Staring Centrum DLO	Wageningen
J. Lourens	National Institute for Coastal and Marine Magagement[1])	RWS RIKZ	The Hague
A. Schuurmans / J. Meijer	Intron (quality assessment institute for the building industry)	Intron	Sittard
G. Sigmond	FODI (federation of aggregates industry associations)	FODI	Nijmegen
M. Spierings	Institute for Inland Water Management and Waste Water Management[1])	RWS RIZA	Lelystad
T. Verstrael	Road and Hydraulic Engineering Division[1])	RWS DWW	Delft
H. Wijnen	Ministry of the Environment	VROM ICB	The Hague

[1]) Divisions of the Ministry of Transport, Public Works and Water Management

Some members of the guiding committee acted as corresponding members. This committee has followed the research and has delivered valuable inputs and criticism. Also, a large international forum of LCA experts were consulted at various stages of the project, asking for comments on draft reports and methodological developments, presenting the nearly final results at an international congress and attending at EU concerted action meetings to communicate on the approach followed. The project lasted for about one year.

One member of the guiding committee finally considered the approach to be insufficiently applicable, due to the general LCA restriction of linearity of impacts versus functional unit (excluding impacts on the last few % of a specific ecosystem habitat) and restricted regionalisation, not including unknown impacts, nor the impacts outside of the actual area (for instance due to fragmentation). As these applications are explicitly excluded from the approach, the authors consider this approach still to have its value, as it is in our view the first consistent, generally applicable, science-based and operational approach to include land use impacts in LCA. Next steps are to be taken to include more aspects, but also land use data need to be gathered to support the use of this approach. Leaving out land use altogether, as is often done in LCA, is considered the worst option.

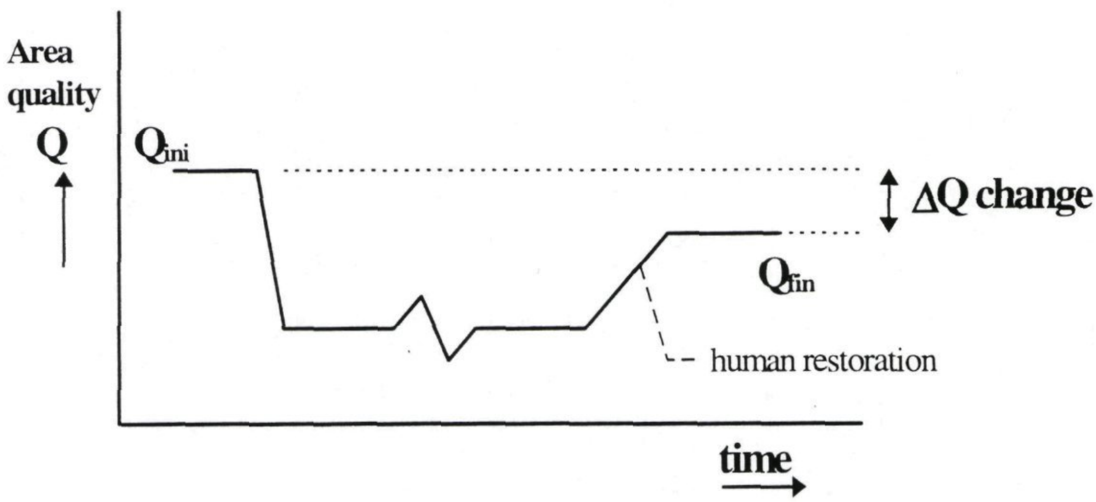
2. Inventory framework for land use

Human activities always imply land use, direct and/or indirect (via use of materials). Either the type of land use is changed (when starting an activity where there was none or there was another type of activity), or the existing land use is continued. We call the first type *land use change* and the second *land occupation*. This distinction is very important and clarifying to make, and both types of land use are illustrated below. The consequence of trying to combine both types of land use is also shortly discussed. This chapter concludes with aspects of dealing with additional inventory information and an inventory format for land use.

2.1 Land use change

The figure below depicts a general view of what happens (in terms of land quality changes) when land is being used for an activity. What land quality we are talking about here is left open, but the quality is related to the natural value in the area.

Figure 1: Land use change



An initial (nature) quality of the land (Q_{ini}) may be decreased upon starting the activity, and remain at a certain low level, during which the activity produces a finite amount of output. After the end of the activity some of the quality may be restored by human measures, leaving probably a lower final quality (Q_{fin}). At the end of the activity, having produced a certain output, a net quality change (ΔQ) results. This net change per unit of output is independent of the time used to produce that output, and is expressed in general terms as:

Land use change = A * ΔQ

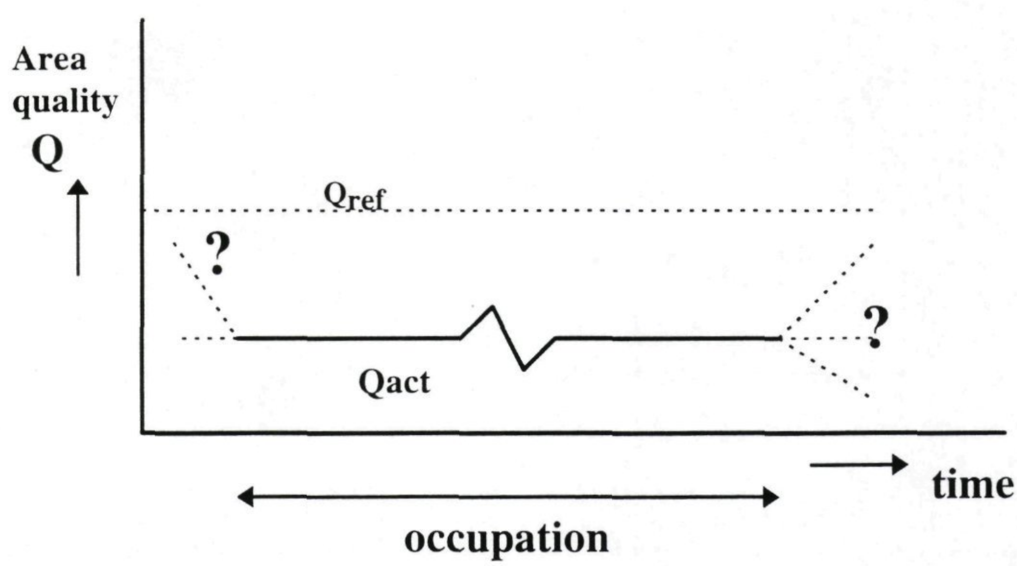
with A the area of land used to produce a unit of output and with units square meters (m^2) when ΔQ has no dimension. By which indicators Q_{ini} and Q_{fin} are expressed and how ΔQ is built up from Q_{ini} and Q_{fin} is subject of the next chapter. Typical activities following this sketch in practice are mining and landfill.

Note that this net change can only be assessed when the situation before and after the activity, and the total output is known! Constructing a building or a road leads to net land use change, but the service or function (delivering a building or road) is linked to the consecutive use of that sealed surface (formally another function). When it is not known how long that use will be or what the total output will be, the net change can not be allocated to the use of the surface (in terms of functional units of output). This relates to other impacts of all capital goods, by the way.

2.2 Land occupation

In situations where the change from one land use type to another has long ago been made, and future changes are not foreseen, it is possible that no direct impact is visible (as in transport over an existing road, use of built area, agriculture and forestry). However, keeping land occupied for human activities implies the continuation of the (generally lower) nature quality of the land caused by an initial land use change, for a certain amount of time. Land occupation for human purposes generally lowers the opportunities for nature outside the considered land area to refuge to that area. It also implies land competition (between man and nature, but also between different human activities). Occupation occurs during temporary changes of land use (see the figure above, the lower horizontal line) but also when the start and/or end of the type of activity is not known or very uncertain (see the figure below).

Figure 2: Land occupation



The land quality during the occupation for human activities (Q_{act}) is generally lower than the maximum quality to be reached in the area (Q_{ref}), and can be related to that. The longer the activity lasts, the larger the total competition and longer period of quality setback. On the other hand, the more efficient the output production, the lower the land occupation impacts per unit of output. Land occupation therefore includes a time aspect and is expressed in general terms as follows:

Land occupation = A * t * Q

where A is the area of land occupied and t is the occupation time. When Q has no dimension, the resulting unit is [m^2.y]. Again, how Q is to be expressed in terms of Q_{act} and Q_{ref} is subject for the next chapter.

2.3 Integration of change & occupation?

The net land use change (x m^2 quality a to x m^2 quality b) due to an activity can be seen as an instantaneous event, whereas the land occupation is by definition spread out over a certain time (using x m^2 during y years). This distinction between land use changes and land occupation (with distinct units: m^2 versus m^2.y) is consistent with the distinction between land transformation and land use as mentioned in [Finnveden, 1996] and with the distinction between land consumption (under the heading resources) and land competition (under the heading availability) [Heijungs & Guinée, 1997]. See box I for some examples of both land occupation and land use change.

Box I: Land use situations

Traffic

To keep trucks driving on roads an initial land use change of x m^2 was necessary once (for instance from agriculture ground to sealed surface, ideally to be allocated over all vehicles ever driving that road). However, as generally neither beginning nor end-point of the activity is known, allocation of this change to a certain amount of activity (functional unit) is impossible.

What can be allocated to a functional unit in an LCA dealing with generic transportation data is the yearly occupation of land in terms of m^2.y land use with the quality of a sealed surface. Also, when for a certain region an average yearly increase in roads can be determined, this change (ΔA in m^2) may be allocated to the yearly increase in use intensity of the road (if adequate) and from that to the functional unit.

Landfill & mining

Land use for landfill is often temporarily. Once the site is closed it is generally sealed off and given another function. There may be no net land use change when the original land quality is retrieved. The only contribution to land use effects is then the occupation of the land during the functional lifetime of the landfill. The land occupied yearly (as z m^2 land with quality c) is then allocated to the amount of waste dumped yearly. When the initial quality is not retrieved, the net land use change can be allocated to all the waste dumped at the site. By the way, soil contamination reducing the retrieval of the initial quality should not be taken into account here to avoid double counting with indicators for soil contamination (such as terrestrial ecotoxicity).

Mining can be described in the same manner, taking mining productivity instead of amount of waste dumped as main functional output. Optional reclamation (restoration) activities can be seen as land occupation with increasing land quality, with a mean land quality during the reclamation time.

Agriculture and forestry

For forestry and agriculture relatively large areas are occupied yearly for a certain mass of product. Land use changes generally have occurred in the far past, or are part of present land planning projects. In the first case, the changes can not be allocated to a functional unit when the original situation, the time of change and the yearly output since is unknown or very uncertain. In the latter case, the change may not be considered as part of generic production activities and requires a specific analysis in terms of land use change (apart from the use of the forest). The change can not be allocated to the use of the forest, as the projected output of the forest over time is unknown, unless it is considered and can be analysed as part of a steady increase in productivity. In general forestry or agriculture practices, land use changes need not be considered (unless the land quality degrades slowly under the land management, see [Swan, 1998]). The continuous occupation of land does require consideration, especially because the amount of land is relatively large. The quality of the land is however higher than during open pit mining, landfilling or transportation over roads, which needs to be taken into account.

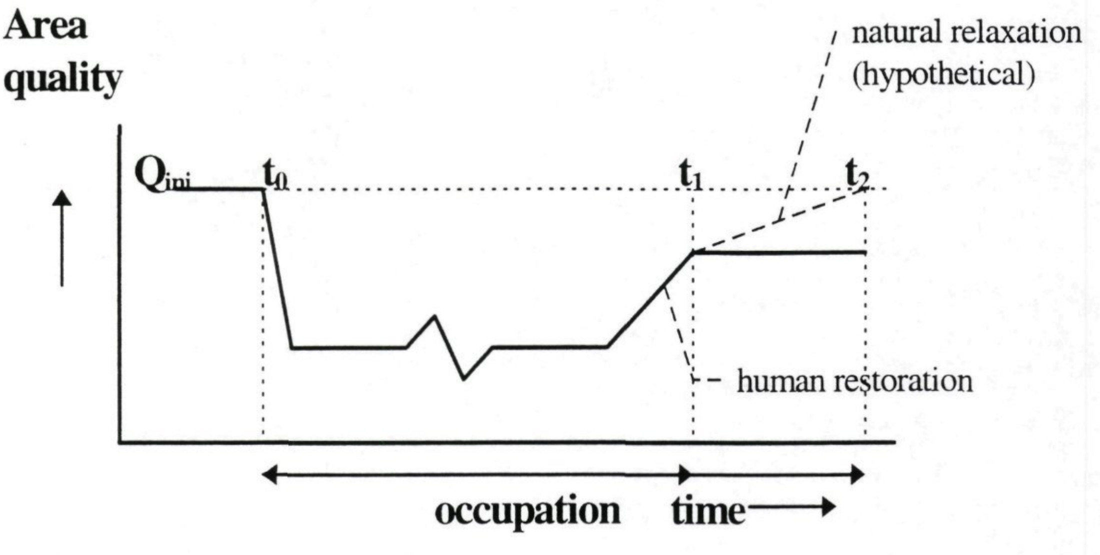
In one LCA land use study [Swan, 1998] land occupation is considered irrelevant with the argument that man has the right to occupy land for his living. This postulates the right of man to dominate over nature unlimited. Other studies ([Wegener-Sleeswijk et al., 1995], [Blonk & Lindeijer, 1995]) state that to assess land use changes is impossible or at least very difficult within LCA, and that the focus should be on the quality of the occupied land. We do not especially adhere to either view, but allow for a weighting of both aspects by suggesting an operationalisation for both aspects of land use. In fact, both aspects of land use are always linked for each activity, although the land use change may be hard to allocate to one specific activity.

One may want to combine the two types of land use, for reasons of simplification. In principle it is possible to integrate over time the land use quality impacts due to an activity, according to the following expression:

land use = A * (t * Q_{ini} - ∫ Q (t) dt)

In order to do this, the quality at the end of the activity must be assumed to be returned to the situation before the activity (Q_{ini}), otherwise integration can not be performed. This implies assuming that no net land use changes occurs; occupation in [m^2.y] is what is left. There are two suggestions to do this. One is from [Baitz et al, 1998] and assumes a cut-off after human restoration, using t_1 in figure 3 for t. The other is applied in the ETH database [Frischknecht et al, 1994]. They include a (hypothetical) relaxation to the original land use quality and set fixed relaxation times per type of activity. In the latter situation, the total occupation time is expanded to t_2 in figure 3.

Figure 3: Possible integration of land use change and occupation



To assume that net change never occurs does not seem realistic. In the solution by ETH this is compensated by including the extra (hypothetical) relaxation time. We prefer to keep both types of land use separate, in order to remain transparent and exclude subjective choices as much as possible.

2.4 Other inventory information

Land use format

Land use impacts are very locally dependent (to be expressed via Q_{ini} and/or Q_{ref}), and also activity-dependent (expressed via Q_{fin} or Q_{act}). Interventions from different parts of a life cycle are generally added up before characterisation is performed. When local dependency is important, the spatial information should be conserved, thus preventing loss of vital information. In most present databases there is only limited space per intervention to record spatial information. Therefore it seems worth while to use a generic framework for the format of land use inventory data. A suggested format is given in appendix 2 to this main text, including optionally the basic regional data, but also (possibly average) quality indicator scores and the activity typology. Based on this format, land use data have been incorporated in the IVAM ER database.

Related interventions

The impacts of land use are on an ecosystem level. Of course there are many more types of interventions with impacts on this level (acidification, eutrophication, terrestrial ecotoxicity and global warming, for instance). Their characterisation is generally not quantified up to this ecosystem level. But there are some interventions that are not often characterised but which do have a direct link to ecosystem impacts.

One intervention close to area-related land use leads to *fragmentation*. Here, the intersection of surrounding area is the cause of an impact rather than the area of land used. Although the fragmentation impact may be expressed in the same quality terms as land use, the intervention can not. Although the area does contribute to the extent of fragmentation, the length of the intersection is more crucial. The circumference of an activity is suggested by [Blonk et al., 1997] to be a measure for this intervention. The quantitative link to impacts may however be difficult due to data restrictions.

Another related intervention is (ground)water extraction. This intervention, which can be expressed in m^3 extracted water (specifying the source), may also lead to comparable impacts as land use, and may be expressed in the same terms. Like with fragmentation, a quantitative link with the area impacted may be difficult, however.

Finally, the extraction of a biotic resource (such as fish from the sea and berries from a tropical forest) may also have impacts on an ecosystem, when over-exploitation occurs or when the extraction activity uses land in a degrading manner. The last intervention should be recorded as land use (as suggested in [Sas et al., 1996]). The biotic extraction itself and its impacts on the biotic resource (potential depletion of that resource) is a different intervention and should be expressed in terms of individuals extracted in a certain time (see [Sas et al., 1996]).

3. Characterisation of land use impacts

In this chapter the concept of land use impacts is discussed, different approaches for quantifying land use impacts are mentioned and one approach is further elaborated. The quantification of the relationship between the LCA inventory data (land use itself, see the previous chapter) and its impacts is called the *characterisation* of land use impacts in LCA. We will first mention some general requirements for characterisation in LCA.

3.1 Requirements for characterisation in LCA

In [Blonk & Lindeijer, 1995] some requirements were given for the characterisation of land use impact characterisation. The commissioner gave additional requirements, also mentioned in chapter 1. Slightly modified according to comments given, the requirements were:

1. adequate description of the impacts characterised and the link with intervention results
2. linear towards functional unit: doubling the functional unit should mean doubling the impact
3. globally applicable, for all kinds of processes
4. a measure as science based and objective as possible
5. valuation should be able to cope with the characterisation results
6. practically feasible within a short notice

Requirements 1) and 4) are in with CD ISO 14042.3 ([ISO, 1998]). Others are additional ones from own experience and from the commissioner. As will be shown, these requirements indicated the type of operationalisation performed in this study. But first, let us consider which types of impacts may occur during land use.

3.2 Land use impacts

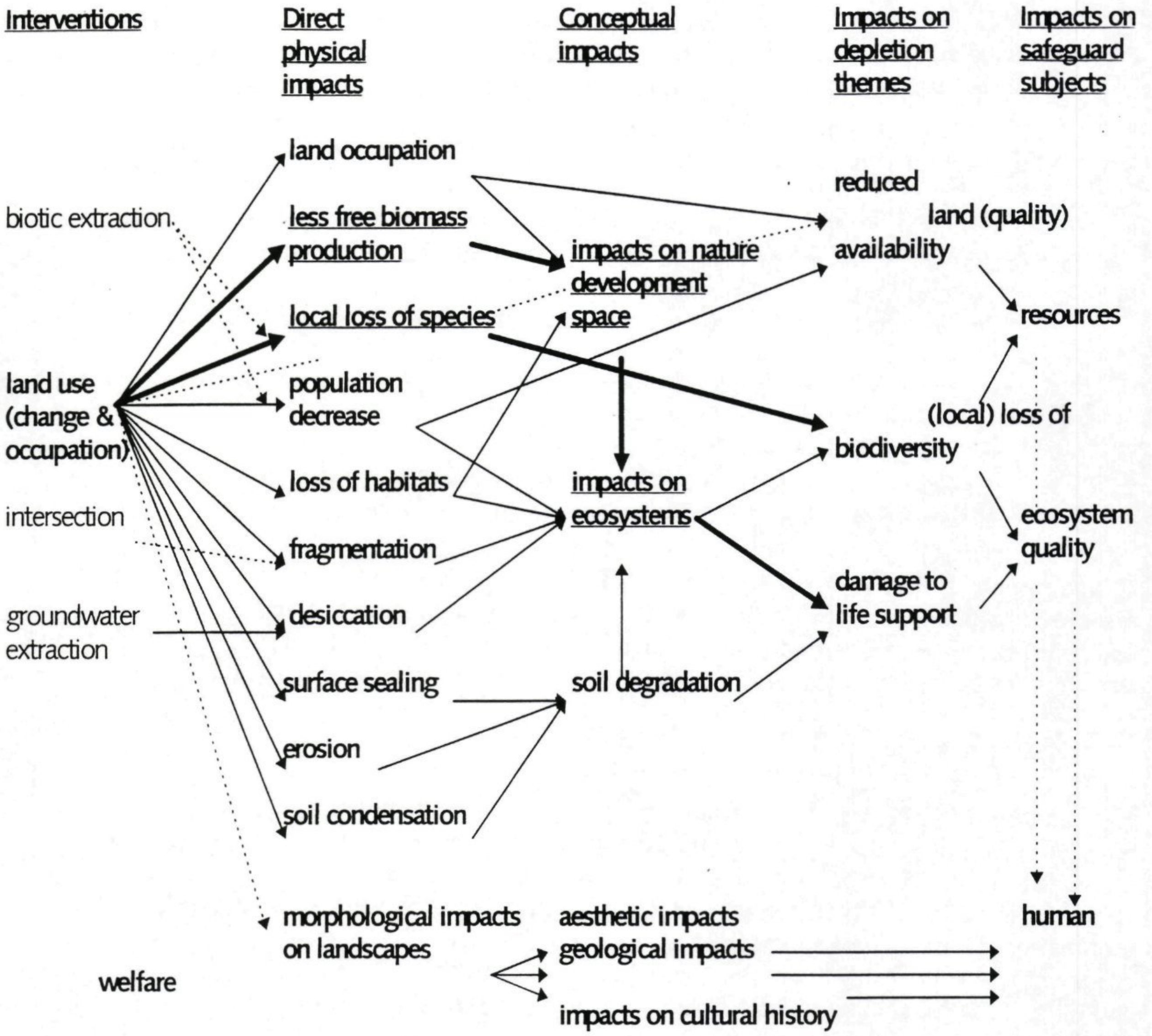
<u>The cause-effect chain and indicators</u>

The impacts of physical interventions in land (land use impacts) on ecosystems can be seen on various levels. The most physical impacts are the occupation of the land by a certain activity itself and possible changes in amount of biomass produced, local loss of species, population changes, erosion etc. These physical impacts lead to changes in functions of nature. On a conceptual level, these functional changes can be grouped under known headings such as 'soil degradation', 'impacts on ecosystems' and 'impacts on nature development space', expressing different value aspects of nature value changes. Note that 'impacts on nature development space' and 'soil degradation' can be seen to contribute to the more general form of 'ecosystem impacts'. Fragmentation and desiccation are different intermediate levels of impacts leading to the same local ecosystem degradation. Landscape impacts are of a very different nature: they relate to human welfare, and hardly to ecosystems as meant above.

Another conceptual level for resources is given by [Heijungs & Guinée, 1997]: that of impacts on the depletion themes 'reduced availability', 'loss of biodiversity' and 'damage to life-support'. These are functional expressions on a different (higher) level again, pointing towards aspects of the LCA Safeguard Subjects Resources and (overall) Ecosystem quality (often addressed with the ill term Ecosystem health).

The above is a rough sketch of the cause-effect chain starting with land use, and is illustrated in the figure below. The figure is not complete nor very precise and has more an illustrative function. The distinction between the various impact levels is not as strict as shown and between these levels of impacts there are more complicated relations to be distinguished. Related interventions are also shown, not in bold. Underlined and connected by thick arrows are the two indicators for land use impacts operationalised in this study. The dotted-lined arrows depict relationships which are not conceptually worked out.

Figure 4 A rough sketch of the cause-effect chain related to land use



At what level should one now operationalise land use impacts? One can focus on the direct physical impacts without considering which indicators make the most important or representative link with higher order impacts. We fear that this so-called bottom-up approach could easily go into too much detail when assessing all physical impacts. On the other hand, the conceptual levels of [Heijungs & Guinée, 1997] would have to be expressed in physical terms to get operationalised. What indicators should one use for that? This last line or reasoning is followed here.

An important term which is often used in this context is loss of *biodiversity*. In essence global biodiversity is a resource, of which the loss is irreversible. As mentioned earlier, such impacts can and should not be assessed with LCA because their relationship with land use is of a clearly non-linear and extreme nature, to be assessed separately. Only the contribution to the potential risk of global biodiversity can be assessed in LCA via loss of local biodiversity (increase of habitat loss). Local biodiversity contributes to global biodiversity, but also contributes to the quality of the natural ecosystem at large (an LCA safeguard subject), through interactions, safety nets and potencies of adaptation. Also, ecosystem and habitat diversity are aspects of biodiversity. As biodiversity plays on different levels and is an important issue in public perception, an indicator should be developed for its local loss. In natural science, various indicators for biodiversity have been developed. The major problem with biodiversity impacts of land use is that when looking closely at critical areas these impacts are not linear with the area occupied or transformed. For LCA, such indicators can therefore only be applied to non-critical areas. It is part of this study to select and operationalise such an indicator.

Land as a *resource* is another often mentioned major issue. We feel that land as such is not a non-depletable flow resource. There is no constant availability of (new) land over time and the total amount of land is fixed (except for minor land claims from the sea as in the Netherlands). However, we acknowledge that the total land surface consists of stocks of different quality which can be transformed into each other by human interference. It is the deliberate choice of man to perform such transformations. These different land qualities can be seen as flows within a pool of limited area. Due to this inherent freedom to move land area from one stock to another, an indicator for reduced availability of each separate quality stock seems senseless. What is gradually but definitely reduced by these human-induced transitions is the area of undisturbed land. An indicator which measures the contribution of an activity to this reduction of relatively undisturbed land would be sensible. In fact, another proposal to deal with land use impacts focuses solely on this aspect [Müller-Wenk, 1998]. We do not go further into the operationalisation of reduced availability of land with distinct qualities, although the scarcity of ecosystems may be incorporated into an indicator for biodiversity, as proposed by Bo Weidema at the SETAC Europe meeting on land use impacts, April 17th 1998. A rough second best indicator for the reduced availability of land would simply be the fraction of land used compared to the total amount of land available. This would be consistent with the way mineral resource depletion is often operationalised in LCA. However, this implies no distinction between different types of land use (see also [Baumann et al., 1992]), so there is only one 'equivalency factor' for all land uses: 1/the earth's surface.

To stay at the same level in the cause-effect chain, an indicator should be developed for the *damage to life support*. When diversity is split off as a separate indicator, life support comes down to the natural substance flows and conversion processes taking place at large. These processes are driven by biota on the one hand and the soil on the other hand. It would therefore be adequate to develop indicators for soil degradation and loss of biomass productivity. For biomass productivity adequate indicators have been developed. Soil degradation may still be caused by different factors. Up till now, no single indicator has been proposed for soil degradation which can readily be used in LCA. The indicator for biomass productivity promoted in [Blonk & Lindeijer, 1995] (fNPP) does give an indication of the soil quality too, as the free available part of the biomass produced is collected in the soil (see also section 3.4 of annex 2).

Within this study, following the cause-effect chain argumentation above, the emphasis is on developing indicators for biodiversity and biomass productivity which can be applied at various levels of detail, and on collecting data for equivalency factors on the most generic level, to be applied for non-specified processes.

It should be emphasised that this argumentation focuses on the ecological values of the land. It therefore excludes functions of land for humans, and it goes beyond a mere aggregation of all possible parameters to express the ecological quality in physical terms. This is called a bottom-up approach. In the top-down approach followed here, choices have to be made on which indicators express most adequately the most important aspects of ecological quality.

3.3 Proposed approaches for operationalisation

A review has been made of all existing methods to deal with land use in LCA. This review is included as appendix 1. Concluding, one can state that there are three major approaches to deal with land use effects:

1. functional aspects
2. land use classes
3. single indicators

The functional approach is operationalised via many physical impact indicators (on the left side of the cause-effect chain in section 3.2) and all indicators may be weighted equally, optionally by a panel. This is a bottom up approach, making no link to the end-points. Such a link is required by ISO CD 14042.3 [ISO, 1998].

The land use classes approach is designed from a top-down view, incorporating a sense of ecosystem quality in the classification of land uses. The concept of 'naturalness'; Hemerobiestufen is used for this. However, without weighting of the classes the LCA results may be insufficient for decision support. Also, a further subdivision of 'naturalness' for specific economic activities (forestry or agriculture) is difficult as naturalness is not clearly defined.

All existing indicator approaches perform a weighting of the above existing land use classes. The results for 5 coarse land use classes are repeated in table 1 below:

Table 1: Formerly proposed weighting systems for 5 land use classes (Hemerobiestufen)

Land use class (Hemerobie-stufe)	Biological accumulation, based on [Whittaker & Likens, 1973]	Regeneration time, mainly based on [Hampicke, 1991]	Panel value, based on [Jarass et al., 1989]	Multi criteria of [Auhagen, 1994]	Diversity & Red Lists Öko-indicator [Felten & Glod, 1995]
A Naturelike, modified systems	1	1	1	1	1
B Thinned forests and extensive cultivation	1	0.17	0.84	0.35	0.85
C Intensive agriculture	0.1	0.0047	0.52	0.18	0.49
D Partly built areas	0.05	0.0004	0.29	0.06	0.15
E Sealed surfaces	0	0	0	0	0

The biological accumulation indicator gives the most coarse indication of land use effects. The panel and the ecological (Öko-)indicator give similar results, although the first is based on social preferences of land use (functions) and the latter is based on mixing biodiversity and scarcity criteria. The indicator based on regeneration times gives the highest relative value to naturelike systems, and the multicriteria indicator gives an intermediate result.

These indicator systems can not be detailed further as data is only available for the distinct land use classes. The data used is often very regional: the panel was performed in southern Germany, the multi-criteria approach was developed for the region of Berlin and the data for the diversity indicator also came from that region.

3.4 Indicator measures for biodiversity and life support impacts

The indicator approaches based on the above land use classes was considered inadequate in the present study, as presently collected scientific data is generally not collected and stored according to these classes, and ecosystem types are not sufficiently detailed to allow for various levels of detail in the assessment. Instead, we propose a new classification based on physical information. According to ecologists the quality of land use is considered to be largely dependent on abiotic (physical) factors. These factors are not to be confused with the physical impact indicators used in the functional approach.

The main abiotic factors determining land use quality on a global scale are geographical latitude, height above/below sea level and rainfall (for tropical rain forests). Each combination of (ranges of) values for these factors determine largely the occurrence of a certain type of ecosystem. Areas with such combinations of abiotic factors are called *physiotopes*. In this study, biodiversity and biomass productivity indicators are operationalised per physiotope. In annex 1 the operationalisation of physiotope scores for each measure is described.

The physiotopes can be detailed for any case, but it is sufficient to determine the indicator scores relevant for the case, either in relation to the global physiotope the activity takes place in or (when data are available) in relation to specific physiotope data expressed in those indicators. We thus developed a set of default physiotope data for application in generic LCA situations (see section 3.6).

Biodiversity, α

As stated in section 3.2 only the local loss of biodiversity can be assessed in LCA. This can give an indication of the potential threat to global biodiversity, however. Biodiversity is here used as a measure of the intrinsic value of the ecosystem at stake. Data limitations on a global scale have strongly determined the choice of a biodiversity measure. According to IBN-DLO (see annex 1) the only measure for which enough data is available on all physiotopes in the world is vascular plant species diversity. This measure (called α, alpha) is expressed as the number of plant species per m^2. Formally, α represents the increase of number of species when the area is increased by a factor 10, but as argued in annex 1 this value is also an adequate indicator for the number of species per m^2 and thereby for biodiversity.

Free net primary biomass productivity, fNPP

fNPP was proposed in [Blonk & Lindeijer, 1995] as an indicator for the nature development space, or as biomass experts from IBN-DLO agree, for the potential of nature development. Here not the intrinsic value of the ecosystem is measured, but the quantitative amount of biomass, from which any kind of nature value may be developed (on the long term) by its own metabolism, expressed in amount of biomass free for development of higher species. This contribution to local and global metabolism cycles is an important aspect of life support (the closing of natural life cycles and the buffering capacity needed for a stable functioning of ecosystems).

Indicators at sea

Unfortunately, these concepts are not readily applicable to changes and occupation of the seabed. An early survey for this has been performed by IBN-DLO Texel (see annex 2). Plant diversity is not a sufficient measure for the seabed, and for other diversity indicators insufficient data are available presently. At least two additional parameters should be used: age distribution within species and the species composition. Due to the high dynamics of the ecosystems and abiotic factors, it is necessary to include the regeneration time for the biodiversity for land use changes. This would be the time necessary for achieving the same level of biodiversity at the location of the activity as at a few randomly chosen equally large areas (reference areas) within the same ecotope. The same level of biodiversity can be specified as an equal variance and median of the measures species diversity, age distribution and species composition. This level may be very different from the situation before the activity started. However, no operational indicator can be derived at present due to lack of data.

Biomass data are only available for isolated and very specific benthic (seabed) ecosystems. Incidentally, as long as an activity is not chronic or not occupying a large portion of the total homogeneous area, no relevant life support impacts are expected due to the large dynamics of ecosystems at sea. The biomass of benthic systems are a factor 10 to 100 lower than at land. See for more details annex 2.

3.5 Quantification of equivalency factors

In practice, the measures α and fNPP do not as such form the indicators for land use impacts. Rather, α and fNPP are made part of formulas to express the biodiversity and life support impacts of land use, respectively. In these formulas, the elements ecosystem value measure, change or occupation and a reference state should be considered.

All formulas have to be linked to an area (in m^2) to really imply land use. Net changes are without time dimension whereas steady state land use implies occupation during a certain time, suggesting a multiplication of area and time.

The reference state is at least necessary for land occupation to include the spatial aspects. But also for land use changes it is important to relate the initial situation to what is the most natural situation (see below). For this reference, the present most 'natural' situation in the region where the land use takes place is taken. In [Blonk & Lindeijer, 1995] the theoretical, would be or undisturbed situation was preferred as a reference, but there appears to be too much discussion on what the undisturbed situation would be, and data can not be collected world-wide consistently for this approach. The reference situation is thus the presently most nature-like land use quality in the area where the activity takes place. This does introduce a bias, as European countries have large areas where ecosystems are structurally less developed. By taking a large enough area (physiotope) to determine the reference values, the most valuable areas are taken as a reference. Whenever more local reference states would be chosen, the problem of international compatibility must be dealt with.

Then, what is the place of the reference situation in the formula? It can be used to express the actual state (Q_{act}) as a fraction of the reference Q_{ref}, Q_{act}/Q_{ref}, or to express an absolute distance between actual and reference situation (Q_{ref} - Q_{act}), proposed in [Blonk & Lindeijer, 1995]). For biodiversity, an indicator showing absolute species density numbers seems inappropriate since not only species diversity but also ecosystem diversity is part of biodiversity. This implies that ecosystems with a low species diversity may be considered just as valuable as one with a large species diversity. Also in (inter)national policy the diversity between high biodiversity and low biodiversity ecosystems is more appreciated. A relative expression is thus more appropriate. Q_{act}/Q_{ref} results in positive scores whereas all other present LCA impact scores have a negative content. $1 - Q_{act}/Q_{ref}$, or $(Q_{ref} - Q_{act})/Q_{ref}$ is therefore a more consistent expression for land occupation biodiversity equivalency factors. For life support, not the intrinsic value of individual ecosystems is at stake, but the physical contribution of biomass to substance cycling and soil structure. Therefore, an absolute measure is more appropriate for life support: $Q_{ref} - Q_{act}$ or $Q_{fin} - Q_{ini}$.

For land use change the situation after the activity is related to the situation before the activity instead of to the reference situation. This would lead to $(Q_{fin}-Q_{ini})/Q_{ini}$. However, it is argued that the relativity argument only applies to reference (natural) ecosystems, since policy statements on conservation relate to the most nature-like ecosystems. Dividing by Q_{ini} is therefore not appropriate, also because a relative change in non-natural ecosystems such as intense agriculture is considered less problematic than a same relative change in a natural ecosystem. By dividing by Q_{ref} this problem is solved. This leads to the expression $(Q_{fin}-Q_{ini})/Q_{ref}$ for equivalency factors related to land use change. Note that the formula for land use change and then one for land occupation now use the same reference value for the indicator!

We must thus allow different formulas for land occupation and land use changes. The above results in the following formulas (ini = initial state, fin = final state, act = actual state, A is area in m^2, t is occupation time in years, α is the biodiversity measure and fNPP is the life support measure):

ECOSYSTEM CHANGE (EC):

$EC = A \cdot (fNPP_{ini}-fNPP_{fin})$ as a measure for life support functionality and
$EC = A \cdot (\alpha_{ini}-\alpha_{fin})/\alpha_{ref}$ for local loss of biodiversity due to changes

ECOSYSTEM OCCUPATION (EO):

$EO = A \cdot t \cdot (fNPP_{ref}-fNPP_{act})$ as a measure for life support functionality and
$EO = A \cdot t \cdot (\alpha_{ref}-\alpha_{act})/\alpha_{ref}$ for local loss of biodiversity due to occupation

Note that the free NPP in natural (reference) situations is equal to its NPP, as no biomass is extracted for human consumption there.

The above results in the following illustrative judgements based on the same area and time of occupation.

Starting from a natural situation, changing from 100 to 20 species per m^2 in tropical rainforests is considered equal to changing from 10 to 2 species per m^2 in boreal forests. A change of 8 species per m^2 in tropical forest will cause much less impact than a change of 8 species/m^2 in boreal forest. A same relative change as above due to industry in former intensive agriculture land (say, from 5 to 1 species/m^2 in boreal area) yields a lower impact than coming from a natural situation. For biomass impacts a change from 10 to 2 Mg free biomass/ha.y in tropical forests is already considered worse than a change from 5 to 1 Mg/ha.y in boreal forests (in absolute terms more contribution to life support is lost in tropical forests - 8 > 4 in this example -). When an activity first reduces the diversity from 10 to 1 species/m^2 and finally restores a situation with 3 species/m^2, this scores better on change than going from 10 to 2 species/m^2 without restoration (the score on occupation is of course worse).

For occupation, using the same data would yield the same results, except that the situation before the activity is not taken into account. The occupation due to a factory in the tropics will always yield the same score, irrespective of the situation before (as that is to be considered under change).

3.6 Link to valuation and decision support

It depends on the valuation system to be used, how the above indicators for life support and local biodiversity should be incorporated, provided weighting is desired. We have the feeling that the indicators can in principle be included in all present valuation approaches, although the weight of land use effects need to be determined for each approach separately. This would mean weighting life support against biodiversity. This could lead to one score for the safeguard subject Ecosystem Quality. See also [Swan, 1998], using the term bioproductivity for life support and bioquality for Ecosystem Quality. Other impacts (for instance ecotoxicity, smog, acidification) will also contribute to Ecosystem Quality (loss), and all should be expressed in the same units if an ultimate aggregation is required. This units problem is a serious one, causing practical problems as long as the operationalisation of all impacts is not pursued in the same project or normalisation can not be performed for all relevant impact categories. This weighting will require value statements on targets, costs or preferences for each impact. Apart from the problem of double-counting and endpoints at different levels of the cause-effect chain (global warming contributes also to life support and biodiversity impacts, for instance), this is in principle possible. We will not go further into detail on this, as in fact not one complete and generally acknowledged weighting approach on this level is available.

Normalisation is possible by dividing by the amount of land use effects induced by a certain region in a certain time period. Presenting normalised impact scores in an appropriate manner already contributes to a better decision support than merely showing impact scores.

3.7 Acceptance of the approach operationalised here

It is difficult to access whether a proposed characterisation approach is acceptable. SETAC-Europe is in a process to design and implement an acceptance procedure [Udo de Haes & Jolliet, 1998]. In attendance for this procedure, comments from the guiding committee, sending draft reports to interested parties, giving presentations and workshops seem a first step towards getting feedback on the acceptance of such an approach. All these actions have been undertaken, culminating in a workshop at the SETAC-Europe Annual conference in Bordeaux, April 1998. The report of this workshop included a survey with the invitation to make statements on the points raised (see appendix 2). 20% of the participants have reacted to this invitation.

The workshop was linked to a presentation on the subject the day before, and appeared to have partly an informative function, as a number of participants had not been intensely involved in the subject before. But also valuable feedback was given on the approach followed:

- It was clear that the intervention framework (land occupation versus land use change and their units for expression) was agreed upon, also from the survey
- According to the reaction on the survey, and no objections during presentations, the three conceptual depletion themes availability, biodiversity and life support were accepted.
- Vascular plant diversity seems a generally accepted measure for biodiversity in general, under the present data restrictions
- There seems to be reluctance to accept the free net primary production as a measure for life support. The biomass indicator was initially seen as another measure for the intrinsic value of ecosystems, and as such it would not give an adequate characterisation for impacts on oligotrophic ecosystems (with low intrinsic biomass productivity). But even after it was made clear that not an intrinsic value but rather life support is measured with fNPP, some reluctance remained. This may also be due to the fact that this measure is rather uncommon with respect to for instance the general biomass productivity indicators. Also the still vague concept of life support contributes to this reluctance. Standing biomass (NPP) was suggested as an alternative measure [Udo de Haes, 1998], more in line with the suggestion from a research on biotic extractions to use NPP_{ini} x biomass regeneration time as an indicator [Sas et al., 1996]. Following these suggestions, the latter indicator is also applied to the cases (see chapter 4).
- A suggestion was given by Bo Weidema to add ecosystem scarcity to the biodiversity indicator, to incorporate rareness.
- Another suggestion was to include the recovery time for biodiversity, or when a worst case assessment would be performed, the geological recovery time (especially for extractions of minerals).

More or less the same reactions were given during a presentation for the Danish LCA community and the Danish EPA on May 26th 1998, confirming some acceptance of the basic ideas presented here.

We participated in two EU Concerted Actions considering land use in LCA (COST E1 and E9). A small scale presentation for E1 (LCA for paper products) in October 1997 was favourably received. At E9 meetings (Impact Assessment for LCA of forestry) there is some tendency to develop more elaborated approaches: more aspects, like the functional approach. Inclusion of the indicators operationalised here can become point of discussion in the future.

3.8 Reference data on vascular plant diversity and free biomass productivity

IBN-DLO has performed an extensive literature search to collect generic plant diversity and biomass productivity data for physiotopes all over the world. The results of this operationalisation are reported in annex 2. The main results for the biodiversity indicator α (number of species per m^2) are repeated in the table below. The α values are derived for the most nature-like ecosystems per physiotope, resulting in present-day 'background' data for plant diversity.

Table 2: Estimates of α for physiotopes (in vascular plant species per m^2). Bold: figure supported by relatively large amount of literature data. Normal: tentative figure.

Latitude

80	<5	0	0
60	10	15	0
40	15	25	0-10
30	10-40	15	10
20	50-75	25	15
0	100	35	20
	0 - 1000	1000-3000	>3000

Altitude

Additionally, a world map of α has been produced, as α is also precipitation- and continent-dependent (see appendix 4 to annex 2). As such, the world map is the best practical reference for determining land use biodiversity impact scores.

Similar results are given for the reference situation measure for life support, NPP. See table 3. Also for NPP a world map is produced, giving a practical guide to biomass reference states world-wide (see appendix 5 to annex 2).

Table 3: Estimates of NPP for physiotopes (in Mg/ha.y; n/a = not applicable)

Latitude

80	<=1	n/a	n/a
60	8	3	0
40	12	8	2
30	<=1	<=1	<=1
20	16	8	3
0	22	12	4
	0 - 1000	1000-3000	>3000

Altitude

Both above tables are only to be used when the maps give inadequate results, for instance because a more specific situation (i.e. a mountain valley) needs to be considered.

These reference data can be applied in the formulas of section 3.5 to make the link between land use and their impacts, including the relation with the reference situation relevant to the case at stake.

For the reference α in Europe the uncertainty, as based on appendix 2 to annex 1 is about 100%; for the reference α in tropical forests the uncertainty is about 50%, based on paragraph 2.4.2 in annex 1. For the reference NPP an indication of the range is given in table 4 of annex 1, resulting in an approximate uncertainty of about 50%. The uncertainty in the global reference data is thus large, due to natural variation. This is not considered problematic when one decides on using the same reference figures for all studies. For forestry and agriculture the natural variation will be much smaller. This will reduce the overall uncertainty to acceptable levels. For roads including verges, wasteland and recreational areas the variation may again be large due to the local situation.

3.9 Related hard-to-get impacts (landscape, desiccation and fragmentation)

In relation to land use impacts, landscape impacts, fragmentation and desiccation are often mentioned or considered consecutively. These impacts were chosen not to be main subjects of this study. Nevertheless, a short comment on the possibilities to operationalise these impacts is given.

Landscape impacts

Aesthetic values relate to another safeguard subject (human health or human welfare) than the one considered here (ecosystem quality) and no general operationalisation method for LCA has been proposed yet. The indicators proposed by [Knoepfel, 1995] (top-height and volume) were the only quantitative landscape indicators developed for LCA this far, but done so with the case of electricity poles as a basis. These indicators were not considered generally valid ones for landscape value [Klijn, 1997].

Apart from ecological impacts on ecosystems landscapes harbour a range of values that belong to the domain of cultural history (archeology, historical geography, historical buildings), earth-sciences (geological, geomorphological and pedological features) and visual aspects (often related to aesthetics) For a reference see [Natuurbeleidsplan, 1990] and [Dijkstra & Klijn, 1992]. So far no generally accepted methods for international or global applications have been developed, although on a national level these exist and are used for e.g. E.I.A. (Environmental Impact Assessments), as is the case within The Netherlands (e.g. [Dijkstra, 1992]).

Approaches that really try to include all kinds of landscape aspects all over the world dealing with various landscapes and landscape values are not yet operational. Approaches to use ecosystem indicators (biomass or other aspects) to serve as indicators for landscape appreciation or recreation values cannot stand fundamental criticism. The only feasible approach is to assess all relevant kinds of impacts within a limited region, where i) data are available and ii) where a minimal consensus exists of values at stake, such as within national boundaries or between countries with comparable classification and valuation methodologies.

The above stresses the fact that further conceptual and methodological progress is urgently needed .

Fragmentation

Fragmentation of ecosystems due to intersection of an area of land is an intermediate impact, causing higher order ecosystem and/or specific fauna impacts. As the most clear impact is on fauna mobility, it can be doubted whether the plant diversity indicator developed here should be used as an indicator for fragmentation. More likely an additional factor should be developed to express fragmentation impacts. This indicator might be linked to the circumference of the area causing the impact (i.e. a road or railway), as suggested by [Blonk et al., 1997]. When collecting land use data, additional information on the circumference of these per m^2 can be given.

Desiccation
The potential impacts of extracting groundwater are most similar to the impacts of land use. In fact, they could be expressed by the same indicators for biodiversity and life support. However, the impacts are not related to the area of land occupied. They have a linear relationship with the amount of groundwater extracted [Beugelink et al., 1992]. In order to operationalise the impacts of groundwater extraction for LCA purposes, a model should be applied to determine where groundwater extraction causes which amount of impact (this is done in the above study of Beugelink et al.) and then express this impact in the required LCA indicator scores. A first very crude attempt has been made in [Van Tilburg, 1997] where the impacts were expressed in Eco Indicator '95 scores.

3.10 Conclusions on the operationalisation of land use impacts

The following conclusions can be drawn from the above sections:

- Two of the three depletion themes in LCA (biodiversity and life support) have been operationalised on a world scale level. This resulted in a complete set of land use impact data for these themes. For the third theme (availability) a very simple operationalisation based on the amount of m^2.y land occupied is readily applicable.
- The reference data gathered contains a rather large uncertainty range. This uncertainty is inherent in the variability of nature, and can only be dealt with by agreeing on one basic set of reference values.
- Vascular plant diversity seems a generally accepted measure for biodiversity in general, under the present data restrictions
- There seems to be some reluctance to accept the free net primary production as a measure for life support. This may also be due to the fact that this measure is rather uncommon with respect to for instance the general biomass productivity indicators. But also the still vague concept of life support will contribute to this reluctance.
- The derivation of the formulas to express indicators based on the measures α and fNPP is not a trivial process. The arguments 'relative, not absolute expressions', 'include a reference to incorporate spatial information' and 'uniformity of the expressions' have largely determined their form. The result is for biodiversity a relative and thus dimensionless score and for life support an absolute measure with the dimension [Mg/ha.y], or [kg/ m^2.y].
- Some other impacts (soil degradation and desiccation) can probably be captured under the same indicator scores. Other related impacts (on aesthetic values and due to fragmentation) need other indicators for their operationalisation.

4. Application to cases

4.1 Assessment of various land use cases

For a number of land use change and occupation cases, values have been determined for α and fNPP (see annex 1). For the biodiversity measure, a Dutch GIS map has been used to determine average reduction factors for each type of activity, relative to the natural background situation. It was assumed that these reduction factors are applicable in whole Europe. These were combined with the reference data of section 3.6. For the life support measure, case-specific literature has been used to determine fNPP, which was linked to NPP data from section 3.6 for the reference situation. Both sets of data are expert guesses at present. Collecting this data on site is another realistic option for crucial cases. The resulting values are given in tables 4a and 4b for changes and occupation, respectively. NPP values are in Mg/ha.y.

Table 4a: α and (f)NPP values for some cases of land use changes

Testcases	alpha (ref.)	alpha (before)	alpha (after)	NPP (ref.)	NPP (before)	NPP (after)
Extracting sand in Europ. Agricultural land	10-15	5	0-8 [1])	8	8	0-8 [1])
Mining aluminium ore in S.-Am. Tropical forest [2])	100	100	80 [2])	17	17	17 [2])
Landfill household waste in European agricultural land	10-15	5	9 [2])	8	8	10 [2])

[1]) recreation ground assumed after occupation; values strongly dependent on local situation (depth of sand pit, steepness of shore)

[2]) secondary forest assumed after occupation

Table 4b: α and (f)NPP values for some cases of land use occupation

Testcases	alpha (ref.)	alpha (occu-pation)	NPP (ref.)	NPP (occu-pation)
Extracting sand in Europ. Agricultural land	10-15	0	8	0
Industrial production in Europ. Agricultural land	10-15	0-10 [2])	8	1
Hydropower in Scandinavian hills	10	0	7	0
Road traffic in Europ. agricultural land	10-15	5	8	2
Mining aluminium ore in S.-Am. Tropical forest [1])	100	0	17	0
Harvesting wood in Scandinavian hills [1])	10	7	7	7
Landfill household waste in Europ. Agricultural land	10-15	0	8	0

[1]) 'pristine' situation assumed before change or as a reference for occupation

[2]) depending strongly on the management of unused unsealed areas

The formulas for occupation from section 3.5 have been used to combine land use data from other literature sources (in this case the IVAM ER database, based on a literature survey published in [Mak et al., 1996]) on above cases with the above data from table 4b to render land use occupation scores for biodiversity and life support impacts. These were compared to impact scores resulting from some formerly proposed land use indicator operationalisations (see table 1, section 3.3 for 'Öko' and 'regeneration time'; to use NPP * biomass regeneration time is a suggestion by [Sas et al., 1996] and table 5 in annex 1 for the biomass regeneration times. The following table is the result:

Table 5: Results of applying various indicators to a number of land occupation situations

cases	1t sand extraction in agriculture Europe	1t bauxite mining in tropical forest	1t pine harvesting in Swedish montane forest	1 MJ hydro power in Swedish montane forest	1 tkm transport in agriculture Europe (incl. verge)	1 t household waste in agriculture Europe
A x t [m^2.y]	0,16-0,18	0,09-1,14	3964-8371	0,001	0,004-0,021	0,002-0,003
land use class (Qact)	D	E	B	D	D	E
ref. land class (Qref)	C	A	A	A	C	C
α_{act} ¦ $fNPP_{act}$ [Mg/ha.y]	0 ¦ 0	0 ¦ 0	7 ¦ 7	0 ¦ 0	5 ¦ 2	0 ¦ 0
ref. physiotope	agriculture	tropical forest	montane plantation forest	montane forest	agriculture	agriculture
α_{ref} ¦ NPP_{ref} [Mg/ha.y]	12 ¦ 8	100 ¦ 17	10 ¦ 7	10 ¦ 7	12 ¦ 8	12 ¦ 8
biomass regen. time [y]	90	60	110	110	90	90
(NPP_{ref} -$fNPP_{act}$) [Mg/ha.y]	**8**	**17**	**0**	**7**	**6**	**8**
(α_{ref} - α_{act})/α_{ref}	**1**	**1**	**0,3**	**1**	**0,6**	**1**
1-Öko	**0,85**	**1**	**0,15**	**0,85**	**0,85**	**1**
1-regeneration time	**0,9996**	**1**	**0,83**	**0,9996**	**0,9996**	**1**
NPP* biomass regen. Time [kg/m^2]	**72**	**102**	**77**	**77**	**72**	**72**
A*t*(Qref-Qact), Q = fNPP	1,3-1,4	1,5-19	0	0,007	0,030-0,16 1)	0,016-0,024
A*t*(Qref-Qact)/Qref, Q = α	0,16-0,18	0,09-1,14	1189-2511	0,001	0,004-0,019 1)	0,002-0,003
A*t*Q, Q = 1-Öko	0,14-0,15	0,09-1,14	595-1256	0,0009	0,003-0,018	0,002-0,003
A*t*Q, Q = 1-regeneration time	0,16-0,18	0,09-1,14	3290-6948	0,001	0,004-0,021	0,002-0,003
A*t*Q, Q = NPP* biomass regen. time	12-13	9,2-117	(31-64)E4	0,077	0,29-1,5	0,14-0,22

1) Assuming 20% verge and 80% sealed surface

Uncertainty and conclusions

In table 5 only the uncertainty due to the intervention data is presented. The basic intervention data for the cases is presented in appendix 3. The uncertainty in the indicator scores is not given. For the reference α in Europe the uncertainty, as based on appendix 2 to annex 1 is about 100%; for the reference α in tropical forests the uncertainty is about 50%, based on paragraph 2.4.2 in annex 1. For the reference NPP an indication of the range is given in table 4 of annex 1, resulting in an approximate uncertainty of about 50%. The uncertainty in the global reference data is thus large, due to natural variation. This is not considered problematic when one decides on using the same reference figures for all studies. It was therefore suggested to just decide upon one fixed set of reference α and NPP values in annex 1. To be consistent with this suggestion, the value 12 is used for α_{ref} in European agriculture land.

For the actual state the uncertainties are important to consider, as different land occupation situations are compared to each other based on these data. We presume these are lower than the data on the reference states because the natural variation will be smaller. Also, the scores are based on reduction factors combined with a closer scrutiny of the GIS data, thus using more detailed information. Uncertainties of about 25% should nevertheless be considered. The biodiversity indicator scores would then range maximally from 0,2 to 0,4 for pine harvesting and from 0,5 to 0,7 for the verge of road transport. The life support indicator would range from -2 to 2 for pine harvesting and from 5 to 7 for road transport (the verge). Assuming at least the same uncertainty for the other indicators applied in table 5, the two indicators developed here show the most significant difference in scores for most cases. This uncertainty can be further reduced when more detailed information is collected, which is not possible with most other indicators.

Thus, when focusing on the impact indicator results themselves (the fat scores in table 5), one can conclude that the two indicators developed in this study show the largest sensitivity to different cases, followed by the Öko indicator of Felten & Glod. The difference between scores is largest for the life support indicator, which uses an absolute scoring system. As in the case of forestry in Sweden the reference biomass productivity is estimated to be the same as in the natural situation, the impact turns out to be zero there. The other biomass indicator NPP * biomass regeneration time shows little sensitivity as neither the reference situation nor the biomass regeneration time show large variations[1].

Still, the differences in scores is generally much lower than the variation in land use in the different cases. As can be seen from the total impact scores including the amount of land used, the large overall differences come from the variation in land use rather than from the variation of the impact scores, even including the uncertainty therein.

The gain of using the indicators developed here is not only their higher sensitivity, but rather their basis in scientific data, the explicit link with relevant endpoints and the possibility to adapt the data to more specific situations.

4.2 Data assessment for resource extraction

In order to be able to use the generic data derived in this study more easily, for resource extractions a literature search has been performed on the location of these extractions worldwide. These are combined with the maps of α and NPP (appendix 4 and 5 in annex 2). The results for α are given in appendix 4 to this main text. These data have again been summarised into table 6.

The above data will facilitate the application of the data generated in this study, as for every LCA a vast amount of different resources is required, for which default impact scores can now be calculated.

Note that when the source of the resource is known more specific, deviating scores for α and fNPP can be found from appendix 4 of the main text or directly from appendix 4 and 5 of annex 2. For instance, the α value for hard wood for the Netherlands is 150 and that for pine wood is 10. Also for extracting gravel, sand and other bulk materials more specific data should be gathered. When assessing specific cases, also data for land use changes can be gathered with acceptable accuracy. More complete data for occupation of a large set of process types is incorporated in a recent update of the IVAM ER database.

1 *The variation in this indicator is actually much larger in the original publication [Sas et al., 1996], as their definition of biomass regeneration time is based on the ratio between biomass in biota and that in the soil, whereas our data are expert guesses of actual biomass regeneration times.*

Table 6 Data on reference biodiversity and life support scores for resource extraction cases

Global Resource Extraction	Average α-value	Average NPP-value [Mg/ha.y]
Energy		
Coal mining	15	8
Oil extraction	11	2
Gas extraction	15	7
Wood		
Forestry of hard wood	60	22
Forestry of pine	19	8
Metals		
Bauxite mining	44	10
Cadmium ore mining	26	10
Chromium ore mining	27	9
Cobalt ore mining	32	10
Copper ore mining	24	5
Iron ore mining	28	11
Ilmenite mining	50	10
Lead ore mining	26	8
Lithium ore mining	25	6
Manganese ore mining	37	11
Mercury mining	16	6
Nickel ore mining	10	8
Platinum group ore mining	29	7
Silver ore mining	55	13
Tin ore mining	62	14
Uranium ore mining	17	8
Zinc ore mining	20	9

4.3 Allocation and the functional approach

In attributing land use scores to cases, the problem of multi-functionality can occur: the land may fulfil different functions. The land use scores should ideally be divided (allocated) between these different functions (outputs), when only one of these functions are considered in a study. In the example of a national road network carrying trucks and cars, the land occupation for existing roads should be allocated between the cars and the trucks driving it, based on their respective performances. Incidental land use changes due to expansion of the road network should also be allocated to these two items, based on their relative contribution to this expansion. This example is illustrated in box II below.

Other examples of such multi-output allocation are other economic outputs during forestry, like berries, pharmaceutics and thinning wood, or during agriculture (manure, straw, hides etc.). Multi-output allocation should be applied not only to the land use, but also to the other interventions related to an activity.

For land use the issue of multi-functionality needs special consideration, because in the case of forestry and agriculture it is generally *not the process* which performs different *natural* functions such as erosion resistance, groundwater protection and water buffer capacity. Rather, the process may allow naturally occurring functions to be performed by the land. If the economic process would not have been there, these functions would have occurred anyhow. In this case, the intervention land use is not to be allocated to different *economic* functions, but the possibility to perform these *natural* functions should ideally be included in the assessment of the quality of the land occupied. This is explicitly done in the functional approach of [Baitz et al., 1998] (see also appendix 1, under 1). The habitat resource function mentioned there could be expressed in terms of biodiversity as operationalised here. In that approach the recreational value is not included; we propose that the extent to which that function can still be performed should be operationalised within an approach to assess the landscape value of an area.

Box II Multi-output allocation for roads

According to statistics, there were 1,096 km^2 paved roads in the Netherlands in 1993 [CBS, 1997]. These roads are used by personal cars, vans, trucks etc. Considering only the major activities, 80^{e}9 car-kilometres have been driven in 1990, and 13^{e}9 km cargo-kilometres, whereas in that period 35,3^{e}12 kmkg cargo has been transported [CBS, 1992]. Now how should the land occupation be divided between these two activities? Some of the possible options are: the amount of vehicle-kilometres, the net weight carried across the country in kmkg (reducing persons to their average weight of 80 kg and 2,25 persons per car), the economic value of the activities and the length of the vehicles. Lacking data on the economic value and taking the two options vehicles-kilometres (14% allocated to trucks) and transport-performance (72% to trucks), the following range in land occupation for truck transport arises: 138 - 709 m^2.s/kmkg (4,4 - 23 e-6 m^2.y/kmkg). When corrected for the extra area trucks and vans consume compared to cars (assuming an average 3 times a car leads to 305 m^2.s/kmkg or 9,7 e-6 m^2.y/kmkg) the amount of vehicles allocation turns out somewhere in between this range. Thus, for land use, the range due to such allocation problems can easily be large.

Next to the land occupation, there are also land use changes occurring for roads: new roads are being built, mainly in former agriculture ground in the Netherlands. The area of paved roads has increased from 98^{e}7 via 107^{e}7 to 110^{e}7 m^2 between 1983 via 1989 and 1993 [CBS, 1997]. Of this 12^{e}7 m^2 change, 8^{e}7 m^2 arise from paving unpaved roads in agriculture area and most of the other 4^{e}7 m^2 is from changing main roads within cities from another statistical category. Most changes are therefore consistent with the case in table 5. How should these land use changes be allocated between the trucks and the cars? Generally, higher needs for attainableness and relief of urban centres is the argument for the road network expansion [Schouten, 1998]. Indicators for the stress leading to this need are traffic jams, accidents and other traffic impact complaints in urban areas, and efficiency complaints from industry and the service sector. Considering the cause of the above change from agriculture ground via unpaved roads to paved roads, this has been a slow process of rural development. Ideally, specific data on increased traffic intensity in those agricultural areas should be used to allocate the land use change to the various vehicles [Schouten, 1998]. This data is however not available. The change of 8^{e}6 m^2 per year can therefore only be allocated to truck transport in the same way as above, using the national mix of vehicle-area-performance (31% to trucks): 7^{e}-8 m^2.y or 2,2 m^2.s per kmkg truck transport. Again, the same range can be put around this score. Note that it is assumed that this change occurs every year again. Finally, also other interventions due to the paving of the road should be allocated.

Concluding, it is necessary to perform multi-output allocation when the land is used for more economic functions. This may lead to large ranges when no single allocation factor can be decided on. Although single land use changes can generally not be allocated to a single process (as the amount of function units profiting from this change is unknown), the yearly change of land use due to these functions can be allocated among the functions.

Concluding, we state that *natural* functions performed by an area should not be credited to an economic activity such as forestry or agriculture. In contrast to multiple use of the land (as with roads) their allotted land use should not be reduced for this. Rather, the possibility for these functions to be still performed during the activity should ideally be included in their land quality assessment. In the approach operationalised here this is not the case.

4.4 Conclusions from the case studies and data collection for future cases

From the previous paragraphs, some general conclusions can be drawn:

- A major conclusion from the above cases is that irrespective of the indicator chosen, the land area used for renewable resources will dominate the land use impact scores, compared to mining or other human activities (assuming that the outputs per functional units are of comparable orders of magnitude). Natural functions simultaneously occurring in the same area can not be used to allocate part of this land use to other than the main functions.
- It depends heavily on the reference situation taken what indicator score will appear. Only for biodiversity impacts due to occupation the reference situation is irrelevant when biodiversity is zero during the occupation. As the score for biodiversity is a relative score, the range is smaller than for other indicators.
- The results from the biodiversity and life support indicators are comparable to results using other indicator systems, although the life support indicator shows the largest sensitivity (no relative score is used).

- The land use quality data collected in this study is not more detailed and hardly more precise than which is proposed in earlier indictor systems. Its main benefit is that the data can be improved by field measurements or from other sources, and that the improved data can fit in the same framework as that for which these crude indicator data are collected.

Data collection for other cases

When the data presented in this study is not considered specific enough, extra effort can be put in the collection of specific data. This data gathering can be based on GIS information as in the Dutch FLORON database or in the Flora Europeae, for European countries. For situations where no adequate literature data is available, data should be collected on site, when considered of sufficient importance. This is not uncommon for many non-LCA environmental assessments. Especially for cases where a land use change is central to the study (as in the production of a resource) information from a consecutively executed Environmental Impact Assessment can be used, or from other EIA studies on related situations. The ambition level of the study is thus decisive for the amount of effort that needs to be put in this regionally differentiated impact category.

In general, data for land occupation cases can be gathered in terms of the number of species S and then transformed to α via the formula $\alpha = (S_{\text{mapping cell}} - 10)/LOG(A_{\text{mapping cell}})$ from annex 1, and in terms of the amount of biomass, subtracting the biomass for human consumption. The related reference values can be found from the world maps in annex 1. For the number of species any selection of species can be taken, as long as the reference data can be transformed to or expressed in that same selection (as that indicator is expressed in relative terms only). For land use changes also information is required on the situation before the activity and the situation after restoration.

Additional inventory data needed for occupation are the amount of output per time unit and the area used for that. For land use changes information is needed on the total amount of output (or input, for landfilling); it should be realised that the change is only allocated to the first user; consecutive economic users are only credited for the occupation, and may get credit for recovering the original state if this occurs.

These brief indications on how to deal with cases not included in this chapter are elaborated in a short manual for including land use impacts in LCA, available as a separate publication of the commissioner.

5. Evaluation, conclusions and further research topics

5.1 Fulfilment of characterisation requirements

As an evaluation of the characterisation approach operationalised here, the requirements set at the start of the project (see chapter 1 and 3.1) will be matched with the results.

Applicability on different scale levels
In principle this method is applicable on all scale levels, from the global to the local level. Of course the exactness of the reference data is the least on a global level, due to the natural variation in ecosystems (50 to 100% uncertainty). At present only reference data on a global level are generated, in order to make a first order application within LCA possible. Assessing data on a more detailed level is possible, using biodiversity databases and more specific biomass literature. Even at a local level data can be gathered, although probably such data has to be collected specifically for the study, unless an Environmental Impact Assessment study has already revealed this data. Species diversity data from any level can be converted to the biodiversity measure α used, by means of a formula given in annex 1.

Provide a readily applicable set of equivalency factors for all main categories of land use
Strictly speaking no equivalency factors have been derived, as the impact scores are not related to one reference. All data are related to their adequate reference, which is regionally dependent. Therefore, no general scores for types of land use can be given. Nevertheless, for 7 different categories of land use common in Europe a first estimate of land use impact scores are given. As the land occupation data are generally applicable, only the adequate reference state data needs to be added to render impact scores. These 7 land use categories are in fact the most important ones; with these most LCAs can be performed including generic land use impact scores. For resource extraction reference data are gathered for all major resources to facilitate the application, as it is generally not known where a used resource comes from. For land use changes, specific information still needs to be gathered.

The land occupation data generated here has already been implemented in the next public update of the IVAM ER LCA database and will be incorporated in the Eco-Quantum software to evaluate buildings. This certainly contributes to the applicability of the data.

Explicitate the cause-effect chain related to land use
In order to judge the value of the indicators used, the link with relevant end-points and the cause-effect chain should be given, according to ISO. This cause-effect chain is rather complicated and quantification of the links is probably not yet possible. Nevertheless, a first attempt is made to give an overview of the cause-effect chain and the indicators and endpoints used. This sketch is performed in paragraph 3.2.

Linearity with respect to the functional unit
This technical requirement is linked to the philosophy of LCA: the assessment should be independent of the actual amount of function selected in any LCA. For the seabed, this requirement can not be met for biodiversity when a relatively large area is affected, according to IBN-DLO Texel (see annex 2). But also for biodiversity on land this is a problem, as the loss of the last 20% of an ecosystem habitat will lead to the loss of an overproportional part of its biodiversity. Therefore, this method (and any one designed according to traditional LCA) is limited to interventions in generic areas and is explicitly not applicable when assessing the

impacts on the last 20% of a specific ecosystem. For such serious impacts a separate Environmental Impact Assessment is more adequate, and should not be replaced by LCA.

As science based and objective as possible
This requirement is also in line with ISO. One of the main improvements by operationalising these biodiversity and life support indicators, with respect to previous land use impact indicators, is that the classification of the scores is based on scientific data on abiotic features relevant to the indicator and not on a subjective criterion such as 'naturalness'. Some subjectivity is however inevitable.

Due to data restrictions, vascular plants had to be selected as indicator species group, and no objective weighting within this group was possible. All vascular plant species are therefore considered of equal importance. Also, the free net primary biomass production was selected as an indicator for life support. The reasons for this were that net biomass production gives an indication of the potential turnover of substances, that the part of the net biomass taken out of an ecosystem by humans do not contribute significantly to life support and that focusing on left leaves, branches and roots also gives an indication of the soil quality. Nevertheless, due to the still meagre scientific knowledge on the concept of life support, another indicator could have been selected too. Therefore data has been gathered to optionally operationalise another indicator too. Finally, the reference state was chosen to be the *present* state of the most natural areas within each land use class. The subjectivity incorporated here is that of areas where much degradation has already occurred in the past, as in Europe. The reference state here has a relatively lower nature value. Comparing land uses to such references will lead to lower impact scores than the same intervention in physiologically comparable regions elsewhere.

Another aspect of a scientific approach is its reproducibility. When applying the default values generated in this study, this reproducibility is considered to be optimal. However, when data on specific land use is gathered anew, some deviation of the generated default values can be expected. This is due to the large variability of biodiversity and biomass productivity in ecosystems. The larger the sample taken, the higher the reproducibility will be.

5.2 Conclusions

In the various chapters conclusions were drawn on the operationalisation of the method. They will therefore only be repeated in short here. See the respective chapters for a background.

- the basic framework is acceptable and applicable; the distinction between land use changes and occupation is a recent development which is not yet incorporated in present databases
- two of the three depletion themes in LCA (biodiversity and life support) have been operationalised on a world scale level, allowing for a complete assessment of land use impacts in general LCAs
- vascular plant species diversity seems a generally accepted measure for biodiversity under the present data restrictions; scarcity is a not yet operationalised aspect of biodiversity, for which solutions seem available
- there is some reluctance to accept the free net primary biomass productivity as a measure for life support; other measures based on the net primary production itself are also suggested, but fNPP seems the best indicator to also express the soil quality
- an official acceptance of this approach can not be expected before some years, when the SETAC-Europe Working Group on Impact Assessment is planned to perform such a possible accreditation
- the uncertainty in the reference data is large (50-100%) due to the natural variability of ecosystems; it is supposed to use one default set of reference data and this set is depicted

on world maps (appendix 4 and 5 to annex 1); the uncertainty in the case data is estimated at 25%
- the present quality of the impact scores for the cases should be improved
- results for cases are comparable to those using former indicators suggested for land use, thus still leading to relatively large scores for the extraction of renewable resources; the sensitivity and the scientific basis of the indicators developed here is however better
- it depends heavily on the reference state selected what the indicator score will be; this is less the case for the biodiversity indicator, as this is a relative score
- major improvement over former indicator scores is that global data have been applied, that the classification is more based on scientific data and made explicit and that field data for specific cases can be fed into the system
- further detailing of the data is possible (within the set of indicators for specific cases and by expanding on the number of impact indictors, for instance for fragmentation and landscape deterioration)
- desiccation can be operationalised using the same impact indicators
- for landscape deterioration and for fragmentation additional impact indicators need to be developed
- for benthic (seabed) land use no single indicator could be selected; also the biodiversity regeneration time needs to be incorporated for these dynamic systems
- no definite guidelines could be given on how to evaluate the relative importance of impact scores for biodiversity and life support due to land use changes and land occupation, leading to 4 impact scores for land use impacts; the suggestion to incorporate the hypothetical regeneration time in the characterisation of land use changes needs closer scrutiny and may lead to a similar indicator for benthic systems
- the indicators for biodiversity and life support are a first crude step towards assessing land use impacts to its full extent; but land use data needs to be generated first in order to show what can be done with this kind of information

5.3 Further research topics

Finally, an overview is given on the possible routes for further developing this first operationalisation step. Again, the reader is referred to the relevant parts of the main text, to chapter 6 of annex 1 and to annex 2.

- improve intervention data related to land use
- generate land occupation impact data for more cases and test/improve the robustness of the reduction factors
- collect land use change data and judge its importance by attempting an integration of occupation and change (see chapter 2)
- develop land use type specific impact models, including validation by expert judgement or more detailed database scrutiny, and/or collect data in the field
- await future data and scientific knowledge on crucial indicators for including biodiversity impacts for the seabed; include the regeneration time in the indicator(s)
- collect appropriate data for desiccation
- integrate land use data in presently popular databases
- apply the data to complete LCA studies
- await more scientific knowledge on the quantitative relationships within life support, to decide finally on the most appropriate biomass indicator for this theme
- expand the impact indicators to fragmentation and landscape deterioration
- include the aspect of scarcity in the biodiversity indicator at the level of ecosystem scarcity
- develop a link to existing valuation approaches

For RWS DWW the most important issue to pursue further seem to be land use change, as this is the most obvious aspect of land use when a new infrastructural work is being developed or when a new aggregate extraction site is planned. Data on site, as collected for an Environmental Impact Assessment and maybe altered slightly for the purpose of inclusion in LCAs according to this approach, needs to be integrated with land use data from other economic processes. It may be too ambitious to collect (site-dependent) land use change data for all these other economic processes all over the world. Therefore, an integration of land use change and land occupation data may be considered, by assuming immediate restoration of the original state after an activity or by postulation a hypothetical restoration time.

Also, for comparison of land use in the sea and on land, a sensitivity analysis should be performed using available (now or in the near future) data on the parameters species diversity, age distribution and species composition for some zoobenthos species to decide how species diversity should be expressed. The regeneration time to be considered according to IBN-DLO may be included by making an expert guess of the hypothetical restoration time.

6. References

T. Aldenberg: ETX 1.3a; A program to calculate confidence limits for hazardous concentrations based on small samples of toxicity data, RIVM report 719102015, 1993

Auhagen et al.: Wissenschaftliche Grundlagen zure Berechnung einer Ausgleichsausgabe, Auhagen & Partners GmbH, im Auftrag der Senatsverwaltung für Stadtentwicklung und Umweltschutz Abt. III, Berlin, April 1994

M. Baitz, J. Kreiβig & C. Schöch: Methode zur Integration der Naturraum-Inanpruchnahme in Ökobilanzen, IKP Universität Stuttgart, Februar 1998

H. Baumann, T. Ekvall, G. Svennson, T. Rydberg & A.M. Tillman: Aggregation and operative units, in Anonymous: Life-cycle Assessment, SETAC, Brussels, 1992

F.F. Beetstra: Kwanificeren van aantasting en het verbruik van ruimte, UCB rapport 301 A, Technische Universiteit Eindhoven, the Netherlands, december 1996

G.P. Beugelink, F.A.M. Claessen & J.H.C. Mülschlegel: Effeten op natuur van grondwaterwinning ten behoeve van Beleidsplan Drink- en indistriewater en MER, RIVM/RIZA, november 1992

T.J. Blonk & E.W. Lindeijer: Naar een methodiek voor het kwantificeren van aantasting in LCA, IVAM ER Universiteit van Amsterdam, Publicatiereks Grondstoffen RWS DWW nr. 1995/15, Delft, the Netherlands, november 1995

H. Blonk, E. Lindeijer & J. Broers: Towards a methodology for taking physical degradation of ecosystems into account in LCA, The International Journal of LCA 2(2): 91-98, 1997

CBS: Statistiek van het bodemgebruik 1993, Voorburg/Heerlen, 1997

H. Dijkstra: Milieu-effectrapportage; 24 Effectvoorspelling VIa Landschap, 1992.

H. Dijkstra & J.A. Klijn: Kwaliteit en waardering van landschappen. Staring Centrum Rapport, 1992

EUROSTAT et al.: Statistical Compendium for the Dobris Assessment, 1995

P. Felten & S. Glod: Weiterentwicklung ökologischer Indikatoren für die Flächenbeanpruchung und für Lärmwirkungen und Aufwendung auf Logistik-Konzepte einer Firma, Semesterarbeit, ETH Zürich, Switzerland, Juli 1995

G. Finnveden: Resources, in Udo de Haes et al., 1996

R. Frischknecht et al.: Ökoinventare für Energiesysteme, 1^{e} Auflage, ETH/PSI, Zürich/Villigen, Switzerland, 1994

J. Giegrich & K. Sturm: Methodenvorschlag Operationalisierung der Wirkungskategorie Naturraumbeanspruchung, IFEU/Büro für angewandte Waldökologie, Heidelberg/Duvensee, Germany, August 1996

M. Goedkoop & R. Spriensma: The Eco-indicator 97: Proposal for the impact assessment methodology, draft version 1.1, PRé Consultants, Amersfoort, 14 April 1997

R.S. de Groot: Functions of nature, ISBN 9001 35594 3, The Netherlands, 1992

T. Hamers et al.: Definition report - Indicator Effects Toxic Substances (I_{tox}), RIVM report 607128001, 1996

U. Hampicke: Naturschutz-Ökonomie, Ulmer, Stuttgart 1991

R. Heijungs et al.: Environmental Life Cycle Assessment of products; Guide and Backgrounds, CML, Rijks Universiteit Leiden, The Netherlands, 1992

R. Heijungs & J. Guinée: Impact categories for natural resources and land use, CML report 18, CML, Rijksuniversiteit Leiden, The Netherlands, 1997

ISO: Evironmental Management - Life cycle assessment - Part 3: Life cycle impact assessment, CD 14042.3, ISO/TC 207/SC5, Februari 10th 1998

IUCN, UNEP, WWF: Caring for the earth, ISBN 2 8317 0074 4, Gland, Switzerland, October 1991

L. Jarass et al.: Von den Sozialkostentheorie zum umweltpolitischen Steuerungsinstrument - Boden- und Raumbelastung durch Hochspannungsleitungen, Nomos, Baden-Baden, Germany, 1989

Jan Klijn: personal comment, Guiding committee of this study d.d. 22/12/1997, IBN Staring Centrum, Wageningen

W. Klöppfer & I. Renner: Methodology of Impact Assessment within the framework of LCA, CAU, Germany, February 1994, also in Methodik der produktbezogenen Ökobilanzen - Wirkungsbilanz und Bewertung -, Texte 23/95, UBA Forschungsbericht 101 01 102, UBA-FB 94-095, ISSN 0722-186X, Berlin, Germany, Juli 1995

I. Knoepfel: Indicatorensystem für die ökologische Bewertung des Transports von Energie, Dissertation ETH nr. 11146, ETH Zürich, Switzerland, September 1995

J.P. Mak et al.: Eco-Quantum, W/E adviseurs duurzaam bouwen/IVAM ER, Gouda/Amsterdam, maart 1996

Th. Mosimann: Ökotope als elementar Prozesseinehiten der Landschaft, Geosynthesis nr. 1, Phys. Geographie und Landschaftsökologie, Universität Hannover, Germany, 1990

R. Müller-Wenk: Safeguard subjects and damage functions as core elements of Life-Cycle Impact Assessment, IWÖ-Diskussionsbeitrag nr. 36, IWÖ-HSG, St. Gallen, CH, draft 29-10-1996

R. Müller-Wenk: Depletion of abiotic resources weighted on base of "virtual" impacts of lower grade deposits used in future, IWÖ-Diskussionsbeitrag nr. 57, IWÖ-HSG, St. Gallen, CH, March 1998

Natuurbeleidsplan, 1990

R. Nijland: personal comment, TNO Bouw, 1997

E.P. Odum: Ecology and our endangered life-support systems, 2nd ed., ISBN 0 87893 634 3, Sinauer Associates, Massachusetts, USA, 1993

RIVM: The environment in Europe: a global perspective, RIVM report 481505001, May 1992

H. Sas et al.: Onttrekking van biotische grondstoffen: ontwikkeling van een methodiek voor inpassing in LCA's, CE/CML, Delft, The Netherlands, september 1996

Schouten: personal comment, RWS AVV, July 1998

G. Swan (ed.): Evaluation of Land Use in Life Cycle Assessment, CPM Report 1998:2, Chalmers University of Technology, Göteborg, Sweden, February 1998

J. van Tilburg: Huishoudwater in Nederland, een duurzame optie?, NWS RUU/DHV Water, december 1997

W.B. Trusty & R. Paehlke: Assessing the relative ecological carrying capacity impacts of resource extraction, Forintek Canada Corp., Vancouver BC, Canada, August 1994

W.B. Trusty & R. Paehlke: The ecological effects of resource extraction in Ontario, Forintek Canada Corp., Ottawa, Canada, March 1997

H.A. Udo de Haes et al.: Towards a methodology for Life Cycle Impact Assessment, report of the SETAC_Europe working group on Life Cycle Impact Assessment (WIA), SETAC-Europe, 1996

H.A. Udo de Haes, personal comment, 1-7-1998

H.A. Udo de Haes & O. Jolliet: Preparation document for SETAC-Europe Working Group on Life Cycle Impact Assessment - 2 (WIA-2); establishment of recommended life cycle impact categories, characterisation factors and damage functions, Bordeaux, 15-17 April 1998

P.M. Vitousek et al.: Human appropriation of the products of photosynthesis, BioScience **36**, 6, p. 368 - 373, June 1986

A. Wegener Sleeswijk et al.: Toepassing van LCA voor agrarische produkten, CML/LEI-DLO/CLM, Den Haag, the Netherlands, 1996

R.H. Whittaker & G.E. Likens: Primary production: the biosphere and man, Human Ecology 1973 (I), Nr. 4, p. 357ff

Appendix 1: Overview of existing proposals to deal with land use impacts in LCA

In this annex, we will focus on the indicator systems for ecosystem degradation and global biodiversity as suggested to be appropriate for LCA. Three groups of indicator systems can be distinguished, although they are to a certain extent overlapping. These are:

1. functional aspects
2. land use classes
3. single indicators

Next to this grouping, indicator systems vary in their dealing with a reference state to relate a certain score to and whether they relate to changes or static land use. These issues will be discussed at the end of the review below.

1. Functional aspects
In [De Groot, 1992] an extensive overview of the various functions of nature is given, in the context of evaluating nature in planning, management and decision making. As stated in [Blonk & Lindeijer, 1995], most of those functions are dependent on the local situation and need a lot of information to be operationalised over a whole life cycle. The problem is that the local situation is often unknown and that the data requirement is enormous for LCA purposes (product policy). Only one system is known to be suggested according to such detailed functions: the functional classes of [Baitz et al., 1998]. As functions they suggest:

- erosion resistance
- alien substance filter, buffer and transformation capacity
- groundwater protection
- buffer capacity for surface water
- protection against physical immissions (noise, dust)
- capacity to improve small-scale climates
- human resort function
- productivity for sustainable agriculture/forestry
- drinking water productivity potential
- landscape quality
- habitat resort function
- All scoring is in three categories: low (0), middle (1), high (2). In their example they add up all scores equally. These scores describe the quality of the land use itself. The quality difference between the situation before and after the activity can also be assessed in this way.
- Abiotic, human and pure nature functions are included here. The main arguments for this approach are that all functions of an area are assessed, that they are coherent with the value system of experts and plan-makers and that the value of an ecosystem is described by more than its nature value.
- Expert knowledge per activity (locality) is necessary for such a detailed approach. A reference or absolute measurements to compare scores from different sites is thus lacking, in spite of the focus on physical parameters. Thereby, consistency in weighting across different activities is not ensured. Also, this functional approach gives very little direct credit to an area as resort for nature (the most prominent effect of land use on ecosystems) when all functions are weighted equally. The same lack of focus on the value

of land as a resort for ecosystems is noticed in [Knoepfel, 1995] about a comparable system referred to there [Mosimann, 1990]. Nevertheless, such systems can be seen as a more detailed and locally oriented elaboration of the classification schemes discussed below.

2. Land use classes

It was proposed in [Heijungs et al, 1992] to use the IUCN ecosystem classification [IUCN/WWF/UNEP, 1991] to classify land use. These were 5 classes:

- I natural systems
- II modified systems
- III cultivated systems
- IV systems dominated by human buildings
- V *systems degraded by pollution and loss of soil and vegetation*

This classification, expressed as the extent of human interference, was based on the free net primary biomass production (fNPP) as elaborated by [Vitousek et al., 1986] in an annex of [IUCN/WWF/UNEP, 1991]. No weighting of the classes (characterisation) was performed. In [Frischknecht et al., 1994] this system was adopted for use in a large database on energy systems. Both references used the classes to express changes. In [Knoepfel, 1995] a similar system was suggested based on the naturalness ("Hemerobiestufen") of a system, naturalness being defined according to the succession theory of [Odum, 1993]:

- A modified systems
- B forests and extensive cultivated systems
- C intensive agricultural systems
- D systems dominated by human buildings
- E sealed soils

This author suggested three possible characterisation (quantification) schemes for land use effects based on this classification (see below under Single indicators). [Klöppfer & Renner, 1995] also used the concept of Hemerobiestufen to distinguish 7 classes (splitting intensive systems into forest monocultures and agriculture, and including the natural systems). They suggest to assess only the land classification during occupation, no changes.

In [Wegener Sleeswijk et al., 1996] the ambition to assess changes was also dropped. They suggest to refine the IUCN class for agriculture for different crops and per crop to distinguish between intensive and extensive cultures by means of a so-called pressure indicator. No operationalisation is given, however.

In [Giegrich & Sturm, 1996] an even further detailed classification scheme was developed for the forestry sector. The classification was based on 3 criteria of naturalness, focusing on the soil, the forest population and the development requirements. This was placed in a larger scheme of naturalness resembling the Hemerobiestufen above:

- undisturbed ecosystems
- almost natural forestry
- reasonably natural forestry and agriculture
- half-natural forestry and agriculture
- reasonably nature-remote forestry and agriculture
- nature-remote agriculture
- sealed and deteriorated surfaces

Forestry and agriculture activities were seen to possibly range from category 2 to 6, with a range of aggregated scores from 1 to 5, respectively. The classification in category 2 to 6 consisted of scoring 6 or 7 indicators per criterion on a scale from 1 to 5, adding all scores equally (for 5 indicators doublecounting score 5 to take into account extreme situations) and calculating the mean score per criterion. The three resulting criterion scores can be added up equally again to obtain one score, giving a very crude overview of the naturalness of the system. In fact, this implies giving an equal value to all 20 indicators and results in a single score, although it is mentioned in [Giegrich & Sturm, 1996] that the scaling is ordinal, with no definition of the distance between 2 scores (!), that aggregation to one score blurs a lot of information and that it is supposed to be not suitable. It is nevertheless observed that a score on naturalness can be compared with targets, whenever these would be set.

As long as no sound argumentation for ranking the classes in either of the above systems can be given, these classifications will often not be sufficient to base decisions on, as it is in general hard for decision makers to decide on the relative value of each land use class from an environmental point of view. As there are many more environmental issues to take into account, leaving this issue disaggregated may cause cognitive or ethical stress for the decision maker or person(s) performing the valuation. It is for this reason that the prelude to this study [Blonk & Lindeijer, 1995] focused on a system where the ranking of scores was performed on a ratio scale, leading to single score results for land use effects (see below). In general it should be realised that for any system the (preferably explicit) reasoning for eventually weighting different land use situations or different criteria should be sound and acceptable.

3. <u>Single indicators</u>
In 1995 the first studies were performed to obtain single indicator scores for effects of land use. In [Knoepfel, 1995] the five above mentioned classes were weighted according to three argumentation scenarios:
I degree of biological accumulation (total biomass / gross primary production)
(a measure for degree of succession, ecosystem stability and plant species diversity according to [Odum, 1993])
II natural regeneration time
(expressing the finally limiting factor for recovering after an activity has ended)
III opinions of a panel
(expressing private perception of the value of each land use class)
The first two argumentations are more ecologically oriented than the last one. They give relatively higher scores for the classes nearer to nature. See table 1. Knoepfel also stated that for tropical forests, deserts and other extreme situations scenario I is insufficient and should be supported by a measure for species diversity. Two more diversity-oriented methods ([Auhagen, 1994] and [Felten & Glod, 1995]) were shortly discussed in [Knoepfel, 1995]. In [Auhagen, 1994] four criteria were applied: naturalness (Hemerobie), species diversity, rareness of biotopes and density of individual plants and animals. According to [Felten & Glod, 1995] there was too much overlap in the criteria. They suggest to operationalise diversity with the Simpson index and with quantification of endangered species through Red Lists. Their scores for the 5 classes are included in table 1, although their classification was more detailed (9 classes).

Table A1: Proposed weighting systems for 5 land use classes (Hemerobiestufen)

Land use class (Hemerobie-stufe)	Biological accumulation, based on [Whittaker & Likens, 1973]	Regeneration time, mainly based on [Hampicke, 1991]	Panel value, based on [Jarass et al., 1989]	Multi criteria of [Auhagen, 1994]	2 Diversity & Red Lists indicator [Felten & Glod, 1995]
A	1	1	1	1	1
B	1	0.17	0.84	0.35	0.85
C	0.1	0.0047	0.52	0.18	0.49
D	0.05	0.0004	0.29	0.06	0.15
E	0	0	0	0	0

Apart from the panel research mentioned in [Knoepfel, 1995], we know of one more panel approach for weighting land use effects. This was limited to biotic and abiotic resource extractions in Canada. In this study, no relation to inventory results of LCAs were made explicit in the questionnaire. Only in the report itself it appears that the related unit was to be 1 m^3 of product. Respondents were assumed to imagine themselves the extent of extraction in terms of area consumed and duration, as they were all Canadian experts in the field. The two other criteria surveyed were impact intensity and significance of the area impacted. The written questionnaire was sent out twice, one in 1994 on a national scale [Trusty & Paehlke, 1994] and another three years later on a regional scale [Trusty & Paehlke, 1997] (23 respectively 21 respondents). The results were presented as a relative index ranging from 1.00 for concrete aggregates extraction to 3.25 for coastal timber extraction. At the regional level the range was smaller (1.00 - 2.56) but there was no further convergence in results.

In [Blonk & Lindeijer, 1995] most of the systems existing at that time were reviewed against the wish to obtain a ratio scale for a single indicator. The link between the above classes and fNPP as ratio measure was seen as promising. The free net primary production was argued to measure the ability of natural development during any activity as the core of land use effects. The suggested formula for quantification was: ED = A . t . (fNPPref - fNPPact), where A is the area affected and t the duration of the activity to produce a certain amount of output, fNPPact relates to the situation during the activity and fNPPref to a reference situation, preferably the undisturbed situation. In the end it was recognised that an additional measure for biodiversity might be necessary [Blonk et al., 1997].

In [Sas et al., 1996] the focus was on the development of indicators for extraction of biotic resources. Due to this focus, only changes are considered and differences between static systems (for instance ecological versus traditional agriculture) are ignored. Ecosystem degradation due to such extractions is expressed in terms of the area affected, net primary production (NPP) of the situation just before the activity and the natural regeneration time t(re) of the NPP: ED = A . NPP . t(re). NPP is here chosen as measure for the life support function regulation (sink for the economic system, climate regulation and cycling of elements such as C and O and water) lost due to the extraction, since NPP closely correlates with carbon immobilisation (energy regulation), water intake and uptake and mobilisation of nutriënts. Life support is seen as one of the two basic functions of nature (the other being keeping and delivering genetic information) according to [Sas et al., 1996]. The biomass production indicator NPP thus indicates loss of life support functions due to a change in land use whereas the fNPP indicator of [Blonk & Lindeijer, 1995] indicates the ability of natural developments during an activity (occupation).

During the development of the LCA method Eco-Indicator '97 [Goedkoop & Spriensma, 1997], the PAF concept was introduced by the RIVM (see [Aldenberg, 1993] and [Hamers et al., 1996]). PAF stands for Potentially Affected Fraction of species and measures which part of the total number of species in an area is potentially affected (in terms of the laboratory test effects as expressed in EC5 results). In the EI '97 approach this concept is extended to land use, by stating that sealed surfaces have a PAF of 100% and that nature resorts (class A and B) have a PAF score of 0%. By measuring the actual PAF due to ecological farming (excluding effects of chemicals) any type of land use can be defined in terms of extent of nature resort, extent of sealed surface and extent of other non-chemical impacts. This approach is not yet operational.

In [Müller-Wenk, 1996] the idea of relating the quality of land use to the risk of impact on scarce habitats was given. The risk is to be measured by the place where particular activities take place, but the approach is not operational yet.

Finally, there are a few monetarisation methods which try to express the effects of land use in terms of money. In [Beetstra, 1996] the weights of [Auhagen, 1994] are used to relate to monetary values. These are based on the highest Dutch land price possible for building in an extensive cultivated system. For this he took Dfl 500,- per m^2 for converting extensive agriculture land to partly built area (a house with a garden), leading to a quality range from Dfl. 0,- for sealed surfaces to Dfl. 2381,- for modified systems. Beetstra has split up class C into C1 (including agriculture with weight 0,21) and intensive agriculture C2 (with weight 0,15). The interest rate r for a long-term commercial loan is included as an extra factor to express that in using land (temporarily) we are loaning from nature, resulting in

$$ED = \Delta(Qmonetary) \cdot r \cdot A \cdot t.$$

In another Dutch study the value of land use is considered a factor 10 to 100 lower. There the costs for nature development after -in this case- producing gravel pits are considered [Nijland, 1997]. This exemplifies the possible range of absolute scores when using different definitions of (monetary) value. Especially when a measure is used which is also used outside of LCA characterisation (such as money) this approach is very tricky since the relative score automatically gets an absolute dimension. Also, all other characterisation results should be expressed in the same monetary values which require a lot of consistent data collection and dealing with data gaps.

Only recently, another system to deal with land use effects using land use classes incorporating monetarisation has been proposed ([Swan, 1998]). In this research, practical LCA land use classes have been developed, which partly coincides with the Hemerobiestufen. Apart from the distance to a natural reference situation (1-P), also the quality of management compared to an ideal management for the land use at stake (1-R) is taken into account, combining to the 'bioquality Q' of a land use case:

$$\text{Land use quality} = Q * (1-P) * (1-R)$$

Land occupation is ignored here, as it is stated that man has the right to occupy land. The bioquality is determined by estimating the costs for preventing natural land to be converted to sealed surface and expressing these as Environmental Load Units, ELU.

We will not further discuss these monetary approaches as in fact no new characterisation approach is developed. It is more the valuation step that is developed here. Any characterisation schema can be fit to these monetary approaches, as was illustrated above.

Appendix 2: Report Workshop on Land Use Impacts (including survey)

(8th annual SETAC-Europe meeting Bordeaux, Friday 17th of April, 8:30-10:30)

Introduction:
This workshop was initiated by Erwin Lindeijer as a discussion workshop, based on his presentation of the IVAM ER research on land use impact methodology the day before. The aim was to further clarify the methodology, to get feedback on the chosen indicators and expressions used and to try to set up a framework to fit land use impact approaches in. The chronology of the discussion is not followed; rather an overview is given of the points raised and discussed. A questionnaire is added, to survey opinions on the conclusions and proposals drawn from this workshop.

Present (23):
Benetsson & Steen (Chalmers, S), Finnveden (FMS/Stockholm Univ., S), Fourcade & Ollivier (EDF, F), Guinee & Udo de Haes (CML, NL), Hansen (COWI, DK), Hofstetter, (UNS-ETH, CH), Huijbregts (IVAM, NL), Jolliet (EPFL, CH), Klöppfer (CAU, Int. J. LCA, D), Kreissig & Baitz (IKP, D), Lindeijer (chair, IVAM ER, NL), Lorenzoni & Powell (CSERGE, UK), Matsuno (NIRE/MITI, JP), Olsson (SIK, S), Philpott (Univ. of New South Wales, AU), Potting & Weidema (IPU-DTU, DK), Seppala (FEI, SF)

Overview of approaches:
IVAM ER
The approach of IVAM ER starts with the focus on an indicator methodology (continuous scale, no weighting of classes, enough data to operationalise at least on a global scale). The indicators 'local loss of species diversity' measured with α for vascular plants and 'loss of free net primary biomass productivity' measured with **fNPP** on a physical (objective) level are (subjectively) chosen to indicate coarsely two 'depletion themes' (as Heijungs and Guinée proposed them in their resources overview for MITI in 1997): loss of biodiversity and loss of life support, respectively. fNPP is also a rough measure of soil quality, as it expresses the amount of biomass left on or in the ground after human harvesting.

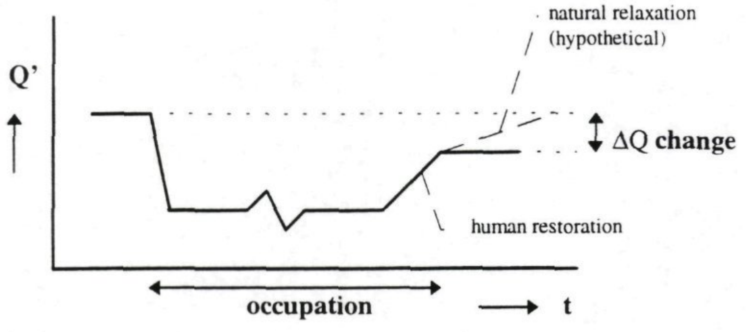
This is a top-down approach to impact assessment for land use impacts, taking into account meaningful endpoints for the safeguard subject Ecosystem Quality and data restrictions. As with all approaches for land use impacts, only potential impacts are indicated. Impacts due to intersection and visual impacts are not included here. Visual impacts (landscape deterioration) need a separate operationalisation, as they lead to another safeguard subject. Double counting with other impacts (f.i. due to biotic extractions or eutrophication) should be avoided.

For both α and fNPP scientific inventory data is available on a global scale. At least for α but probably also for fNPP the data on the situation without activity can be shown on a world map, and can be related to physiotopes (areas with the same range of relevant physical parameters, such as geographical latitude, altitude above sea level and amount of rainfall). The situation during and after the activity can be measured too. At present, preliminary generic values for α and fNPP during a number of case activities have been generated by expert guesses.

Both indicators are used as quality loss parameters (Q) in expressions linked to two aspects of land use as an intervention: land use **change** and land **occupation**:

land use CHANGE = $m^2 * \Delta Q$ land OCCUPATION = $m^2 * y * Q$

When shown in one graph of an imaginary quality variation over time due to an activity, both aspects are visualised as follows (here Q' is the positive counterpart of the quality loss above):



IVAM ER proposes to include the time of starting up an activity and of human restoration in the occupation time, whenever relevant. The quality can either be averaged over the whole occupation time (one Q') or be split in 3 parts with average Q's. The net quality change ΔQ is computed between the situation just before the activity and the situation at the end of an activity, or when relevant after human restoration.

There will be situations where the change to the present land use activity occurred long ago and/or a future change is not yet foreseen and can therefore not be reasonably allocated to one functional unit. In this case, one can only compute land occupation. When the situation before the activity would be completely restored, ΔQ is zero, and only land occupation needs to be recorded. On the other hand, whenever an activity is performed in a short time period, both land use change and occupation can be computed. In fact, every land using activity is a mixture of both occupation and change.

ETH

In the ETH database, inventory data are recorded in the format of land occupation (m^2*y) with additional information on the various possible conversions between five Hemerobie ('naturalness') land use classes also used by the IUCN, CML and CAU. No weighting of th classes is performed. This way, in principle the quality level before the activity, the quality level during the activity, the quality level after human restoration (relative to the nature-like situation) and the occupation time can be recorded separately. However, Patrick Hofstetter admitted that the present inventory format is not transparant enough.

Additional in the ETH database is data/estimates of the (often hypothetical) relaxation time: the time to return to the situation before the activity. By including this factor, the land use change is theoretically set to zero, reducing the amount of expressions in the inventory to those on occupation.

IKP

IKP has developed an approach to express land use impacts via 12 indicators for the various natural functions of the land. The different selected functions were identified by landscape ecologists. Functions for which no objective measure is available (such as recreational/scenic value) are left out. The set of indicators have been selected with a generic view of all possible activities, but the methodology was developed within a project for building materials where mining activities are important. Each indicator can be scored on a five-point scale (by using existing EIA data, expert judgements or -hopefully in future- GIS data from internet). In the first operationalisation, all indicators are weighted equal, giving one score for the land use quality during the activity. Land use is measured as occupation (m^2*y), and the quality level during all phases of the activity can be integrated over time (using the same figure as IVAM

ER above). Generic (default) data for each type of activity stills needs to be provided, in case no specific data can be found.

UNSW
In Australia every impact assessment approach needs to include impacts on ecosystems and indigenous areas. The approach developed for LCA by UNSW classifies activities and gives scores from 1 to 10 for these activities based on expert guesses and activity-specific impact variables (comparable to IKP's functions, but including aesthetic values). In an appraisal step (part of valuation) spatial variables and national heritage values are introduced as additional weighting criteria.

SIK
This Norwegian research group are also developing a detailed approach based on direct physical impact indicators but especially for agriculture, such as organic matter, soil loss and chemical properties for the function soil fertility.

IFEU
Although nobody from IFEU was present, it was mentioned that their methodology used a kind of functional approach for the forestry sector, with a possible link to the Hemerobie-stufen.

IPU
Within the EUREKA project LCA-GAPS the feasibility was assessed of taking the scarcity of the ecosystem into account. Although no data is collected up till now, based on the assessment Bo Weidema suggests that the scarcity of the ecosystem should be taken into account and that it may fit in the generic approach from IVAM ER. It is difficult to measure the scarcity of the ecotope, but it may be done on a global level.

Discussion on indicators for land use impacts:
Biodiversity
The indicator for local loss of biodiversity proposed by IVAM ER is in fact a measure of the relative change of biodiversity from before the activity (α_{ini}) to after it (α_{fin}): $(\alpha_{ini} - \alpha_{fin})/\alpha_{ini}$, or the actual diversity during the activity relative to a reference diversity $(\alpha_{ref} - \alpha_{act})/\alpha_{ref}$ for occupation. Thus, for putting an industrial site in agriculture land (change) the α of agriculture land is α_{ini} and the α of the industrial site is α_{fin} (assuming no return to a higher α). For the occupation of land due to this industry $\alpha_{act} = \alpha_{fin}$ is the α of the industrial site, and the reference α can be taken from the world map on reference α's (showing average diversity scores based on recent diversity measurements and averaged out over large areas).

It is a relative measure because not only species diversity per ecosystem, but also ecosystem diversity (ones with large and ones with low diversity) is appreciated. Assessing this relative difference was agreed upon in the workshop. Also the use of vascular plants diversity as an indicator for total species diversity (and a measure for local loss of biodiversity) was accepted, as it is the only indicator for which data is available on a global level. Only the local loss of biodiversity can be directly taken into account in LCA as the arbitrariness of the functional unit does not allow assessments of final (global) biodiversity losses, although local loss of biodiversity may be an indirect indicator for the potential loss of global diversity, albeit not linearly

It was suggested that ideally species could be weighted according to their scarcity, when sufficient data becomes available. Bo Weidema suggested that a measure of scarce ecosystems might be added to this indicator to include the scarcity aspect.

Life support
To measure loss of life support the free net primary biomass productivity is proposed by IVAM ER. This is the amount of biomass left on or under the ground (whenever harvesting occurs) and it is the amount of biomass free for use by natural biomass consumers such as fauna and decomposers such as funghi. As such, it is an indicator for the development potential of nature (according to IBN-DLO) and as it measures the amount of biomass left on the land, it may contribute to the fuelling of biochemical cycles.

Prof. Helias Udo de Haes was reluctant to agree on this indicator for life support; this needed further thinking. Also, the situation of oligotrophic (biomass-poor) ecosystems was discussed: for these ecosystems an increase of biomass would be unwanted and not to be awarded. First, the issue of eutrophication should be kept separate (this impact is due to emissions which is another type of intervention, leading to the same safeguard subject ecosystem quality as biodiversity and life support do). Is was argued by Mark Huijbregts that a land use change resulting in a higher biomass productivity due to a eutrophic system is likely to reduce biodiversity (the intrinsic ecosystem quality indicator) but may at the same time increase life support. Olivier Jolliet's suggestion was to use only non-negative figures for the difference in fNPP, giving any change a negative impact value. No agreement was reached on this issue, however.

Naturalness
According to prof. Helias Udo de Haes this should also be an indicator for ecosystem quality. According to Erwin Lindeijer the Hemerobiestufe and IUCN naturalness classes as used by CML and CAU indicate the severity of the land use by the type of land use (actually linked to the fNPP concept in the IUCN report from 1991). The type of activity does not really express an impact. As sole indicator these Hemerobie-classification is too coarse and not linked to available scientific data, as argued by IVAM ER. No final conclusion was drawn on this issue, however.

Natural relaxation time
As an additional characterisation approach the natural relaxation time was supported by various persons. Incorporating the time nature would need to return completely to the situation before the activity expresses a more egalitarian viewpoint according to Erwin Lindeijer and would thus answer to the need expressed several times this Annual Meeting to incorporate different viewpoints. This may be added as a separate factor to the characterisation formula.

Functionality indicators
Erwin suggested that the functional approaches are more detailed ways to express land use impacts than his own approach, and apparently produce different sets of indicators for different situations of land use. These different levels of detail should ideally fit in one land use impact framework.

In IKP's functionality approach aesthetic values such as recreational value were excluded, just as in the IVAM ER approach, as it seemed difficult to express these values in measurable indicators. Using simple indicators as the top-height or and volume of natural elements were not acceptable for landscape specialists according to Erwin Lindeijer. Nevertheless, these values were accepted to be important, but leading to another endpoint than ecosystem quality: they relate to human welfare aspects. For Western European countries these landscape values are more often driving the public dispute than intrinsic natural values, according to Jane Powell. She suggested that mere the amount of 'green' could be a valid indicator for this. Also in the Australian approach aesthetic values are included. It thus seems important to develop a framework for these aspects, but maybe in separation of the ecosystem quality aspects of land use impacts.

Framework proposal:
The time was too short to discuss a general framework. However, Erwin mentioned some elements of this framework:

- decide on one generic format for inventory data related to land use
- allow for a very generic approach via indicators based on direct physical impacts to cope with short term data problems
- there are two lines for more detailed approaches: 1) by making a limited number of ecosystem quality indicators more location-specific via allowing nonlinear relationships between intervention and impacts and indicator measurements per case and 2) by extending on the number of indicators via the functional approach and replacing default values per detailed type of activity by actual data whenever possible. To allow application of the latter line of detail in regular LCAs, an agreed total list of these detailed functionality indicators needs to be drawn up and a preliminary default scores per type of activity is required. In both cases, impacts due to intersecting should be included and impacts due to for instance noise and emissions should be computed separately, although the same endpoint will be affected.
- include a separate indicator (set) for aesthetical values related to land use

Survey on the methodology of land use impacts:
As most of the workshop time was used to explain and discuss the various approaches and aspects related to land use impacts, no time was left to discuss a possible framework, state preferences or draw conclusions. The following survey is meant to attempt some steps in this direction. It is part of our research to consult a number of experts on the acceptability of the ideas put foreward. We hope you are willing to express your opinions and argumentations for it, and will include them in the IVAM ER report on the land use impacts methodology. The form of the survey is a questionnaire, to be filled in by computer and sent back to IVAM ER by email (elindeijer@ivambv.uva.nl). In order to allow listing of participating organisations and persons, we would like you to mention your name and affiliation at the end of the survey below. If you prefer remaining anonymous, that is of course also possible. Also, do not hesitate to leave questions blank when you have no clear opinion on the subject.

1) Do you agree on the distinction between land use change and land occupation (with the consequence of the two different units)?

2) Should species diversity, biomass productivity and eventually all possible land use indicators be related to both types of interventions (change and occupation), as suggested by IVAM ER?

3) Do you accept the three depletion themes related to land use and resources as proposed by Heijungs and Guinée in 1997 (reduced availability, loss of biodiversity and impacts on life support functions)?

4) Do you accept species diversity (α) and more specific vascular plants diversity as a measure for potential impacts on biodiversity due to land use in non-critical areas (areas where there is no acute need for policy measures or where measures are already taken to conserve local biodiversity)?

5) Do you accept free Net Primary biomass Productivity (fNPP) as a possible measure for impacts on life support functions (contribution to biochemical cycles and buffering against other impacts) due to land use?

6) In order to develop equivalency factors, measures as the above need to be expressed as indicators and related to interventions in a meaningful way. General expressions based on

the distinction between land use change and occupation are respectively: $m^2.\Delta Q$ and $m^2.y.Q$. For biodiversity IVAM ER proposes expressions for ΔQ and Q based on the following arguments for a most generic approach:

- potential impacts on biodiversity need to be expressed relatively (not in absolute numbers), as ecosystems with lower absolute diversity (as in Nordic areas or in deserts) do not have a lower intrinsic value. This relative expression implies that the simplest expression for occupation Q (quality loss) = $1/\alpha_{act}$ is not appropriate. Because α_{act} may become zero, also $Q = \alpha_{ref}/\alpha_{act}$ is not possible.
- for land use changes only the net change in land quality from just before (Qini) to just after (Q_{fin}) an activity (including actual human restoration) needs to be expressed
- for land occupation two reference states (Q_{ref}) are possible to obtain a relative expression with the quality during the activity (Q_{act}): the situation as measured presently in the region of the activity or the 'would be' situation, if man had not started using land in the (far) past. For instance, in Western Europe the general biodiversity as measured now is lower than that at the end of the last century, and again different when another reference time had been chosen. As there is a large dispute on the possible would-be situations, as only recently measured data are available worldwide and as recent measurements are then consistently used for both change and occupation, we propose to use the more recent diversity data as a reference.
- select the most simple yet adequate expressions for potential loss of local biodiversity.

This results in the expressions for potential biodiversity impacts (PBI) due to land use:

$PBI_{change} = m^2.(1-\alpha_{fin}/\alpha_{ini}) = m^2.(\alpha_{ini}-\alpha_{fin})/\alpha_{ini}$ and

$PBI_{occupation} = m^2.y.(1-\alpha_{act}/\alpha_{ref}) = m^2.y.(\alpha_{ref}-\alpha_{act})/\alpha_{ref}$

Do you agree with these argumentations and the resulting expressions for local biodiversity losses due to occupation and change?

7) For fNPP as a measure of local contributions to life support functions, a comparable reasoning is followed. We follow the suggestion to use a relative expression for fNPP. The main reasoning for leaving our original idea of an absolute expression is that we assume that it is useless for local ecosystems when for instance an unnatural high or low amount of biomass is produced during an activity, compared to the surroundings. On the other hand, we argue that according to natural dynamics it is not especially 'good' to conserve oligotrophic ecosystems, as its natural tendency may be to evolve towards a system with higher fNPP (remember that important is the *free available* amount of biomass!). This is why we accept fNPP as an indicator for life support, also for oligotrophic systems. We do not follow the suggestion for expressions such as $|Q|$, as we need an indicator which gives an unequivocal direction of what is positive for life support; in our view fNPP provides this already, and a higher free available amount of biomass is thus considered better. This results in the following expressions for potential life support impacts (PLI) due to land use:

$PLI_{change} = m^2.(1-fNPP_{fin}/fNPP_{ini}) = m^2.(fNPP_{ini}-fNPP_{fin})/fNPP_{ini}$ and

$PLI_{occupation} = m^2.y.(1-fNPP_{act}/fNPP_{ref}) = m^2.y.(fNPP_{ref}-fNPP_{act})/fNPP_{ref}$

Do you agree with these argumentations and the resulting expressions for local life support losses due to occupation and change?

8) Do you agree that the functional approaches are more detailed ways to express land use impacts than the above approach based on a few indicators?

9) *We argue that the Hemerobiestufe approach is a more limited impact assessment approach for land use, due to its limited link to scientific data and its coarse classification. Do you agree on this?*

10) As the Hemerobie classes have been linked to biodiversity indicator scores, and the biodiversity and life support indicators can be used within the scope of a functinal approach, we believe that it is possible to fit all these approaches within one framework for land use impact appraoches. Do you share our optimism here?

11) The inventory format to be used for land use impact needs further improvement and harmonisation, as spatial information needs to be included in the name to allow comuterised impact assessment. We suggest different levels of detail, related to the various approaches. For each level the inventory units are m^2 and m^2.y for land use changes and land occupation, respectively. For the name (restricted to 20 characters for easy use in present LCA software) we suggest the following format structure:

NAME

Activity type [classification];[indicator scores for actual/final resp. reference/initial states, separated by ;][criteria ranking separated by ;][detailed information for scoring indicators or criteria separated by ;]

The characteristics in [] are optional, depending on the approach used. It is our experience that it is difficult to interpret and manage land use inventory data in characterisation if the activity type is not given in the name, especially in the build-up phase of these data. Therefore we suggest to always start with a general typology of activities such as the following, with an optional further detailing (all within 10 characters):

mining:	minin	[pit],[shaf],[Al],[Cu],[Fe] etc. (pit or shaft mining, mineral type)
drilling:	drill	[land],[sea]
transport:	trans	[road],[rail],[cana] (manmade canals)
hydropower:	hydro	[rese],[curr] (reservoirs, currency)
agriculture:	agric	[trad],[orga] (traditional, organic)
forestry:	fores	[cert] (certified)
buildings usage:	build	
dumpsites:	dump	
other industrial activities:	indus	(incl. general energy production, incineration etc.)

Also a country code could be added in the typology. The activity typology certainly needs harmonisation for database exchanges and interpretations.

For the IVAM ER approach a name for occupation could look like: minin;0;0;15;12 including average biodiversity and fNPP scores, respectively. For the ETH approach the names might look like mininpitAl;II>IV (during occupation) and mininpitAl;II>III (for human restoration). For the functional approach of IKP the name could be mininpit;14;5;8;1;6 for 5 functional indicators.

We have experienced that it is virtually impossible to include all relevant detailed infomation in one name. For instance, aluminium is mined in various countries with various country-specific biodiversity and biomass characteristics. However, only one set of average indicator scores can be incorporated in a process describing the mining of aluminium for the worldmarket. This is why the level of indicator and criteria scores are included. Ideally, spatial characteristics such as latitude, longitude, altitude or just country should be given per specified activity. Considering the present level of land use data in databases, for many processes additional efforts are required to reach this level.

What is your opinion on this proposal for the format of land use interventions?

What are your name and affiliation?

-End of survey-

Appendix 3: Inventory data for the cases

unit	production	source	time	area	a x t	top height	volume	remarks
			years	m^2	m^2yr	m	m^3	
1 kmT	transport	cbs97	1	1,10E+10	0,0043	5	5,48^{E}+10	data valid for 1993, based on solid roads
1 kmT		lin95	1	1,10E+10	0,0044	5	5,52^{E}+10	1990; allocation between cars and trucks based on number of vehicles
1 kmT		lin95	1		0,0212	5		1990; allocation between cars and trucks based on weight transported
		eth94	1	3,06E+12	0,0076	5	1,53E+13	1993 for Germany
		eth94	1	4,58E+11	0,0097	5	2,29E+12	1989 for Switzerland
1 ton	iron ore	lin95	10	0,011	0,111	10	0,11	
1 ton	bauxite	lin95	75	0,001	0,090	10	0,01	hee87, min. Value lin95
1 ton	bauxite	lin95	10	0,114	1,143	10	1,14	ETH or resource, max. vlue lin95
1 ton	bauxite	vri94		0,21				
		vri94		0,06				
1 ton	coniferous wood NL	lin95	1	8942	8942	30	268265	low value, dep. On spec. Weight
1 ton	coniferous wood NL	lin95	1	15348	15348	30	460426	high value, dep. On spec. Weight
1 ton	coniferous wood EU	lin95	0	3964	3964	30	118920	low value, dep. On spec. Weight
1 ton	coniferous wood EU	lin95	1	8371	8371	30	251130	high value, dep. On spec. Weight
1 ton	sand	zan96	1	0,156	0,022	3	0,47	
1 ton	sand	lin95	10	0,180	1,798	3	0,54	ETH or. Resource
1 ton	yeast	gis93	1	5442	0,079	5	27210	68687 t yeast/y
1 MJ	hydropower	lin95			0,001		0,00	ETH or. Resource, storage
1 MJ	hydropower	lin95			0,001		0,00	ETH or. Resource, flow
1 kg	household waste	lin95	25	0,0000	0,002	30	0,002	data 1989 low val. spec. Wght
1 kg	household waste	lin95	25	0,0001	0,003	30	0,003	data 1989 high val. spec. Wght

Sources:
cbs97: CBS Bodemstatistieken, 1997
eth94: R. Frischknecht et al.: Ökoinventare für Energiesysteme, ETH/PSI, 1994
gis93: Aanvraag revisievergunning Gist Brocades, 1993

lin95: E.W. Lindeijer: Onderbouwing data landgebruik, in Mak et al.: Eco-Quantum, W/E adviseurs duurzaam bouwen/IVAM ER, Gouda/Amsterdam, maart 1996
vri94: S. de Vries: *Mijnbouw en duurzaamheid*, IVEM-studentenrapport 81, Groningen, 1994
zan96: Data zandindustrie d.d. 1996

Appendix 4: Data on NPP and α reference values for resources

Metals

Aluminum			
origin	production 1994 % of world total	mean value α	mean value NPP
Australia	37,6%	40	9
Guinea	15,3%	50	9
Jamaica	10,4%	50	16
average		44	10

Nickel			
origin	production 1994 % of world total	mean value α	mean value NPP
USSR	30,3%	10	8
Canada	18,7%	10	8
average		10	8

Cadmium			
origin	production 1994 % of world total	mean value α	mean value NPP
Japan	14,2%	40	12
Canada	12,0%	10	8
Belgium	8,7%	15	12
USSR*	8,2%	10	8
China*	7,1%	10	9
US	6,0%	22	6,6
Germany	6,0%	15	12
**1993 figures*			
average		26	10

Tin			
origin	production 1994 % of world total	mean value α	mean value NPP
China	27,2%	10	9
Indonesia	18,1%	150	22
Peru	11,8%	55	12
Brazil	10,0%	50	16,6
average		62	14

Metals continued

Copper

origin	production 1994 % of world total	mean value α	mean value NPP
Chile	23,3%	25	3
US	18,9%	22	6,6
average		24	5

Zinc

origin	production 1994 % of world total	mean value α	mean value NPP
Canada	14,6%	10	8
Australia	13,7%	40	9
China	13,1%	10	9
average		20	9

Lead

origin	production 1994 % of world total	mean value α	mean value NPP
Australia	18,9%	40	9
China	13,6%	10	9
US	13,5%	22	6,6
average		26	8

Iron ore

origin	production 1994 % of world total	mean value α	mean value NPP
China	23,7%	10	9
Brazil	15,3%	50	16,6
Australia	12,2%	40	9
average		28	11

Antimony

origin	production 1994 % of world total	mean value α	mean value NPP
US	?	22	6,6
Bolivia	6,8%	46	13
Mexico	2,7%	50	12
South Africa	5,5%	25	7
Other	84,9%		
average		?	?

Silver

origin	production 1994 % of world total	mean value α	mean value NPP
US	11,7%	22	6,6
Mexico	15,1%	50	12
Peru	11,2%	100	22
average		55	13

Metals continued

Chromium			
origin	production 1994 % of world total	mean value α	mean value NPP
Kazakhstan	31,2%	10	7
South Africa	30,5%	25	7
India	11,5%	75	16
average		27	9

Platinum group			
origin	production 1994 % of world total	mean value α	mean value NPP
South Africa	59,8%	25	7
Russia	29,5%	10	8
average		29	7

Manganese			
origin	production 1994 % of world total	mean value α	mean value NPP
Australia	12,7%	40	9
Brazil	10,3%	50	17
China	15,5%	10	9
Gabon	9,9%	75	22
India	9,3%	75	16
South Africa	16,9%	25	7
Ukraine	15,5%	12,5	9
average		37	11

Titanium (ilminite)			
origin	production 1994 % of world total	mean value α	mean value NPP
Australia	10,2%	40	9
Norway	21,4%	7,5	4,5
USSR	10,7%	10	8
Malaysia	9,0%	150	22
average		50	10

Metals continued

Mercury				Uranium			
origin	production 1994 % of world total	mean value α	mean value NPP	origin	production 1994 % of world total	mean value α	mean value NPP
China	24,1%	10	9	Russian Federation	26%	10	8
Algeria	13,8%	10	1	Canada	21%	10	8
Spain	10,3%	40	7	Australië	11	40	9
Kyrgyz Rep	10,3%	?	?	USA	9%	22	6,6
average		16	6	average		17	8

Cobalt				Lithium			
origin	production 1994 % of world total	mean value α	mean value NPP	origin	production 1994 % of world total*	mean value α	mean value NPP
Canada	25,8%	10	8	Chile	36,0%	25	3
Zambia	23,9%	50	9	Australia	23,4%	40	9
Russia	14,9%	10	8	Russia	14,4%	10	8
Zaire	11,1%	62,5	16	Canada	10,6%	10	8
average		32	10	average		25	6

* *excl. USA*

Sources:
Metals Al, Cd, Cu, Pb, Hg, Ni, Sn, Zn, Fe ore, steel:
WRI, World Resources. A Guide to the Global Environment. The Urban Environment, 1996-97
Metal Ti (ilminite) and Uranium (U): Hargreaves, Eden-Green & Devanay: World index of resources and population, Dartmouth PC, Vermont USA, 1994
Other metals: NTIS, Mineral Commodity Summaries 1995, Bureau of Mines, Washington DC, Jan. 1995

Fossil fuels

Crude oil			
origin	production1991 % of world total	mean value α	mean value NPP
Middle East	26,2%	10	1
Saudi Arabia	15,7%	10	1
USA	13,6%	22	6,6
ex-USSR	16,4%	10	8
average		11	2

Gas			
origin	production 1991 % of world total	mean value α	mean value NPP
ex-USSR	35,9%	10	8
USA	25,1%	22	6,6
average		15	7

Coal			
origin	production1991 % of world total	mean value α	mean value NPP
USA	25,4%	22	6,6
China	23,3%	10	9
ex-USSR	12,5%	10	8
average		15	8

Wood

Roundwood *coniferous* origin	production 1991 % of world total	mean value α	mean value NPP
US	23,9%	22	6,6
Former USSR	22,4%	10	8
Canada	12,6%	10	8
China	10,5%	10	9
Sweden	3,4%	10	8
Germany	2,6%	15	12
France	1,8%	15	12

Roundwood non *coniferous* origin	production 1991 % of world total	mean value α	mean value NPP
US	9,2%	22	6,6
Nigeria	5,1%	50	9
Brazil	9,1%	50	17
China	7,4%	10	9
India	12,9%	75	16
Indonesia	8,6%	150	22
Europe	5,5%		
Former USSR	3,3%	10	8

Fuelwood + charcoal origin	production 1991 % of world total	mean value α	mean value NPP
Africa	25,6%		
US	4,7%	10	8
Brazil	10,4%	50	17
China	10,5%	10	9
India	13,9%	75	16
Indonesia	7,9%	150	22
Europe	2,8%		
Former USSR	4,4%	10	8

Sawnwood coniferous origin	production 1991 % of world total	mean value α	mean value NPP
US	24,5%	22	6,6
Canada	15,6%	10	8
Indonesia	15,6%	150	22
Japan	7,7%	40	12
Germany	4,0%	15	12
Austria	2,1%	15	12
France	2,2%	15	12
Sweden	3,5%	10	8
Finland	1,8%	12,5	10
China	3,6%	10	9

Wood continued

Sawnwood non coniferous origin	production 1991 % of world total	mean value α	mean value NPP
USA	18,4%	22	6,6
Africa	4,7%		
S America	11,3%		
China	6,7%	10	9
India	11,4%	75	16
Indonesia	6,9%	150	22
Malaysia	6,8%	150	22
Japan	2,4%	40	12
Germany	1,7%	15	12
France	3,0%	15	12
Australia	1,1%	40	9
Romania	1,0%	15	12
Former USSR	9,9%	10	8

Source: FAO Yearbook, Forest Products

Import wood in The Netherlands

Roundwood (excl. fuelwood)	origin	% of total import	mean value α	mean value NPP
Coniferous wood	Begium/Lux.	54	15	12
	Germany	44	15	12
Non-tropical wood	Germany	48	15	12
	Begium/Lux.	43	15	12
Tropical wood	Cameron	47	75	22
	Gabon	19	75	22
	Eq. Guinee	12	50	9
	Ghana	10	63	16
mean average			72	21
Sawnwood				
Coniferous wood	Sweden	33	10	8
	Finland	25	12,5	10
	former USSR	10	10	8
	Germany	9	15	12
	Begium/Lux.	8	15	12
mean average			12	10
Non-tropical wood	Begium/Lux.	25	15	12
	USA	23	22	6,6
	Germany	19	15	12
	France	19	15	12
mean average			17	11
Tropical wood	Malaysia	68	150	22
	Africa	10		

Source: SBH, Landenorientatie Bos en Hout, Basisgegevens over bos, bosbeleid en houtmarkt in 24 landen (1992), Wageningen, 1995

Aggregate mining in Europe

Aggregate mining within Europe	% with α value 40 (estimate)	% of land with α value 15 (estimate)	mean value α	% with NPP value 7 (estimate)	% with α value 12 (estimate)	mean value NPP
Sand	20	80	20	20	80	11
Gravel	20	80	20	20	80	11
Clay	10	90	17,5	10	90	11,5

Annex 1

Biodiversity and productivity parameters as a basis for evaluating land use changes in LCA

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1 Introduction

Effects of the land use for production of the raw material for a product have usually been neglected in LCA. This was mainly because of a lack of a consistent and widely accepted methodology. LCA's of products have usually concentrated on the energy use and emissions of pollutants during processing, the life span and disposal part of the product chain. When a product originates from a raw material which is produced in a management system in e.g. agriculture or forestry, the energy use and emissions during that production part of the chain were usually not the problem either. However, the degree of degradation due to that specific form of land use was always difficult to quantify because these effects of land use consist of many aspects like fragmentation, erosion, loss of biodiversity etc. which are hard to quantify. Blonk and Lindeijer (1995) define this degradation of ecosystems as 'the diminishment of development space for nature'. They operationalise this through the quantification of loss of free Net Primary Production (NPP) thereby assuming that the free NPP represents that part of the NPP which remains for nature.

Apart from this methodology, land use degradation has usually been assessed in a semi-quantitative way, by defining classes of naturalness and the closeness to nature, e.g. Giegrich and Sturm (1996) define seven parameters (e.g. soil disturbance, continuation of vegetation development) each with five classes of degradation. Also Trusty (1995) defines semi-quantitative classes for intensity and extent of land use. Already since 1992 the use of five to seven distinct classes of land use have been proposed for LCA (e.g. Heijungs et al., 1992, Knoepfel, 1994, Klöppfer & Renner, 1995). Within the framework of a study carried out by IVAM BV on behalf on the Ministery of Transport, Public Works and Water Management, we attempted to operationalise two aspects of the effects of land use quantitatively, on a continuous scale:

1. biodiversity as a basis for evaluating land use changes;
2. loss of (free) productivity as a measure for degradation.

2 Biodiversity as a basis for evaluating land use changes

The aim of this section is to describe the development of an indicator that quantifies the effect of human land use on (local) biodiversity. This indicator will be used to quantify the loss of natural values within the scope of LCA. This quality loss will be related to LCA input (so called intervention) data on land use change in terms of m^2 area changed and land occupation in terms of m^2.y (area used during a certain amount of time).

2.1 Rationale behind the use of only species diversity as a measure for biodiversity

Biodiversity is '*the variability among living organisms from all sources, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*' (Agenda 21).

The function of biodiversity for mankind is acknowledged on three levels:

1. ethical function, i.e. the existence of species for their own sake (this includes possible future use of species e.g. for medical purposes or as gene reservoirs, and thereby also has an economical component);
2. life support function, i.e. the maintenance of essential cycles of matter and energy;
3. production function, i.e. of biomass as food, industrial raw material or energy.

It follows from the above that loss of biodiversity can be defined in terms of (1) loss of species, (2) loss of ecosystem functions (life support, 'regulation' function), (3) loss of productivity. Here, only (1), the ethical aspect of biodiversity, will be considered, although some attention is given to functional aspects. Loss of ecosystem functions will usually be due to the loss of functional groups of species, and thereby related to (1). Production functions of nature (3) are not generally assessed directly in LCA studies as their aim is to assess the impacts of the economic system on the environmental system, and therefore have to keep both clearly separated. However, biomass productivity available for the development of nature itself is considered as a quantitative indicator for life support. This indicator will be discussed in section 3.

Following above reasoning, the ethical aspect of biodiversity can be approximated by the number of species. In view of the available data, loss of species seems a simple and practically feasible measure, with the number of species as a quantitative parameter for loss of biodiversity.

One could argue that some species have a larger contribution to biodiversity than others; or, that measures based on a selection of species are more practical than those based on all species. This leads to the idea of species weighting: biodiversity is not simply measured as the number of species, but as:

$$\Sigma_{i=1,n}(spec_i * weight_i) \qquad (1)$$

with n as the total number of species.

The following criteria for weighting might be considered:
1. no weighting, i.e. all species are equally important;
2. weighting as to 'importance' from a nature conservancy point of view (i.e. considering rareness, decline); the IUCN concept of 'Red Lists' and the Dutch concept of 'target species' (which is derived from the Red List concept) are such forms of weighting;
3. weighting as to functional groups; e.g. primary producers should be present anyway, an ecosystem is only 'complete' if large carnivores are present, or, in a forest, mycorrhiza-forming mushrooms should be present, etc.
4. weighting as to taxonomic groups; often only mammals, birds and vascular plants are considered. This criterion is related to criterion (2), but is also used in response to practical

constraints (e.g. availability of data);
5. weighting as to endemism, as has been proposed in Sas et al. (1996). This criterion is related to (2), but an advantage might be that data are more easily available, at least for some groups;
6. *weighting as to abundance of individual per species. Some of the classical biodiversity* measures, e.g. the Simpson and the Shannon-Weaver index (see e.g. Huston 1994) use this form of weighting.

For the development of a method that is applicable world-wide, the availability of data is a huge constraint. Distribution maps are available for a limited number of species only, especially outside Europe. Red Lists as defined by IUCN give the probabilities of extinction per species. Red Lists have the problem that their geographical extent is defined politically and not on ecological grounds (e.g. per physiotope). Moreover, Red Lists are available for only a small number of countries (mostly western Europe excl. the Mediterranean countries). This makes their application to global LCA's impossible at present.

If species-weighting is to be applied, data availability is a larger problem than when using species diversity sec. It could be decided to map functional groups if data are unavailable at the level of species. But also for functional groups data are scarce, and in practice knowledge will be limited to a small number of 'interesting' ecosystems (e.g. savannah, tidal areas, temperate forest). Besides, the functional groups will have to be defined beforehand, and it will have to be decided which groups are present in the 'natural' state, which may also constitute a problem.

The biodiversity measures using abundance-weighting partly represent a measure called 'evenness', this is the extent to which all species are present in about equal quantities in a given ecosystem; or, in other words, whether an ecosystem is composed of a few very common species, or of many rare species. The most popular measures for abundance-weighted biodiversity are the Shannon-Weaver index:

$$H' = -\Sigma[p_i \log(p_i)]$$

and Simpson's index:

$$\lambda = \Sigma p_i^2$$

in which p_i is the percentage of species i (measured as biomass or number of individuals) in the total sample. It can be easily seen that H' is a weighted number of species, with $-\log(p_i)$ as a weighting factor, i.e. rare species contribute more to the diversity than abundant species. This is in line with the common feeling among ecologists in which ecosystems with many rare species are higher valued than ecosystems with a few abundant species. The Simpson's index simply uses the abundance p_i as a weighting factor, and therefore works the opposite way compared to the Shannon-Weaver index. It is therefore sometimes expressed as Simpson's diversity D with

$$D = 1 - \lambda$$

Some ecological theories (that now have largely become obsolete, however; e.g. Hurlbert 1971) state that a system's evenness is related to its stability. In spite of their popularity (especially in the past) the ecological relevance of the abundance-weighted measures is unclear (Huston 1994). For the present application, however, their greatest drawback is of a pratical nauture, namely that data on abundance are generally lacking.

In conclusion, most forms of species weighting will lead to methods that are strongly limited by data availability. Besides, some of these methods (especially 3. and 5.) will be prone to under-evaluate the temperate zone, especially Europe (where, over geological time scales, a large loss of species has occurred not only due to human activities, but also due to glaciation). In these regions the number of endemic species is low, and some functional groups may be lacking (e.g. large carnivores). Weighting species as to taxonomic groups (i.e. by only considering the groups that are most intensively studied) seems an acceptable and practically feasible method. It is important to note that in doing so, the taxonomic group studied is considered to be representative for the biodiversity as a whole, i.e. for all taxonomic groups. A debate is going on in literature whether or not biodiversity is correlated among different taxonomic groups. There are a number of indications that this question may be answered confirmatory (Hansson 1997, Monkkonen & Viro 1997, Kerr 1997, Crisp et al. 1998).

2.2 'Alpha' as a measure for species diversity

Species diversity data

In practice the number of species per unit area will only be known in certain regions, and for certain taxonomic groups. Both these constraints cause practical problems. As to the taxonomic groups, data acquisition is limited to vascular plants in this study. It is hypothesised that the number of species in this group is indicative for the total number of species. Other taxonomic groups which could be feasible indicators in the terrestrial environment are mammals and birds. For these groups there are probably sufficient data to allow a test of the above hypothesis of correlated diversity among taxonomic groups. This test is outside the scope of the present study, however.

For other taxonomic groups data are usually highly defective and regional species numbers are not even known by order of magnitude (cf. Aptroot 1997).

As to the regions, it is proposed to make a rough division in 'ecosystem types'; these types are defined on the basis of abiotic parameters and will be termed **physiotopes** following Kemmers & Van der Bolt (1997). When physiotopes are geographically delimited, the average species density can be determined for each physiotope on the basis of data available from species mapping programmes etc. It is expected that such data are available for at least part of each physiotope. If a physiotope has strongly degraded parts, these parts should be excluded when determining species density for a generic application in LCA's where the exact situation before the land using activity is not known. A general 'background' value for species diversity in the average surroundings of the activity should then be applied, together with specific data on the situation during and/or after the activity.

To determine a basic set of biodiversity indicators for generic (background) situations the mapping data to determine species diversity indicator values per physiotope need to be harmonised. Different mapping schemes have used different unit areas for their inventories; usually between 1000 and 1.000.000 m^2 in size. In order to allow comparison of these data, the relationship between numbers of species and area valid for these data sets should be determined. There is an ongoing debate in ecological literature as to the shape and nature of the so-called 'species-area curve'. Its principle can be easily understood if one imagines an area A with S species; if a larger area is considered, the same species will be met over and over again. Therefore the number of species increases more slowly than the area considered. The form that is most suitable to express this principle for areas that are not too large is:

$$S = \alpha^* \mathrm{LOG}(A) \qquad (2)$$

with:
S = number of species
A = mapping area for determining species diversity
α = constant

This form is shown graphically in Figure 1.

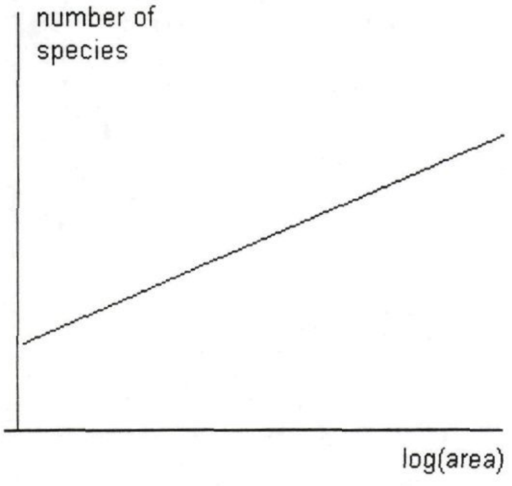


Figure 1: Species - area relationship

This form is often encountered in literature, and is supported by many data, at least for values of A that are not too large (say, < 1000 km^2) (Fisher et al. 1943, May 1975, Sugihara 1980, Tokeshi 1993; Figure 1). Other forms that have been postulated (e.g. by Arrhenius 1921) are akin to the one proposed here (Pastor et al. 1996), but may function better for very large areas.

If parameter α in this relationship can be determined, it can be used as a measure for the regional species density that is independant of the size of the original mapping cell. These can be used as background data to relate to the influence of specific activities on present biodiversity. In order to find α, species density has to be known for at least two values of A. However, mapping schemes always use a fixed A. A second value will therefore have to be artificially added. An obvious value may seem {A=0;S=0}, but this is not realistic for vascular plants in the tropical, temperate and lower boreal zone: species from this group are present almost everywhere, therefore the probability of hitting at least one species on a 'pinpoint' area (A=0) is close to 100%. Therefore the practical solution is proposed to use {A=1 m^2 ; S=10) as a second point. Ecological data with respect to vascular plants are often collected on areas in this order of magnitude, and these data show that the number of species on that area is usually around 10, even in tropical areas (in a pioneer vegetation, 10 species may be growing on 1 m^2 without touching each other; in a tropical rainforest, a single tree with >1 m^2 basal area may carry some 20 epiphyte species). So the formula to determine α then becomes:

$$\alpha = (S_{\text{mapping cell}} - 10) / \text{LOG10}(A_{\text{mapping cell}} / A_{\text{reference cell}}) \qquad (3)$$

with:
S = number of species
A = area; $A_{\text{reference cell}} = 1\ m^2$
mapping cell = standard area used in mapping scheme

α is here the species richness for a given area, which by statistical procedures based on an extensive literature research could be converted to average α values for a number of physiotopes. When the number of species on 1 m^2 is less than 10 the formula cannot be directly applied but α still has a valid interpretation (namely, the number of extra species when expanding the surveyed area from 1 to 10 m^2 (the increase in number of species might for instance be from 5 to 15).

2.3 Linearity of land use with an indicator based on α

If (unweighted) species number for selected taxonomic groups is accepted as a useful measure for biodiversity, the next question is whether the loss of species is proportional to the occupied or changed area.

When land has been occupied for years and the present occupation is to be assessed in terms of its average biodiversity value compared to the surrounding region, it is clear that this average biodiversity value per m^2 during the activity will remain constant, whether the activity lasts for long or not and whether it uses a large area at a time or not. The same is true for the possibility of biota to reside to these areas from outside the occupied area: a long occupation time or a large occupation area both contribute linearly to the absence or change of this resort space. Using a larger area thus implies a linearly higher impact due to land occupation, as does a longer occupation. Less trivial is the situation of changing the type of land use. When a net change in land use is assessed, a small part of an ecosystem which itself covers a larger area is changed to (a different kind of) human usage. The local loss of species may be large, but globally no species may be lost. To what extent global biodiversity is influenced by a local activity can generally not be assessed within LCA, as specific site information is generally lost and the relation between the amount of land changed and a functional unit is generally not clear-cut. It can therefore not be predicted whether the last habitat for a certain species will be transformed, causing real extinction on a global scale. Rather, the contribution to an undesired diminishment of habitats for species, with an increase of potential risk for global loss of biodiversity, is what can be assessed in LCA. As expelled species may thrive outside the area changed, the linearity between the land use change and the change in biodiversity is not obvious. The following argumentation is used:

As long as the area changed is a small portion of the total ecosystem/physiotope, the transformation is contributing to the risk that specific habitats are destroyed, making life potentially harder for species dependent on those habitats. In fact, one could say that not the risk of expelling species but of hitting their habitats is determining the value of the transformation. Species density (i.e. the number of species per unit area) is strongly dependant upon abiotic diversity. In most ecosystems environmental factors are more or less uniform over large areas, but have deviating values in small areas within the large uniform areas. Usually these small areas with deviating environmental factors are 'hotspots' of biodiversity. Examples are small pools in heathland, rock outcrops or ant-hills in grassland, fissure zones in mountain areas. The hotspots can be caused by natural non-random variation in environmental factors, but by human activities in the past as well. It is because of these hotspots that the number of species will continue to increase when increasing the area inventoried, even when going from large to very large areas within a given physiotope.

It follows from the above that in order to quantitatively estimate loss of biodiversity when transforming a given area, one should estimate the probability to 'hit' a 'hotspot' when randomly transforming land use in a certain larger area. This probability is proportional to the density (number per unit area) of hotspots, and the altered area in that larger region. It is therefore necessary to estimate the density of hotspots. If this is possible, (density of hotspots * transformed area) is a linear predictor for local loss of biodiversity and for the risk of decreasing global diversity. Here the working hypothesis is made that on the large scale of making species diversity inventories **the number of species per unit area is a measure for the density of hotspots**.

In order to force linearity between the change in biodiversity and area transformed we assume each species to be potentially present everywhere (i.e. each species is distributed world-wide, but in densities that vary geographically). This means that species turnover is assumed to be zero. Species turnover is the rate at which the distribution areas of new species are included on expanding a certain area of interest (Harte & Kinzig 1997). A high species turnover therefore means a large number of endemic species. The above assumption is permissible for the present study because biodiversity is only assessed within a given physiotope, i.e. within an area with more or less uniform abiotic conditions, and therefore with a low species turnover within its limits.

The local density (=probability of occurrence) is governed by the parameter α. However, large changes in critical areas (c. 10-20% of a large physiotope for instance) will almost certainly lead to global extinctions and can therefore not be evaluated sufficiently according to this method. For assessing the biodiversity during or after an activity, actual measurements or estimates per type of activity must be performed. See chapter 5 for first expert estimates for α values for a number of cases.

2.4 Data collection for implementation of biodiversity criterion in LCA

2.4.1 Definition of physiotopes

In order to be practical, areas with a comparable level of biodiversity have to be treated as entities for which a single α value has to be estimated. This system should (1) be applicable world-wide, (2) be simple enough for practical usage, and (3) allow the collection of sufficient data for each entity. Therefore the world's vegetation zones have been taken together into a number of 'physiotopes' which are combinations of height above sea level and geographical latitude. Tentative ecosystem types for the physiotopes are given in Table 1. Ideally, a three-dimensional matrix should be constructed with as its axes: (1) geographical latitude, (2) elevation above sea-level, (3) level of human interference. In practice it is not possible to fill all resulting 'cells'. In the following, the three-dimensional matrix is reduced to three planes of human interference: 'zero human interference', 'human exploitation', and 'urban'.

In the process of data collection, the physiotopes have only been used as a guideline. In practice, a single physiotope as defined here may have different α values according to its geographical position. Therefore the α values are presented both per physiotope and on a world map.

Table 1: Definition and ecosystem types of physiotopes. The horizontal axis represents meters above mean sea level, the vertical axis represents geographical latitude.

A. zero human interference

80	ocean	coastal sea	tidal area, saltmarsh etc.	arctic tundra	bare	bare
60	I	I	I	boreal forest	arctic-alpine tundra	bare
40	I	I	I	temperate forest	montane forest	alpine tundra
30	I	I	I	desert	desert	desert
20	I	I	I	savannah		
0	ocean	coastal sea	tidal area, saltmarsh etc.	tropical rainforest	tropical montane forest	alpine tundra
	<-100	-100	ca 0	0 - 1000	1000-3000	>3000

B. 'human exploitation'

80	ocean	coastal sea	tidal area, saltmarsh etc.	arctic tundra	bare	bare
60	I	I	I	plantation forest	arctic-alpine tundra	bare
40	I	I	I	agricultural area	montane plantation forest, montane agricultural area	alpine tundra
30	I	I	I	desert	desert	desert
20	I	I	I	savannah		
0	ocean	coastal sea	tidal area, saltmarsh etc.	tropical secondary forest, tropical agricultural area	tropical montane forest	alpine tundra
	<-100	-100-0	ca 0	0 - 1000	1000-3000	>3000

C. Urban

80						
60		harbour	canal	urban, industrial	urban, industrial	urban, industrial
40		I	I	I	I	I
30		I	I	I	I	I
20		I	I	I	I	I
0		harbour	canal	urban, industrial	urban, industrial	urban, industrial
	<-100	-100-0	ca 0	0 - 1000	1000-3000	>3000

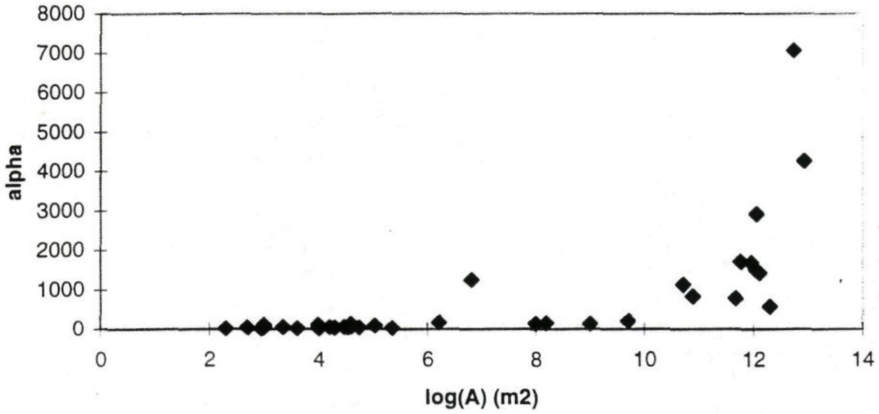
2.4.2 Estimation of α values

Tropics

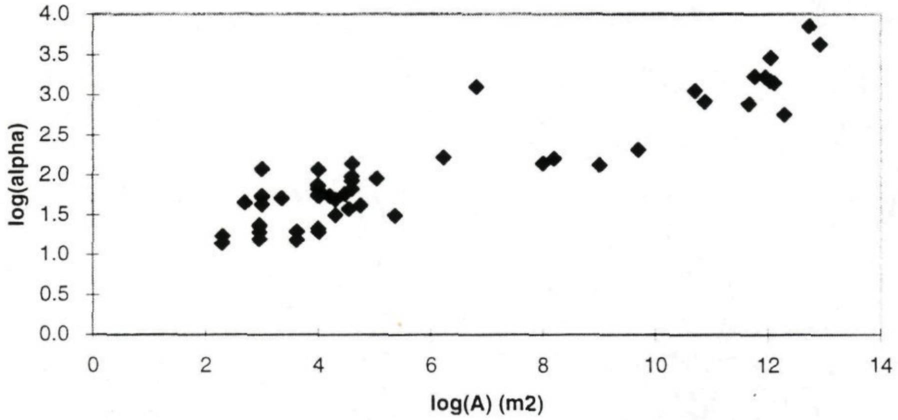
The publicly available scientific literature had been used as the primary source of data. Emphasis has been on syntheses of data and reviews. The sources scanned are enumerated in Appendix 1. Values for the biodiversity indicator α were calculated on the basis of these data according to Eq. (3).

The working hypothesis that α is independent of A (the area used to determine α) was tested by plotting α versus log(A) (Figure 2). On the basis of this plot, the data could be divided into two groups:

- 'small' area (< 10^{10} m^2 =100*100 km), α approximately constant between 10 - 200;
- 'large' area (> 10^{10} m^2), α >> 500 and strongly increasing with LOG(A).



(a)



(b)

Figure 2: relation between α and surface area determined on the basis of Appendix 1: (a) α on a linear scale, (b) α on a log scale.

Data in the first group are usually from individual reserves, whereas data in the second group are from whole countries or even (parts of) continents. The explanation for this phenomenon is probably that the linear relationship between α and LOG(A) only holds when species turnover is close to zero. This is the case if the considered area does not contain large discontinuities in abiotic circumstances, i.e. within a sinlge physiotope. If the considered areas are so large that they do not contain overlapping sets of species, each new area added will also add a new set of species (Harte & Kinzig 1997), and the assumption of a zero species turnover does no longer hold.The data from the second group are therefore not suitable for use in our concept, as they can not be used to determine α for a single physiotope. Therefore this group was further left out of consideration.

The data from the first group ('small' areas) were used to calculate a mean α for tropical forest. It should however be realised that these data have a number of strong limitations that may require further study before they can be used for generalised applications:

- not all data are related to primary forest, and this is not always stated in their sources;
- most data only apply to trees, but this not always clear from the sources. Often there are statements like 'only woody individuals with a diameter breast-high (dbh) > a certain value (see Appendix 1).

Omitting herbaceous species strongly affects α. This limitation is dealt with in the next section.

Estimation of total species number on the basis of the number of woody species. An attempt has been made to estimate the total number of species on the basis of the number of woody species. From Appendix 1 sources were selected in which total number of species and number of woody species were both mentioned. From the total number of species about 33% are woody species (Table 2). Jacobs (1981) also mentions that the number of tree species is about one third of the total number of species. Therefore the total number of species was derived as a function of the number of woody species An α value has been calculated on the basis of Appendix 1, using only the data from 'small' areas ($<10^{10}$ m^2), and corrected for the effect of using woody species only (total number of species = 3 * number of woody species). This yielded an average alpha for tropical rainforest of 103 (n=55).
Jacobs (1981) gives the following estimates for number of species per ha:
'normal' : 200 (α = 48)
'high' : 400 (α = 98)
'exceptional' : 600 (α = 148)
In summary, for the tropics an average alpha of 100 can be used; for exceptional species-rich rainforests (such as Asia) the alpha can go up to 150.

Table 2: Total number of species as a function of number of tree species (woody species) in the tropics

source	number of tree species	total number of species	%	nr. in App. 1
Kapelle, Kennis en De Vries 1995	52	176	30	5
Kapelle, Kennis en De Vries 1995	43	145	30	6
Kapelle, Kennis en De Vries 1995	43	130	33	7
Kapelle, Kennis en De Vries 1995	39	96	41	8
Meijer 1959. In : Jakobs 1981	78	331	24	40
Prance, In: Hawksworth 1995	291	1318	22	64
Prance, In: Hawksworth 1995	154	1033	15	65
Prance, In: Hawksworth 1995	784	1119	70	66
			mean: 33	

Relationship between annual rainfall and species diversity in tropical rainforests. Species diversity in tropical forests is determined by many variables. Among these, annual rainfall is often stressed in literature as the most important one. According to Wright (1996), '*rainfall is maybe the most critical abiotic variable in tropical forests*'. Recent work has shown that precipitation is strongly (direct and indirect through soil fertility) correlated with species diversity in Costa Rican forests for trees >10 cm dbh (Huston 1980), and for all vascular plants, including trees, understorey plants, and epiphytes over a range of Neotropical (Gentry 1982) and African (Hall and Swain 1976) forest sites. The species richness of trees increases five or sixfold as annual rainfall increases from 1000 to 4000 mm in the Neotropics and from 750 to 1750 mm in Ghana (Hall and Swain 1981; Gentry 1988).

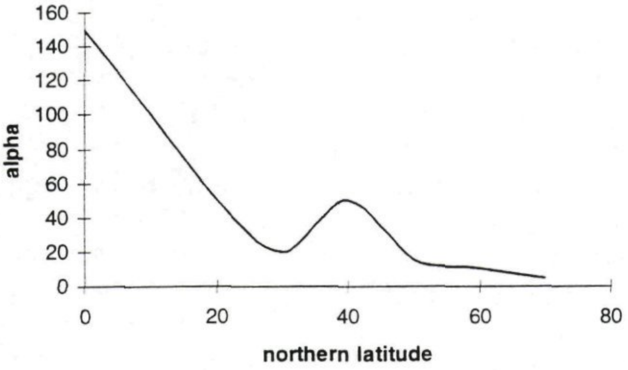


Figure 3: plot of α vs. geographical latitude at sea level (after Orians et al. 1996 and data in Appendices 1 and 2)

There are a number of classification systems for tropical forest on the basis of rainfall. Tropical rain forest occurs wherever precipitation is sufficient, generally above 1500-2000 mm/yr (Walter 1973, Holdridge 1967). The distribution of rainfall throughout the year is at least as important as the total amount (Walter 1973, Stephenson 1990). Tropical rain forest (sensu Schimper 1898, 1903) is the evergreen forest of the humid lowlands where rainfall is distributed evenly throughout the year. With increasing seasonal variation in precipitation, most trees shed their leaves for part of the year. These forests are variously called monsoon forests or tropical dry forests. Many other types of (sub)tropical forests can be defined, based on variation in precipitation, elevation, etc. One of the systems classifying tropical forests is the Holdridge Life Zone System (Holdridge 1947, 1967). However, this system contains too much detail to be useful for LCA in its present form.

Non-tropics

Outside the tropics a distinction can be made in Mediterranean, temperate and boreal areas, with different levels of species richness (Figure 3). Data availability is far better in Europe than outside Europe, but the general level of species richness is lower in Europe than in other areas. Appendix 2 gives an overview of data scanned. The 'small area vs. large areas' problem is encountered here like in the tropic. Data in Appendix 2 have been divided according to this criterion. Values for α seem to be far lower here than in the tropics; a tentative value of c. 10 ± 5 for the temperate and boreal region, and c. 30 ± 10 for the Mediterranean can be derived from Appendix 2. These values have been incorporated into the map in Appendix 4.

2.5 Syntheses of biodiversity component in LCA

Table 3 gives tentative figures for α per physiotope in the 'pristine' situation, i.e. without human interference. **Appendix 4 is a rough map of α on a world scale**, taking account of (a) precipitation, and (b) general between-continent differences in biodiversity. For the tropics, the values are determined based on the review studies in Appendix 1.The data on which Table 3 and Appendix 4 are based are summarized in Appendices 1 and 2.

The α values in Table 3 are without incorporation of rainfall- and continent-specificity. The map of Appendix 4 includes these and is therefore to be preferred for practical use. Data from table 3 are to be used when application of the map seems invalid, for instance due to large height differences within an area.

Table 3: Estimates of α for physiotopes. Bold: figure supported by relatively large amount of literature data; Normal: tentative figure.

Latitude

80	**<5**	**0**	**0**
60	**10**	**15**	**0**
40	**15**	**25**	0-10
30	10-40	15	10
20	50-75	25	15
0	**100**	**35**	20
	0 - 1000	1000-3000	>3000

Altitude

3 Productivity as a measure for degradation

The productivity of the land use system under study might be a measure for the degradation of the natural system. However, productivity can be assessed by different parameters and it is likely that one single parameter will not give satisfactory results for all circumstances. We therefore test different parameters and discuss the results and applicability of each. This evaluation will especially concentrate on feasibility of two methods: the methodology proposed by Blonk et al. (1997) expressing the loss of free NPP.and the methodology suggested by Sas et al. (1996), expressing the loss of NPP and including the regeneration time. To be feasible, the required data have to be 1) existing, 2) accessible, 3) widely accepted, 4) reliable and 5) the natural variation for specific sites should be small (± 10%).

3.1 Definitions of Net Primary Production (NPP)

In order for a parameter to be feasible, one widely accepted definition should exist for it, and data to assess this parameter should be collected in a standard way. However, even though one widely accepted definition exists on NPP, there is no single answer to the question: 'What is the productivity of the ecosystem at study site A?', rather there may be a range of estimates of NPP, depending upon what data were actually collected in which ecosystem and how these data were processed.

NPP (Net Primary Production) = Gross Primary Production minus respiration of vegetation per unit area. For a given period, this is equal to the change in plant mass plus any losses due to death and decomposition, measured for both above- and belowground plant parts. This can be quantified in the field by summing up the net increase (or change) in plant biomass plus litterfall (assuming that an equal amount of litter is added to the litterfloor as decomposes during the time interval) plus consumption (by e.g. herbivores).
Earlier estimates of NPP of grasslands were based on peak standing dry matter only, and the estimates of the International Biological Programme (IBP) (Reichle 1981) in the late 1960's and early 1970s were based mainly on above-ground biomass changes, with few estimates of belowground production.
Peak aboveground biomass (or in some cases the difference between maximum and minimum biomass) has been used as an estimate of net primary production (usually where only one or two estimates per growing season were carried out) in grasslands.

Aboveground NPP = max {Aboveground Biomass} (4)

Assumptions:

- Any standing dead matter or litter was carried over from the previous year, and death in the current year in negligible;
- Live biomass was not carried over from the last year;
- Belowground production is ignored or roughly estimated from a root/shoot ratio.

This method may be applicable to annual crops, but is clearly a poor estimate of production in perennial vegetation. It may be useful for comparison of seasonal temperate grasslands but has little meaning for tropical grasslands and should definitely not be used to compare tropical and temperate grasslands.

Two variations have been in use for the above given method:

Aboveground NPP = max {Aboveground clipped biomass} (5)

(In this method all the plant material is collected instead of just the live biomass in the first method)

Assumptions:

- any standing dead matter was formed by death in the current year, hence counts as part of the production of this year;
- no standing dead matter has yet fallen as litter or decomposed;
- *neither live nor standing dead mater were carried over from the previous year;*
- Belowground production is ignored or roughly estimated from a root/shoot ratio.

This method may give a slightly better estimate of NPP where significant death occurs during the growing season.

Aboveground NPP = max {Aboveground biomass} - min {Aboveground biomass} (6)

Assumption:

- In this way, any live standing biomass which may have been carried over from the previous year is excluded.

Conclusion: substraction of minimum biomass during a cycle of seasons may be a useful correction.

The 'IBP Standard Method' assumes that where live biomass increases between successive samples, production equals this increase; where biomass decreases or remains the same, production is assumed to be zero. In particular the peak biomass method and variations on the IBP method underestimate production by not accounting for simultaneous growth and death.

NPP = sum {positive increments in Aboveground biomass} (7)

Assumptions:

- most growth occurs between successive sample intervals, i.e. simultaneous growth and death do not occur;
- NPP is never negative during a sample interval;
- belowground production may be similarly measured, ignored or estimated based on a root/shoot ratio.

Conclusions: this method allows for distinct phases of growth within a year but still fails to account for new shoot growth during periods of high mortality.

Modified IBP standard method:

NPP = sum {growth increment} (8)

where growth increment = positive increment in aboveground biomass, unless aboveground total dead matter increases for that sample interval in which case the growth increment = positive increment in aboveground biomass plus positive increment in aboveground dead total.

Assumptions:

- simultaneous growth, death and decomposition does not occur;
- NPP is never negative during a time interval;

- belowground production may be similarly measured, ignored or estimated based on a root/shoot ratio.

Conclusion: the correction for material lost by death during periods of biomass increase will reduce the underestimation of NPP

'UNEP project' method:

NPP = sum {change in aboveground biomass + change in aboveground total dead matter + (aboveground relative rate of decomposition * aboveground total dead matter)} (9)

Assumptions:
- measured changes are statistically significant over each time interval;
- decomposition rate is independent of the composition of organic matter;
- losses of aboveground biomass and aboveground total dead are negligible;
- belowground production may be similarly measured, ignored or estimated based on a root/shoot ratio.

Conclusion: this is the only method which incorporates all components required for an accurate estimate of NPP.

The outline given above of varying definitions of NPP and the way to assess it in the field, shows how variable data may be only because of different ways of collecting. As soon as data have been scaled up and given the parameter name 'NPP' it may be almost impossible to retrieve the original way of collecting data. It is unclear how important this methodology caused variation is compared to the natural variation.

3.2 Natural variation in NPP

Another cause in possible variation in NPP and biomass lies in the existing natural variation. This natural variation which is caused by differences in soil fertility, climate etc. can be very large and contributing to uncertainty about the quality of the data. Even when a site is very well (narrowly) defined natural variation of as much as 10% may still exist. Figures 4 and 5 present standing biomass and NPP data for one ecosystem type (mature tropical lowland rainforest) from Cannell (1982).

The figures show that within a rather well defined ecosystem, the variation is still very large, even within a single region. Largest variation is found in the biomass data which vary between 179 and 872 Mg/ha, but also the NPP data vary with almost a factor 3 between 11 and 31 Mg/ha.yr. It is uncertain to what degree this variation is caused by methodological differences and to what degree by natural variation. However, it is certain that the site quality within one such defined ecosystem is very variable. Therefore, general compilations of NPP and biomass data per ecosystem always present a wide variation in possible values (Table 4).

Even though these data on NPP per ecosystem type present a large variation, the uncertainty for a specific site can be rather small. But, to be sure about the local circumstances local data would have to be assessed, because it is unlikely that data for a very specific site exist. In Figure 4, only 34 data were available from a thorough compilation for a biome which may cover 1.7 billion ha. It is clear that the data can easily be biased.

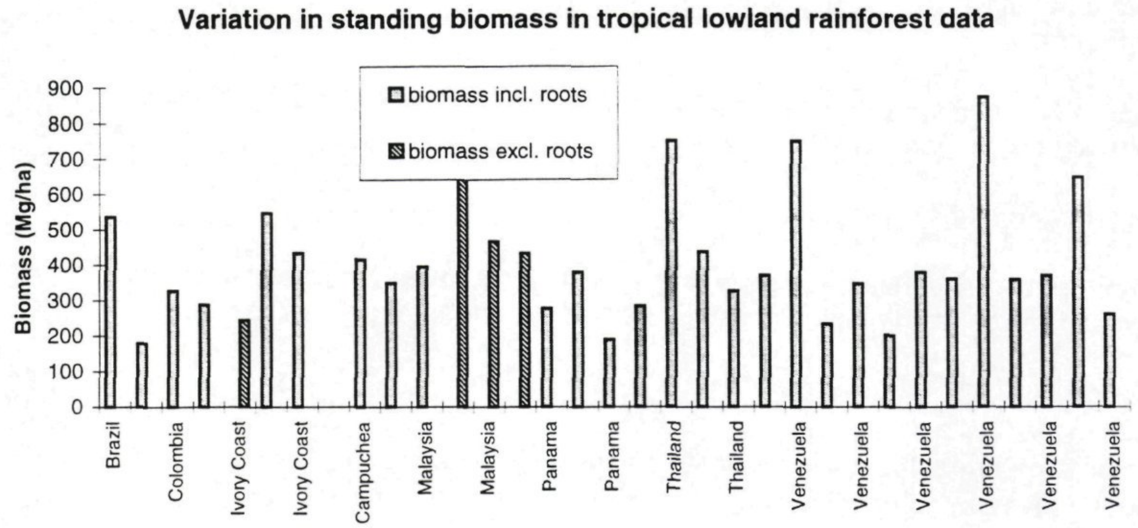


Figure 4. Biomass data for mature tropical lowland rainforest (Cannell 1982).

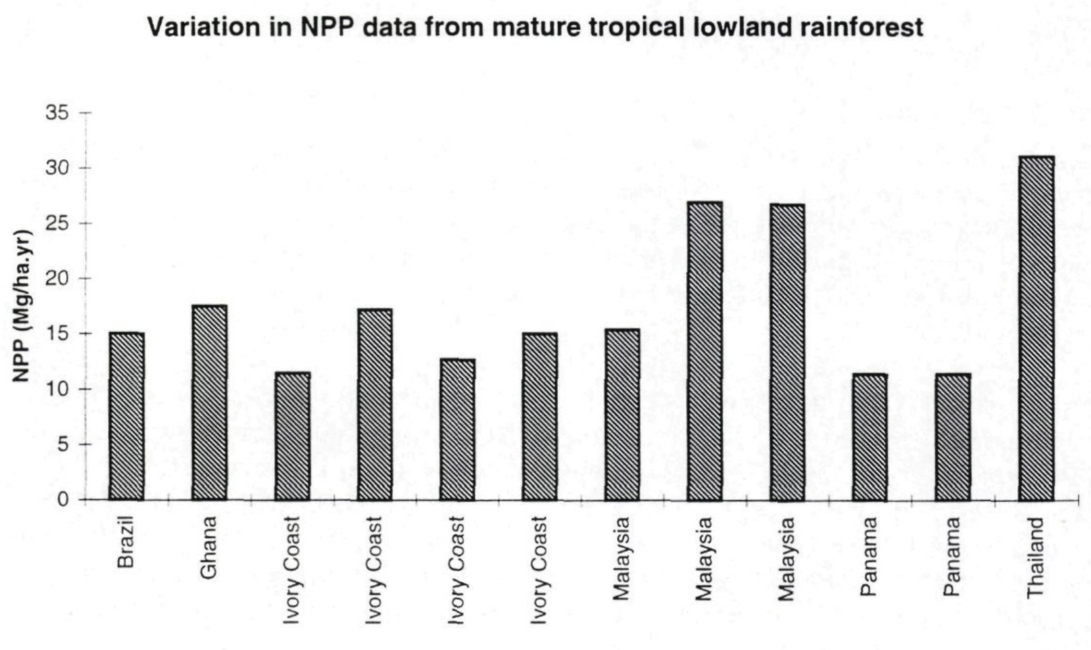


Figure 5. Net primary production data for mature tropical lowland rainforest (Cannell 1982).

Table 4: NPP in natural ecosystems (Mg $ha^{-1}.yr^{-1}$) (Ajtay et al.1977)

Ecosystem	Range in Net Primary Production	Mean Net Primary Production
tropical rainforest	10 - 35	22
tropical seasonal forest	10 - 25	16
temperate evergreen forest	6 - 25	13
temperate deciduous forest	6 - 25	12
boreal forest	4 - 20	8
woodland and shrubland	2.5 - 12	7
savannah	2 - 20	9
temperate grassland	2 - 15	6
tundra and alpine	0.1 - 4	1.4
desert and semidesert scrub	0.1 - 2.5	0.9
extreme desert	0 - 0.1	0.03
cultivated land	1 - 40	6.5
fertilised meadow	3 - 65	9.1
swamp and marsh	8 - 60	30

3.3 Data availability, and accessibility

A few thorough compilations of biomass and NPP data exist and are widely accepted and used. These are Olson et al. (1983) and Milleman and Boden (1985) with 1545 records of carbon in live vegetation of world ecosystems, Cannell (1982) a compilation of biomass and NPP data per plant compartment of 1200 forest stands in 46 countries, and DeAngelis et al. (1981) with the compilation of the data obtained in the International Biological Programme carried out in 116 forest stands. These compiled sets are generally accepted as being the best sets of data available although it is usually acknowledged that the choice of the location of the sample plots is not always a good representation of the variability between the sites. This is because the aim of many ecological sampling procedures was not to find the most representative spot but to find the high stocked, impressive mature forests.

Apart from these compilations it is very likely that many more biomass measurement studies have been carried out, but were never published and therefore never included in one of these compilations. It will be a difficult task to find and obtain those unpublished records. Another source for related biomass data can be found in growth and yield tables for forest management and national forest inventory results. Although these studies never cover the total tree biomass (they usually only contain stem volume and increment data), they are based on long term monitoring of a representative set of sample plots.

Especially the sample plots for growth and yield tables (management guidelines) have been monitored for a long time and the guidelines represent the variety in site qualities. These data are therefore highly reliable and present growth and volume data per site class. The variety within the site class is thus small, but the data are always measured on managed stands. These tree volume data can easily be converted to total tree biomass using conversion coefficients based on the above mentioned sets of biomass data. Published growth and yield tables are available for many (usually developed) countries.

Reliable national forest inventory data exist for almost all European countries, North American countries, countries of the Former Soviet Union, a small number of developing countries and e.g. countries like Australia, New Zealand, Chile, Argentina. These national forest inventory data are based on a much larger number of sample plots per country, but have not been monitored as intensively as the plots for the growth and yield tables. For many countries the plots have only been measured once or twice. For few countries only, the national forest inventories date also back to the beginning of this century.

3.4 Possible methods to deal with productivity in land use as a component of LCA

Here we will deal with two proposed parameters as a measure for land use impacts via biomass productivity. These are: 1) annual production in terms of NPP linked to regeneration times (according to Sas et al, 1996) and 2) the loss of free NPP (according to Blonk & Lindeijer,1995).

3.4.1 Annual production in terms of NPP and biomass regeneration times

Two factors are proposed in the approach of Sas et al: the biomass productivity and the regeneration time. The two factors will be discussed consecutively.

NPP

In this method the long-term average level of Net Primary Production is valued as such. NPP is not regarded as a measure for natural values, but it can be compared to the natural situation. A possible operationalisation (characterisation in LCA terms) can be found by expressing the degree of deviation from the natural situation. In this method we rely on the limited number of available data on natural systems like in Cannell (1982) and compare them to the modified system. We assume that the natural system is in equilibrium which results in a stable 'background' NPP.

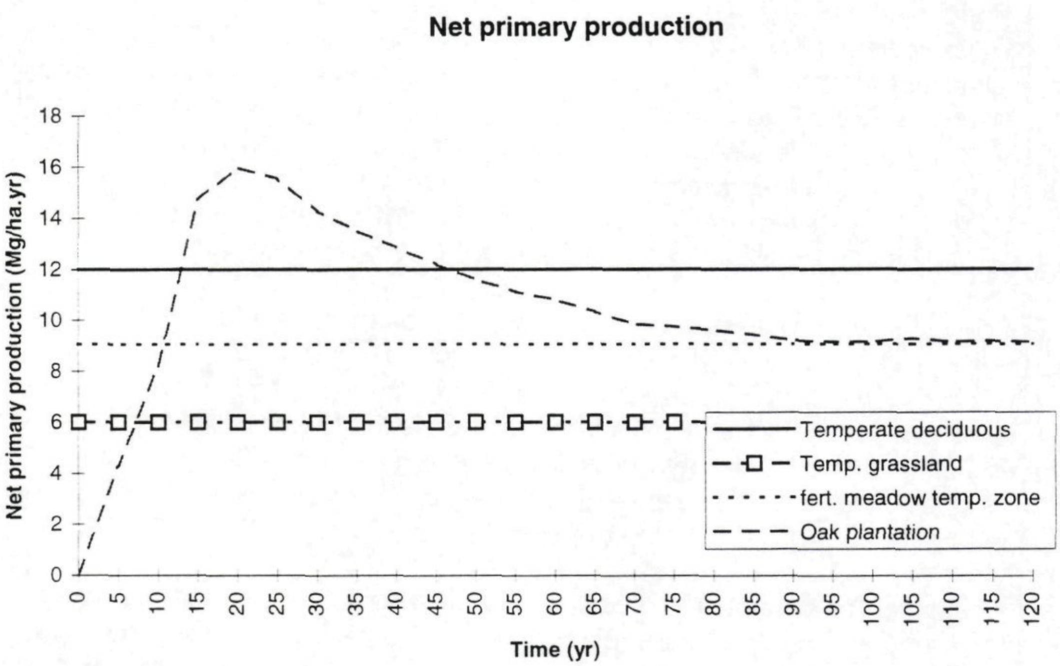


Figure 6. Comparison of Net Primary Production in two natural situations (straight line and dashed line with symbols) with one modified forest situation (oak plantation) and a modified grassland situation, respectively.

Figure 6 gives a general outline of what will usually be found when a natural system is compared to the modified situation. The growth of oak plants reaches a peak value around the age of 20 years. After that, the NPP decreases, but stabilises. The average NPP for the oak plantation over a rotation of 120 years amounts to 10.3 Mg/ha.yr and is less than in the natural situation, but the peak value is higher. The NPP of the fertilised meadow is clearly higher than in the natural situation and is the same every year.

However, the largest uncertainty lies in the values for the natural situation. As discussed earlier, the NPP values of the modified systems can be assessed rather accurately. Based on the large number of growth and yield measurements, for every specific site relevant data can be found and accuracy will be rather high, while the variation within the site is small. However, for the natural situation, only few data exist and general averages will have to be chosen. To function in an LCA system, for these natural situations generally accepted values which are to be used for all studies, would be required. A suggestion for such values is given in the map of Appendix 5.

Regeneration time
Next to the loss of NPP as a measure for the extent of deterioration of an ecosystem, the regeneration time (time required to reach the maximum potential biomass) can be used as a measure for inelasticity of an ecosystem (Table 5). It also expresses the vulnerability of a system. In Sas et al. (1996) these two factors are suggested to be combined in one formula to express the loss of nature value (in their study as side effect of the extraction of biotic resources; here treated separately). For the sake of this combination, the biomass regeneration time is estimated.

Table 5 gives tentative numbers for the regeneration time required to reach the maximum potential biomass after a system has been 'cleared', under the assumption of no soil degradation.. These numbers cannot always be given in general for a system, because they depend on the type and degree of deterioration. Therefore care should be taken in using the data below as generic numbers for the inelasticity of a system. Another remark to be made is that these numbers just represent the time to reach the full stocked biomass again, they do not represent the time to reach full maturity in biodiversity aspects (which might take up to about 300 years).

Table 5: Regeneration times (years) to reach potential biomass in physiotopes by altitude and latitude (column).

Latitude

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

Altitude

3.4.2 Loss of free NPP method (fNPP; method Blonk & Lindeijer (1995)).

Blonk and Lindeijer (1995) define degradation of ecosystems as 'the diminishment of development space for nature'. They operationalise this through the quantification of loss of free Net Primary Production (NPP) thereby assuming that the free NPP (fNPP) represents that part of the NPP which remains for nature e.g. through additions to the soil organic matter. They suggest that fNPP is a measure of the possibility to contribute to natural life support functions (via freely available biomass in or on the soil). Additionally, they suggest to take into account the fragmentation of nature by calculating the ratio between the circumference of the area which is affected by the land use and its minimum possible circumference of the same surface i.e. a circle (Blonk et al., 1997).
The loss of free NPP is calculated as follows:

$$fNPP_{(loss)} = NPP_{(ref)} - (NPP_{(act)} - \text{yielded biomass}) \quad (10)$$

where
$fNPP_{(loss)}$ = the loss of free NPP in the land use system compared to the natural situation;
$NPP_{(ref)}$ = average Net Primary Production in the reference situation (natural situation);
$NPP_{(act)}$ = average Net Primary Production in the particular land use situation
yielded biomass = the amount of the total Net Primary Production which is harvested for human use.

Appendix 3 gives for a number of case studies the NPP in the reference situation and in the land use system. This appendix gives all relevant basic data used to derive the free NPP values in Table 6. The loss of free NPP as NPPref-fNPP is also given.

Table 6: NPP in reference situations, in land use systems on comparable sites and the (loss of) free NPP (Mg/ha) for those cases

Land use system	Commer-cial forest of Norway spruce in Central Europe	Commer-cial forest of Norway spruce in boreal zone	Commer-cial oak forest	Shorea robusta plantation, India	Eucalypt plantation	Tectona grandis, Indonesia	Mine	Agricultur e (wheat)
No of years	50	90	100	38	40	70	100	50*)
NPPref	605	378	1060	657	280	1211	90	605
Land use NPP	750	324	1030	448	432	1008	0	650
Actual used NPP (thinning, stems, harvest etc.)	257	268	455	350	191	660	0	350
Free NPP under land use	493	56	575	98	241	348	0	300
Loss of free NPP	112	322	485	559	39	863	90	305
Annual loss of free NPP (Mg/ha.yr)	2.24	3.58	4.85	14.71	0.98	12.33	0.9	6.1

*) Figures are 50 times annual NPP etc. (for the sake of comparison with forest)

3.4.3 Discussion of approaches for including biomass productivity measures in LCA

Like for the 'Annual production of NPP' method, the uncertainty in the 'loss of free NPP method' lies also in the reference ('present background') situation. For that part we rely again on the limited set of data available. Because of uncertainty in that part of the method, the quantification of loss of free NPP becomes uncertain as well. Another choice for the natural system could yield significantly different results. The uncertainty in the land use systems is far less. For that part we rely again on the growth and yield tables, and e.g. agricultural data.

Apart from the uncertainty in the reference situation (which could be taken away by selecting a widely accepted set of reference situations; see Appendix 5 for suggested data on world-scale), the loss of free NPP method seems to work very well. In general: where the land use system yields a lower NPP than the reference and where a large part of the land use NPP is used, the free NPP will be reduced significantly. In case of mining all free NPP is lost. This implies that mining is 'worse' in tropical rainforest than in desert, because in the first case, more free NPP is lost.

In theory it could be possible to have more free NPP in the land use system than in the reference situation. E.g. if in the land use case, we manage to produce more NPP than in the reference situation and we harvest only part of the land use NPP, we may be able to have more free NPP in the land use case than in the reference situation: we increase free NPP. Such a situation is hard to imagine as maximum biomass production for human consumption is what is generally strived for. However, the case of changing an oligotropic ecosystem (with low natural biomass production) to a high-production land use system leaving more free NPP than in the natural ecosystem is a clear example of this contra-intuitive situation. This does not mean that the methodgives wrong results; they could still be a good representation ofthe change in nature development potential. This is not the same as the total change in nature value, as the intrinsic nature value as judged by biodiversity indicators may show a strong decrease in the case of changing an oligotropic ecosystem to a human biomass production system. Thus, the productivity parameters do not seem to be the sole solution to characterise the degradation of land under certain land use. They must always be seen as part of a complete LCA. Although the free NPP method seems promising (assuming that the uncertainty in the reference situation can be reduced), no single system parameter can give a 'good' measure of the degree of land degradation. These production parameters must always be part of a broader set of criteria and can as such contribute to a complete LCA.

3.5 Estimates of NPP for physiotopes

Table 7 gives figures for NPP in natural ecosystems per physiotope (Mg $ha^{-1}.yr^{-1}$) (NPP=fNPP). This table is comparable to the one for biodiversity in terms of α (Table 3).The data are based on the literature searched for the cases and for drawing the map in Appendix 5, which is a rough world map of NPP, comparable to the world map of α (Appendix 4). The data for the map are derived from the following sources: Milleman and Boden (1982), Deangelis et al. (1981), Cannell (1982), Smith et al (1992), Prentice et al. (1992) and King et al. (1997).

Table 7: estimates of NPP (Mg/ha/y))for physiotopes

Latitude			
80	<=1	0	0
60	8	3	0
40	12	8	2
30	<=1	<=1	<=1
20	16	8	3
0	22	12	4
	0 - 1000	1000-3000	>3000

Altitude

4 Height and aboveground biomass as evaluation criteria for effects on landscape.

For the evaluation of effects on landscape, it is proposed to use height (m) in combination with the area and aboveground standing biomass (tonnes / ha) as crude evaluation criteria. A first approximation of these figures is given in Table 8.

Table 8: Analysis of landscape per physiotope

physiotope	height (m)	aboveground biomass (t/ha)
aquatic (all forms)	0	-
saltmarsh	0.5	5
tundra (all forms)	0.2	3
boreal forest	15	100
temperate forest	30	300
montane forest	20	100
tropical rainforest	40	400
temperate plantation forest	30	200
tropical secondary forest	30	300
desert	0	0
savannah	20	50
agricultural	1	10
urban/industrial	10-50	0-10

5 Data for land use cases

Table 9 and Table 10 give the estimated α and NPP values for the testcases to be evaluated for occupation and change, respectively. The 'pristine' or reference α and NPP values have been taken from Table 3 and Table 6, respectively. Some activities reduce α and NPP to zero values by removing *all* vegetation. However, other activities allow a 'residual' α or fNPP, because they do not completely remove vegetation but only lead to a 'degraded' vegetation. Therefore, reduction factors for α in such degraded situations had to be estimated, due to lack of more specific measured data for these cases. For this, the situation in The Netherlands, which is rather well-known, was taken as an example. Here the number of species per km^2, from which α can be computed, is known over large areas (Witte & Van der Meijden 1995). By comparing grid cells (of 1 km^2) that are almost completely in agricultural use with grid cell that are almost 'pristine', a reduction factor for α of 0.4 for intensive agriculture was estimated (Figure 7). Intermediate reduction factors were estimated for intermediate levels of human interference on the basis of expert judgement. These estimates are summarised in Table 11. It should however be stressed that these estimates are based on rather scanty data and deserve further validation.

The values for α and (f)NPP for the situation before the activity in table 9 are derived from reference values or from the cases in table 10, depending on the most probable situation before such an activity in Europe.

With the data of table 9, formulas to determine land use effects in LCA can be filled (see main text of this report).

Table 9: Evaluation of α and (f)NPP (Mg/ha/y) for some land use changes

Testcases	alphax (ref).	alpha (before)	alpha (occu-pation)	alpha (after)	NPP (ref)	NPP (before)	NPP (occu-pation)	NPP (after)
Extracting sand in Europ. agricultural land	10-15	5	0	0-8 [1])	8	8	0	0-8 [1])
Mining aluminium ore in S.-Am. Tropical forest	100	100	0	80 [2])	17	17	0	17 [2])
Landfill household waste in Europ. agricultural land	10-15	5	0	9 [2])	8	8	0	10 [2])

[1]) recreation ground assumed after occupation; α (after) strongly dependant on local situation (depth of sand pit, steepness of shore)
[2]) secondary forest assumed after occupation

Table 10: Evaluation of α and NPP for cases of land occupation

Testcases	alpha (ref.)	alpha (occup.)	NPP (ref.)	NPP (occup.)
Industrial production in Europ. agricultural land	10-15	0-10	8	1
Hydropower in Scandinavian hills	10	0	7	0
Road traffic in Europ. agricultural land	10-15	5	8	2
Harvesting wood in Scandinavian hills	10	7	7	7

Table 11: Reduction factors of α in degraded ecosystems

degradation stage	level of human influence	reduction factor of α
'pristine'	*none*	1
secondary forest plantation forest extensive agriculture etc.	intermediate	0.8
urban industrial road & railroad verges recreation ground military area etc.	strong, but locally low; abiotic diversity high	0.6
intensive agriculture	strong, abiotic diversity low	0.4
sealed	no vegetation possible	0

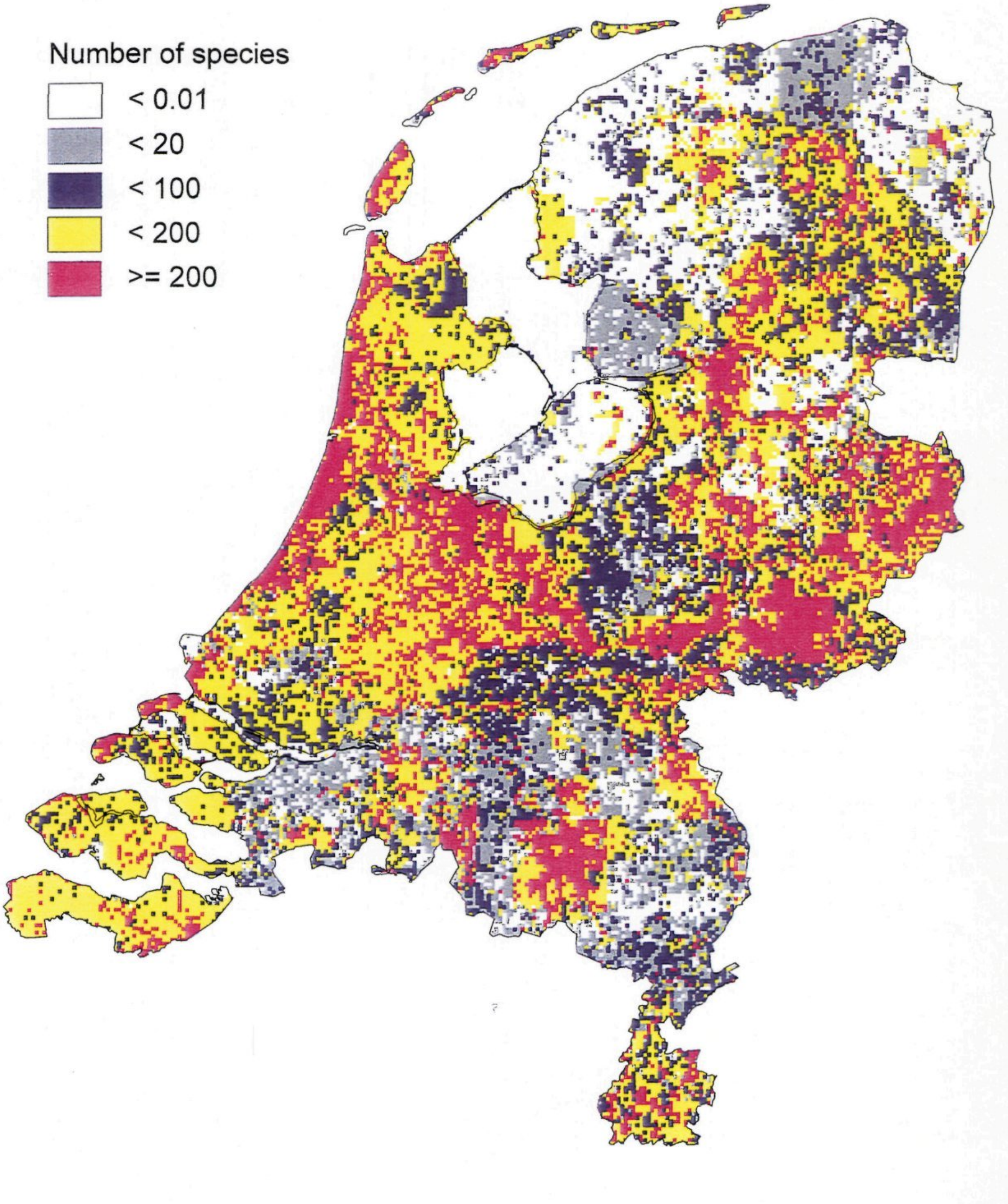


Figure 7: number of vascular plant species per Km^2 in The Netherlands. Taken from: Wite & Van der Meijden (1995). This map is derived from FLORBASE. FLORBASE is a database containing observations of plant species on a 1x1 km scale. The database consists of data from provinces, private persons, terrain management organizations and institutes. Data originate from a wide variety of sources, often inventories by volunteers. The quality of the data is variable, but probably quite good in the species-rich areas. Areas with <100 spp/km^2 (gray, blue, white) are probably insufficiently inventoried and should not be considered. 'Red' contains natural areas (dune, NW Overijssel, Strabrecht, etc.); yellow mostly contains intensive agricultural areas, although in The Netherlands some natural areas are quite specie-poor as result of human degradation in historic times (heathland etc.). For natural areas in 'red', 250 spp./km^2 was assumed, for agricultural areas in 'yellow', 100 spp./km^2. The reduction factor for α in agricultural areas can then be estimated as $\{(100\text{-}10) / \{(250\text{-}10) / \mathrm{LOG}(10^6)\} \approx 0.4$.

6 Topics for further research

An important drawback for the application of the characterisation approach developed in this study is that not for all cases data are readily available. In this section some directions are given how to deal with this problem for α as a measure of biodiversity.

One approach is to further refine the reduction factors used to derive the α values for the cases (Table 11). Together with the regional α values as estimated in the present study, these reduction factors determine the residual α during an activity. The following could be used as a starting point to derive reduction factors:

The most important impact of land use is that land is (temporarily) made unsuitable as a habitat for a large variety of biota. Other impacts have an economic component (e.g., making land unsuitable for certain (other) human activities) and should not be part of an LCA, as LCA focuses on impacts of human activities on nature. There are two factors that may make land unsuitable for a variety of biota:

1. setback of succession: 'later' succession stages are usually richer in species than earlier stages due to simpler vegetation (examples of artificial earlier stages are forestry and agriculture); moreover, late succession stages are harder to create artificially or naturally than earlier stages;
2. changes in the abiotic environment; the most common changes are:
3. fertilisation
4. desiccation
5. use of pesticides
6. habitat fragmentation
7. soil degradation & compaction
8. (pollution);

Ad 1. The setback of succession should be incorporated in generic reduction factors as those in Table 11. The factors derived there are first estimates. Based on available GIS data the robustness of these factors can be tested. Using data from other countries the validity of their use outside the Netherlands can be checked.

Ad 2. All changes in the abiotic environment that have been assessed in this study are related to the general intensity of land use. The level of fertilisation (e.g., in kg N or P $ha^{-1}.y^{-1}$) is a good measure for this intensity. This is a figure that is probably available world-wide on a geographical basis. This figure can be used to distinguish between different levels of agriculture or forestry intensity, thus further specifying the reduction factors to be used. Loss of biodiversity as a function of fertilisation level can be estimated from literature. A first approximation might be:

kg N /ha/j	reduction factor of α
< critical load (ca. 5-15)	1
<50	0.8
100	0.6
...linearly decreasing....	
400	0.2

These reduction factors could again be slightly altered when information on extent of desiccation or extent of pesticide use is known, based on studies on these issues. Remember that such factors are suggested to derive α values where no more specific data are available. They should not be used when double counting of impacts may occur. For instance, when ecotoxicity impacts of pesticides on species in ditches is included via other impact categories in LCA, the correction factor for pesticide use should not be used. Also, when desiccation or fragmentation is taken into account separately this should not be included here.

When land is used for biomass production, the method using reduction factors (outlined above) can be used to determine the residual biodiversity. Other cases are for instance mines, industrial and urban areas, roads etc. In principle, these are 'sealed surfaces' with biodiversity = 0. However, such areas usually contain small patches of wasteland with a low level of human interference (not harvested, no fertilizer etc.), and therefore have a biodiversity > 0 and sometimes even larger than intensively used agricultural land (cf. Table 11)! When detailing the α values for cases, the percentage of 'wasteland' areas should be determined incorporated in the general α value for each case.

Taking all these aspects together requires a fairly complicated model to evaluate the impact of land use on biodiversity. This would yield land-use-type specific models. For example mining: low or zero biodiversity in the areas where activities take place but a certain percentage of wasteland patches with a higher biodiversity. A realistic model should be developed by fine-tuning all these factors. Validation may take place by (1) expert judgement, or (2) existing data [e.g. in The Netherlands: FLORON database on individual species distribution (cf. Figure 7), or IBN-DLO vegetation database].

Finally, measurements in the field can be performed. For specific land use changes this is a realistic possibility which is occasionally applied in Environmental Impact Assessments. This can also be applied to collect data to fill in gaps in the above generic models. Direct measurement of species densities is rather easy to accomplish. For the purpose of LCA, direct measurement of the density of hotspots (in the sense of this chapter) would be even more useful, but is probably hard to accomplish. The reduction factors for degraded situations can also be validated in the field. Alternatively, a comparison of numbers of species inside and outside the Ecological Main Structure (EHS) can be made on the basis of data for The Netherlands available with FLORON. This will also give an indication for the effect of human activities on local biodiversity.

References

Agenda 21: programme of action for sustainable development : Rio declaration on environment and development : statement of forest principles: the final text of agreements negotiated by Governments, at the United Nations conference on environment and development (UNCED), 3 - 14 June 1992, Rio de Janeiro, Brazil [1993?]

Ahton, P.S. 1993. Species richness in plant communities. In: P.L. Fiedler & S.K Jain (eds.), Conservation Biology. Chapman & Hall, New York. 4-22.

Aptroot, A. 1997. Species diversity in tropical rainforest ascomycetes: lichenized versus non-lichenized; foliicolous versus corticolous. Abstracta Botanica 21:37-44.

Arrhenius, O. 1921. Species and area. Journal of Ecology 9:95-99.

Ashton, P.S. 1964. Ecological studies in the mixed dipterocarp forests of Brunei State. Oxford Press, Oxford. 75 p.

Balslev, H., J. Luteyn, B. Øllgaard & L.B. Holm-Nielsen 1987. Composition and structure of adjacent unflooded and floodplain forest in Amazonian Ecuador. Oper. Bot. 92: 37-57.

Beentje, H.J., B. Adams & S.J. Davis 1994. Regional overview: Africa. In: S.D. Davis & v.H. Heywood (eds.), Centres of plant diversity. In Press.

Black, G.A., T. Dobzhansky. & C. Pavan 1950. Some attemps to estimate species diversity and population density of trees in Amazonian forests. Bot. Gaz. 111: 413-425.

Blonk, H. and E. Lindeijer 1995 Naar een methodiek voor het kwantificeren van aantasting in LCA. Blonk Publikatiereeks grondstoffen Nr 1995/15. Ministerie van Verkeer en Waterstaat, Dienst Weg- en waterbouwkunde. IVAM, Environmental Research. Delft en Amsterdam. 43 p.

Blonk, H., E. Lindeijer, and J. Broers 1997 Towards a methodology for taking physical degradation of ecosystems into account in LCA. The International Journal of Life Cycle Assessment 2(2): 91-98.

Bongers, F., J. Popma, J. Maeve del Castillo & J. Carabias 1988. Structure and floristic composition of the lowland rain forest of Los Tuxtlas, Mexico. Vegetatio 74: 55-80.

Boom, B.M. 1986. A forest inventory in Amazonian Bolivia. Biotropica 18: 287-294.

Boyle, T.J.B. & B. Boontawee (eds.)1995. Measuring and monitoring biodiversity in tropical and temperate forests. Proceedings of a IUFRO Symposium held at Chiang Mai, Thailand. August 27th-September 2nd, 1994. 395 p.

Briggs, J.C. 1996. Tropical diversity and conservation. Conservation Biology , vol.10, no 3: 713-718.

Campbell, D.G., D.C. Daly, G.T. Prance & U.N. Maciel 1986. Quantitative ecological inventory of terra firme and várzea tropical forest on the Rio Xingu, Brazilian Amazon. Brittonia 38: 369-393.

Cannell, M.G.R. (ed.) 1982 World forest biomass and primary production data. Natural Environment Research Council. Institute of Terrestrial Ecology. Academic Press London New York. 391 p.

Chiarucci, 1996. Species diversity in plant communities on ultramafic soils in relation to pine afforestation. Journal of Vegetation Science 7: 57-62.

Crisp, P N, Dickinson, K J M, Gibbs, G W. 1998. Does native invertebrate diversity reflect native plant diversity? A case study from New Zealand and implications for conservation. Biological Conservation 83:209-220.

Croat, T.B. 1978. Flora of Barro Island. Stanford University Press. Stanford. 943 p.

DeAngelis, D.L., Gardner, R.H. & H.H. Shugart. 1981 Productivity of forest ecosystems studied during the IBP: the woodlands data set. In: Reichle, D.E. (ed.) Dynamic properties of forest ecosystems. International Biological Programme 23. Cambridge University Press. Cambridge etc. pp. 567-672.

Dodson, C. & A. Gentry 1978. Flora of the Rio Palenque Science Center. Marie Selby Botanical garden, Sarasota.

Emanuel, W R, Shugart, H H, Stevenson, M. 1985. Climatic change and the broad-scale distribution of terrestrial ecosystem complexes. Climatic Change 7: 29-43.

Faber-Langendoen, D & A.H. Gentry 1991. The structure and diversity of rain forests at bajo Calima. Chocó region, western Colombia. Biotropica 23: 2-11.

Fisher, R, Corbet, A S, Williams, C B. 1943. The relation between the number of species and the number of individuals an a ramdom sample from an animal population. Journal of Animal Ecology 12:42-58.

Gentry, A.H. 1988. Tree species richness of upper Amazonian forests. Proc. Natl. Acad. Sci. U.S.A. 85: 156-159.

Gentry, A.H. & C. Dodson 1987. Contribution of nontrees to species richness of a tropical rain forest. Biotropica 19: 216-227.

Gentry, A.H. 1982. Neotropical floristic diversity. Phytogeographicalconnections between central and South America, Pleistocene climatic fluctuations or an accident of the Andean orogeny? Ann. Missouri Bot. Gard. 69: 557-593.

Giegrich, J. and K. Sturm 1996 Methodenvorschlag - Operationalisieriung der Wirkungskategorie Naturraumbeansprachung. Institut fuer Energie- und Umweltforschung Heidelberg GmbH. 19 p + app.

Guariguata, M.R., R.L. Chazdon, J.S. Denslow, J.M. Dupuy & L. Anderson 1997. Structure and floristics of secondary and old-growth forets stands in lowland Costa Rica. Plant Ecology 132: 107-120.

Hall, J.B. & M.D. Swain 1981. Distribution and ecology of vascular plants in a tropical rain forest: forest vegetation in Ghana. Junk, The Hague. Geobotany 1: 383.

Hansson, L. 1997. Environmental determinants of plant and bird diversity in ancient oak-hazel woodland in Sweden. Forest Ecology and Management 91:137-143.

Harte, J., Kinzig, A. P. 1997. On the implications of species-area relationships for endemism, spatial turnover, and food web patterns. Oikos 80:417-427.

Hawksworth, D.L. (ed.) 1995. Biodiversity; measurement and estimation. Chapman & Hall, London. 140 p.

Heikkinnen, R.K. & S. Neuvonen 1997. Species richness of vascular plants in the subarctic landscape of northern Finland: modelling relationships to the environment. Biodiversity and Conservation 6: 1181-1201.

Holdridge, L.R. 1967. Life zone ecology. Tropical Science Center, San Jose, Costa Rica.

Hoogmoed, M.S. & R. De Jong (eds.) 1992. Tropisch regenwoud; schatkamer van biodiversiteit. Nationaal Natuurhistorisch Museum. 206 p

Hurlbert, S.H. 1971. The noncencept of species diversity: a critique and alternative parameters. Ecology 52:577-586.

Huston, M.A. 1980. Soil nutrients and tree species richness in Costa Rican forests.

Huston, M.A. 1994. Biological diversity; The coexistence of species on changing landscapes. Cambridge university press. 681 p.

Jacobs, M. 1981. Het tropisch regenwoud; een eerste kennismaking. Dick Coutinho, Muiderberg. 318 p.

Jansen-Jacobs, M.J. 1992. Neotropische flora en vegetatie. In: M.S. Hoogmoed & R. De Jong (eds.). Tropisch regenwoud; schatkamer van biodiversiteit. Nationaal Natuurhistorisch Museum. 69-83.

Johns, R.J. 1992. *Biodiversity and conservation of the native flora of Papua New guinea*. In: B.M. Bechler (ed.), Papua New Guinea conservation needs assessment. Papua New Guinea: Department of Environment and Conservation. vol. 2: 15-32.

Kapelle, M., P.A.F. Kennis & R.A.J. de Vries 1995. Changes in diversity along a successional gradient in a Costa Rican upper montane *Quercus* forest. Biodiversity and Conservation 4: 10-34.

Kemmers, R H, Van der Bolt, F J E. 1997. Fysiotopentypologie voor beekdallandschappen. Rapport SC-DLO 502. Wageningen. 37 p.

Kerr, J T. 1997. Species richness, endemism, and the choice of areas for conservation. Conservation Biology 11:1094-1100.

King, A W, Post, W M, Wullschleger, S D. 1997. The potential response of terrestrial carbon storage to changes in climate and atmospheric CO2. Climatic Change 35: 199-227.

May, R M. 1975. Patterns of species abundance and diversity. In: M L Cody & J M Diamond (eds.): Ecology and evolution of communities, 81-120. Belknap / Harvard University Press, Cambridge MA.

Medail F. & P. Quezel 1997. Hot-spots analysis for conservation of plant biodiversity in the mediterranean basin. Annals of the Missouri Botanical Garden 84: 112-127.

Meijer, W. 1959. Plantsociological analysis of montane rainforest near Tjibodas, West Java. Acta Botanica Neerlandica 8: 277-291.

Millemann, R.E. & T.A. Boden. 1985 Major world ecosystems complexes ranked by carbon in live vegetation: a database. Carbon Dioxide Information and Analysis Centre. Oak Ridge National Laboratory for the US Department of Energy. NDP-017. 164 pp.

Miller R.I. (ed.) 1994. Mapping the diversity of nature. Chapman & Hall, London etc. 218 p.

Monkkonen, M, Viro, P. 1997. Taxonomic diversity of the terrestrial bird and mammal fauna in temperate and boreal biomes of the northern hemisphere. Journal of Biogeography 24:603-612.

Mooney, H.A., J.H. Cushman, E. Medina, O.E. Sala & E.D. Schulze (eds.) 1996. Functional roles of biodiversity; a global perspective. J. Wiley & Sons, Chichester etc. 493 p.

Murphy, P.G. & A.E. Lugo 1986. Ecology of dry tropical forests. Ann. Rev. Ecol. Syst. 17: 67-88.

Neilson, R.P. 1995 A model for predicting continental scale vegetation distribution and water balance. Ecological Applications 5: 362-385.

Nooteboom, H.P. 1992. In: M.S. Hoogmoed & R. De Jong (eds.). Tropisch regenwoud; schatkamer van biodiversiteit. Nationaal Natuurhistorisch Museum. 103-119.

Olson, J.S., Watts, J.A. & L.J. Allison. 1983 Carbon in live vegetation of major world ecosystems. US Departement of Energy, Oak Ridge, TN. DOE/NBB-0037.

Orians, G.H., R. Dirzo & J.H. Cushman (eds.) 1996. Biodiversity and ecosystem processes in tropical forests. Springer Verlag Berlin Heidelberg, New York. 229 p.

Parga I.C., J.C.M. Saiz, C.J. Humphries & P.H. Williams 1996. Strengthening the Natural and National Park system of Iberia to conserve vascular plants. Botanical Journal of the Linnean Society 121: 189-206.

Parthasarathy, N. & R. Karthikeyan 1997. Plant biodiversity inventory and conservation of two tropical dry evergreen forestson the Coromandel coast, south India. Biodiversity and Conservation 6: 1063-1083.

Pastor, J, Downing, A, Erickson, H E. 1996. Species-area curves and diversity-productivity relationships in beaver meadows of Voyageurs National Park, Minnesota, USA. Oikos 77:399-406.

Peet, R.K. & N.L. Christensen 1988. Changes in species diversity during secondary forest succession on the North Carolina piedmont. In: H.J. During, M.J.A. Werger & H.J. Willems (eds.), Diversity and pattern in plant communities. SPB Acad. Publ., The Hague. 233-245.

Pires, J.M., T.H. Dobzhansky &. G.A. Black 1953. An estimate of species of trees in an Amazonian forest community. Bot. Gaz. 114: 467-477.

Poore, M.E.D. 1968. Studies in Malaysian rain forest. I. The forest on Triassic sediments in Jengka Forest Reserve. Journal of Ecology, 56: 143-196.

Prance, G.T. 1995. In: D.L. Hawksworth (ed.) 1995. Biodiversity; measurement and estimation. Chapman & Hall, London. 89-101

Prance, M.E.D. 1978. Floristic inventory of the tropics: where do we stand? Ann. Missouri Bot. Gard. 64.: 659-684.

Prentice, I C, Cramer, W, Harrison, S P, Leemans, R, Monserud, R A, Solomon, A M. 1992. A global biome model based on plant physiology and dominance, soil properties and climate. Journal of Biogeography 19: 117-134.

Purata, S.E. 1986. Floristic and structural changes during old-field succession in the Mexican tropics in relation to site history and species availability. Journal of Tropical Ecology 2.: 257-276.

Ramakrishna, N, Running, S W. 1996. Implementation of a hierarchical global vegetation classification in ecosystem function models. Journal of Vegetation Science 7: 337-346.

Raven, P.H. 1976. Ethics and attidudes. In: J. Simmons et al. (eds.), Conservation of threatend plants. Plenum Press, New York & London. 155-179.

Rejmánek, M. 1996. Species richness and resistance to invasions. In: G.H. Orians, Dirzo & J.H. Cushman (eds.) 1996. Biodiversity and ecosystem processes in tropical forests. Springer Verlag Berlin Heidelberg, New York. 153-172.

Saldarriaga, J.G., D.C. West, M.L. Tharp & C. Uhl 1988. Long-term chronosequence of forest succession in the upper Río Negro of Colombia and Venezuela. Journal of Ecology 76: 938-958.

Sas H. et al. 1996. Onttrekking van biotische grondstoffen: ontwikkeling van een methodiek voor inpassing in LCA's. CE/CML, Delft.

Schimper, A.F.W. 1898. Pflanzengeographie auf physiologischer Grundlage (ed. 2). Jena.

Schimper, A.F.W. 1903. Plant-geography upon a physiological basis. Translation by W.R. Fischer, E.P. Groom & I.B. Balfour. Oxford.

Schulz, J.P. 1960. Ecological studies on rain forest in northern Suriname. North-Holland, Amsterdam. 267 p.

Simmons, M.T. & R.M. Cowling 1996. Why is the Cape Peninsula so rich in plant species? An analysis of the independent diversity components. Biodiversity and Conservation 5: 551-573.

Smith, T M, Shugart, H H, Bonan, G B, Smith, J B. 1992. Modelling the potential response of vegetation to global climate change. Advances in Ecological Research 22: 93-116.

Stephenson, N.L. 1990. Climatic control of vegetation distribution: the role of water balance. American Naturalist 135: 649-670.

Stohlgren T.J., G.W. Chong, M.A. Kalkhan & L.D. Schell, 1997. Multiscale sampling of plant diversity: effects of minimum mapping unit size. Ecological Applications 7: 1064-1074.

Sugihara, G. 1980. Minimal community structure: an explanation of species abundance patterns. American Naturalist 116:770-787.

Tokeshi, M. 1993. Species abundance patterns and community structure. Advances in Ecological Research 24:111-186.

Trusty, W.B. 1995 Assessing the ecological carrying capacity effects of resource extraction. IN: A. Fruehwald and B. Solberg (eds.), Life cycle analysis - a challenge for forestry and forest industry. EFI Proceedings No 8. International workshop organised by the European Forest Institute and the Federal Research Center for Forestry and Forest Products. Held in Hamburg, May 3-5 1995. p 41-51.

Turner, I.M., K.S. Chua, J.S.Y. Ong, B.C. Soong & H.T.W. Tan 1996. A century of plant species loss from an isolated fragment of lowland tropical rain forest. Conservation Biology, vol 10. No. 4: 1229-1244.

Valencia, R., H. Balslev & C. Paz y Miño 1994. High tree alpha-diversity in Amazonian Ecuador. Biodiversity and Conservation 3: 21-28.

Walter H. 1973. Vegetation of the earth in relation to the eco-physiological conditions. Springer Verlag. New York.

Whitmore, T.C. 1990. An introduction to tropical rain forests: i-xi, 1-226. Clarendon Press, Oxford.

Whrite, S.J. 1996. Plant species diversity and ecosystem functioning in tropical forests. In: Orians et al. (eds.), Biodiversity and ecosystem processes in tropical forests. Springer-Verlag Berlin Heidelberg, New York. 11-32.

Witte J.P.M. & R. van der Meijden 1995. Verspreidingskaarten van de botanische kwaliteit in Nederland uit FLORBASE. Gorteria 21, 1/2 : 3-60.

Appendix 1: Summary of data for tropical forest

If only the number of tree species are known, the total number of species were calculated with the function: total number of species = 3 * number of woody species. Only 'small areas' were taken into account
Physiotopes: TR=*tropical rainforest*, TMF=Tropical Montane Forest, MF=Montane forest
* for reference: cites elsewhere, not seen

nr.	S aantal	A m^2	LOG(A)	alpha: as given	alpha: incl. non-woody	alpha: final	physiotope	source	remarks
1	448	4.0E+04	4.60	95.2		95.2	TR	Turner et al. 1996	Singapore (130° 50' E. 1° 20' N); 4 ha lowland tropical forest; protected since 1859
2	314	4.0E+04	4.60	66.1		66.1	TR	Turner et al. 1996	inventarisation vascular plants '94; orig. : 448 native species (no 1: herbarium); left 220, 94 'new' native species (no 2: 220+94),
3	394	4.0E+04	4.60	83.4		83.4	TR	Turner et al. 1996	80 *introduced* (*no* 3: 220+94+80); as as many extincted not documented species as documented species then
4	639	4.0E+04	4.60	136.7		136.7	TR	Turner et al. 1996	original flora: 639 species ((220+94)/0.491): no 4.
5	176	1.2E+04	4.08	40.7		40.7	TMF	Kapelle, Kennis en De Vries 1995	Costa Rica (9°35'40''N, 83°44'30''W); upper montane *Quercus* forest (3000m); 1991, '92; opp. 12*0.1 ha.; 176 spec.(nr. 5),12 gen.,75 fam.
6	145	1.2E+04	4.08	33.1		33.1	TMF	Kapelle, Kennis en De Vries 1995	terr. Vascular plants; 3 succ. fases.: early secondary, late secondary en primary forest. total number of species per fase 145 (6), 130 (7)en 96 (8)
7	130	1.2E+04	4.08	29.4		29.4	TMF	Kapelle, Kennis en De Vries 1995	
8	96	1.2E+04	4.08	21.1		21.1	TMF	Kapelle, Kennis en De Vries 1995	density #srt/log(A); selected plots 31.4, 27.5, 23.2
9	39	2.0E+02	2.30	16.9		16.9	TR	* Purata 1988	Mexico; Tropical lowland forest; 11 years, secundary; ferns and fern-allies excluded
10	42	2.0E+02	2.30	13.9		13.9	TR	* Purata 1988	Mexico; Tropical lowland forest; 20 years, secundary; ferns and fern-allies excluded
11	234	1.0E+04	4.00	56.0		56.0	TR	* Bongers et al. 1988	Mexico; Tropisch laagland bos; mature, primary; ferns and fern-allies excluded
12	56	9.0E+02	2.95	15.6	46.7	46.7	TR	* Saldarriaga et al. 1988	Colombia-Venezuela, Rio Negro; tree-stems min. 1 cm dbh.; 14 yr. sec.
13	77	9.0E+02	2.95	22.7	68.0	68.0	TR	* Saldarriaga et al. 1988	Colombia-Venezuela, Rio Negro; tree-stems min. 1 cm dbh.; 30 yr. sec.
14	79	9.0E+02	2.95	23.4	70.1	70.1	TR	* Saldarriaga et al. 1988	Colombia-Venezuela, Rio Negro; tree-stems min. 1 cm dbh.; 80 yr. sec.
15	66	9.0E+02	2.95	19.0	56.9	56.9	TR	* Saldarriaga et al. 1988	Colombia-Venezuela, Rio Negro; tree-stems min. 1 cm dbh.; mature prim.
16	137	1.0E+03	3.00	42.3	127.0	127.0	TR	* Faber-Langendoen & Gentry 1991	Colombia-Choco region; woody species, stems: min. 2,5 cm dbh

nr.	S	A	LOG(A)	α given	incl. herbs	final	physiotope	source	remarks
19	54	2.0E+04	4.30	10.2	30.7	30.7		Parthasarathy & Karthikeyan 1997	India (south), Trop. dry evergreen forests. Kuzhanthaikuppam (KK) and Thirumanikkuzhi (TM). Woody species, min. 10 cm gbh; 31 fam. and 47 gen.; 1m^2: ca. 5-10 spec.)
20	42	1.0E+04	4.00	8.0	24.0	24.0		Parthasarathy & Karthikeyan 1997	KK (11°45'N Lat., 79°38'E. Long.), *stunted forest* (*avg.* tree height: ca. 6m); 26 fam. And 37 gen.)
21	38	1.0E+04	4.00	7.0	21.0	21.0		Parthasarathy & Karthikeyan 1997	TM (11°43'N Lat., 79°41'E. Long.), tall forest (*avg.* height : ca. 10m); 26 fam. and 35 gen.)
22	35	1.0E+04	4.00	6.3	18.8	18.8		* *Murphy & Lugo 1986*	*range* dry tropics: 35-90 species (as no 19 t/m 21)
23	90	1.0E+04	4.00	20.0	60.0	60.0		* Murphy & Lugo 1986	
24	80	4.1E+03	3.61	19.4	58.1	58.1	TR	Guariguata et al. 1997	Costa Rica (North-East), Heredia province (10°26'N, 84°00'W); Tropical Wet Forest); vulcanic origen. 6 areas. 40-200m in elevation); dominancy: *Penthaclethra*
25	65	4.1E+03	3.61	15.2	45.7	45.7	TR	Guariguata et al. 1997	*macroloba* & palm-trees *Welfia regia*, *Socratea exorrhiza*, *Iriartea deltoidea*
26	180	2.3E+03	3.35	50.7	152.1	152.1	TR	Guariguata et al. 1997	(max. numbers and areas from graphics. 16-18 yr. secondary stands.
27	130	5.0E+02	2.70	44.5	133.4	133.4	TR	Guariguata et al. 1997	min 10 cm dbh (no 24), >5 en <10 cm dbh (*no* 25), *min 1m* high and < 5 cm dbh, min. 0.2m and < 1m high)
28	50	1.0E+04	4.00	10.0	30.0	30.0		Wright 1996; in: Orians et al.1996	in general: 50 tree species per hectare in a dry tropical forest and more than 300 species per ha in a wetter forest
29	300	1.0E+04	4.00	72.5	217.5	217.5	TR	Wright 1996; in: Orians et al.1996	
30	365	1.0E+03	3.00	118.3		118.3	TR	* Gentry & Dodson 1987	Ecuador; vascular plant species
31	169	1.0E+03	3.00	53.0		53.0	TR	* Gentry & Dodson 1987	Ecuador; vascular plant species
32	173	1.0E+03	3.00	54.3		54.3	TR	* Gentry & Dodson 1987	Ecuador; vascular plant species
33	300	1.0E+04	4.00	72.5	217.5	217.5	TR	* Gentry '88, Ashton '93, Valencia et al '94	Western Amazonia and Borneo more than 300 tree species were found in 1 ha plots
38	460	1.1E+05	5.04	89.3		89.3	TR	Jakobs 1981	Malakka; 460 Tree species, dia. min. 10 cm.; 5 plots 2ha and 5 plots 0.2 ha. Dipterocarpaceae 1-14%, Euphorbiaceae 5-16 %, *Burseraceae* 5-9%,
40	331	1.0E+04	4.00	80.3		80.3	TMF	* Meijer 1959. In:Jakobs 1981	West-Java, Cibodas; 1450-1500m; p94 Jakobs; distribution trees, scrubs etc.
41	206	5.6E+04	4.75	41.3	123.8	123.8	TR	* Schulz 1960 In: Jakobs 1981	Suriname, Mapane;
43	227	1.0E+04	4.00	54.3	162.8	162.8	TR	* Ashton 1964 In: Jakobs 1981	Malesia,Malakka, Rengam Forest Reserve; Trees >10 cm dia.
44	220	2.0E+04	4.30	48.8	146.5	146.5	TR	* Ashton 1964 In: Jakobs 1981	Brunei; Andulau Forest Reserve, trees min. dia. 10 cm.
45	144	2.0E+04	4.30	31.2	93.5	93.5	TR	* Ashton 1964 In: Jakobs 1981	Brunei, Badas
46	174	2.3E+05	5.36	30.6	91.7	91.7	TR	* Poore 1968 In: Jakobs 1981	Malesia, Malakka, Trees min. 30 cm. dia.; distribution families: p.97 Jakobs 1981
49	300	1.0E+04	4.00	72.5	217.5	217.5	TR	Whitmoore 1990 In: Hoogmoed & de Jong '92	
51	240	1.6E+04	4.20	54.7	164.1	164.1	TR	* Whitmore 1985 In: Nooteboom '92 in: id.	Kalimantan, Wanariset. boomsoorten (0.3m)

nr.	S	A	LOG(A)	α given	incl. herbs	final	physiotope	source	remarks
64	1318	1.6E+08	8.19	159.6		159.6	TR	Prance, In: Hawksworth 1995	Panama, Barro Colorado I (data from Croat 1978); lowland rain forest; incl. Trees >10 cm dbh, small trees and shrubs, herbs and subshrubs, epiphytes,
65	1033	1.7E+06	6.22	164.4		164.4	TR	Prance, In: Hawksworth 1995	Ecuador, Rio Palenque; Data from Dodson & Gentry 1978; as no 64
66	1119	1.0E+08	8.00	138.6		138.6	TR	Prance, In: Hawksworth 1995	Brazil, Reserva Florestal A. Ducke; Hopkins (personal comm.); as no 64
67	87	1.0E+04	4.00	19.3	57.8	57.8	TR	Prance, In: Hawksworth 1995	Brazil, Belem; Black et al. 1950; gen. 65, fam. 31. **No. 67-76: only tree taxa!!!**
68	265	3.0E+04	4.48	57.0	170.9	170.9	TR	Prance, In: Hawksworth 1995	Brazil, Rio Xingu; Campbell et all 1986; ge. 127, fam. 39.
69	179	3.5E+04	4.54	37.2	111.6	111.6	TR	Prance, In: Hawksworth 1995	Brazil, Castanhal; Pires et al. 1953; gen. 130, fam. 47.
70	473	1.0E+04	4.00	115.8	347.3	347.3	TR	Prance, In: Hawksworth 1995	Brazil, Manuas; Valencia et al. 1994;gen. 187, fam. 54.
71	307	1.0E+04	4.00	74.3	222.8	222.8	TR	Prance, In: Hawksworth 1995	Ecuador, Cuyabeno; Valencia et al. 1994; gen. 138, fam. 46.
72	228	1.0E+04	4.00	54.5	163.5	163.5	TR	Prance, In: Hawksworth 1995	Ecuador, Anangu; Balslev et al 1987; gen. 126, fam. 53.
73	275	1.0E+04	4.00	66.3	198.8	198.8	TR	Prance, In: Hawksworth 1995	Peru, Mishana; Gentry 1988; fam. .
74	300	1.0E+04	4.00	72.5	217.5	217.5	TR	Prance, In: Hawksworth 1995	Peru, Yanamono; Gentry 1988; fa. 58
75	94	1.0E+04	4.00	21.0	63.0	63.0	TR	Prance, In: Hawksworth 1995	Bolivia, Alto Ivon; Boom 1986; gen. 61, fam. 28.
77	200	1.0E+04	4.00	47.5		47.5	TR	Jacobs, 1981	estimation of total number of species; normal (gangbaar); p. 95
78	400	1.0E+04	4.00	97.5		97.5	TR	Jacobs, 1981	estimation of total number of species; high
79	600	1.0E+04	4.00	147.5		147.5	TR	Jacobs, 1981	estimation of total number of species; exceptional
					mean 119.1	**mean 103.1**			

Appendix 2: Summary of data for temperate areas.

nr	S	A	LOG(A)	alpha	physiotope	source	remarks
Europe							
1	100	1.0E+06	6.00	15.0	agr.area	Witte & van der Meijden 1995	The Netherlands; avg. number of species; adjust?
2	75	1.0E+06	6.00	10.8	pl.forest?	Heikkinen & Neuvonen 1997	Finland; open uplands with forests of subalpine mountain birch
3	25	1.6E+01	1.20	12.5	pl.for.	Chiarucci 1996	Italie, 43 NB, pine-trees
4	20	1.6E+01	1.20	8.3	agr.area	Chiarucci 1996	Italie, 43 NB, no forests
5	500	2.5E+09	9.40	52.1	bergbos	Parga et al. 1996	Spanje, montane areas, avg. number of species after correction: 2133 species of 7000
6	250	1.0E+08	8.00	30.0	bergbos	Parga et al. 1996	Spanje, montane areas, avg. number of species after correction: 2133 species of 7000
7	100	2.5E+09	9.40	9.6	riviervlakte	Parga et al. 1996	Spanje, montane areas, avg. number of species after correction: 2133 species of 7000
8	25	1.0E+08	8.00	1.9	riviervlakte	Parga et al. 1996	Spanje, montane areas, avg. number of species after correction: 2133 species of 7000 correction 801 species of 7000
outside Europe							
9	88	4.0E+03	3.60	21.7	mon.pl.for.	Stohlgren et al. 1997	pine-forest, Rocky Mountains, Colorado, 2500-3000 m
10	150	4.0E+03	3.60	38.9		*Stohlgren et al.* 1997	*populierenbos*, Rocky Mountains, Colorado, 2500-3000 m
11				18.1	geen fynbos	Simmons & Cowling 1996	S-Africa, Cape Peninsula
12				28.1	struikgewas	Simmons & Cowling 1996	S-Africa, Cape Peninsula
13				16.9	duinbos	Simmons & Cowling 1996	S-Africa, Cape Peninsula
'large areas'							
24	2285	471000000	8.67	262.3		Simmons & Cowling 1996	S-Africa, different veg. Types
25	25000	2.3E+12	12.36	2021.6	med.gebied	Medail & Quezel 1997	mediterranean area
26	4450	324000	5.51	805.7		Medail & Quezel 1997	California
27	2900	140000	5.15	561.6		Medail & Quezel 1997	mediterrane Chili
28	8000	112260	5.05	1582.1		Medail & Quezel 1997	SW Australia
29	8600	90000	4.95	1733.9		Medail & Quezel 1997	Cape S-Afrika

Appendix 3: Literature data & calculations to derive fNPP values

(see table 5 in chapter 3)

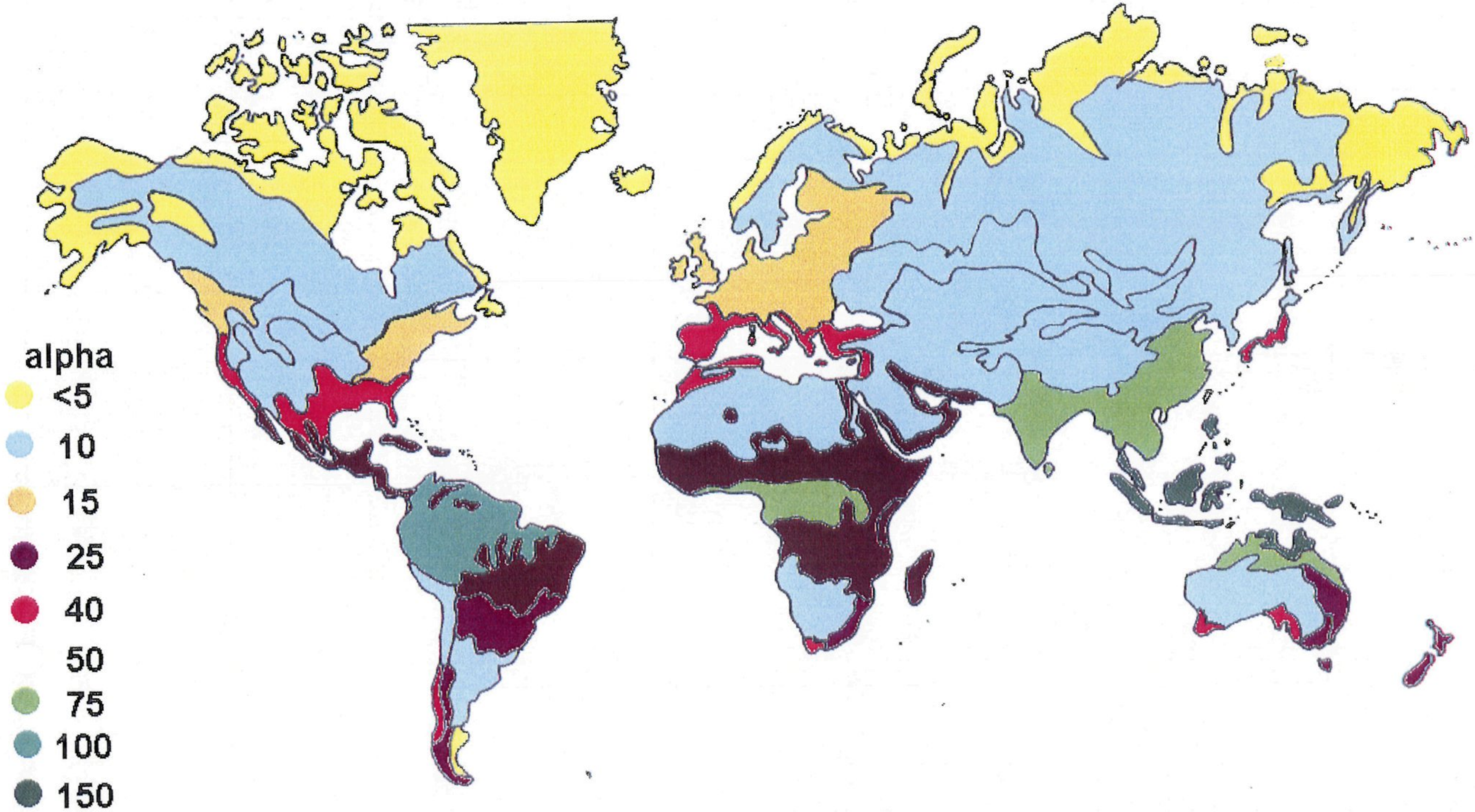
Natural system	Mixed deciduous forest in Central europe	Natural spruce forest in boreal zone	Beech forest Fontainebleau (central Europe)	Lowland Dipterocarp rainforest	Mediterranean shrubland	Lowland dipterocarp rainforest	desert and semi-desert shrub	mixed deciduous forest temperate zone
Rotation	mature	mature	mature	mature	mature	mature	mature	mature
NPP (Mg/ha.jr)	12.1	4.2	10.6	17.3	7	17.3	0.9	12.1
NPP in rotation period	605	378	1060	657.4		1211	90	605
Standing stock at end of rotation (m3/ha)								
Mean volume increment (m3/ha.yr)								
standing biomass at end of rotation	450	128.7	342.93	380.9		380.9	7	450
stem	318.5	67.3	232	287.1		287.1	0	318.5
branches	50.8	17.1	58	59.4		59.4	0	50.8
foliage	3	6.8	3.93	4.8		4.8	0	3
roots	74	37.5	49	29.6		29.6	0	74
forest floor	*		14	12.6		12.6		*
dead material	1.8							1.8
herb and shrub layer	*	0.13	0.12	42.9		42.9		*

Land use system	Commercial forest of Norway spruce in Central Europe	Commercial forest of Norway spruce in boreal zone	Commercial oak forest	Shorea robusta plantation, India	Eucalypt plantation	Tectona grandis, Indonesia	mine	Agriculture (wheat)
Rotation	50	90	100	38	40	70	100	50*)
NPP (Mg/ha.jr)	15	3.6	10.3	11.8	10.8	14.4	0	13
NPP in rotation period	750	324	1030	448.4	432	1008	0	650
Standing stock at end of rotation (m3/ha)		251	302	345	350	630	0	
Mean volume increment (m3/ha.yr)		6.2	7.8	8	15	15	0	
standing biomass at end of rotation	260	144.6	227	293.9	175.35	530	0	11
stem	170	90.4	142	207	105	378	0	7
total volume production		750	780			1050	0	
thinned volume (m3)		499	478	157	250	420	0	
thinned biomass			263	110			0	
branches	16.6	10	50	33	20	30	0	0
foliage	16.1			6.4			0	3
roots	31			47.5			0	2
forest floor	*			4.9				
dead material	7.8			*				
herb and shrub layer	*	0.2		*				

*) Figures are 50 times annual NPP etc. (for the sake of comparison with forest)

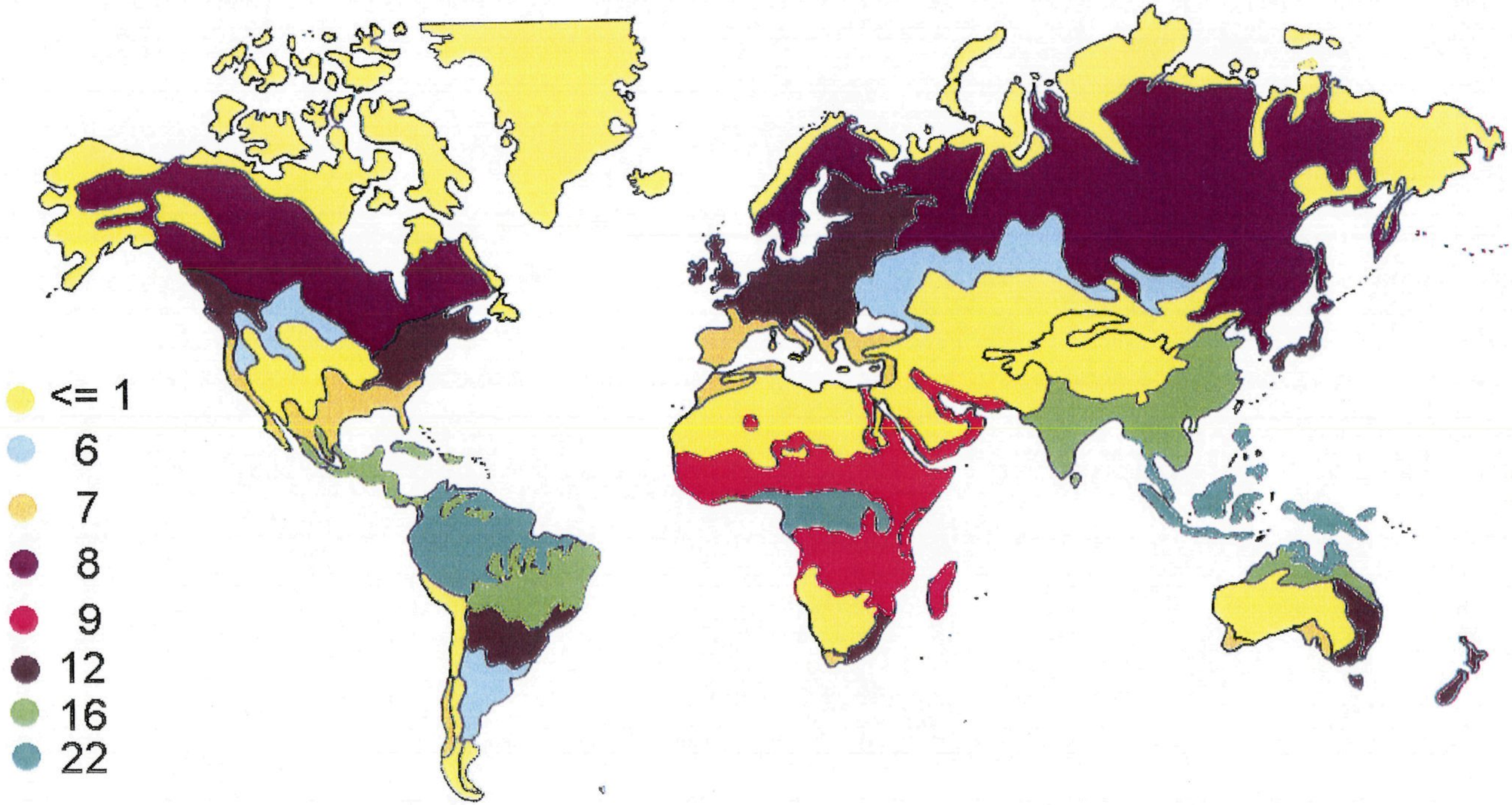
Appendix 4 Reference data on α for biodiversity indicators

(compiled on the basis of data in Appendices 1 and 2)



Appendix 5 Reference data on NPP for life support indicators

(compiled on the basis of data in Appendices 3)



Annex 2
Biodiversity as a basis for evaluating impacts in marine benthic systems in LCA

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1. BIODIVERSITY AND BIODIVERSITY LOSSES DUE TO HUMAN IMPACTS

Introduction
The aim of this study is to investigate the possibilities of developing and using a biodiversity index for the quantification of the impacts of human interventions (dredging, mining, dumping, bottom-trawl fisheries etc) on marine benthic (seabed) systems.

The definition of biodiversity according to the Rio-declaration is:

> *The variability among living organisms from all sources, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.*

In the broad definition, all components of natural systems can be included. Diversity can be approached on the level of species, habitats, communities, biotopes, ecosystems and maybe even landscapes.
The loss of biodiversity can be defined in terms of (1) loss of species, (2) loss of ecosystem functions, (3) loss of productivity.

In first instance we will investigate whether loss of species is an important issue in marine ecosystems on different scales (global or regional). For measurements of species diversity a variety of parameters has been developed, each having its advantages and disadvantages.

Loss of ecosystem functioning will often be due to loss of functional groups of species, changes (increase or decrease) of functional groups in some trophic level, changes in population parameters or disturbance of ecologically important processes (physical, chemical, biological). Loss of specific habitats or ecotopes will often also lead to changes in ecosystem functioning.
Parameters for the measurement of ecosystem functioning will largely still have to be developed.

Loss of productivity can occur if a target species decreases due to direct impacts, or due to changes in ecosystem functioning. Because of changes in ecosystem functioning total productivity may increase, as is often the case if a community is disturbed, and pioneer species take over from species characteristic for a climax community. For some users this may be advantageous. The aspect of loss of productivity will therefore be included in the discussion on changes in ecosystem functioning.

First a general overview will be given of the differences with the situation on land, and biodiversity of marine ecosystems will be discussed in general.

Where the seas differ from the land

Terrestrial landscapes are often described in terms of surface area of habitat, where the borders between different habitats may be more or less clear. The perception of a landscape is usually rather ptwo-dimensional, although a third dimension (altitude) may be incorporated in mountanous landscapes or in forests. The skies are often ignored, although air-quality or air pollution may be important and the air holds a high diversity of animals, particularly insects and birds. The latter are, however, usually mapped in terrestrial rather than in aireal terms.

In the seas and oceans there is a similar situation, but with two important differences. The third dimension, the water column over the seabed is usually considered far more important than the third dimension over land. Many animals and plants spend their entire life in the water column. Like air, water is highly mobile, and different water masses, each with their own variety of life may be discerned in the seas. A major problem with describing these 'seascapes' is exactly their mobility. They are not fixed in time or place and local or temporal biodiversity descriptions need to take this into account.

The second difference between marine and terrestrial environments is, that plants dominate the earth, but not (or not so obviously) the oceans. The structural component added by plants to the earth's surface and biotopes, is largely missing in the oceans. Marine vegetation is mainly found in the (microscopically small) plankton, and again, these plants are unlike their terrestrial counterparts, not fixed by roots. There are some underwater 'forests', like Laminaria or Kelp beds along the margins of the oceans, while in other places reefs of corals have a similar function. However, the seabed is largely devoid of fixed plants, compared to the land.

A third, but less structural difference between land and sea is the difference of data-density. While terrestrial plants and animals have been mapped and studied for many decades or even centuries, this is often not the case at sea. Exceptions are (commercially important) fish and to a much lesser extent, plankton and seabirds. Even groups that seem relatively easy to study, like benthic shellfish of worms, have not been followed in large areas of sea. Even in the relatively well-studied Dutch part of the North Sea, the benthic community can only be described in detail on the basis of one or at best a few, years of surface. This means that year to year variation and trends are poorly understood, hampering good description of biodiversity or LCA's

The sea: uniform or diverse?

To most people, the sea is just a vast mass of salt water, with only the shoreline as a distinct structural element. However, just like landscapes exist in the terrestrial environment, 'seascapes' exist in the marine world, with their own fysiotopes, as well as ecotopes with habitats and niches for specific plants and animals. Marine fysiotopes exist at different scales. The open ocean is distinctly different from shelf seas, coastal waters or estuaries, but within each of these, many smaller fysiotopes exist, for instance canyons or seamountains at the ocean floor. Moreover, the marine physical world is typically three-dimensional, and fysiotopes that are based on water column properties may or may not overlap with fysiotopes based on bottom topography or sediment type. For instance, water entering the sea from an estuary with mix into a distinct 'coastal water mass' in the nearshore, coastal zone of that sea and may then travel as a physical unity over a large area, but interacting with many different bottom types and/or depth zones on the way. The interplay of traveling watermasses through the sea and their interactions with different seabeds and associated life forms result in a wide variety of ecotopes in the sea. Through all this move the fishes, the seabirds and the marine mammals (seals, dolphins and whales). These animals are highly mobile, capable of travel from one watermass to the next, and even of leaving the marine environment altogether. Yet, at sea, they usually have strong preferences for particular ecotopes.

Biodiversity of marine systems

There is a great lack of knowledge on the taxonomy of (benthic) marine organisms. Approximately 200,000 marine species have been described (May, 1994). Until recently it was assumed that the greatest diversity occurred in shallow tropical seas, and the deep sea was considered to have a low diversity. Gradually the concept of stability was included, and mainly on the basis of theoretical considerations it was concluded that systems with low dynamics possibly contained more species. This was confirmed when samples were taken. The deep sea proved to contain many species (Hessler and Sanders, 1967, 1968). Since then several studies have been carried out and estimates of the total number of species vary between 500,000 and 10 million (Wolff, 1998). The number of species found in inventories was proportional to the area sampled. There also seemed a great variety between regions (Poore & Wilson 1993, Gray 1994, Coleman et. al 1997, Wolff, 1998).

According to Wolff (1998) only 19 of the described 200,000 species have disappeared because of human impacts. Nine of these were birds which were impacted on their terrestrial breeding habitats and 5 were marine mammals. Locally, for example in the Waddensea more species can disappear from the area. Wolff (1998) mentions 43 species. Fishing or hunting was the major cause (28 species) and habitat loss the second (18 species). As the majority of these species are still present in the adjacent waters, they may come back if conditions become favorable again.

Marine ecosystems are characterised by large fluctuations in the abundance of individual species. Rare species may (locally) become abundant for many years, and common species may become almost extinct for many years. The causes for these fluctuations are not understood.

These considerations indicate that species diversity may not be a suitable parameter indicating human impacts. There are several types of indices for species diversity. One type of indices indicates species richness, the number of species in a community, while the other type indicates 'evenness' parameters, and gives a measure for distribution of individuals among the species. The 'Shannon-Weaver' index is an example of an index which is sensitive to changes in the rare or less abundant species of a community, while the 'Simpson' index is an example of indices which are sensitive to changes in the more abundant species.

For the North Sea the Working Group on Ecosystem effects (ICES, 1996) investigated changes in biodiversity of commercial fish stocks in the North Sea. They conclude that diversity according to the Shannon-Weaver index has increased. This is due to reduction of the abundance of dominant (common) species and more evenness in the population. The different diversity indices are also very sensitive for the occurrence of rare species. In marine monitoring programs rare species can not be sampled in a quantitative way.

In marine ecosystems distribution of larvae and adults does not pose many problems. Currents cross oceans, and tidal movement can transport organisms over many km's per tide. There are hardly barriers for swimming or flying organisms. However, in some areas there have been indications that despite the good distributional capabilities, local species losses have not been compensated. Also, there are often clear differences in small scale distribution between species of genetic strains. Recent investigations indicate that within a habitat there are differences in species or genetic composition related to depth, period of emersion etc. This diversity may be important for the long term survival of a species, as the different genetic strains may have an better survival in a fluctuating, unpredictable environment. Conservation of a variety of habitats and biotopes will assure that the (unknown) biological diversity is maintained.

Parameters responsible for differences in biodiversity on a global scale

In the last 10 years the debate on marine biodiversity and the parameters influencing it has boomed. Bouma (1996) mentions a > 1000 fold increase in the popular and professional literature.

In marine biodiversity, several gradients have been recognized. Highest diversity was found in the tropics, and diversity decreased towards the arctic. Recent investigations indicate that this is less clear when going from the tropics towards Antarctica. Biodiversity is therefore not solely explained by latitude.
There also exists a longitudinal gradient with the highest biodiversity in the Indonesian Archipellago. The great variety in island types, archipels, depth gradients, isolation and long undisturbed development times, are held responsible for the development of a high biodiversity.
The existence of a biodiversity gradient with depth is a matter of debate. Early investigations found high diversities at greater depths, but others found high values at intermediate depths (for a review see Stam 1996, Bouma 1996 and Gray 1995)

In all studies is became clear that an important aspect was the surface area sampled. With an increase in sample size the number of species increased. Even more important seemed to be the area of the biotope sampled. The larger the area, the more habitats are present. Some of these might be very specific and contain few species in high densities, others may be rare, but contain many species. In his 'public lecture' Wolff 1998 argues that we seem to spend much effort into the explanation of the causes for differences in biodiversity by looking at evolutionary and biogeographical hypotheses, while we seem to overlook the most important issue of aerial extent.
This remark is important if we want to relate human interventions and their impact on biodiversity. It would mean that impacts occurring over large areas should be considered as being potentially more threatening than small scale impacts. Therefore area should be part of the assessment, although it is not yet clear how.

Of the many human interventions, only pollution, eutrophication and fisheries can be considered to influence areas of the size of estuaries, coastal seaes, continental shelves, oceans and deep-sea, and therefore potential threats for changing biodiversity on a global scale. On the other hand many species are restricted to very specific habitats, and these habitats may be influenced. (examples are coral reefs, mangrove forests, salt-marshes, seagrass and kelp-beds , mussel and oyster reefs etc. Continuous (low) pressure over the whole distribution range of each specific (sub)type may cause their extinction or decline, resulting in a decrease in biodiversity. This means that a chronic relatively low impact over a large area should be viewed with care as it may have far greater impact than a catastrophic event at a local scale.

In many instances food-chain or community relationships are responsible for an increase in species diversity. Clear and well described examples are coral reefs, oyster-reefs, mussel-beds, mangrove-forests, salt-marshes, seagrass-beds, kelp-beds etc.. Many of these 'biogenic structures' maintain themselves by a combination of physical, chemical and biological processes, and might therefore be considered as 'super-organisms' (Dankers 1993). Disruption of any of these processes will result in disappearance of these communities and their associated species. This is especially serious if the conditions for new development are not existing any more. Coral reefs have developed in areas with rising sea level or land subsidence. When destroyed they will not develop in the same areas as these will be too deep, and may not be able to redevelop in other areas as these may be too strongly influenced by human activities. The same may be true for the other self-maintaining communities mentioned earlier.

Ecosystem functioning is governed by ecosystem processes which are to a large extent initiated by physical and chemical processes. For an undisturbed ecosystem functioning, presence of all trophic levels and if required presence of all succession-stages is essential. In the marine environment the majority of the physical processes can not be influenced on a

major scale (tides, currents, seasonal temperature cycles etc). Pollution, eutrophication and fisheries can influence complete marine systems on a global scale by altering or removing trophic levels, changing population characteristics, species composition or having impacts on the genetic level within a species. The widespread loss of top predators may encourage the selection of opportunistic pests and pathogens across a wide taxonomic ranges of plants and animals, the so called 'environmental distress syndrome' (Epstein 1995).
On a smaller scale (for example the North Sea) the chronic disturbance by beam-trawl fisheries may destroy a variety of benthic habitats (Lindeboom et al 1998) and therefore have a lasting impact on biodiversity.

2. SUITABILITY OF INDICATORS EXPRESSING BENTHIC CHANGES AND OCCUPATIONS BY HUMAN INTERVENTIONS.

Biodiversity

Typical parameters often used to express the (biodiversity) quality of marine ecosystems are: presence or absence of certain species, number of individuals or area covered by certain species, the occurrence of selected biotic and abiotic processes, or parameters indicating the quality of a system such as the ratio between organisms in different trophic levels, a normal age structure of a population, or the ratio between long and short living species. The impact on biodiversity should therefore be measured in different parameters. This type of parameters requires further study before they can be made operationable. The selection of just one parameter, as suggested for the land, seems too premature considered the lack of scientific knowledge.

The occurrence of specific habitats is an important issue in maintaining species diversity. Important habitats are those which are formed and maintained by an interplay of physical and biological processes. If only physical processes are important, the regeneration ability will in general be good, unless the physical processes are prevented to act. Therefore it is important to describe which processes are considered to be essential. In appendix 1 an elaboration is given of processes important to North Sea biodiversity.

As has been said before, there is much uncertainty on biodiversity on a global scale. When sufficient research is carried out new areas with record breaking species diversity are being found. The present state of knowledge is not sufficient to select a single parameter such as species richness for equations calculating the severity of human interventions. At least two additional parameters should be used: age distribution within species and the species composition.

Due to the high dynamics of the ecosystems and abiotic factors, it is necessary to include the regeneration time for the biodiversity for land use changes. This can not be the regeneration of the undisturbed system before the activity as the whole surroundings will have changed too, but rather the levelling to the surrounding situation at that time. In detail the definition of the regeneration time would become:

"The time necessary for achieving the same level of biodiversity at the location of the activity as at a few randomly chosen equally large area's (reference area's) within the same ecotope."

The same level of biodiversity can be specified as an equal variance and median of the measures species diversity, age distribution and species composition. This level may be very different from the situation before the activity started. The reference area's should be chosed such that no interference by the activity is possible.

Life support
In the approach of IVAM-ER primary production measured by fNPP or NPP is an important measure for life support in LCA. This approach is not suitable for the majority of benthic marine communities. The majority of these ecosystems do not receive sufficient light for primary production. Growth of benthic organisms is almost completely based on import of algae, detritus or animals (or their larvae) that are produced elsewhere. The equations should therefore be broadened to the production or biomass of zoobenthos (seabed dwelling animals). The majority of zoobenthos is not wanted for human consumption, but important as food item for fish (both commercial as other species). Changes in zoobenthos and non commercial fish species should be considered of importance in evaluating anthropogenic impacts.

Although some general figures for benthic biomass can be given for estuaries, shallow coastal seas, continental shelfs, and the deep-sea, within these regions small areas with much higher biomasses can be found. As these areas have a patchy, and often random, distribution, the impact of human activities is not easy to predict. Nor are these area's to be considered in the non-specific approach developed in this study.

In coastal zones where mean biomass ranges between 0 and 75 grams ash free dryweight (AFD) per m^2 , assemblages of bivalves can occur with biomasses of 1 kg AFD/m^2. In the Danish Waddensea a musselbed with a density of 75 kg freshweight/m^2 (3.75 kg AFD) has been described (REF). As these areas with high biomass often vary spatially and temporally, it is not yet possible to use this type of information in the equations. On the other hand it is possible for general areas of the North Sea to indicate different areas with different *mean values, up to 10 g AFD per m2. For the algea in the watercolomn above the seabed the* biomass productivity is much lower than that. Impacts on these scores by exploiting the seabed are very difficult to give. Nevertheless, as a comparison with the situation on land, the mean biomass productivity is about 1 kg/m2 for temporate regions (see annex 1). The contribution of the seabed to biomass productivity in the strict sense of annex 1 will thus be limited, and its life support impacts will not easily dominate LCA studies not especially focusing on seabed activities as in non-specific area's the biomass productivity is up to a factor 100 lower than at land.

As a last comment, in heavily impacted areas 'mature' benthic communities are often replaced by pioneer communities consisting of fast reproducing and fast growing species. The production may therefore increase and according to the equations used be considered positive for life support, while biodiversity has decreased. This is the same discussion as with oligotrophic ecosystems on land (see annex 1, paragraph 3.4.3).

3. IMPACTS OF SEDIMENT DISTURBING ACTIVITIES ON BIODIVERSITY IN THE NORTH-SEA

Evaluating impacts of human activities on different fysio- and ecotopes.

As explained in appendix 1, the impact of a human activity should be evaluated against three relevant parameters: the area impacted (a) relative to the total area of that particular fysio- or ecotope present (A) and the extent of quality loss (Q). We will here compare two different human activities that both are destructive to life at and in the seabed, but do so at different temporal and geographical scales. These activities are removal and associated depositing of sand on the one hand, and beam-trawl fisheries on the other hand. These two have in common, that life is effectively destroyed in the path of the equipment (high values of Q). They differ, however, in the extent of area affected: removal and deposition of sand is done at specific locations, fishery occurs throughout the Dutch part of the North Sea (DNS). Does this mean that sand removal has a lower impact than fisheries? Not necessarily, as the extent to which the total area A is affected may differ, but also values of Q are different. Consider first a scenario, where both activities take place and are to be evaluated in the Southern Bight. This is a large area, roughly half of the DNS, say A = 20.000 km^2. Fishing vessels are found all over this part of the North Sea and can fish nearly anywhere. Still, there is dispute as to how large a proportion of the seabed is actually touched by the gear. In some estimates, each square meter is touched at least once a year, in others, many square meters are missed in any one year and these could act as sources for fished trawl tracks (sinks). The area impacted (a) is thus not easily estimated, but probably large in relation to A. Compared to this, an area of sand extraction is relatively well-defined and small in relation to an A of 20.000 km^2. The severity of the two impacts is high in both cases, although in fishery more animals escape ($Q<1$), than in sand-digging where all animals in the removed sand will be killed or at least displaced ($Q=1$). In this comparison, A is equal for both activities, while a and Q differ. This means, that **locally** (scale of a) sand extraction has the greater impact, but **globally** (scale of A) the fishery has the greater impact. A sand pit left after extraction is over will fill again and be re-stocked with animals. Likewise, fishing tracks will be restocked from the surroundings and probably at a faster rate as the impact was smaller (smaller Q). However, considering the large geographical scale at which this fishery takes place, some benthic animals that live long and reproduce slowly may be heavily impacted ($Q = 1$ or nearly 1). Animals that need to remain undisturbed for a considerable period of time before they can reproduce, run a high risk of being hit by passing fishing gear each year (large a), and summed over several years, this probability may approach 1. For such animals the impacted area may approach one if the impacts over several years of exploitation need to be summed and such animals may go extinct globally, while short-lived animals may experience a much smaller impact (smaller a) and they may even benefit from fisheries, through reduced competition with long-lived species. Hence, an activity like bottom-trawling may change biodiversity globally and long-term, while a locally more destructive activity like sand extraction will never do so.

Now, consider sand-extraction and fishing in the coastal zone of the DNS. If densities of fishing vessels are equal in the coastal and offshore parts of the DNS, filling in the equation of annex 1 $(a/A)*Q$ will yield similar results in both inshore and offshore fisheries, unless animals with very different contributions to the total Q are present. This may be the case for young fish, that are bycaught in vast numbers. For this reason exactly, large parts of the coastal zone are practially closed to fishing. For sand extraction, the outcome of the equation is quite different. Impacted area a and the values for Q will remain the same, weather the sand-pit is dug inshore or offshore. However, the total area available A is much smaller in this case, making the relative impact far worse. Moreover, sand extraction is often followed by sand dumping elsewhere in the coastal zone, increasing a. Values for Q may be extremely high ($Q=1$) in inshore digging and dumping, if unique or non-renewable physical features are destroyed in the process. This happens if particular sands that have been put in place during the ice-ages are moved by the extraction or dumping. This may be the case if heavy minerals are moved, if stony or gravely sands are moved or if special features like the Pettumer Polder are used as a storage area (so-called 'punaise'). Beaches that are supplied with sand dug up from the sea may not look the same for a long time. For instance, the beaches of Texel now contain cliffs of gravely sands, while thousands of pebbles, stones and boulders now litter the formerly sandy beaches. This new feature is the result of nourishing these beaches with sediments that differed considerably from those that made up the original beach.

4. CONCLUSIONS

To assess impacts of land use two parameters have been proposed to indicate impacts on biodiversity and life support, vascular plant species diversity α and free net primary biomass productivity (fNPP). The extent to which impacts on both themes can be indicated for benthic ecosystems have been discussed in this annex.

Life support

In the approach of IVAM-ER the (free) net primary biomass production is operationalised as an indicator for life support (the function of starting, closing and maintaining substance cycles).

In most seas the primary production occurs in the surface layers, and is governed by the availability of light and nutrients. Anthropogenic influences on nutrients may occur through runoff from land. Light availability can be influenced by increased runoff and erosion, as well as dredging and dumping. When this occurs chronically, the impact on fNPP (=NPP) may be included in LCA assessments. In general dredging only occurs for a limited time in a restricted space, and the influence on NPP can be neglected when no specific valuable region is concerned. When the latter is the case, an Environmental Impact Assessment including detailed local data should be performed. Resulting data on NPP may then be included in LCA's, but do not cover the whole issue.

The primary production approach is not suitable for the majority of benthic marine communities. The majority of these ecosystems do not receive sufficient light for significant primary production. In non-specific regions the NPP is a factor 10 to 100 lower than in average regions on land. The contribution of marine benthic ecosystems to life support is therefore limited when seen per m2 or m2.y.

Biodiversity

To assess biodiversity analogously to what has been done for land ecosystems, more knowledge is needed to determine which species are indicative for marine benthic biodiversity. Zoobenthum (animals dwelling in or on the seabed) should be included here. But also more complex indicators are presently used to give an indication of marine nature value. The impact on biodiversity should therefore be measured using different parameters. Several projects are under way to develop parameters that can be used to give better indications of biodiversity in exploited marine ecosystems (GONZ-report, Natuurdoeltypen approach).

One possible combination of parameters is to use the regeneration time of biodiversity, relative to reference states in the same ecotope and measured as variance and median of species diversity, age distribution and species composition.

LITERATURE

Baptist H.J.M. & Wolf P.A. 1993. Atlas van de vogels van het Nederlands Continentaal Plat. Rapport DGW-93.013, Rijkswaterstaat, Dienst Getijdewateren, Middelburg.

Bergman M.J.N., Lindeboom H.J., Peet G., Nelissen P.H.M., Nijkamp H. & Leopold M.F. 1991. Beschermde gebieden Noordzee - noodzaak en mogelijkheden. NIOZ rapport 1991-3, 195p.

Bouma, H. (1996) Geographic Paterns of Biodiversity and Diversity Generating Mechanisms. Dept. Marine Biol. Univ. Groningen. 25 pp

Camphuysen C.J. 1996. De verspreiding van zeevogels in de Noordzee: naar een beter begrip van patronen en verbanden. Sula 10: 1-48.

Camphuysen C.J. & Leopold M.F. 1993. The harbour porpoise *Phocoena phocoena* in the southern North Sea, particularly the Dutch sector. Lutra 36: 1-24.

Camphuysen C.J. & Leopold M.F. 1994. Atlas of seabirds in the southern North Sea. IBN Research report 94/6, NIOZ Report 1994-8, Institute for Forestry and Nature Research, Netherlands Institute for Sea Research and Dutch Seabird Group, Texel.

Coleman, N., A.S.H. Gason & G.C.B. Poore, 1997. High species richness in the shallow marine waters of south-east Australia. Marine Ecology Progress Series 154: 17-26.

Dankers, N. (1993) Integrated estuarine management - Obtaining a sustainable yield of bivalve resources while maintaining environmental quality. In: R.F.Dame (ed) Bivalve filter feeders - Estuarine and Coastal Ecosystem Processes. pg 479-511 Springer-Berlin

Epstein, P.R. (1995) Emerging diseases and ecosystem instability: New threats to public health. Amer. J. Public Health 85:168-172

Gray, J.S. (1994) Is deep-sea species diversity really so high? Species diversity of the Norwegian continental shelf. Marine Ecology Progress Series 112: 205-209.

Gray, J.S., G.C.B. Poore, K.I. Ugland, R.S. Wilson, F. Olsgaard & Ø. Johannessen, 1997. Coastal and deep-sea benthic diversities compared. Marine Ecology Progress Series 159: 97-103.

Gray, J.S. (1995) Marine Biodiversity: patterns, threats and development of a strategy for conservation. Concept-GESAMP-report 31 pp, final version in press (1997?).

Hessler R.R & H.L. Sanders, 1967. Faunal diversity in the deep sea. Deep-Sea Research 14: 65-78.

Holtmann S.E., Groenewold A., Schrader K.H.M., Asjes J., Craeymeersch J.A., Duineveld G.C.A., van Bostelen A.J. & van der Meer J. 1996a. Atlas of the zoobenthos of the Dutch continental shelf. NIOZ, NIOO & RWS-DNZ.

ICES, (1996) Report of the working group of ecosystem effects of fishing activities. ICES C.M. 1996/Assess/Env:1

ICONA 1992. Noordzeeatlas. (zie NVK)

Knijn R., Boon T., Heessen H. & Hislop J. 1993. Atlas of North Sea fishes. Cooperative research report no. 194, Intern. Council Explor. Sea, Copenhagen.

Leopold M.F. 1994. Walvisachtigen in de zuidelijke Noordzee: twee survey methoden vergeleken. Sula 8: 207-225.

Leopold & Dankers 1997. NVK-zout

May, R.M.1992. Bottoms up for the ocean. Nature, London 357: 278-279.

May, R.M (1994) Biological diversity: differences between land and sea. Phil. Trans. R. Soc. Lond. B. 343: 105-111

Ministerie V&W 1996b. Achtergrondnota Toekomst voor Water - Watersysteemerkenningen. RIZA Nota 96.058 en Rapport RIKZ-96.030.

Poore, G.C.B. & G.D.F. Wilson, 1993. Marine species richness. Nature, Lond. 361: 597-598.

Sanders, H.L.1968. Marine benthic diversity: a comparative study. American naturalist 102: 243-282.

Stam, M. (1996) Marine Biodiversity: A Literature Update. Dept Marine Biol. Univ. Groningen. 20 pp (+appendix)

Wolff, W.J. (1998) De zee zal ons een zorg zijn. Inaugurele rede Universiteit Groningen

Zevenboom W. 1993. Assessment of eutrophication and its effects in marine waters. Nota 93.6-NZ, Directie Noordzee.

Zevenboom W., Rademaker M., Backus L.C., Kamphuis J.E. & Orth R.C. Surface algal blooms on the Dutch continental shelf, 1979-1992. Nota 93.8-NZ, Directie Noordzee.

Appendix 1

Elaboration of important biodiversity features in the Dutch part of the North Sea

The occurrence of specific habitats is an important issue in maintaining species diversity. Important habitats are those which are formed and maintained by an interplay of physical and biological processes. If only physical processes are important, the regeneration ability will in general be good, unless the physical processes are prevented to act. Therefore it is important to describe which processes are considered to be essential. Before discussing these, a general overview will be given of the main characteristics of the Dutch part of the North Sea.

Fysiotopes and ecotopes in the Dutch part of the North Sea

The North Sea has a total surface area of circa 572.000 km² (ICONA 1992), of which some 10% (57.000 km²) belongs to the Netherlands. Directly linked to this part of the North Sea is the Wadden Sea, (2800 km² in the Netherlands), and the remaining marine parts of the Delta area of the rivers Rhine, Meuse and Scheldt. These include 'Veerse Meer' (40 km²), 'Grevelingen' (140 km²), Eastern Scheldt (350 km²) and Western Scheldt (300 km²). Between these land-locked parts of the Delta and the North Sea proper lies the so-called Outer Delta ('Voordelta'), a still developing shallow part of the nearshore North Sea of about 900 km² in size. To put these figures in perspective: the total surface area of the NetherLANDS measure only 37.000 km².

Considering these large areas of sea, it is hardly surprising that large differences exist between different parts. Here, we will only discuss the Dutch part of the North Sea and not dwell on its associated waters that are encompassed by land. The large-scale fysiotopes that are commonly distinguised in our study area have recently been summarize in several reviews, eg. Bergman et al. 1991, ICONA 1992, Ministerie V&W 1996 and Leopold & Dankers 1997. Here, we will just give a brief summary, more background data and ideas can be found in the reports listed above.

Abiotic factors

depth In general, water depths in the North Sea increase from the south-(east) to the north(west). Hence, the Dutch part of the North Sea (**DNS**) is relatively shallow, on a North Sea scale. The deepest parts are found in the northwest of the DNS. Waters deeper than 30 meters are found in two distinct areas: in a depression in the central southern half of the DNS, an extension of The Channel, while most of the northern half also is over 30 m deep. In the northwestern corner the Dogger Bank rises up from the sea floor, and water depths decrease to less than 20 m. The deepest part of the Netherlands is located next to the Dogger Bank, to its southwest. Here, the so-called Silver Pit goes down to -63 meter.

water Seawater is salty by definition, but several watermasses of different origin and salinity flow through the DNS. Nearshore, there is a large input of riverine waters that only slowly mix with saltier, offshore waters. The result of this process is a distinct, *coastal watermass*, with typical low salinity and distinct temperatures, reminiscent of its riverine origin (relatively high in summer and low in winter). Around the mouths of rivers, *riverine watermasses* are often distinctly present, fenced in by *river plume-fronts*. In winter, when plankton growth is minimal, the riverine and coastal watermasses have remarkably elevated concentrations of nutrients (phosphorus and nitrogen) that are discarded by the rivers. Finally, the coastal watermass has a high turbidity and low levels of light penetration. This is due to plankton growth in summer and suspension of sand and silt throughout the year, powered by wind and tidal currents running over the shallow bottom. Further offshore off the Dutch mainland coast runs a current of *Channel water* through the deeper part (central gully) of the Southern Bight. This water has an Atlantic origin, is highly saline, relatively cold in summer and warm in winter. On calm days, Channel and coastal watermasses are clearly separated by a distinct *coastal front*. Both Channel water and coastal waters move gradually north, powered by the currents of ebb and flow, the latter being on average stronger (residual, northward current). 'Around the corner' of the Dutch mainland coast, the influence of Channel water ceases. In the wide space of the German Bight another watermass forms, the *'continental coastal water'* characterised by a large residence time and inputs from the central North Sea, the Channel as well as the Wadden Sea and through her, the rivers Rhine, Eems, Weser and Elbe. At its seaward side, the continental coastal water turns into 'proper' seawater, with low tidal currents, little influence from rivers, high salinities and vertical stratification of waterlayers in summer. These waters belong the the *'central North Sea watermass'*, fed by input from the Atlantic and northwestern North Sea. At its southern borderline, it is bordered by the *Frisian Front*, a transition zone between stratified waters to its north and tidally mixed waters in the south, in summer. The different watermasses are clearly distinct, both by chemical and physical properties, and for instance visible by satellite telemetry using false colour temperature imaginary. Each has its own community of associated life forms ranging from minute planton that drifts passively with the currents, up to highly mobile animals like seabirds or dolphins.

Bottom In general, the seabottom in the southern half of the DNS exists of sand, while the northern half is saltier. At the transition zone (Frisian Front) an distinct area of clay is found. Here small particles are deposited as tidal currents slow down to a critical level. Such small particles can never settle long in the Southern Bight, where tidal currents (the influence of The Channel and ebb and flow running through a relatively narrow basin), but are deposited further north. The Frisian front receives large quantities of silt, but the bottom of large areas north of the front also have small particle sizes. However, the Dogger Bank in the northwest is shallow and sandy again, but the deep parts next to it (Silver Pit) have a bottom of clay. In the south of the DNS the *'Zeeland, Hinder- en Flemish Banks'* are found, large ridges of hard sands running in the direction of the watercurrents, from SW to NE. Large, but less defined sandbanks are also found in the outer Delta, and parts of these emerge during low tide. West of IJmuiden, near the border with the British part of the North Sea, a 30 km long, narrow ridge of hard clay is found, rising over 10 meters up from the sandy seabed. This *'Brown Bank'* has rather deep gullies (> 30 water depth) at either side and this area is an important site for fossil ice-age and pre ice-age animals such as whales and mammoths. Nearshore, off the mainland coast of North-Holland a rather small, shallow area is found with very course sands, the *'Pettummer Polder'*. Very coarse sands are also found nearshore further north, in the outer deltas of the Wadden Sea. The coarsest sediments are found in the *'Texel Stones'* to the north of the isles of Texel and Vlieland, where end-morenes of the last ice-age have formed a bottom of coarse sands interspersed with gravel and large boulders. In several patches in the nearshore zone, rather large deposits of heavy minerals appear to be present, while along other stretches of the coast these do not occur. These deposits have not yet been properly surveyed.

Biota, watermasses and bottom-characteristics
Large differences have been found in the occurrence of plantonic (see eg. Zevenboom 1993, Zevenboom et al. 1993 or Leopold & Dankers 1997), in the communities of bottom-dwelling animals (benthos), fishes, seabirds and mammals in the different fysiotopes described above. At the moment of writing, distribution maps covering the whole DNS exist for most major animal groups, the zoobenthos (animals like worms and shellfish that live in or right on top of the bottom-sediments): Holtmann et al. 1996; fishes (Knijn et al. 1993), seabirds (Baptist & Wolf 1993 and Camphuysen & Leopold 1994, Camphuysen 1996) and marine mammals (see eg. Camphuysen & Leopold 1993, Leopold 1994). Examples of indicator species or complexes of species associated with the different fysiotopes can be found in each of these publications.

Obviously, as all plankton and the vast majority of animals (or their larvae) are mobile, as are the watermasses that they live in, there is not a single species that is endemic to a particular environment in the North Sea or even to the North Sea as a whole. Such an absolute habitat compartimentation does not exist in the DNS. However, some species thrive in specific areas, and not in others. Here we cannot go into this in depth but we will give some examples for each large fysiotope.

The coastal zone, including outer Delta, with its sandy bottoms, low salinity, high input of nutrients and high current velocity and wave action has a particular plankton community (large incidence of blooms) and fauna. The benthic community is highly distinct from areas further offshore that have lower dynamics. Rather few species occur here, but often in large quantities, like banks of the through shell *Spisula subtruncta.* Many species of fish have their nurseries in these parts and the coastal zone is thus particularly rich in small fish, while larger fish are comparatively scarce. The shallowness and rich food supplies lure vast numbers of near-arctic seabirds to winter in this zone, like shellfish eating seaduck or fish eaters like divers and grebes. Harbour seals venture into the coastal North Sea throughout the year from the Wadden Sea, but rarely go beyond this area while on the other hand, dolphins and 'pelagic' seabirds avoid this zone.

The Southern Bight, with its high salinity and currents and its sandy bottoms, is an area of large underwater, moving sand-dunes. The area is highly dynamic and biomass of benthos is relatively low, as is biodiversity. The fish community is different from that of the northern part, with sand-loving species like the lesser weever being dominantly present here. In the wedge of Channel water entering from the south, dolphins are comparatively common. The *coastal front*, the border between this Channel water and the coastal waters, acts as a collection zone for floatsom and small animals in the water column. As such, it is a prized foraging ground for several species of seabirds, and often used as a 'migration highway' for many animals, including these seabirds. Porpoises avoid the Southern Bight in summer, but move in in winter for reasons not yet understood.

The Frisian Front, with its bottom of clay and transition from turbid to clear water, has a very rich benthic fauna with brittle-stars as one of the dominating species, and associated bottom and pelagic fish and seabirds. All larger animals here live on the productivity of the Southern Bight, that is carried off by the tidal currents and deposited at the front. The deposited material and reworked by 'deposit feeders' in and on the bottom that start a whole new food-chain.

The central North Sea, north of the Frisian Front, with its deep, clear waters and rich, clayish bottom that is not disturbed (much) by waves or currents, has its own bottom fauna. In analogy to terrestrial situations, this zone may be seen as having a little disturbed, climax fauna, compared to the much more disturbed, pioneer fauna in the Southern Bight. Relatively high biomasses and diversity are found in the north, with relatively high densities of top-predators such as pelagic seabirds, porpoises and dolphins.

Physical processes
For the development and maintenance of each characteristic marine ecosystem different processes play a key role. The relative importance can differ greatly whether one considers a coral reef, a mangrove area, a continental shelf, the open ocean, the deep-sea or upwelling regions. As an example the important processes for the Southern North Sea and the adjoining Waddensea will be worked out.

Both in the North Sea and in the estuaries, tides and waves are responsible for the mixing of the water column, the transport of sediment and the transport of nutrients and organisms. Light and temperature are responsible for the primary production.
In estuarine and coastal regions the tidal amplitude is responsible for the large-scale morphology of the coastal area. Sandy coasts with a small tidal amplitude develop into a system of barrier-islands with intertidal flats between the islands and the mainland. Areas with a large tidal amplitude form an open coast with intertidal sand banks. The tidal currents together with wind and waves are responsible for the maintenance of gullies and tidal flats. Wind, and the availability of sand are the primary factors in dune and island formation.

Physical processes which have taken place since the last ice ages are responsible for the large-scale morphology of the North Sea. The southern part is characterized by a coarse sandy sediment in a shallow sea. The sediment is continuously moving because of tidal currents and waves. In the northern part of the North Sea current speeds are lower and the water column is stratified. Therefore fine silt can settle. In the so-called Frisian Front region between the southern and northern North Sea, the water column is seasonally stratified and both organic and inorganic matter will settle.
In the tidal areas along the coast several characteristic structures have developed. A Wadden Sea system is characterized by complete gully systems. That means, a tidal inlet, ebb- and flood systems and main channels, which branch into small gullies and creeks in sandy or salty areas or salt marshes. Within a Wadden Sea system there is a diversity of tidal flats with sediment of different silt content and different exposure times.
In an interplay of physical and biological processes, salt marshes and dunes are formed. Erosion and sedimentation processes cause small islands to move into the direction of sand transport.
Typical structures have a biogenic origin such as oyster and mussel beds, reefs of tube building polychaetes or eelgrass fields.

The North Sea is a relatively young system. The sea has developed because of a sea-level rise during the last 10 000 years. It is uncertain whether present changes in sand banks and coastline are due to the fact that the system has not reached a balanced climax situation or whether changes are due to changing natural conditions which may occur in long-term cycles (more than 100 years).
In the Wadden Sea geomorphological developments can be observed on even shorter time scales. The Wadden Sea developed after inundation of freshwater marshes in the 12th century. Because of sedimentation and reclamations the area has been reduced considerably since then.
The area near the tidal inlets is very dynamic. Sand banks in the tidal inlet migrate in a clockwise direction and cross over to the next island. The sand moves along the island as a wave. The tip of the island shows a cycle growth and erosion, depending on the availability of sand. On some high sandflats vegetated dunes may develop. Occasionally these islands move in the direction of the sand transport, and may eventually disappear.
In quiet places under favourable conditions a salt tolerant pioneer vegetation may develop on tidal flats. When the pioneer vegetation is succeeded by a vegetation of the next successional

stage, the young, low-lying saltmarsh will maintain itself by enhancing sedimentation. In a period with sea-level rise, the marsh will grow higher but if the tidal flat lags behind, a cliff will form along the marsh. Subsequently the marsh will erode until a new vegetation will develop on the bare gently sloping tidal flat.
Although the system as a whole will contain the major elements of marine or estuarine system, any specific part of the system may not always contain all elements.

It is essential to determine which elements of the ecosystem must be regarded as 'critical capital'. Critical capital are elements which are not replaced when destroyed. It can be processes, geomorphological or biogenic structures, species, or elements like scenic beauty. Apart from the critical capital, 'other capital' may also be impacted by human activities. The distinction between 'capital' and 'other' is not always clear: this may be a matter of taste, time-scale of definition. Several features in the North Sea are clearly unreplacable at times scales that are relevant to present day humans. These include geomorphological structures that have been present at least since the formation of the North Sea and that cannot be replaced, unless for instance, a new ice-age completely reshapes the landscape. Examples of such features are: the gravel beds at the Klaver Bank, the Brown Ridge and the system of the Hinder Banks. A similar argument can be followed for biota. If a species, a race of even a gene is made extinct, this feature is clearly lost.

For both abiotic and biotic critical capitals, there is the problem of scale, however. For instance, the Klaver Bank may be the only gravel bed in the Dutch part of the North Sea, is is not unique if the whole North Sea is considered. Likewise, a certain species may have become extinct in the Netherlands, while still thriving elsewhere. It is thus important to properly define the physiotopic region with area (A) that is to be considered in any impact assessment. If A is known (or defined) this may be compared to the area (a) impacted chronically by a given human activity. The total area A may be a specific habitat, depth zone, subtratum, etc, within the sea, relevant for any indicator species for biodiversity. The ratio of a/A may then be used in an impact assessment. As chronic impacts are likely to be important, the time should also be an important factor here. Also, a weighting factor should be included to assess the extent of impact. An impact in area a may be total, i.e. lethal for all members of a given species or removing the total amount of an abiotical feature, or the impact may be partial (some animals escape, some abiotic elements are spared). This weighting factor corresponds to the quality factor Q in the main text:

$$\text{Impact} = f(a / A, t, Q)$$

The impact runs from 0 (Q equals 0), *via* nearly 0 (a and/or Q nearly 0, or $a \ll A$) to 1 ($a=A$ **and** $Q=1$). In theory, this simple equation should provide a simple yardstick for impact assessment of human activities. In practice, a and A must be known, measured or defined, and Q must be evaluated. Note that Q may be very different for different features/biota within area a and a sensible aggregation of individual feature contributions to Q should be applied. Also, A should be determined consistently as with different interpretations of what A should be, different scores will arise. Finally, a should be determined for the whole period t in order to take the importance of the chronic situation into account.

PUBLICATIEREEKS GRONDSTOFFEN (prijzen gelden niet voor RWS-ers)

Nr.	Titel: subtitel (cursief)	DWW-Nummer	Prijs
1995/01	Zuinig omgaan met granulaire grondstoffen: *Voorstudie*	W-DWW-95-505	*f* 20,00
1995/02	De stand van het zand II: *Beton- en metselzandverbruik per provincie 1991-1993*	W-DWW-95-512	*f* 15,00
1995/03	Proefprojekt AVI-slakken in rijksweg 15: *Covernota*	W-DWW-95-513	*f* 10,00
1995/04	Proefprojekt AVI-slakken in rijksweg 15: *Basisrapport*	W-DWW-95-514	*f* 20,00
1995/05	Het Bouwstoffenbesluit en de Rijkswaterstaat	W-DWW-95-523	*f* 25,00
1995/06	Onderzoek naar de verkitting van AVI-bodemas: *stand van zaken*	W-DWW-95-520	*f* 10,00
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1995/09	Prototype Simulatiemodel Matflow: *Eindrapport*	W-DWW-95-522	*f* 7,50
1995/10	Richtlijn voor de toepassing van licht verontreinigde grond	W-DWW-95-528	*f* 20,00
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1995/14	Gebruik van Secundaire Grondstoffen bij de Rijkswaterstaat: *1993-1994 Evaluatie*	W-DWW-95-538	*f* 10,00
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1996/04	Proefproject metselwerkgranulaat dam Ventjagersplaat (eindconclusie)	W-DWW-96-053	*f* 10,00
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1997/10	Opnamecapaciteit van de wegenbouw voor secundaire materialen *Bepaling van de maximaal mogelijk vraag naar funderingsmaterialen en naar ophoogmaterialen van categorie 2 en de bijzondere categorie*	W-DWW-97-037	*f* 25,00
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