Ministerie van Verkeer en Waterstaat

j k l m n o p q

Dienst Weg- en Waterbouwkunde

Improving and testing a land use methodology for LCA

Including case-studies on bricks, concrete and wood

May 2002

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Colofon

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Voorwoord

Milieugerichte levenscyclusanalyse (LCA) is van groot belang geworden voor het beoordelen van producten en diensten op hun milieuaspecten. In de LCA-traditie ligt de nadruk op milieuthema's die samenhangen met procesemissies en in mindere mate op grondstoffengebruik, terwijl directe ingrepen in de omgeving zoals landgebruik, straling en geluid weinig aandacht krijgen. Er zijn gestandaardiseerde werkwijzen ontwikkeld en databases met dergelijke milieugegevens van honderden processen. Directe ingrepen in de omgeving zijn van dominant belang voor de mondiale biodiversiteit, maar blijven hierbij veelal buiten beschouwing. Dit leidt ertoe dat LCA-uitkomsten een onvolledig beeld geven van de milieueigenschappen van producten, wat in sommige gevallen kan leiden tot onjuiste keuzes.

Daarom heeft de Dienst Weg- en Waterbouwkunde (DWW) een methode laten ontwikkelen om landgebruik in LCA mee te kunnen nemen. Het voorliggende rapport is het eindresultaat van een traject dat in 1995 is begonnen met een voorstudie[1]. De oorspronkelijke reden om dit onderzoek op te starten was dat het thema 'aantasting van ecosystemen en landschap' zoals genoemd in de CML-handleiding (1992)[2] niet operationeel was en dat dit van belang is voor LCA's die worden uitgevoerd binnen het werkterrein van Verkeer en Waterstaat; gedacht werd aan grondstoffen, afvalverwerking en infrastructuur. Steeds is voor ogen gehouden dat een algemene methode moest worden ontwikkeld, die voor alle soorten processen over de hele wereld toepasbaar is, net als de andere karakterisatiemethoden in de CML-handleiding. In 1997-1998 is een methode uitgewerkt[3]. Het beproeven van deze methode op een case was de voorziene vervolgstap in de ontwikkeling.

Parallel hieraan besloot het kabinet bij de vaststelling van het Structuurschema Oppervlaktedelfstoffen (I), deel 3, dat er een LCA van baksteen en beton moet worden gedaan omdat de winning van grondstoffen voor baksteen minder nadelige effecten zou hebben dan de winning van grondstoffen voor beton. Vrij kort daarna is door het CML in opdracht van de DWW een studie uitgevoerd waarin is aangetoond dat een dergelijke LCA nog niet uitgevoerd kon worden, onder andere omdat een methode om effecten van landgebruik te kwantificeren ontbreekt[4].

In 1999 is door overheid (Ministerie van Verkeer en Waterstaat) en bedrijfsleven (Nederlands Verbond Toelevering Bouw, Koninklijk Verbond van Nederlandse Baksteenfabrikanten en Betonplatform) gezamenlijk besloten om, in aanvulling op de beschikbare LCA-gegevens, LCA-landgebruikgegevens te verzamelen voor baksteen, beton en hout. Hierdoor kon de wens van het kabinet worden gecombineerd met de wens om de ontwikkelde methode te beproeven. Dit heeft in 2002 heeft geresulteerd in voorliggend rapport.

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1 H. Blonk en E.W. Lindeijer, Naar een methodiek voor het kwantificeren van aantasting in LCA, Publicatiereeks Grondstoffen 1995/15, IVAM i.o.v. DWW, W-DWW-95-545, 1995.

2 R. Heijungs (red.), Milieugerichte levenscyclusanalyses van produkten: handleiding en achtergronden, Centrum voor Milieukunde, Leiden, 1992.

3 E.W. Lindeijer e.a., Biodiversity and life support indicators for land use impacts in LCA, IVAM i.o.v. DWW, W-DWW-98-059, 1998.

4 [L. van Oers], Voorstudie LCA keramische producten en betonproducten, CML i.o.v. DWW, W-DWW-97-070.

Enkele constateringen die kunnen worden gedaan zijn de volgende:

1. het is gelukt om een methode te ontwikkelen waarmee het thema landgebruik in LCA's kan worden gekwantificeerd; de methode sluit aan bij de LCA-traditie, ondanks het feit dat landgebruik een heel ander karakter heeft dan de emissies van stoffen die in LCA de toon zetten;
2. de methode is data-intensief en vereist data die in de praktijk veelal niet voor milieudoeleinden verzameld worden. Toch bleek het mogelijk om een database op te bouwen met een groot aantal (basis)processen, wat de methode praktisch toepasbaar maakt;
3. de uitkomsten blijken sterk afhankelijk van methodische keuzen die bepaald worden door waardeoordelen, met name de keuze voor een referentieniveau voor biodiversiteit;
4. het in de CML-handleiding van 1992 benoemde thema 'aantasting van landschap' heeft betrekking op beleving en is daarom sterk subjectief, op mensen gericht, en past niet in het kader van milieugerichte LCA;
5. de indicatoren voor effecten op ecosystemen zijn grof;
6. de methode levert voor de casestudie geen contra-intuïtieve resultaten op.

Op enkele van deze punten zal kort worden ingegaan.

Ad 3. Afhankelijkheid methodische keuzen
LCA is een model van de werkelijkheid, waarbij methodische keuzen soms grote invloed hebben op de uitkomst. Daardoor kan een abstracte methodische keuze de uitkomsten sterk beïnvloeden en bepaalde producten bevoordelen of benadelen. Dit geldt niet alleen voor landgebruik, maar ook voor andere milieuthema's. Er is bijvoorbeeld significante invloed van keuzen die men maakt over de tijdsperiode bij het milieuthema klimaatverandering en de toerekening van milieueffecten aan individuele producten.
Dergelijke methodische keuzen zijn vaak subjectief, waarbij verschillende mensen afhankelijk van hun wereldbeeld en belangen verschillende opties aanhangen. In dit geval blijkt dat als men natuurgericht (ecocentrisch) denkt en 'pure natuur' als referentie hanteert bosbouw – en dus hout – negatief scoort. Bij een meer mensgerichte (antropocentrische) denkwijze is de gemiddelde kwaliteit de referentie en scoort bosbouw positief.
In beleidsgerichte LCA's kan men hier vaak mee omgaan door beide opties naast elkaar te zetten, maar indien men LCA wil inzetten in de ontwerppraktijk zal een daartoe bevoegde instantie een keuze moeten voorschrijven. In de beleidspraktijk wordt er vaak gewerkt met het standstillbeginsel – de kwaliteit moet niet slechter worden – maar zelden gestreefd naar 'pure natuur'. Gesteld kan dan ook worden dat in dit geval de mensgerichte benadering beter aansluit bij de in het Nederlandse beleid rond hout gangbare denkwijze.

Ad 4. Verandering van landschappen (belevingswaarde)
Indien men veranderingen in landschappen in de besluitvorming over producten wil betrekken is gangbare milieugerichte LCA daarvoor klaarblijkelijk niet het juiste kader. Dit deelaspect heeft betrekking op mensen en hun beleving van de omgeving en laat zich niet in een natuurwetenschappelijk kader dwingen. Wellicht past dit aspect beter in een andere reeks aspecten die soms bij de beoordeling van producten een rol spelen, maar ook niet in het LCA-milieuprofiel of de kosten tot uitdrukking komen: arbeidsomstandigheden, cultuurbehoud, dierenwelzijn, kinderarbeid, mensenrechten en sociaal-economische ontwikkeling. Men zou kunnen denken aan een sociaal-

economisch-culturele levenscyclusanalyse. Dit is geen volledig nieuw concept. Op academisch niveau is hier al wel over gedacht, en er is ook wel ervaring met combinaties van LCA en meer op mensen gerichte hulpmiddelen zoals maatschappelijke kostenbatenanalyses. Een praktische uitvoeringspraktijk zoals deze er is voor milieugerichte LCA ontbreekt echter volledig.

Ad 5. Grofheid indicatoren
Om LCA's mogelijk te maken moeten extreme vereenvoudigingen van de werkelijkheid worden gemaakt. In de in voorliggend rapport beschreven methode worden de effecten op ecosystemen uitgedrukt in slechts twee indicatoren: aantallen soorten groene planten en de netto primaire biomassaproductie. Alleen langs deze weg is het nog redelijkerwijs mogelijk iets te zeggen over de mogelijke milieueffecten die een product in de keten van wieg tot graf kan veroorzaken. Wanneer men naar een individueel milieueffect van een individueel proces kijkt, is de LCA-indicator veel te ruw; deskundigen zullen deze dan ook aanvechten. De 'wet van de grote getallen' maakt echter dat voor complexere producten de LCA-indicator een bruikbare indicatie geeft. Er kan een parallel worden getrokken met andere thema's in LCA, zoals toxiciteit. Ook hier zijn grove versimpelingen gemaakt; dit geldt nog sterker voor de methoden uit de CML-handleiding van 1992, die toch zo'n 8 jaar veel gebruikt werden en pas rond 2000 vervangen zijn.
Deze vereenvoudigingen leggen aan de LCA-praktijk beperkingen op. Een eerste beperking is het in normen en standaarden benadrukte vereiste dat men zich goed bewust is van de vereenvoudigingen en eventuele gevolgen daarvan voor de uitkomsten. De opkomst van routine-LCA's en LCA's door ontwerpers maakt het echter vrijwel onmogelijk dat dit ook daadwerkelijk gebeurt in de praktijk: de verantwoordelijkheid ligt nu bij degenen die hulpmiddelen aanbieden voor dergelijke LCA's.
Een tweede beperking is het toepassingsgebied. De vereenvoudigingen zijn alleen gemaakt omdat men LCA's wil doen: de bijdragen van honderden processen aan een product verzamelen en op een systematische manier optellen. Waar men zich vooral bezig houdt met één proces, bijvoorbeeld de locatiekeuze voor zandwinning in Europa, biedt LCA te ruwe informatie om zinvolle uitspraken te kunnen doen. Voor dergelijke keuzes kunnen locatie-gerichte methoden zoals milieueffectrapportage (MER) worden gebruikt.

Ad 6. Resultaten
Het is nooit de bedoeling geweest om uitspraken te doen over de drie onderzochte bouwproducten, maar alleen om de methode te beproeven. De wijze van dataverzameling en de berekeningen hebben tot doel gehad deze test te kunnen uitvoeren en niet het doel gehad om uitspraken te doen over de producten. De test heeft veel geleerd over het praktisch toepassen van de methode, geleid tot enkele aanpassingen aan de methode en een grote hoeveelheid data opgeleverd.

Ondanks het feit dat geen uitspraken over de bouwproducten beoogd zijn, is het nuttig om de resultaten te bestuderen teneinde zich een beeld te vormen van mogelijke uitkomsten van het gebruik van de methode. Bij de gehanteerde case blijkt dat baksteen en beton in dezelfde orde van grootte uitkomen. Dit is min of meer te verwachten omdat beide producten uit oppervlaktedelfstoffen worden vervaardigd. Belangrijk om te zien is dat winningen met veel aandacht voor herinrichting een positieve bijdrage leveren aan het milieuthema ecosystemen. Een andere relevante constatering is dat de grondstofwinning slechts een beperkte bijdrage levert aan het landgebruik in de levenscyclus van deze producten. Met name de energieopwekking voor het bakken van baksteen en het maken van cement levert een belangrijke bijdrage. Ook de

ruimte voor de bedrijfsgebouwen en de infrastructuur voor het vervoeren van grondstoffen en producten is significant.
Het derde onderzochte bouwmateriaal, hout, wijkt sterk af. Voor bosbouw is een hoog fysiek landgebruik nodig dat andere processen in de keten in de schaduw zet. Bosbouw kan afhankelijk van methodische keuzes (zie punt 3) gunstig of ongunstig worden bevonden, wat ertoe leidt dat hout veel slechter of juist veel beter kan scoren dan de steenachtige producten.

Uit het bovenstaande kunnen de volgende conclusies worden getrokken:

- De methode is niet bedoeld en dus ook niet geschikt voor het beantwoorden van vragen over de plaats waar men het beste bepaalde grondstoffen kan winnen, ook niet op een wat grotere schaal (zoals Europa). Indien men zich dergelijke vragen stelt, dan kunnen een MER of een maatschappelijke kostenbatenanalyse relevante antwoorden geven.
- Uit de indicatieve uitkomsten blijkt dat goed heringerichte grondstoffenwinningen een gunstig effect hebben op het ecosysteem. De maatschappelijke en politieke weerstand tegen ontgrondingen zijn van een karakter dat zich niet laat meten met een natuurwetenschappelijk georiënteerde, kwantitatieve methode als LCA.
- De methode is, ondanks beperkingen, geschikt voor beleidsvragen met een abstracter karakter, waarbij naast landgebruik ook andere milieuthema's een rol spelen. Gedacht wordt bijvoorbeeld aan keuzes over de vraag hoe we met bepaalde afvalstoffen om willen gaan.
- De methode is, ondanks beperkingen, ook praktisch bruikbaar in LCA in de bouw. Voor toepassing in de dagelijkse LCA-praktijk zal nog veel werk moeten worden verricht, met name op het gebied van dataverzameling; ook is meer praktijkervaring nodig voordat men overgaat op dagelijks gebruik.

Op basis van het rapport, na overleg met de begeleidingscommissie, doet de DWW de volgende aanbevelingen:

Aan beleidsmakers:
- hanteer de methode voor landgebruik bij eventuele beleidsstudies waar landgebruik een thema is, zoals LCA-studies op het gebied van afval- en grondstoffenbeleid;
- gebruik naast LCA ook andere instrumenten in die gevallen waarin mensgerichte aspecten ook relevant zijn voor de besluitvorming;
- zorg voor financiering van completering van de LCA-methode en dataverzameling;
- financier over enkele jaren een evaluatie over het gebruik van de methode en bezie vervolgens, op basis van die evaluatie, of verdere ontwikkeling van de methode dan wel praktische invoering in de dagelijkse LCA-praktijk de volgende stap moet zijn.

Aan LCA-uitvoerders:
- hanteer de methode voor landgebruik in casestudies waar landgebruik een relevant thema is en er voldoende tijd is om er op een verantwoorde wijze mee om te gaan;
- verzamel data;
- besteed voor dit milieuthema extra aandacht aan de kritische evaluatie van de uitkomsten in het licht van de gehanteerde methode, gezien de nog beperkte praktijkervaring;

- koppel terug over de ervaringen en evalueer over enkele jaren de verrichte casestudies.

Aan LCA-wetenschappers:

- concentreer de methodiekontwikkeling op de milieuaspecten die nog niet of zwak met LCA tot uitdrukking kunnen worden gebracht en wel relevant zijn voor de besluitvorming over producten, zoals geluid, verdroging, versnippering en ingrepen in aquatische ecosystemen;
- ontwikkel methoden voor ‘mensgerichte LCA’ voor de aspecten die niet goed in het kader van milieugerichte LCA passen.

Aan personen die betrokken zijn bij standaardisatie en richtlijnen voor LCA:

- uniformeer, waar nodig, het gegevensformaat en de werkwijze;
- neem de methode, waar mogelijk, als optie op in standaarden en richtlijnen.

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mei 2002

Preface

Environmental life cycle assessment (LCA) has become of great importance for assessing the environmental impacts of products and services. So far, environmental themes related to process emissions and to a lesser extent resource use were emphasised in LCA, whereas direct interventions in the environment like land use, radiation and noise got little attention. Standard practices have been developed and databases have been filled with such environmental data of hundreds of processes. Direct interventions in the environment have a dominant interest for the global biodiversity, but are usually disregarded. As a consequence, LCA results give an incomplete picture of the environmental properties of products, which may occasionally lead to incorrect choices.

In order to integrate land use in LCA, the Road and Hydraulic Engineering Institute (DWW) has initiated the development of a new method. This report presents the final results of a project which started in 1995 with a preliminary study[5]. The original reason to start this research was that the theme 'degradation of ecosystems and landscapes' as mentioned in the CML guideline (1992)[6] was not operational and that this theme is relevant for LCAs within the working field of the Ministry of Transport, Public Works and Water Management, including topics like construction raw materials, waste management and infrastructure. The focus was to develop a general method which can be used for all types of processes all over the world, just like other characterisation methods in the CML guideline. In 1997-1998 a method was elaborated[7]. Testing this method was the next step in the plan.

Parallel to this project, the Dutch cabinet decided in the Structure Plan on Surface Raw Materials (I), part 3, that an LCA for brick and concrete should be undertaken. The reason for it was an assumption that the extraction of raw materials for brick might have less adverse effects than the extraction of raw materials for concrete. Shortly after the cabinet decision the CML performed a study for the DWW which demonstrated that such an LCA could not yet be performed, partly because a method to quantify the effects of land use was missing[8].

In 1999 the government (Ministry of Transport, Public Works and Water Management) and industry (Dutch Association for the suppliers in the building sector, Royal Association of Dutch Brick Manufacturers and the Concrete Platform) decided to collect LCA land use data for brick, concrete and wood, in addition to existing LCA data. In this way the cabinet's wish could be combined with testing the recently developed method. In 2002, this resulted in the report at hand.

The following conclusions can be drawn from the report:

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5 H. Blonk en E.W. Lindeijer, Naar een methodiek voor het kwantificeren van aantasting in LCA, Publicatiereeks Grondstoffen 1995/15, IVAM i.o.v. DWW, W-DWW-95-545, 1995.

6 R. Heijungs (ed.), Environmental Life Cycle Assessment of Products: Guideline and Backgrounds, Centrum voor Milieukunde, Leiden, 1992.

7 E.W. Lindeijer e.a., Biodiversity and life support indicators for land use impacts in LCA, IVAM i.o.v. DWW, W-DWW-98-059, 1998.

8 [L. van Oers], Voorstudie LCA keramische producten en betonproducten, CML i.o.v. DWW, W-DWW-97-070.

1. it was possible to develop a method to quantify land use in LCA; the method joins in with the LCA tradition, in spite of the fact the land use has a completely different character than the substance emissions which set the tone in LCA;
2. the method is data intensive and asks for data which are usually not collected for environmental reasons. Nevertheless it was possible to build a database containing a large number of (basic) processes, which enables practical application of the method;
3. the results appear to be strongly dependent on methodical choices which are determined by value judgements, especially the choice for a reference level for biodiversity;
4. the theme ‘degradation of landscapes’ which was mentioned in the CML guideline of 1992 relates to perception and is strongly subjective, focussed on humans, and does not match the framework of environmental LCA;
5. the indicators for effects on ecosystems are rough;
6. for this case study, the method does not lead to contra-intuitive results.

In the next section some of these conclusions will be elaborated on.

Ad 3. Dependency on methodical choices
LCA is a model of reality, where methodical choices can significantly influence the results. Therefore, a theoretical methodical choice can strongly influence the results and lead to a bias for or against certain products. This applies not just to land use, but also to other environmental themes, for instance the time period for the environmental theme climate change and the allocation of environmental effects to individual products.
Such methodical choices are often subjective, where different people will adhere to different options based on their own world views and interests. In this case it appears that forestry – and therefore wood – has a negative score if one thinks ecocentrically and uses ‘pure nature’ as a reference. However, it has a positive score in a more anthropocentric way of thinking, where the average quality is used as a reference.
In policy-oriented LCAs one can usually handle this by contrasting both options in the LCA report. However, when LCA is used in the design practice a competent institution will have to prescribe a choice. The stand still principle is commonly used in the policy practice: the quality should not become worse. Only seldom ‘pure nature’ is aspired. It can be stated that the human-oriented approach is more in keeping with the way of thinking in the Dutch policy around wood.

Ad 4. Change of landscapes (perception value)
The framework of environmental LCA appears to be inappropriate for changes in landscapes. This aspect involves people and their perception of the environment and does not fit into a framework which is based on natural sciences. This aspect might better fit in a series of other aspects which are sometimes considered in the assessment of products, but are not taken into account in the environmental profile or the costs: animal welfare, child labour, culture conservation, human rights, labour conditions and social-economic development. One could consider a social-economic-cultural life cycle assessment. This is not a completely new concept, since it has been considered on an academic level and some experience exists with combinations of LCA and other tools such as social cost-benefit analysis. However, there is no practical experience such as exists for environmental LCA.

Ad 5. Roughness of indicators
Reality needs to be simplified extremely to enable LCAs. Hence, the effects on ecosystems are measured with only two indicators in the method described in the report at hand: the number of green plant species and the net primary biomass production. Such simplifications are inevitable if one aims to say something about the possible environmental effects which a product can cause in the chain from cradle to grave. When one focuses on an individual environmental effect of an individual process, the LCA indicator is too rough; experts will therefore fight it. However, the 'law of great numbers' makes that for complex products the LCA indicator will give a useful indication.
A parallel can be drawn with other themes in LCA, such as toxicity, where gross simplifications have been made; this applies even stronger for the methods in the CML guidelines of 1992, which nevertheless have been used much during 8 years until they were updated around 2000.
The simplifications lead to restrictions for the LCA practice. A first restriction is the requirement that each practitioner is well aware of the simplifications and their consequences for the results, as required by norms and standards. However, the rise of routine LCAs and LCAs by designers makes it virtually impossible that this also happens in practice: the responsibility now lies with those that offer tools for such LCAs.
A second restriction is the field of application. The only reason for the simplifications is the LCA framework: collect the contributions of hundreds of processes to a product and add them systematically. Where one focuses on one process, such as a location choice for sand extraction in Europe, LCA offers too unprecise information to do significant assertions. Location specific methods such as Environmental Impact Assessment (EIA) can be used for such choices.

Ad 6. Results
It was never aimed to make assertions about the three building raw materials at hand, but only to test the method. The way of data collection and the calculations were primarily focussed at the ability to test and not at the ability to make assertions about the products. The test provided useful knowledge about the practical applicability of the method, led to some adaptations of the method and resulted in a large collection of data.

Although no assertions about the building products were intended, it is useful to study the results as a means to form an image of possible results of the method. In this case study it appears that brick and concrete end up in the same order of magnitude. This result can be expected more or less because both products are manufactured of surface raw materials. It is important to see that raw materials extractions with much attention for restoration lead to a positive contribution to the environmental theme ecosystems. Another relevant conclusion is that raw materials extraction has a limited contribution to the land use in the life cycle of these products. Especially energy production for baking brick and producing cement has a large contribution. Also the space for factories and infrastructure for transporting raw materials is significant.
The third building product which was studied here, wood, strongly differs from brick and concrete. Forestry requires a very high physical land use, which overshadows the land use of other processes in the chain. Depending on methodical choices (see point 3) it can score favourable or unfavourable, which means that wood can score either much worse or much better than the stony products.

The following conclusions can be drawn from the above:

- The method is not designed nor suitable for answering questions on the optimal location for extracting certain raw materials, even not on a somewhat larger scale (like Europe). If one asks such questions, than an Environmental Impact Assessment or a social cost-benefit analysis can give relevant answers.
- The indicative results suggest that well-restored mineral extraction sites have a beneficial effect on the ecosystem. The societal and political opposition against mineral extraction are of a character which can not be measured with a scientifically oriented, quantitative method like LCA.
- The method is, despite its restrictions, applicable for policy questions of an abstract character, where next to land use other environmental themes play a role, for instance choices on the question how to deal with certain waste materials.
- The method is, despite its restrictions, practically applicable in LCAs in building and construction. However, before application in the daily LCA practice, a lot of work needs to be done, especially in the field of data collection. It is recommended to gain more practical experience before the method is taken into daily use.

Based on the report, after consultation of the guiding committee, DWW has the following recommendations:

To policy makers:
- apply the method for land use in policy studies where land use is a theme, such as LCA studies in the field of waste and raw materials policy;
- use next to LCA other instruments in those cases where human-oriented aspects are relevant for decision making;
- finance completion of the LCA method and data collection;
- finance an evaluation of the use of the method in a few years time and use this evaluation to consider whether the next step should be further development or practical introduction in the daily LCA practice.

To LCA practitioners:
- apply the method for land use in case studies were land use is a relevant theme and enough time is available to take this into account in a responsible way;
- collect data;
- pay extra attention to the critical evaluation of the results for this environmental theme in the light of the method which was used, given the limited practical experience;
- give feedback on the results and evaluate the case studies which have been carried out in a few years.

To LCA scientists:

- focus methodology development on the environmental aspects which can not be expressed in LCA, or only weakly, and are relevant for decision making on products, like noise, desiccation, fragmentation and interventions in aquatic ecosystems;
- develop methods for 'human-oriented' LCA for the aspects which do not match the framework of environmental LCA.

To persons who are involved in standardisation and guidelines for LCA:
- uniform, where needed, the data format and the work practice;
- include the method, where possible, as option in standards and guidelines.

Joris Broers
project manager Ministry of Transport, Public Works and Water Management,
Road and Hydraulic Engineering Institute
May 2002

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Samenvatting

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Inleiding

Het onderhavige project komt voort uit de behoefte van de Dienst Weg- en Waterbouwkunde van Rijkswaterstaat (RWS DWW) om effecten van landgebruik te kunnen aangeven bij bijvoorbeeld het gebruik van baksteen of beton, (tropisch hard)hout, gebruik van grondstoffen uit zee of van land, of meer houtgebruik in Nederland binnen het kader van de milieukundige levenscyclusmethode (LCA). LCA is een methode waarbij de milieu-effecten over de gehele levensloop van een product of dienst worden geanalyseerd. In 1998 was er al een beoordelingsmethode voor landgebruik in LCA ontwikkeld door IVAM Environmental Research, voor RWS DWW. Sinds dat onderzoek zijn internationaal meerdere methodische studies gedaan naar landgebruik in LCA. Daarnaast werd door Rijkswaterstaat en bedrijfsleven besloten dat er LCA-landgebruiksgegevens verzameld moesten gaan worden, ter aanvulling van emissie-gegevens over producten en processen. Om deze redenen is onderhavige studie uitgezet.

Doel

Doel van het huidige onderzoek is:
1. Het aanpassen van de IVAM methodiek aan de laatste methodische ontwikkelingen;
2. Het verzamelen van relevante landgebruiksgegevens voor het uitvoeren van LCAs in de bouwsector in Nederland;
3. Het testen van de bruikbaarheid van de TNO methodiek voor een aantal cases.

Oorspronkelijk stonden de laatste twee doelen voorop. Door extra financiering van Delft Cluster kon het eerste doel meer nadruk krijgen. Hiermee zijn de doelstellingen in het oorspronkelijk onderzoeksvoorstel aangepast.

Aanpak

De studie is uitgevoerd in 2 stappen. Eerst is een voorstudie gedaan om in te kunnen schatten hoe lastig de gegevensverzameling zou zijn en welke methodische problemen er zijn. Hieruit bleek dat de gegevensverzameling zeer lastig is. De daaropvolgende tweede stap –de gedetailleerde studie- is derhalve verder voor een belangrijk deel gericht op de incorporatie van de verzamelde gegevens in databases. In beide stappen is aandacht gegeven aan methodische keuzen (zie ook annex 1). Er is niet ingegaan op de inbedding van de methode in de Nederlandse LCA-praktijk. In deze studie stonden dus methodische verbeteringen en een uitgebreide gegevensverzameling centraal. Het project is uitgevoerd van augustus 2000 tot april 2002.

Resultaten methodische aanpassingen

De uitgevoerde methodische verbeteringen zijn:
- Het opnemen van effecten op (mondiale) ecosystemen naast effecten op plantensoorten (in de biodiversiteits-indicator);
- Het ontwikkelen en operationaliseren van een methodiek op basis van het principe dat de mens centraal staat in het onderwerp landgebruik, naast de meer ecologisch gerichte methodiek uit 1998;
- Het aanpassen van de (locale) plantensoortenindicator uit 1998 aan recente voorstellen inzake de LCA landgebruiksmethodiek en aan recent gepubliceerde datasets;
- Een methode die sterker reageert op regionale differentiatie (verschillen tussen waar het landgebruik plaatsvindt);
- Een algemeen toepasbare manier van het beschrijven van landgebruiks-ingrepen in databases.

De verbeteringen betreffen tevens meer analyse-opties (dus ook meer interpretaties), meer data beschikbaar en meer geharmoniseerd met huidige ontwikkelingen. In hoofdstuk 7 worden verdere methodische onderzoeksrichtingen op basis van dit onderzoek kritisch beoordeeld.

Resultaten gegevensverzameling

Na de conceptuele ontwikkeling van de TNO-methode voor landgebruik in LCA, zijn fysieke landgebruiksgegevens verzameld voor:

- ontgrondingen (winning van oppervlaktedelfstoffen) in Nederland;
- bosbouw wereldwijd;
- diverse landbouwpraktijken;
- diverse mineraalwinningen wereldwijd;
- de energie-sector in Europa (middels de zgn. ETH database);
- transport over de weg, over het spoor en over kanalen in Nederland;
- industriële productie algemeen in Nederland;
- wonen in Nederland;
- afvalverwerking in Nederland.

Bovenstaande gegevens zijn ingevoerd in een operationele LCA database. De gegevens voor oppervlaktedelfstoffenwinning zijn door de begeleidingscommissie overigens alleen geauthoriseerd voor de toepassing in de cases.

Het proces om tot de landgebruiks dataset inclusief beoordelingsmethodiek te komen heeft ruim een manjaar werk gevergd. De meer dan 250 afzonderlijke landgebruiks ingrepen zijn verbonden met meer dan 20 beoordelingsklassen. Deze omvatten voor de biodiversiteitsindicator: aparte minimum, maximum en gemiddelde waarden, en voor beide indictoren aparte waarden voor het bezet houden van land, voor veranderingen in landgebruik en voor de herstelperiode na een ingreep. Daarnaast zijn diverse gevoeligheidsanalyses uitgevoerd, naar het gebruik van een gemiddelde of een maximale referentiewaarde, en naar het al dan niet meenemen van de gehele herstelperiode.

Bij de effectbeoordeling op biodiversiteit blijkt een grote onzekerheid te zitten in de globale soortendichtheid per biome. Meer gedetailleerde gegevens verzamelen is niet mogelijk echt mogelijk, omdat er een grote natuurlijke variatie in de data zit. Per type landgebruik is meestal gebruik gemaakt van Zwitserse en Duitse gegevens over de soortendichtheid.

Voor biomassa als indicator voor het ecologisch draagvermogen van de aarde, is het de vraag of niet beter staande biomassa gekozen kan worden dan NPP, omdat dat de gegevensverzameling zou vergemakkelijken. Onderzocht zou moeten worden of staande biomassa eenzelfde relatie met het ecologisch draagvermogen heeft. Dit is in deze studie niet nader onderzocht.

Resultaten cases

Met de TNO landgebruiks methode en –database is het landgebruik in Nederland geanalyseerd en vergeleken met de analyse volgens de Eco Indicator 99. Ook zijn case studies uitgevoerd voor 1 m^2 buitenwand van voornamelijk baksteen, of beton, of hout. Voor elke case zijn steeds twee winningstechnieken of -plaatsen vergeleken. Voor ontgrondingen is gekeken wat het effect van moderne (ecologische) herstelwerkzaamheden is, en voor hout is het verschil tussen Europees en Amerikaans hout bekeken.

De conclusies die getrokken kunnen worden uit de ervaringen met de case studies zijn:

- De meest gevoelige methodische keuze blijkt die van de referentie bij occupation te zijn. Hierdoor kunnen effectscores ineens negatief worden (met als interpretatie “goed voor het milieu”).
- De onzekerheid in beoordelings-scores zijn groot door de natuurlijke variatie in biodiversiteits- en life support indicator waarden; het vergt een uitgebreide dataverwerkingsstap om deze onzekerheid mee te kunnen nemen met de huidige LCA software.
- De verzamelde gegevens zouden moeten worden opgenomen in andere algemeen toegankelijke LCA databases zodat LCA uitvoerders elders de verzamelde data ook kunnen gebruiken.

- De interpretatie van landgebruikseffecten is niet eenvoudig, door de grote onzekerheidsranges, optredende negatieve scores en methodische keuzen die inherent aan de methodiek zijn. Alleen als een aantal basale keuzen van tevoren gemaakt zouden worden (gemiddelde of maximale referentie, met of zonder herstelperiode etc.) zou er een simpel plaatje ontstaan met twee effectscores per alternatief.
- Het meenemen van de herstelperiode na ontgrondingen heeft een grote invloed op de effectscores, en kan zelfs leiden tot negatieve scores.
- Hout heeft een veel hogere danwel een veel lagere effectscore voor het *bezet houden* van land dan andere grondstoffen, afhankelijk van de keuze van het referentie-niveau (maximaal of gemiddeld, respectievelijk). Methodologische keuzen zullen dus de resultaten van landgebruiks-beoordelingsmethoden bepalen, tenzij deze keuzen expliciet worden gemaakt.
- Bij de cases met oppervlaktedelfstoffen is de bijdrage van het energie-systeem meer dan de helft van de totale landgebruiks score. Verbeteringen in de energie-efficiency kan dus mogelijk een even goede bijdrage leveren als herstelaktiviteiten in dergelijke gevallen.
- Transformatie effect scores (door het *veranderen* van landgebruik) zijn minder dan een factor 10 hoger voor hout dan voor andere materialen; door het lagere gebruik van per functionele eenheid in de cases komtde effectscore voor hout in dezelfde ordegrootte als voor andere materialen. Dit impliceert dat de keuze tussen hout, beton en baksteen in de praktijk niet bij voorbaat vast ligt in deze methode (voor zover deze keuze beïnvloed wordt door landgebruikseffecten).
- De huidige methodiek kan een bijdrage leven aan het in (mondiaal) perspectief plaatsen van milieukundige discussies rondom landgebruik door er voor te zorgen dat er consistent over de gehele levenscyclus geredeneerd wordt vanuit dezelfde aannames, omdat alle details in de resultaten uitgelegd kunnen worden en normatieve keuzen beheerst kunnen worden in de beoordeling.
- De Eco Indicator 99 landgebruiksmethode [Goedkoop & Spriensma, 1999] kent geen regionale differentiatie, en kan dus niet zo goed het effect van bijvoorbeeld het gebruik van tropisch hout aantonen als de TNO methode.

De voordelen van deze genuanceerde beoordelingsmethode staan tegenover het risico van het zich verliezen in details en de veelheid van mogelijkheden. Versimpeling is ook riskant, maar ten behoeve van een betere communiceerbaarheid zou de mogelijkheid onderzocht moeten worden van:

- ✓ Het weglaten van minmale en maximale effectscores, en alleen gebruiken van gemiddelde waarden;
- ✓ Het optellen van gegevens over bezet houden en van de herstelperiode; idem dito voor verandering van landgebruik;
- ✓ Een verdere aggregatie met andere milieu-effecten om tot enkelvoudige scores te komen, voor toepassing in een beperkt aantal situaties.

Het uitvoeren van een landgebruik-beoordelingsmethodiek in LCA-kader heeft onvermijdelijk beperkingen ten gevolge van:

- de natuurlijke variatie in indicatorscores en benodigde regionale differentiatie;
- de onzekerheid betreffende aspecten van de herstelprocessen;
- waarde-oordelen met betrekking tot de keuze van indicatoren en details in de formules voor de effectfactoren.

Daarnaast zijn er beperkingen die samenhangen met de TNO methode, die overigens ook gelden voor alle bestaande landgebruiksmethoden:

- Effectgerichte data voor onttrekkingen zijn momenteel nog beperkt beschikbaar;
- Landschappelijke aantasting (de esthetische component) wordt niet meegenomen;
- Verdroging wordt niet meegenomen (wel in een parallel project, [Lindeijer et al., 2002]);
- Versnippering wordt niet meegenomen;
- Alleen plantensoorten worden meegenomen bij soortenbiodiversiteit;
- Zee-ecosystemen worden momenteel nog niet meegenomen bij biodiversiteit.

De laatste beperkingen worden nader besproken in hoofdstuk 2.

Bij gebruik moeten deze beperkingen in het oog worden gehouden. Hierdoor zal een LCA-studie over landgebruik alléén niet voldoende zijn voor het beoordelen van individuele cases.

Conclusies

De TNO methode voor landgebruik is aangepast aan de laatste methodische ontwikkelingen. De methodiek blijkt goed toepasbaar op de drie cases, met een groot onderscheidend vermogen. De verschillende uitkomsten zijn goed uitlegbaar, en illustreren de grote gevoeligheid voor methodische keuzen. De methodiek laat diverse keuzen toe, maar vereenvoudigingen zijn ook mogelijk.

Summary

Introduction

For the technical and environmental advisory office for road- and waterworks of the Dutch Ministry of Transport, Public Works and Water Management (RWS DWW) the lack of including land use impacts adequately in LCA was seen as a major flaw to LCA studies performed over the last decennia. LCA is a method to assess the environmental impacts of products or services from gradle to grave or gate. Decisions about the use of tropical wood, application of bricks or clay, increased use of wood, inland sand versus sand from the sea, landfilling or recycling could not be supported with the LCA methodology due to this flaw. In 1998 a methodology for land use impacts in LCA was developed for RWS DWW. The present study was meant to improve that method, and to test whether sufficient data is available to use the method.

Goal

The aim of the present study is to:

- improve the existing land use method from 1998;
- to collect relevant land use (impact) data for performing LCAs in the Dutch building sector;
- to test the feasibility of the application of this method in LCA case studies.

Originally the last two goals were the focus. Thanks to additional funding from Delft Cluster the first goal received more emphasis, adjusting the original aims of the study.

The approach taken was tiered. First a screening was performed to assess how difficult data collection would be, and which methodological problems remain. From this screening it became clear that data collection was very problematic. The following detailed study was therefore mainly aimed at incorporating the collected data in databases. In both subprojects there was attention to methodological choices (see also annex 1). The integration of the method in Dutch LCA practice (origionally a goal of the study) was finally not addressed. In this study, methodological improvements and an extensive data collection were therefore central. The project has been performed from August 2000 to April 2002.

Methodological improvements

The methodological improvements made in this project by TNO to the 1998 land use method are:

- Including the (global) ecosystem level of biodiversity in the land use impact assessment (IA);
- Developing a human-oriented LCIA method use, and harmonising the ecological-oriented 1998 method;
- An adapted (local) biodiversity factor allowing the use of recent extensive biodiversity data for LCA;
- A more sensitive regional differentiation than in [Lindeijer et al., 1998];
- A universally applicable format for land use interventions.

The improvements are thus more interpretation options, more data available and a more harmonised method. Recommendations for LCA practioners and for further methodological research are given in chapter 7.

Data collection

Based on this TNO method, physical land use intervention data has been collected for:

- aggregate extraction in the Netherlands;
- forestry worldwide;
- various agriculture processes;
- various mineral extraction processes worldwide;

- the energy production sector in Europe[9];
- transport by roads, railways and canals in the Netherlands;
- industrial production in the Netherlands;
- housing in the Netherlands;
- waste disposal in the Netherlands.

Above data has been recorded in an operational LCA database. The data for surface aggregate extraction are only authorised by the guiding committee for use in the case studies of this project, by the way. The process of arriving at the land use data set and the impact assessment data set for land use has been a task of over 1 personyear of worktime.

These over 250 separate interventions are linked to the over 20 different impact categories (including minimum and maximum values and sensitivity analyses on the use of an average or a maximum reference, and on including only part or all of the renaturation process).

The biodiversity indicator shows a large uncertainty on the global species density per biome. More detailed data collection on this does not seem possible, due to the large natural variation. Often Swiss or German impact data was used for individual land use types.

The NPP as biomass indicator for life support function impacts may after all be changed into standing biomass, as this would facilitate data collection a lot. It should be determined whether standing biomass has a similar relationhip to life support functions as NPP.

Cases

Using the TNO method and land use database, the normalisation data for the Netherlands have been analysed and compared to the Eco Indicator 99. Also case studies have been calculated, using bricks, concrete or wood as main material for a m^2 of outer wall.

A main conclusion is the importance of the choice of a reference level. Whether an afverage or a maximum reference is chosen can make the difference between a positive and a negative score (bad respectively good for the environment). Using an average reference challanges the mainstream LCA paradigm of a zero background level, as it seems to be the most consistent with opinions on renaturation practices. Nevertheless, this reference level choice is open and very important.

Conclusions from the experience with the case studies are:

- Uncertainties in IA factors are large due to the natural variability in biodiversity and life support indicator values; taking this uncertainty even roughly into account required extensive data treatment.
- The collected data should be incorporated in LCA databases for others to use.
- The interpretation of land use impacts is not straightforward due to the large uncertainties, the negative scores (“good for the environment”) and the value choices inherent to the methodology. Only when some basic choices would be set on beforehand, a rather simple picture with two scores per alternative would emerge.
- Taking renaturation after extraction into account has an important influence on the impact scores, and may even lead to negative impact scores (“good for the environment”).
- Wood has much higher or much lower impact scores for *occupying* land than other materials, depending on the choice of reference level (maximum or average, respectively). Methodological choices therefore will influence the results of land use impact assessment in LCA when silviculture (or agriculture) processes are a considerable part of the functional unit, unless these choices are made explicit.
- Transformation impact scores (due to *changing* land quality) are less than a factor 10 higher for wood than for the other materials; due to the lower input per functional unit in the cases, the impact scores become of the same order of magnitude for all materials. This implies that for real life situations, the choice between for instance wood, concrete and brick as main materials is not obvious on beforehand according to the TNO land use methodology (inasfar as this is influenced by land use impact assessment, of course).

[9] Interpretation of the ETH database of 1996

- The present impact assessment method can assist to put environmental discussions in (global) perspective and ensure consistent reasoning all along the life cycle attached to a certain case study, because all details in the (differentiated) results can be explained and value choices can be controlled in the assessment.
- The Eco Indicator 99 method has no regional differentiation and can thus not cope with a case like the use of tropical hardwood as adequately as the TNO method.

Performing any land use impact assessment has inevitably limitations due to:
- ♦ the natural variation of background data and regional differentiation;
- ♦ uncertainty about renaturation processes;
- ♦ value choices involved in the choice of indicators and details of the impact factors used.

Next to these, there are limitations related to the present methodology (as with all existing land use IA methods):
- ♦ For many extraction processes only expert judgements on impacts are available presently;
- ♦ Landscape degradation is not included;
- ♦ Dessication is not included;
- ♦ Fragmentation is not included;
- ♦ Only plant species are included on the species level of biodiversity;
- ♦ Marine ecosystems are not assessed for biodiversity presently.

The latter limitations are discussed more in detail in chapter 2; some are considered more or less inevitable.

Due to these limitations case studies will seldom be assessed adequately by applying only this LCA method. When using it, above methodological limitations should be considered.

Conclusions

The TNO method for land use is adapted to the most recent methodological developments. The methodology was well applicable to the three cases, showing a large resolution for differences between cases. The various results are well explainable, and illustrate the large sensitivity for methodological choices. The methodology allows for various choices, but simplifications are also possible.

1 Introduction

………………………………………………..………………

1.1 Project history

Land use is an important environmental issue, but has hardly been included in LCA studies to date. The importance of land use as an environmental issue is expressed in policy documents such as the Dobris Assessment [EUROSTAT et al., 1995] and in scientific studies such as [Sala et al., 2000].

In 1995 the technical and environmental advisory office for road- and waterworks of the Dutch Ministry of Transport, Public Works and Water Management (RWS DWW) started a framework study (performed by IVAM Environmental Research, [Blonk & Lindeijer, 1995]) to develop a systematic overview and possibilities to include land use impacts in LCA. In 1997 DWW commissioned IVAM ER to perform a research project to develop a land use (impact) methodology, which was reported in 1998 [Lindeijer et al., 1998]. IBN-DLO (a Dutch national research organisation on agriculture and forestry) was subcontracted to provide specialist contributions on the issues of biodiversity and biomass. This report presented a methodology based on two indicators to express impacts of land use on biodiversity and life support functions. It was presented to the sector and to the commissioner in a workshop on November 19th, 1998. The methodological issues were also presented and discussed during a workshop at the SETAC-Europe congress in Bordeaux, April 17th, 1998 (see appendix 2 in [Lindeijer et al., 1998]).

During this period, several public issues related to land use impact assessment arose: where to extract aggregates such as sand and gravel, whether or not to use more bricks at the expense of concrete, and the proposal to use more wood. Such societal issues were the offset for this third land use project commissioned by DWW. The initial aim of the project was to merely test the land use methodology developed in 1998, and to compare it to the Eco-Indicator 99 method. However, due to the possiblity to contribute in a Delft Cluster research programme, the scope could be broadened to include also recent methodological improvements.

This project is carried out by TNO Industrial Technology, with the help of CML (University of Leiden) and the Technical University of Delft. Two foundations (Floron and Ark) have contributed further on data collection.

The reason for DWW to commission this project is to ensure that a feasible land use method is applied, and that the required inventory data is as much available as possible. The underlaying motive is the aknowledged importance of land use impacts.

1.2 Project structure and aim

The TNO project on land use within the Delft Cluster framework consists of 2 phases: Phase 1 deals with improving and testing the applicability of the proposed land use methodology for LCA applications. Phase 2 is a study on the possibilities to include dessication impacts in LCA (see the project report [Lindeijer et al., 2002]). Results from the two phases could influence each other, as they were performed partly at the same time.

The primary aim of phase 1 of the total Delft Cluster project is to include land use impacts in LCA studies of building materials. The results of this study may also be used by DWW to support decisions in the field of land use. The (adapted) aim of the present phase 1 of the Delft Cluster project is:
To improve where necessary the developed land use methodology of 1998, and to test the proposed method in order to judge the land use effects as part of LCA studies, with the focus on the materials brick, concrete and wood.

In order to test the developed land use methodology for LCA, phase 1 has finally been broken down into the following parts two parts:

A. Screening of IVAM-methodology.
B. Specification of methodology and data collection and input in a database for building materials.

Phase A was carried out to estimate roughly the most important land use impacts in the process chain of building materials, the time consumption for data collection and major methodological problems to be solved. As a result, the priority of the various process for the data collection has been determined. On the basis of these results, data collection was focussed and the methodology has been developed in more detail during phase B. This report presents the results of both phases A and B.

In the present phase 1 of the Delft Cluster project DWW was the main commissioner, with additional financial support from TNO Industrial Technology and from Delft Cluster (financed by the revenues of the Dutch gas exploration). The project partners and their roles were:

Partners	Role
- TNO Industrial Technology, Delft	project leader & main performer
- Centre for Environmental Science (CML RUL), Leiden	methodological counterpart
- Design for Sustainability (DfS TUD), Delft	methodological contributor
- Stichting Floron, Leiden	species level data provider
- Stichting Ark, Heilige Landstichting	renaturation data provider

The guiding committee consisted of the following persons:

Persons	Representation
Ruud Nijland	RWS DWW (commissioner, product group manager, chairman)
Joris Broers	RWS DWW (commissioner, project manager)
Gijs Sigmond	Federation of aggregate extractions industry (FODI)
Monique de Moel	Royal Dutch Brick Association (KNB)
Pieter Lanser	Concrete platform, and Dutch Cement Industry's Association (VNC)
Paul van den Heuvel	Netherlands Timber Trade Association (VVNH)
Rogier Goes	Dutch association for the suppliers in the building sector (NVTB)
Adrie de Groot-van Dam	TNO Building and Construction Research, Delft

1.3 Guidance for the reader

The present report builds on the previous reports on land use impact assessment in LCA. LCA terminology is only briefly discussed in the text. See appendix 9 for a glossary of both terminology and abbreviations. Chapter 2 of this report gives an overview of the methodological issues. Methodological details for the operationalisation of the method (including comparison to some other land use impact assessment methods) and the relation to the LCA framework are discussed in annexes 1 and 2, respectively. Chapter 3 presents the gathered information on the physical land use related to functional units, and regional land use (normalisation) data. Chapter 4 shows the available impact assessment data. Chapter 5 shows the results on linking the physical land use and the impact assessment in LCA case studies. Chapter 6 gives the main project conclusions in relation to the goals of the study, followed by recommendations in chapter 7. The main report ends with the references.

The annexes are listed after the references. After annex 1 and 2 the traditional stuation of land use due to aggregate extraction in the Netherlands is discussed (including a report from a meeting with the sector) is presented in annex 3. The contributions of Stichting Floron and Stichting Ark are presented in annex 4 and 5 respectvely. Annexes 3 to 5 are in Dutch. An overview of the conversion of the major processes of the ETH 1996 database to the present format is given in annex 6. In annex 7 an overview of the impact assessment data related to the various interventions is given. Annex 8 contains the detailed case study results, and annex 9 is a glossary of used terms and abbriviations.

2 Land use methodology

In this chapter we give an overview of the land use methodology as developed in this report. For an overview of the methodological choices on recent proposals for land use methodology in LCA we refer to annex 1. A more in-depth discussion on land use in LCA is performed by CML (annex 2). In this chapter the main conclusions are presented.

The method which is presented in this report distinguishes between physical land use (the intervention of humans in the environment) and land use impacts (its consequences and its interpretation in LCA). The interventions, related to the so-called inventory phase of LCA and the impacts (LCIA) are discussed in paragraphs 2.1 and 2.2, respectively.

2.1 Physical land use and its impacts: occupation & transformation

2.1.1 Occupation and its impacts

The basic type of land use related to processes in LCA is the information on the intervention '**occupation of land**", as illustrated in figure 2.1.

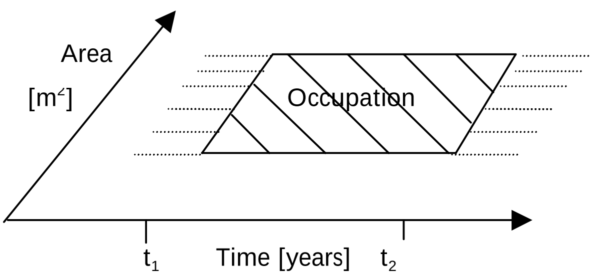

Figure 2.1 Occupation during a land use activity.

The situation before and after the activity (depicted by the dotted lines) is not considered at first instance here. The occupation has a duration and an area aspect, and occupation is therefore expressed in area times time [m^2.y] per functional unit of activity output. This functional unit is based on the total performance during the year(s) which are accounted for.

During the activity occupying the land, the land has a certain quality. This quality determines the **occupation impacts**, as illustrated in figure 2.2a.

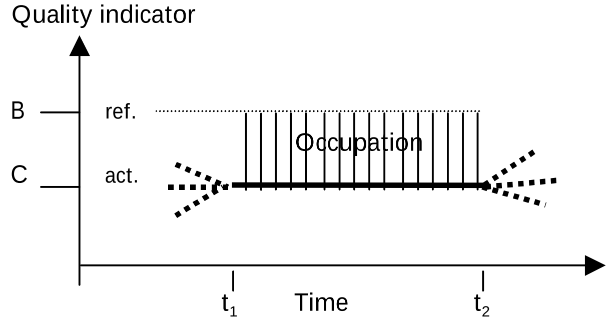

Figure 2.2a Occupation impacts during a land use activity (adapted from Lindeijer et al., 2001), related to an equilibrium reference.

In figures 2.2, the area is left out for simplicity; it is considered on average constant during the land use activity, for simplicity. Occupation impacts are calculated by multiplying the m^2.y of an occupation by the chosen nature quality indicator (see section 2.2). This results in the striped area in figures 2.2. This land occupation is independent of the situation before or after the activity (illustrated by the various thick dotted lines to the left and right of the actual occupation). The striped area is defined by the actual quality level C and some kind of reference level B. It then becomes very important which level is chosen for B and what that choice implies.

Definition of the reference state

Level B is called the reference state because it is the level to which all land use activities are related to. The reason why a reference state is used is threefold:

1) The no-impact situation (related to no human land use) does not have an indicator value zero (in contrast to emissions: no emission means: zero toxic concentration), but a maximum (or average) nature value. The reference state expresses this no-intervention value.
2) Biodiversity impacts should be expressed in relative terms, as climatological of soil characteristic differences should not be used to prefer one ecosystem over another[10].
3) To assist in the decision support function of LCAs including land use. Discussions on how to judge the promotion of the use of wood versus the use of clay bricks or concrete become more transparant when it is made clear what the consequences are of a certain normative standpoint, expressed by the choice of reference state (see below).

The maximum as reference

In figure 2.2a the actual quality during an activity (C) is related to the potential quality after a possible recovery to an equilibrium with its environment (with value B). Land occupation is here interpreted as 'the area prevented to return to the regional equilibrium during a certain time'. For the quality indicator biodiversity, this equilibrium is generally the climax stage of the regional ecosystem type. A land use activity hardly ever reaches this equilibrium value. For biomass this may occur more often, resulting in a negative environmental burden (good for the environment) for these cases.

The average as reference

It is also possible to choose for a different definition of the reference state. For instance, [Köllner, 2001] chose the average quality in a region as a reference. This leads to a different interpretation of the occupation impact score[11]. When the average quality in a region is chosen as a value to relate to, the above interpretation no longer holds: there is no tendency in nature to return to an average quality. In this case, the interpretation could be: 'the area having a quality level at a certain distance from the regional average during a certain time'. In this case, several human activities will have quality levels above this average (level B in figure 2.2b), thus positively contributing to the environmental quality. This will result in a negative LCA score for these activities (good for the environment), as illustrated in figure 2.2b.

[10] These are official statements from the UNEP Rio declaration [UNEP, 1992], and of national governmental policies of (for instance) the Netherlands.

[11] A note for LCA experts:
For emissions, the corresponding equilibrium for an average reference is the background concentration, whereas for a maximum reference it is clean air.

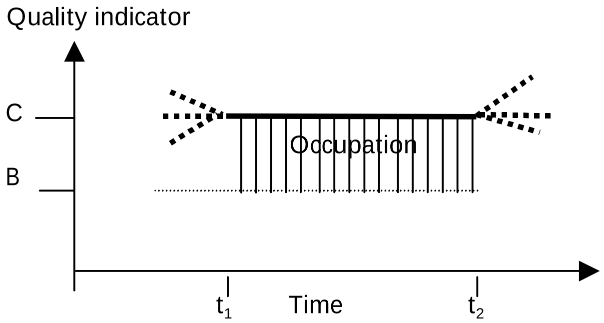

Figure 2.2b Occupation impacts during a land use activity with high quality, related to an average reference.

Renaturation

When the human activity is followed by an active human contribution to changing the final state of that activity, there will be occupation during this process of renaturation too. The final renaturation level may be below, at or above the reference state[12]. This then adds to the overall occupation (see the dotted area in figure 2.2c, in addition to the occupation of figure 2.2a). When the final state is above the reference, the upper part of the occupation impact will result in a negative LCA score (good for the environment). This type of occupation is called renaturation occupation. Renaturation occupation may attributed to the occupying activity until the point in time where the economic responsibility ends (t_3) or until an equilibrium situation is arrived. Both options are investigated in this project.

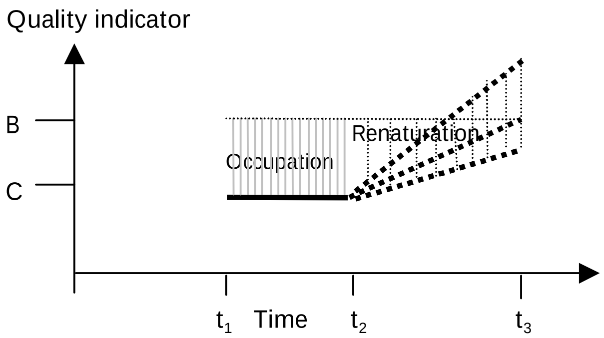

Figure 2.2c Occupation impacts during renaturation after a land use activity with different possible final quality levels, related to a reference.

2.1.2 Transformation impacts

An additional type of information on land use is on the net change of land quality over time. 'Net change' implies comparing only the situation before a change to the situation afterwards[13]. The resulting net change is called **transformation impact**. It is measured in m^2 times quality changed [m^2.Δquality] per functional unit.

Here, the functional unit is determined by all process output taking benefit of the transformation. This may be difficult to assess, and we will deal with this problem below. Two possible cases of transformation (A_1 to B_1 and A_2 to B_2) are presented in figure 2.3.

[12] This will also depend whether the equilibrium or the average reference value is chosen.

[13] Sometimes a renaturation may occur. In that case, the final situation should be measured after this renaturation, when the economic activity or process performing the change is also responsible for the renaturation.

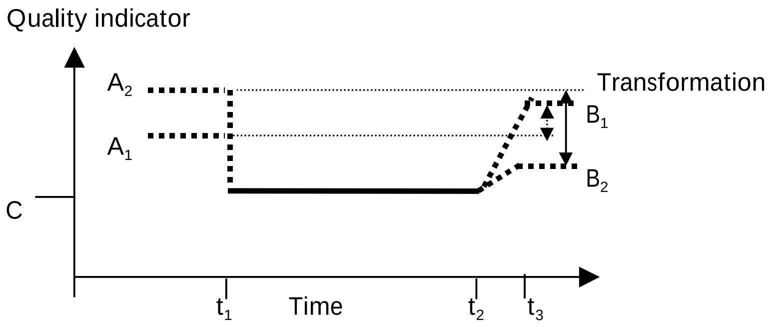

Figure 2.3 Transformation impacts related to a land use activity (adapted from Lindeijer et al., 2001).

In figure 2.3, the contribution of a single case to the trend in land transformations is shown as the length of the vertical arrows. This contribution can be positive or negative, as the final quality B may also be above the initial quality A (as in situation A_1 - B_1). Transformation thus deals with the difference between the initial and the final quality, and does not deal with the occupation time. This is in contrast to occupation impacts, and is clearly complementary.

Recovery to a level B may not take place at all. In that case, level C is the final state, and the net transformation is higher (as illustrated by the difference between levels A and C).

Each case of **land transformation** (changing land quality from before to after an activity) contributes to a regional (national, European) trend in land transformation impacts. The gross yearly transformation for all activities in the region add up to the net regional trend in land use quality[14]. This can be illustrated by the simplified example where the average quality change in a region is only determined by a regional increase of road area, due to various cases of road building, as illustrated in figure 2.4.

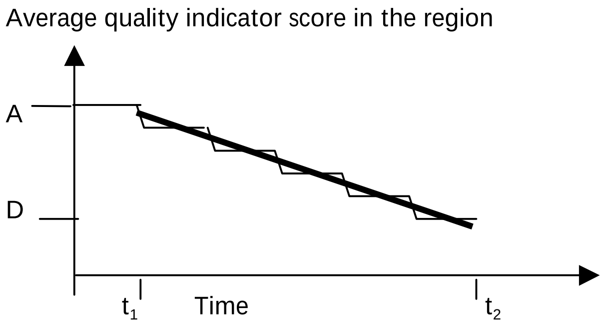

Figure 2.4 Transformation due to a trend in land use changes.

In figure 2.4, the successive transformations of agriculture land to roads lead to a stepwise reduction in average quality in the region. The resulting trend is depicted by the thick line, and the quality loss over the period t_1 to t_2 is given by the difference between A and D. This trend is the context in which an individual land transformation should be interpreted. A single case can contribute positively or negatively to this trend. Situation A_1 to B_1 as demonstrated in figure 2.3 will contribute against a

[14] Note for LCA experts:
This regional net change gives the normalisation score for transformation. It can be interpreted as the net quality change over the relevant period.

decreasing trend, whereas the situation A_2 to B_2 will contribute to the trend line in figure 2.4. In figure 2.3 situation 1 is thus ‘good for the environment’ and situation 2 is ‘bad’, in this framework.

In for instance the case of road building, the number of (future) users is unknown. Hence, it is not possible to determine the attribution of land transformation due to a road building case. In such cases (e.g. railways, industrial estates, housing) the only way to attribute transformations to the total performance is to assume the existence of a trend like the above. This can be assessed by analysing land use statistics, and combining them with performance statistics. The procedure used in this project to do this is given in section 3.1.

2.1.3 Occupation and transformation of aggregate extraction in the Netherlands

In the case of aggregate extraction in the Netherlands, again another situation occurs. As this is the main focus of this project, it merits a separate discussion. Modern aggregate extractions in the Netherlands are only allowed if an extensive renaturation programme is followed. According to Dutch law, this extraction process is linked to the obligation of an infinite maintenance. Aggregate extracting companies therefore either choose to keep the area as their own property and use funding programmes to ensure future maintenance, or to buy off this obligation. Nature conservation organisations then take over the maintenance. During the enduring maintenance the nature quality may increase slowly to a certain ‘climax’ value (see annex 4 for an expert judgement on this; in Dutch). Whether the aggregate extracting company is responsible for this whole renaturation process depends on detailed agreements made in the extraction permit.

Also, the renaturation often starts already at the beginning of the extraction process on part of the area. In a case of modern clay extraction coupled to nature development, the stripped area is left untouched already after a few months. This leads to a new situation, shown in figure 2.5. Here, the quality already increases when the extraction starts (t_1). The changes in quality are generally the highest in the beginning. After a long time (50-200 years) the climax situation may have been reached (t_2). When the average quality in a region is the reference (B), the final state may be above this reference (upper dotted line). When the maximum regional quality is the reference, the final state will be this reference (situation with lower dotted line). The starting situation (A) is often agriculture, and the final quality is then always higher than the initial one. This leads to a negative transformation impact (good for the environment). In this type of process, occupation and renaturation occur simultaneously. The occupation impacts (the dotted area between the curve and the reference) are positive when related to a maximum reference, and may be net negative when compared to an average reference. In practice, some linearisation of the dynamic impact curve will be performed to facilitate calculations (see chapter 3).

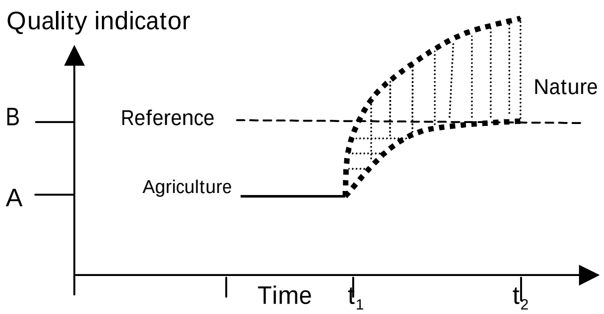

Figure 2.5 Occupation and transformation related to (aggregate) extraction with renaturation.

2.1.4 Additional comments on occupation and transformation in this report

To include trend information such as transformation is not common in LCA. However, it is valuable information for land use, because the percentage of nature value loss through (previous) land transformations is very high world wide, and dominates for instance biodiversity impacts of man [Sala et al., 2000].

Due to the fact that we distinguish between occupation and renaturation, a similar distinguishment between transformation without renaturation and the transformation due to renaturation is necessary. In practice, this leads to 4 inventory types. In practice however, the occupation and renaturation scores can be added up, as can the two transformation scores. Due to the various impact indicators, still a number of impact scores result, both for occupation and for transformation.

The indicators used in this project to assess the quality impacts of occupation and transformation are discussed below in section 2.2.

2.2 Land use impact indicators: biodiversity & life support

From paragraph 2.1 it was concluded that occupation is about preventing the natural recovery (renaturation) of land to an equilibrium with the environment (or being at a certain distance of the average land use quality). This equilibrium (or the average) is called the reference state. This reference state and the related impacts can be expressed in various terms (or indicators).

In this report, two indicators are distinguished (one of which without reference, as explained below). Based on previous research, the reference state and thus also land use impacts in this report are expressed in terms of the biodiversity indicator 'plant species density / ecosystem quality' and the life support indicator biomass. The biodiversity indicator is expressed relative to the relevant region[15], and the life support indicator is expressed in absolute figures. The two indicators express different things and may not point in the same direction for a certain case. They express different aspects of ecosystem quality: the intrinsic value of biodiversity versus the functional value of the global life support system (see also [Udo de Haes & Lindeijer, 2001]).

The biodiversity indicator is a combination of the proposals from IVAM ER [Lindeijer et al., 1998] and from LCAGAPS [Weidema, 2001]. The plant species diversity is expressed relative to the reference state in the regional biome[16]. The species density is standardised to an area of 0.01 ha, according to the proposal from [Köllner, 2001]. As a result, the biodiversity data as collected for his PhD study can directly be used. The reference species diversity data come from a study of the University of Bonn [Barthlott, 1997]. This relative species density factor is multiplied with three factors expressing ecosystem scarcity, ecosystem vulnerability and ecosystem quality on the biome level, as proposed in [Weidema, 2001]. See for more details annex 1. For the value choices involved in this operationalisation, and the resulting formulas see paragraph 2.3. Impact assessment data is given in chapter 4.

On using species density as local indicator for forests, in the final report of the COST E9 (concerted action on LCA for forestry (products) in land use impact assessment [Schweinle et al., 2001]), using only species density as an indicator for biodiversity in forests is rejected. Exergy (the potential energy content) as a measure of the extent of buffering energy fluxes is proposed instead as an evaluating framework. Maximum exergy as an ecosystem goal is used as a kind of reference, with various indicators to assess this exergy, amongst whch species density. The concept of an indicator basket is mentioned, allowing the use of the appropriate set of indicators for a specific ecosystem. A reference state is still required. This concept could be applied in the present methodology, as a percentage of the reference quality (maximum exergy) is sufficient. Using an average value has less sense, as the interpretation of an average exergy value is not clear. Unfortunately no data was collected in the COST E9 project.

In the BEAR project [Hansen, 2000] a large number of indicators are proposed to assess forest biodiversity. Also here, no operationalisation is available at present, hampering a better quantification of the biodiversity value in production forests. This reduces the value of a continuous impact scale, until better data will become available.

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[15] It is internationally agreed that both ecosystems with large and those with low species density should be valued on their intrinsic value. Thus, only relative species density changes within the region itself should be assessed, not the absolute ones.

[16] A biome is the highest level of ecosystem classification on earth.

For life support functions, the most feasible indicator seems biomass, as global data is available [Lindeijer et al., 1998].The net primary production (NPP) of biomass is an indicator for various life support functions (potential for food production, biochemical substance and energy cycles, topsoil formation and preservation [Weidema, 2001]). It is calculated as the net yearly plant (or carbon) mass production per unit area. NPP addresses mainly the life support for human life.

In contrast, in the proposal from IVAM ER [Lindeijer et al., 1998], the free net primary production (fNPP[17]) is used. The fNPP is more appropriate an indicator for non-human life support (see also under 2.3 Value choices). Using fNPP has the drawback that additional data is required, whereas for NPP data is more easily available. Also other authors argue that NPP would be a better indicator than fNPP [Weidema, 2001]. Therefore we have chosen NPP as life support indicator. For more details we refer to annex 1. The resulting formulas are given in section 2.3, and the impact assessment data applied is given in chapter 4.

2.3 Methodological choices made for the operationalisation of land use impacts

Developing an impact assessment method generally implies methodological choices. These may be more or less important. Some may be hard to acknowledge, let alone to assess the model uncertainty resulting from making such choices. A different choice may also lead to a different associated data uncertainty. Calculating with sets of indicators based on different choices would be the best way to assess these. This was considered only feasible for a few choices in this project, as this project mainly focussed on data collection. However, it is important to express the choices involved and to show which alternatives could be chosen based on the same data availability.

For the operationalisation of the biodiversity indicator a large number of choices were implied, as illustrated in table 2.1. They relate to questions such as: which items to address, which formula to use, and which data to apply. See section A1.4 of annex 1 for more background information. Most choices can be well argumented but some are value choices which can not be made based on objective or objectified arguments alone. The main remaining value choice is the use of a maximum or an average reference level (cf. IVAM ER respectively Köllner). Both options are left open. The maximum level is more consistent with the ‘less is better’ paradigm under which most LCIA methods are developed[18]. The average level seems to be more in line with predominant opinions about humans right to use land.

Table 2.1 Methodological choices for the biodiversity indicator.

Subject	Choices made
Biodiversity levels included	Ecosystem and species level
Species included	Only vascular plant species included; equal weighting of all species
Relation of actual state to reference state for occupation	1 – {actual state/reference state}, or {reference state – actual state}/reference state
Species diversity reference state	Maximum or average score for regional biome
Species diversity standardisation	To 0.01 ha via Arrhenius formula using a = 4.1 and b = 0.2
Reference state data	Taken from [Barthlott, 1997]
Relation between initial and final state for transformation	1 – {final state/initial state}, or {initial state – final state}/initial state
Biome definition for ecosystem level	Taken from [Leemans et al., 1998]
Ecosystem quality aspects	Scarcity, vulnerability and value/quality

[17] fNPP is the biomass production left to nature after subtraction of the amount used by humans.

[18] The ‘less is better’ paradigm implies that less intervention is always better, even when this is below the ‘background level’. The reference state is thus for emissions generally a situation with zero emissions, comparable to the maximum biodiversity level.

Ecosystem scarcity factor	Largest potential biome area/potential area of biome i
Ecosystem vulnerability factor	{Existing area of biome i/potential area of biome i}$^{b-1}$
Ecosystem value/quality	Species richness$_i$ / minimum biome species richness

More details on these methodological choices (including some of their alternatives) are given in annex 1. The resulting formulas are:

EO (Ecosystem Occupation) = A x t x SR_i x ES_i x EV_i x SD (1)

with Species Density (SD) = $(1 – S_{act}^{stand} / S_{ref}^{stand})$ (SD ≤ 1); (2)

and SR_i (relative Species Richness of biome i) = S_i / S_{min} (SR ≥ 1);
ES_i (Ecosystem Scarcity of biome i) = $A_{pot,max} / A_{pot,i}$ (ES ≥ 1);
EV_i (Ecosystem Vulnerability of biome i) = $(A_{exi}/A_{pot,i})^{b-1}$ (EV ≥ 1).
S_{act}^{stand} = the actual average (local) species density during occupation
S_{ref}^{stand} = the species density of the reference state (regional average or maximum)

ET (Ecosystem transformation): A x SR_i x ES_i x EV_i x $(S_{ini}^{stand} – S_{fin}^{stand}) / S_{ini}^{stand}$ (3)

with S_{ini}^{stand} = the initial species density before transformation, standardized to 0.01 ha;
S_{fin}^{stand} = the final species density after transformation, standardized to 0.01 ha.

Considering the life support indicator, the methodological choices identified are given in table 2.2.

Table 2.2 Methodological choices for the life support indicator.

Subject	Choices made
Basic indicator	Biomass
Biomass indicator	NPP
Relation of actual state to reference state for occupation	Reference NPP – actual NPP (not relative)
Relation between initial and final state for transformation	initial NPP – final NPP

The resulting formulas are:

LO (Life support occupation) = A x t x $(NPP_{act} – NPP_{ref})$ (4)

LT (Life support transformation) = A x $(NPP_{ini} – NPP_{fin})$ (5)

In conclusion, quite some choices have to be made when operationalising indicators for both biodiversity and life support impacts of land use. Data availability is considered the decisive criterion for selecting an indicator, although its relevance for the impact category involved should be evident too. One major choice remains unargumented, since an important value choice is involved: (1) a maximum or (2) an average value as biodiversity reference state. This choice represents either a more biocentric (ecological) or a more anthropocentric (humans-oriented) viewpoint, respectively. The maximum biodiversity reference is more consistent with references in most present LCIA methods. A final choice is not made here, and both options should be calculated to allow value choices to be included in the decision support process. The case studies are used to assess the consequences of this choice too (see chapter 5).

2.4 Methodological limitations

There are various limitations to the assessment of land use impacts in LCA in general. Additionally, there are limitations to the assessment as developed here. We will discuss both below. Methodological improvements should be seen in this context. They are discussed in detail in the recommendations (section 6.3.2).

2.4.1 General limitations to the assessment of land use impacts in LCA

There are some general limitations to assessing land use in LCA.

Regional differentiation
LCA is a tool to assess impacts of product and service systems expanding all over the globe. The data requirements to assess all impact types along these systems is very large due to this regional differentiation on the level of the life cycle intervention. Regional differentiation on the level of impact assessment requires linking of each LCI process to its appropriate impact score, according to a regionally differentiated impact assessment model. As much impact scores are required as there are different regions in the inventory, but only when it matters where an intervention takes place. For land use this is certainly the case (see chapter 4). The extent to which this differentiation is pursued is limited by the impact assessment model applied and the time available to collect specific data and perform the LCA.

Natural variation
Land use impacts are very dependent on the place and time of the intervention. Not only are temporal aspects generally expelled from LCA studies, but for land use in general the nature value will vary very much from one location to another as well as over time. All biodiversity and life support indicators for land use will suffer from this natural variation to a certain degree. When an indicator is applied in a very generic manner, this variation may be ignored or neglected, but the potential impacts will inevitably contain this variation.

It must be noted that natural variation in space and time is not restricted to land use impacts. In fact all LCA characterisation models are gross simplifications of the real impacts which occur from a given intervention in the environment at a certain point in time and space.

Adequacy of impact indicators
The cause-effect network between a type of land use and its potential impacts is complex (see [Lindeijer et al., 1998] and [Lindeijer et al., 2001]). Many different impacts may result from a single land use, and many relationships may exist between those impacts. Therefore, there is an inherent limitation in the extent to which these impacts can be expressed in LCA. Either many indicators are applied to express initial stressor responses to land use, or a few indicators are chosen to indicate the impacts on high-level impacts on areas of protection such as biodiversity. In the first case the interpretation of different scores is hard, because the ultimate impacts can not be perceived. In the second case the interpretation may be easier, but maybe less valid due to various side-effects on excluded aspects of the area of protection. The balance between completeness of impact types and relevance to areas of protection determines the percieved adequacy of the impact indicators chosen.

Inherent uncertainty on future impacts
Especially for renaturation processes (often performed in mining practice nowadays) the uncertainty on the final state after renaturation is large. Although this uncertainty can be estimated (as done in this project), it remains to be consdered when performing such LCA studies.

2.4.2 Limitations to the TNO impact assessment method for land use

All above limitations should be accepted if the quest is to assess land use impacts in LCA at all. In the TNO method, regional differentiation and natural variaton are dealt with by choosing a moderate regionalisation, including minimum, average and maximum impact scores for each region to show the possibly resulting uncertainty range. Regarding the adequacy of impact indicators, only a few indicators have been chosen to express the impacts of land use. Related additional limitations are:

Landscape degradation not included
Following the conclusions drawn in [Lindeijer et al., 1998] we have not attempted to include impacts of land use on landscape degradation. Landscape degradation is an issue related to esthetics, in this case human well-being due to perception of the change in the pattern of the local environment. In

[Lindejer et al, 1998] it was already stated that this aspect is very hard or impossible to quantify. Although some aspects of landscape degradation may be covered by the indicators for biodiversity and life support functions, an aspect like changing landscape elements is not included. Digging away parts of a mountain is also not included, although this has not only esthetical implications but also a human resource aspect (cf. [Udo de Haes & Lindeijer, 2002]). Such aspects should thus be expressed in a different manner, separate from this land use method.

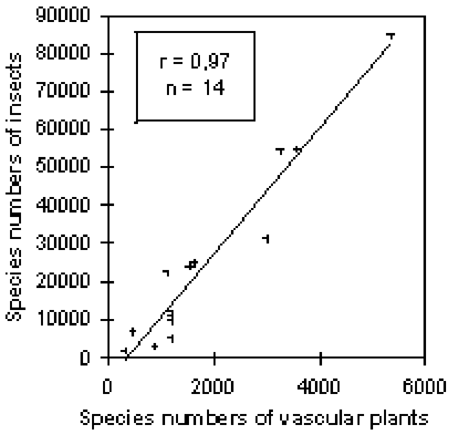
Dessication not included
Although land use and (ground)water extraction are in principle separate interventions, the impacts can be of a similar manner, and often a certain land use process implies impacts on water resources. It may thus seem obvious to include dessication impacts in this methodology. This is not done in this project, because a separate project within the same research programme was dedicated to this issue [Lindeijer et al., 2002].

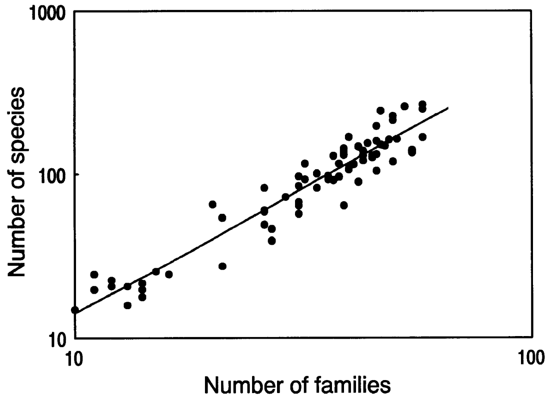
Fragmentation not included
Local land use differences often imply an intersection of larger areas of the same land use. This may be considered positive, as with agriculture patches in vast forests areas[19]. But a road or railway cutting through a forest or meadow may not have a positive influence, when it cuts up habitats and prevents migration of species or individals. Especially where nature area is becoming scarce, this fragmentation becomes an important issue (mainly for fauna), as in the Netherlands. Such regional intersection impacts are not included in this project. Only the area of the land use itself is considered, because including knowledge on the surrounding area for all land use types is not feasible in generic LCAs.

Only plant species included on the species level of biodiversity
Impacts on fauna (fish in artificial lakes, birds and mammals on land) are not included. Although statistical relationships can be acknowledged between various species types on a large scale (see figure 2.6), this does not hold for many situations. Even for Europe the maps on birds, reptiles, mammals and plants do not overlap: different species types have different areas where they are abundant, scarce or absent [EEA, 1998]. This shows not only a limitation to using plant species as sole indicator on the species level, but also that nature value is very much dependent on which species type is considered most valuable. Of all species types, vascular plant species are nevertheless often considered representative of the presence of other pecies, due to their basic role in the maintainance of habitats. For fauna the relationship between area occupied/transformed and impact will at least be much less clear, whereas vascular plants are fairly representative for lower species. See the relationships between vascular plant species and insects, and species and families (a higher order of species aggregation) in figure 2.6. Finally, in this project, we use water plant species diversity as a measure for species level diversity in artificial lakes (see section 4.2). In fact, any species type could be taken as an indicator when considered more adequate than vascular plant species, as long as the average and/or maximum reference states are expressed in the same (relative) terms.

Figure 2.6 Relationship between vascular plant species and insects species numbers, and between species numbers and family numbers, taken from [Barthlott, 1997].

[19] The borders of forests induce higher levels of biodiversity than the forest itself, see the impact scores according to [Köllner, 2001], table 4.4. in this report.





Marine ecosystems not assessed for biodiversity

Although biomass and species density for aquatic ecosystems related to modern aggregate extraction is included in the TNO method, marine ecosystems are not included for the biodiversity indicator. This is because the assessment of marine ecosystem quality on the level of species has the same inherent limitation as assessing fauna in general. They are not fixed to a specific area, and changes are quick and recoverable on a certain scale level, whereas this may not be the case on a higher scale level (see also [Lindeijer et al., 1998]). Coral reefs however could be included in this method rather simply, as they are not very mobile, and are even slower than vascular plants to react to area-related impacts. This has not been done due to the different focus of this project. For the life support indicator NPP the marine ecosystem has been included. Other possible impacts due to changes in the sea floorbed related to life support functions (coastal safety for humans, breeding disturbances for fish etc.) have not been included.

3 Data collection: physical land use

…………………………………………………………………

In this chapter the procedure to collect land use data as applied in this study is described. All calculated land use and land transformation data are described and major background data for the calculations are given. All occupation and transformation data gathered are summarized in the end of the chapter.

3.1 Data collection & treatment strategy

For a specific case of land use different life cycle phases can be distinguished: building, production or use, demolition and finally renaturation (recovery to a more natural land type). As in figures 2.1 and 2.2, these phases should preferably be considered together, but for occupation differently than for transformation.

Approach – Occupation
Occupation is expressed as m^2.years per functional unit. This time period is multiplied with the area occupied, and divided by the functional unit. See for the calculation procedure for this functional unit (output) steps 2 and 3 of procedure A under *Transformation* below.

Thus, the time period is important. During the building, demolition and/or renaturation period of most human activities the land is partly occupied during the change-over. A linear quality change (or a limited sequence of linear quality changes) is assumed during this change-over to another type of land. For this reason, the average quality during these periods is considered. The renaturation occupation is calculated separately, because it depends on the situation whether this should be included or not. Also, in this case either the average quality is taken, or half of the renaturation time is taken. This is especially relevant for the ETH database [Frischknecht et al., 1994, 1996]. There, renaturation is always assumed to happen, whereas this is not always the case. For Dutch cases based on national statistics the occupation data are mainly based on the year 1995.

When no case-specific data is available, yearly statistics can be used. The time period is then 1 year, multiplied with the total area for a certain land use type, and divided by the yearly output of the sector occupying that land type. When more than one sector uses the same land area, an appropriate attribution rule should be applied. See table 3.1 for an overview of the detailed approach and data source used per functional unit.

Approach – Transformation
Transformation is expressed in m^2 per functional unit of that type of land. The aim of transformation is to express the trend of claiming extra land from other types of land use, for a certain functional unit. This transfomation may be assessed for single cases when the situation is known, giving a positive or negative contribution to the net trend. But often the situation before and after a transformation is not known in detail or attributable to a functional unit (for instance road building). In the latter case, the gross trend over a certain time needs to be determined, and attributed to the number of functional units in this period. The gross trends of all land type specific transformations together contribute to the net trend again. The net trend in a region can be expressed as normalisation scores.

In this report the transformation data are based on the periods 1993-1996 and 1989-1996. The gross transformation data for land use in the Netherlands are based on national statistics [CBS, 1997a, 2000a]. The origine of the land which contributes less than 1% to the total, is not taken into account. For aggregate and wood extraction and energy-related processes, the sources mentioned in table 3.1 are used.

Whenever possible, the gross transformation is determined, since only the land transformed towards the land use under consideration is caused by the demand for that land use type. Transformations from the land use under consideration to other land user types should be accounted to users of those other land types.

Table 3.1 Data collection and treatment approach.

Functional unit	Main data source		Attribution rule
	Time	Area	
[t] sand/gravel/clay NL	Governmental data compilation	Governmental data compilation	Economic, between types of aggregate
[t] cement NL	Industrial data	Industrial data	-
[t] wood extraction	FAO FRA 2000	FAO FRA 2000	-
[t] industrial production general NL	National statistics	National statistics	Added value per kg, only for missing data on yearly output for industrial subsectors
[tkm] road transport NL	National statistics	National statistics	Vehicle km, between personal traffic and trucks/vans
[tkm] railway transport NL	National statistics	National statistics	Vehicle km, between personal transport and freight trains
[tkm] canal transport NL	National statistics	National statistics	-
[MJel] electricity NL/UCPTE	ETH (specific sources), Dutch energy statistics	ETH (specific sources)	Specific, per process
[tkm] road transport EU	ETH	ETH	no attribution; very generic data used
[tkm] railway transport EU	ETH	ETH	no attribution; very generic data used
[t] aluminium	Specific	Specific	-
[t] copper	Specific	Specific	-
[t] other materials	ETH (specific sources)	ETH (specific sources)	-
[y] living, [t] house NL	National statistics	National statistics	60 t per house, 50 y living [Beetstra, 1998]
[t] waste disposal NL	National statistics	Nat. Statistics/specific	-

In order to determine the transformation of land through activities or land use types the two following procedures are developed, depending on whether or not the situation before and after the activity is known.

Procedure A: the situation before and after the activity is known (e.g. mining of raw materials)

1. Determine the specific land area used for the activity, including access roads etc.
2. Determine the delivered output(s) during the considered period. When no physical data is given in statistics, try to determine a ratio between economic output and physical output to fill in these gaps.
3. If necessary, allocate to the output under consideration with the most appropriate allocation rule, or use different rules (leading to a range in results)
4. Calculate the transformation by dividing 1. by 2. or 3.
5. Determine the situation before and after the activity. If the land is recovered by the same user to a certain quality level, then the last quality level is the situation after the activity. If the recovery is performed by a different organisation/user, the quality level at the moment of transfer of ownership/use is decisive.

Procedure B: *situation (before and) after the activity is unknown (e.g. road-construction)*

1. Determine a relevant, increasing trend in land use for the concerning land use type using land use statistics[20]. Apply various time ranges to determine the variability of the trend over time.
2. Determine via land cover change statistics the type(s) of land use before the change –only in case of a relevant trend-, and their percentage of contribution. When there are stock-type land users (storage, barren building area, etc.) look further back to determine what was the previous user.
3. Calculate the national (or regional) transformation from each previous user in m^2/y. Whenever possible, leave out the transformations from the user to other users (gross transformation data are preferred over net transformation data, see annex 2)
4. Determine the productivity[21] (e.g. in either kg/y or tkm/y) using statistics.
5. Allocate the total productivity over the period to the relevant output use in case of multi-functionality. General rules are not available (see under 3. of procedure A)
6. Divide the land transformation for each previous user separately by the (allocated) productivity of that activity to arrive at transformation results per functional unit.

3.2 Results aggregate extraction (sand, gravel, clay)

For traditional cases of aggregate extraction in the Netherlands (without renaturation), data were gathered from statistical data [Nieuwland, 2001] and from a PhD study on land use impacts of building materials [Beetstra, 1998]. See for the calculation procedure annex 3 (in Dutch). The here presented data have only been authorised by the guiding committee for the sake of the cases to be performed.

In annex 3 a conservative method was used to determine the uncertainty due to representativity of the data: representativity uncertainty = 1 – representativity for the Duch situation (both uncertainty and representativity expressed in %; representativity determined by the percentage of total extractions in the Netherlands covered by the cases considered). Later, it was decided to use representative model calculations as a basis for the case studies. The representativity of the model calculations was agreed upon by participants of the aggregate sector meeting (see the meeting report in Dutch, concluding annex 3), see table 3.2. The national averages (each second row) calculated from the statistical data are proposed for further use in the case of aggregate extractions without renaturation in the Netherlands.

Table 3.2 Model calculations to verify representative cases of traditional aggregrate extraction in NL.

	Raising sand	Limest. sand	Contr. sand	Gravel	Forel. clay	Indyke clay	Limest. / marl
Net area pit [hectare]	**25**	**40**	**30**	**185**	**2**	**20**	**25**
Depth [m]	**20**	**40**	**20**	**10**	**4**	**4**	**4**
Volume [m3]	4000000	12800000	4800000	14800000	64000	640000	800000
Density [t/m3]	**1.7**	**1.7**	**1.7**	**1.7**	**2.2**	**2.2**	**1.7**
Total output [kt]	6800	21760	8160	25160	141	1408	1360
Period of extraction [years]	**35**	**10**	**20**	**10**	**1**	**5**	**1**
Gross area [hectare]	**30**	**50**	**37**	**210**	**2.4**	**25**	**30**
Net/gross area	0.8	0.8	0.8	0.9	0.8	0.8	0.8

[20] If there is no relevant trend, calculation of land use change to that activity is not relevant either (as a result procedure B ends).

[21] The output or functional unit could be based on:

1. Performance in the first year of the considered period.
2. Average performance during the considered period.
3. Performance in the last year of the considered period.

When a transformation is based on trend data, the second definition seems the most adequate, and is used by us. Whenever possible, two different time periods are considered, leading to slightly different results. This gives an idea of the range in outcome through this approach.

Occupation [m2.years/kt]	1544	230	907	835	170	888	221
National average	1560	220*	890	730	126	1130	230
Transformation [m2/kt]	44	23	45	83	170	**178**	**221**
National average	40	20	50	50	126	25	10

* According to industry data [Schuur, 1996] this value was supposed to be 10 times lower, but a calculation error of a factor 10 has occurred.

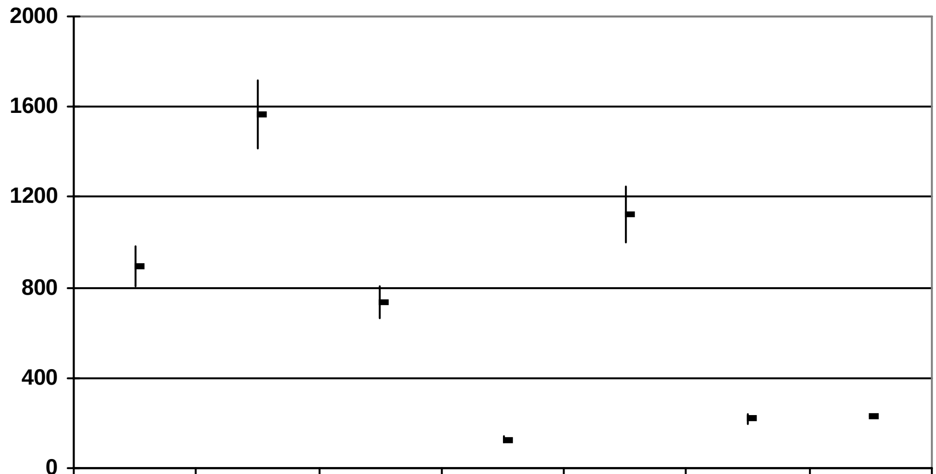
The approach to use a representative model case is in accordance with the procedure to determine the uncertainty in related cases for determining the environmental burden of buildings proposed in a yet unpublished Dutch draft standard (the Material-related Environmental Burden of Buildings, MMG in Dutch). It resulted in much lower uncertainty ranges, which gives a different picture than in annex 3. See for the results below (3.2.1 and 3.2.2.).

See for modern aggregate extraction cases in the Netherlands section 3.2.3 on renaturation for aggregate extractions.

3.2.1 Occupation for aggregate extractions

For aggregate extractions without renaturation, see figure 3.1 for the occupation values resulting from annex 3. This data is slightly different from that of table 3.2 due to rounding off.

Figure 3.1 Occupation values for aggregate extraction without renaturation [m^2.y/kt].



	Construction sand	Raising sand	Gravel	Foreland clay	Indyke clay	Limestonesand	Limestone/marl
high	981	1713	801	139	1250	242	
low	801	1413	661	113	1004	198	
weighted average	891	1563	731	126	1127	220	230

In the ETH database [Frischknecht et al., 1994] for Switzerland the following data are given for aggregate extraction:

sand/gravel: 1.8 m^2.y/t
limestone: 0.05 m^2.y/t

From the 1996 edition of that database [Frischknecht et al., 1996], the following data for sand for building was found: **1,0 m^2.y/t**.

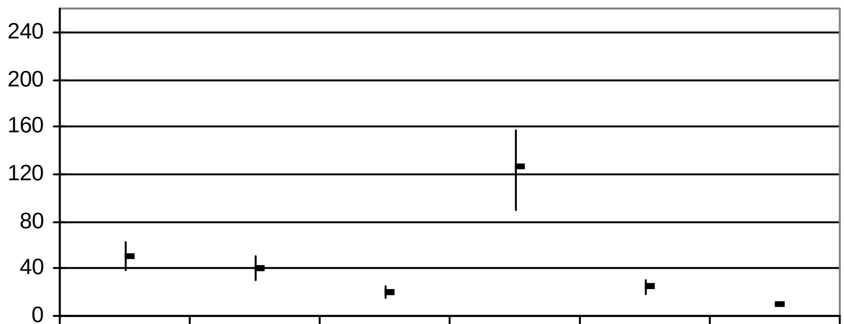
The cases for sand are thus in general accordance with the data for the Netherlands. The limestone figure of ETH is a factor 4 lower than the one from the Netherlands, which may be due to higher hills from which limestone is extracted in Switzerland.

In [Schweinle, 2000] a case of limestone extraction is mentioned. The total area is 239’900 m², with a yearly extraction of 350.000 t limestone. This results in an occupation of **0,685 m².y/t**. Of this area, 40% is the actual pit at the time of inventory (including a wall of 2 ha), 34% is post-extraction area to be renaturated, 3% is now a lake and 14% are roads. Other parts are surrounding green areas (5%) and renaturated arable land (3%). The occupation in this case is thus a factor 2 higher than that of the Netherlands. This may be due to the fact that here the pit is an open, rather flat area, whereas in the Netherlands it is a hill.

3.2.2 Transformation for aggregate extractions

For aggregate extractions without renaturation, see figure 3.2 for the transformation values resulting from annex 3.

Figure 3.2 Transformation values for aggregate extraction without renaturation [m².y/kt].



	Construction sand/gravel	Raising sand	Limebrick sand	Foreland clay	Indyke clay	Limestone / marl
high	62	50	25	157	31	
low	38	30	15	90	19	
▬ weighted average	50	40	20	126	25	9,6

In the ETH database [Frischknecht et al., 1994/1996] the following data are given for aggregate extraction:

sand/gravel: 0.18 m²/t
limestone: 0.005 m²/t

From the 1996 edition of that database [Frischknecht et al., 1996], the following case data for sand for building was documented: **0,04 m²/t**. Only the latter figure is in accordance with the Dutch data. No explanation can be given for the differences here.

3.2.3 Renaturation after aggregate extractions

Renaturation may occur after or during use of the extraction pit. In the Netherlands, a human contribution to such a renaturation process is more common than exception nowadays. However, this renaturation is seldom a recovery to the situation before the extraction. For gravel and sand, often recreational area is developed, including a large water area (because of the high groundwater table).

Sometimes nature is ‘developed’, as with most present clay extractions. This development process does not have a clear endpoint; a more or less natural development process is managed to a certain succession stage. Depending on the land owners and –managers different succession stages may be selected as endpoints. These (often not yet decided upon when starting renaturation) endpoints have different characteristics with respect to renaturation times, biodiversity and biomass.

In the Netherlands, many renaturation processes after extractions are initially managed by Stichting Ark, aiming at nature- and landscape development in various land typologies in the Netherlands. They have experience with about 60 renaturation processes after extractions over the last 10 years. Stichting Ark was asked to make quantitative estimates of the different succession stages after extractions in the Netherlands. This report is included as annex 4. Here, a short summary is given of their findings.

The influence of a resource extraction may be positive or negative or just different (in terms of resulting ecological potentials). For aggregate extractions, one can hardly speak of ecological recovery, as the situation before and after is so different. What ecological (development) potentials will result depend mainly on:

- the remaining soil type (the 'substrate');
- the depth of the pit after extraction has ended (relative to surrounding ground- or surface water);
- the remaining relief and steepness of the talud;
- whether a natural development is allowed or whether recreation, recultivation or a specific nature type is pursued.

In the report of Stichting Ark ([Helmer, 2001], annex 4), estimations of characteristics are made based on spontaneous successions according to natural processes (albeit supported by humans). The final succession stage (climax) is actually a dynamic equilibrium where composition and decomposition phases coexist with all possible intermediate stages. Which succession and climax 'stage' (and thus characteristics) results depends on various factors:

- the clearness and quality of the water;
- the supply of sediment;
- the presence of seed-sources;
- the (more or less accidental) sequence of plant species establishment;
- the grazing intensity;
- the influence of the dash of waves;
- accidental weather type, water level etcetera when the 'renaturation management' starts.

Due to these many variables, a whole range of possible characteristics exist during and after renaturation.

In table 3.3 an overview of the quantitative estimates of three distinct succession stage characteristics for the major Dutch situations from Stichting Ark is given. The characteristics given are: time for a certain succession to start, time for it to transform to the next stage or to become mature, the number of vascular plant species per 10 hectare and the biomass-productivity (in tonnes wet mass per ha). The plant species and biomass include plants in the water! See annex 4 for more details.

When these developments occur after the economic activity on that spot has seized and the responsibility for the nature development has been handed over to for instance Stichting Ark, we argue that not the whole renaturation process may be attributed to the extraction, as the following process is performed partly by new land "users" sharing the responsibility. Attribution is done in LCA basically on economic responsibility (related to economic processes). In other cases, the extraction company may remain owner of the extracted land, and remain contributing actively to the renaturation process. Also in that case, part or all of the renaturation process should be attributed to the aggregate extraction process, al least until the climax stage is reached, or until the economic depreciation time (depending on the reference state chosen[22]). In general, the land occupation during

[22] Note for LCA experts: This is a situation where an infinite time horizon may enter and render attribution impossible, unless a (more or less arbitrary) temporal limitation is set. In other terms, until when shall we take impacts into account? When an average reference state is chosen (related to human use of land by economic processes) the most consistent choice would be the

nature development, the increase of species diversity and the concommittant biomass productivity is part of the societal process of nature and landscape development[23], and can not be simply attributed to single economic processes. However, due to lack of the possibility to perform such an attribution, we attribute 100% of the renaturation to the extraction until the climax state is reached, and show what happens when only 30 years from the start of the activity is accounted (as postulated depreciation time). See table 3.4 for a few selected representative cases.

Table 3.3 Overview of renaturation times, species density and biomass per succession stage for 5 types of soil in the Netherlands, according to [Helmer, 2001] (annex 4).

Substrate soil level to water level other factors	Succession stage	starting time [y]	ending time [y]	S per 10 ha	wet biomass per ha [t]
Clay					
>> 2 m deep	sedimentation	0	50-300	see pioneer 2-3m deep	
< 2-3 m deep	pioneer	1	50	10-50	5-10
clay sedimentation	shrubs	2	50	25-50	15-25
organic deposition	peat-bog	10	200	50-100	10-15
clay sedimentation	climax softwood	15	100	50-100	15-20
organic deposition	climax hardwood	100	300	50-100	10-15
just at water level	pioneer	0	5		
immediate grazing	grassland	2	20	50-100	10-15
grazing after some years	willowbush	2	50	50-100	20-30
fluctuating groundwater level	softwood	5	50	50-100	10-20
grazing after some years	climax hardwood	50	200	100-150	15-20
just above water level	pioneer	0	2	100-150	5
grazing	grassland	2	50	100-150	5-10
grazing	climax	50	100	150-200	15-20
above water level (dry)	pioneer	0	5	50-100	5
grazing	bushes, grassland	5	50	50-100	5-10
grazing	climax	50	200	100-200	10-15
Sand					
>> 2 m deep	pioneer	0	5	10-20	<5
	climax	10	50	25-40	<5
< 2-3 m deep	pioneer	0	10	20-30	<5
clay sedimentation	shrubs	2	50	25-50	15-25
organic deposition	peat-bog	10	50	30-100	5-10
idem, with exit seepage	peat-bog	50	100	50-150	5-10
clay sedimentation	climax softwood	15	100	50-100	15-20
organic deposition	climax moorland	50	200	20-50	5-10
sand deposition	climax	50	200	200-300	15-20

economic depreciation time. When the maximum quality state is chosen as a reference, the climax stage of ecosystem succession is the most consistent choice. In the latter case, the impact after reaching this climax stage is always zero, and all time to reach it is included in the occupation.

[23] There are many actors involved in this process. One way to give a fair attribution of the renaturation process to these actors is to determine the share in effort (in time or capitalised time) to contribute to this process. However, as long as the final goals are not stated, even this is not possible. We therefore reside to optionally attributing the total time to reach the climax state to the extraction process.

Table 3.3 ***Continued***

Overview of renaturation times, species density and biomass per succession stage for 5 types of soil in the Netherlands, according to [Helmer, 2001] (annex 4).

Substrate soil level to water level other factors	Succession stage	starting time [y]	ending time [y]	S per 10 ha	wet biomass per ha [t]
Sand just at water level	pioneer	0	5	150-250	<5
immediate grazing	grassland	5	20	200-300	15-20
grazing after some years	popularbush	5	50	200-300	15-20
strong sand sedimentation	grassland, bushes	10	50	150-250	5-15
grazing	climax hardwood	50	200	200-300	15-20
just above water level	pioneer	0	30	50-100	<5
grazing	grassland, bushes	10	50	150-250	5-15
grazing	climax mixed	50	200	300-400	15-20
above water level (dry)	pioneer	0	5	10-20	<5
intensive grazing	heather, grassland	5	50	50-100	5-10
extensive grazing	softwood	5	50	20-50	5-10
extensive grazing	hardwood	20	200	20-50	10-15
grazing	climax mixed	100	>200	50-150	10-15
Gravel					
>> 2 m deep	pioneer	0	5	10-20	<5
	climax	10	50	25-40	<5
< 2-3 m deep	pioneer	0	5	10-20	<5
low alluvion	marshroot colonisation	5	20	10-50	5-10
grazing	climax	20	100	50-100	10-15
just above water level	pioneer	0	10	25-75	<5
grazing	grassland	10	50	150-250	5-10
grazing	hardwood	10	50	150-250	10-15
Periodic flooding	climax mixed	50	200	300-400	15-20
above water level (dry)	pioneer	0	30	10-20	<1
Extensive grazing	grassland, bushes	20	100	20-100	5-10
	climax mixed	100	300	100-300	5-10
Lime					
>> 2 m deep	hardly pioneer possible; for borders: see <2-3 m deep				
< 2-3 m deep	pioneer	0	5	50-100	5-10
	lime march	5	50	100-150	5-10
	climax	30	100	100-150	10-15
above water level (dry)	pioneer	0	5	100-150	5-10
	lime grassland	5	50	150-250	5-10
	hardwood	5	50	150-200	10-20
	climax mixed	50	200	250-400	10-20
Peat					
< 2-3 m deep	pioneer	0	10	5-20	<5
	peat-bog	10	50	100-150	5-15
	woodland	10	50	50-100	5-15
	climax peat/moor	50	200	20-50	5-10
just at water level	pioneer	0	1	25-75	<5
Mineralisation	reed/marshplants	1	50	50-100	10-20
clay sedimentation	shrubs	2	50	25-50	15-25
	climax mixed	50	100	100-150	10-15

Table 3.3 ***Continued***
Overview of renaturation times, species density and biomass per succession stage for 5 types of soil in the Netherlands, according to [Helmer, 2001] (annex 4).

Substrate soil level to water level other factors	Succession stage	starting time [y]	ending time [y]	S per 10 ha	wet biomass per ha [t]
Peat just above water level	pioneer				
above water level (dry)	pioneer (dry,oxid.)	0	1	10-50	<5
	pioneer (wet peat)	0	1	10-25	<1
oxidised soil	grassland, bushes	1	50	25-75	5-15
wet peat	peat-moss	1	50	10-25	5-10
oxidised soil	climax mixed	50	100	50-100	10-15
wet peat	climax moorland	50	200	25-50	5-10

Table 3.4 **Transformation and occupation data (right of thick line) for a few cases of modern aggergate extraction in the Netherlands; including renaturation**

Indyke sand/gravel extraction until clay layer; >> 2 m water, allowing for sedimentation & softwood climax				
Transformation [m^2/t]	20 y extraction [m^2.y/t]	+200 y sedimentation [m^2.y/t]	+100 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.037	0.74	4.42	6.26	1.10
Riverbed gravel extraction until water level; periodic flooding, pioneer, grassland (grazing), mixed climax				
Transformation [m^2/t]	10 y extr/pioneer [m^2.y/t]	+40 y grassland [m^2.y/t]	+150 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.074	0.37	1.84	7.35	2.21
Borders for riverbed sand/gravel extraction (dry); pioneer, grassland/bushes (grazing), mixed climax				
Transformation [m^2/t]	20 y extr/pioneer [m^2.y/t]	+80 y grassland [m^2.y/t]	+200 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.0099	0.10	0.50	1.49	0.30
Clay extraction, undeep; until water level, pioneer, grazing after some years (+ willows), hardwood climax				
Transformation [m^2/t]	5 y extr/pioneer [m^2.y/t]	+45 y grassland [m^2.y/t]	+150 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.14	0.36	3.56	14.2	4.3
Borders for undeep clay extraction (dry); pioneer, grazing after some years (+ bushes), hardwood climax				
Transformation [m^2/t]	5 y extr/pioneer [m^2.y/t]	+45 y grassland [m^2.y/t]	+150 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.0086	0.02	0.21	0.86	0.26
Lime pits (dry); pioneer, lime grasslands, mixed climax				
Transformation [m^2/t]	5 y extr/pioneer [m^2.y/t]	+45 y grassland [m^2.y/t]	+150 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.22	0.55	5.51	22.1	6.6
Pond in limepit; pioneer, lime marsh, climax				
Transformation [m^2/t]	5 y extr/pioneer [m^2.y/t]	+45 y grassland [m^2.y/t]	+50 y to climax [m^2.y/t]	*30 years depreciation* Renaturation [m^2.y/t]
0.018	0.05	0.46	0.92	0.55

From table 3.4 it can be seen that the total renaturation including the recovery time until the climax state (suggested to be used in combination with a maximum reference) is higher than when only 30 years depreciation would be taken.

3.3 Results wood extraction

For wood extraction, the FAO Forest Resource Assessment 2000 [FAO, 2001] is used as the basic source of information. For occupation, mainly the information on wood removals (volume harvested in m^3) was used, supplemented by information on the percentage of exploitable forest from the UN/ECE [UN/ECE, 1997]. For transformation, mainly the Forest Resource Assessment data on forest cover change from 1990 to 2000 is used. Data on forest cover change between 1990 and 1995 is used to assess the variability in this trend in Western countries.

The land use data per country is converted to data per wood type, as the latter is required for more detailed LCA studies. In order to do this, conversions of data from 'm^3 over bark' (o.b.; including the bark) to 'under bark' and then from m^3 to tonne had to be performed. This land use per tonne is then multiplied with the fraction of pine or broadleaved wood harvested in each country. The distinction between pine and broadleaved wood was the most detailed distinction that could be made based on the present statistics. Finally, a weighted average results when the amount of tonnes imported from each country is divided by the total amount imported. See the formula below:

$$\text{Land use per wood type} = \frac{\Sigma_i\ (\text{land use per tonne}_i \times \text{fraction}_i \text{ of woodtype harvested} \times \text{tonnes imported}_i)}{\Sigma_i \text{ tonnes imported}_i \text{ to NL}}$$

(where i is a country label)[A]

Due to natural variability in the coversion data, ranges of values have been used. See tables 3.5 and 3.6 for the values used for these conversions. Conversion of land use data to kg of wood product (as published consistently for the last time in 1987 [ECE/FAO, 1991]) is not performed, as this is to be done for each LCA separately. The data in this section are thus excluding losses and reuse of waste material, which are to be inserted in process data for the extraction of wood when the extraction process is in kg or tonnes of product. To allow use of the resulting land use data in process data with the output in m^3 under bark, also the land use data per m^3 under bark are given.

[A] The overall procedure expressed in formulas is as follows:

For Western countries:

- Forest land used = m2 total forest area x fraction exploitable forest[24]
- Land use (o.b.) in m2/m3 = forest land used / volume harvested

For other countries:

- Volume harvested = m2 actually harvested x harvesting intensity in m3/m2
  (high and low value for harvesting intensity; 3 gaps filled with continent average)
- Forest land used = m2 total area under timber harvesting scheme
  (4 gaps filled with continent average for % of forest under timber harvesting scheme[24])
- Land use (o.b.) in m2/m3 = forest land used / volume harvested

For all countries:

- Land use (u.b.) = land use (o.b.) x ratio over bark/under bark
- Land use per tonne = land use u.b. / density (high and low value)
- Land use per wood type = Σ_i (fraction$_i$ of woodtype x land use per tonne$_i$ x tonnes imported$_i$) / Σ_i tonnes imported$_i$ to NL

Table 3.5 Data on wood types for building used in the Netherlands.

	Density fresh [kg/m^3] source: [Wiselius, 2001]			Ratio over bark/under bark
Wood type	Average	Low	High	[vDielen, 2001][1)]
Spruce (Picea Abies) Scandinavia	810	520	1100	1.12
Pine (Pinus) Scandinavia	800			1.18
Spruce Eastern Europe	810	520	1100	1.12
Spruce Central-Western Europe	810	520	1100	1.12
Spruce North-America	950	900	1000	1.12
European hardwood(data from oak)	1050	1000	1100	1.19
North-American hardwood (oak)	1050	1000	1100	1.19
Meranti S.E.Asia	850	750	950	1.15
Merbau S.E.Asia[2)]	1000	875	1125	1.15
Iroko Africa	1075	950	1200	1.15
Azobé W.Africa[2)]	1150	1000	1300	1.15
Amburana S.America	775	700	850	1.15
Wane S.America[2)]	970			1.15

1) Data for Dutch wood also applied to other countries, although tropical wood has a thinner bark.
2) Also applied in waterworks [Huizen, 2001].

Due to the large ranges in the conversion figures of table 3.5, large uncertainties remain in the results. For transformations, even larger uncertainties appear due to large variations between countries, and some results render insignificant.

Table 3.6 Data on roundwood equivalents, used to convert import data to roundwood for comparison.

	Roundwood equivalents
Wood product	[vDielen, 2001]
Sawn coniferous wood	1,67
Sawn hardwood	1,82
Plywood and veneer	2,30
Particleboard	1,30
Fiberboard	1,45

The weighted averages are determined for wood types imported from different countries to the Netherlands. The share of imports of each wood type from the various countries based on [VVNH, 2000] are given below in table 3.7. They are determined by adding up roundwood, sawn wood and plywood imports for that wood type and usage (excluing wood for paper products). Other wooden products are neglected, as they do not contribute significantly to the main input for the building sector. For countries from which more than one of the wood types of table 3.7 are imported (Finland, Norway, Sweden, Indonesia and Brazil) equal shares imported of each are assumed for simplicity. Only countries with a contribution of >1% to the Dutch imports are included. Variation in these data (as observed while performing various calculations during the project) showed no relevant influence on the land use results, as they are mainly detemined by the basic data source (FAO) and calculations steps within these.

Table 3.7 Data on shares of wood sources for Dutch building wood [VVNH, 2000].

Wood type	Contributing countries	Import share 1999	
		[m^3 round-wood equiv. over bark]	[% per woodtype and region]
Spruce (Picea Abies) Scandinavia	Finland	5,65E+05	45,2%
	Norway	1,06E+05	8,5%
	Sweden	5,78E+05	46,3%
Pine (Pinus) Scandinavia	Finland	5,65E+05	45,2%
	Norway	1,06E+05	8,5%
	Sweden	5,78E+05	46,3%
Spruce Eastern Europe	Belarus	8,62E+04	7,5%
	Estonia	1,19E+05	10,3%
	Latvia	2,00E+05	17,3%
	Lithuania	6,49E+04	5,6%
	Russia	6,86E+05	59,3%
Spruce Central-Western Europe	Belgium & Luxembourg	2,50E+05	33,3%
	Germany	4,23E+05	56,4%
	Poland	7,67E+04	10,2%
Spruce North-America	Canada	8,19E+04	
Coniferous wood for NL	**All above**	**4,48E+06**	**100%**
European hardwood	Belgium & Luxembourg	1,30E+05	20,9%
	Belarus	1,44E+04	2,3%
	Czech Republic	9,10E+03	1,5%
	Finland	6,50E+04	10,4%
	France	7,04E+04	11,3%
	Germany	1,94E+05	31,2%
	Hungary	9,81E+03	1,6%
	Latvia	6,09E+04	9,8%
	Lithuania	1,66E+04	2,7%
	Poland	1,50E+04	2,4%
	Russia	2,51E+04	4,0%
	Ukraine	1,26E+04	2,0%
North-American hardwood	United States	3,15E+04	40,0%
	Canada	4,73E+04	60,0%
Non-tropical hardwood for NL	**All above**	**7,02E+05**	***100,0%***
Meranti S.E.Asia	Indonesia	3,70E+04	
Merbau S.E.Asia[1]	Indonesia	3,70E+04	10,2%
	Malaysia	3,24E+05	89,8%
Iroko Africa	Cameroon	1,52E+05	
Azobé W.Africa[1]	Gabon	3,70E+04	
Amburana S.America	Brazil	3,85E+04	
Wane S.America[1]	Brazil	3,85E+04	
Tropical hardwood for NL	**All above**	***6,64E+05***	**100%**

[1] Also applied for waterworks [Huizen, 2001].

3.3.1 Occupation for wood

Data from the FAO Forest Research Assessment 2000 [FAO, 2001] are used as mentioned above. For various countries, the area under harvesting scheme, the area actually harvested and the harvesting intensity could be used to calculate the land occupation for forestry per m^3 over bark. For European and North American countries however, only data on total forest area and volume harvested (coniferous, non coniferous) was available. Fortunately, for these countries the UN/ECE gives information on the area of exploitable forest [UN/ECE, 1997]. This was used to determine the land actually used within a forest harvesting scheme. The volume of wood harvested is then conversed from ‘over bark’ (o.b.) via ‘under bark’ (u.b.; without bark) to tonne fresh wood. Finally, of the used forest land only the percentage attributable to the specific wood type is to be considered, and weighted averages of land use have to be determined, based on amounts imported to the Netherlands.

The resulting occupation figures are given in table 3.8.

Table 3.8 Data on land occupation for wood types for building used in the Netherlands; insignificant data are given in italics.

Wood type	Occupation (fresh wood under bark)	
	[$m^2.y/m^3$]	[$m^2.y/t$]
Spruce (*Picea Abies*) Scandinavia	2’920 - 3’080	7’500 ± 2’000
Pine (*Pinus*) Scandinavia	3’080 - 3’080	6’300 ± 1’000
Spruce Eastern Europe	14’700 - 15’500	41’000 ± 24’000
Spruce Central-Western Europe	5’200 - 5’480	6’600 ± 4’000
Spruce Canada	3’390 - 3’390	17’000 ± 12’000
Coniferous wood for NL	**5’530 – 5’820**	**16’000 ± 11’000**
European oak	2’150 - 2’150	*1’300 ± 3’000*
North-American oak	1’200 - 1’200	1’300 ± 500
Non-tropical hardwood for NL	**2’040 - 2’040**	**1’300 ± 300**
Meranti S.E.Asia	16’300 - 29’500	39’000 ± 30’000
Merbau S.E.Asia	12’700 - 22’900	30’000 ± 6’000
Iroko Africa	16’700 - 24’800	29’000 ± 20’000
Azobé W.Africa	2’670 - 5’280	5’400 ± 4’400
Amburana S.America	8’760 - 26’600	24’000 ± 17’000
Wane S.America	8’760 - 26’600	34’000 ± 24’000
Tropical hardwood for NL	**12’800 - 23’100**	**28’000 ± 22’000**

The occupation data are generally of the same order of magnitude as those from [Mak et al., 1996]. The data given (only for Finland) in [EU-DG XII, 1999] are also of the same order of magnitude as those calculated here.

For the upcoming EcoInvent 2000 database some land use data for Swiss forestry have been calculated [Jungbluth et al., 2002]. The area for wood production in Switzerland is on average 5921 m^2y/t, which also falls in the above range. However, assuming not the real output but the maximum potential production (assumed to be the net annual increment), this value would have been 3285 m^2y/t. Additionally, of this theoretical land use only 66% is allocated to the wood production, as the rest is attributed to other functions of the forest[25]. This leaves only 2177 m^2y/t average wood (and 1054 m^2y/t wood for firing based on its specific economic value). This is about half of the value we would find for Switzerland when using the appropriate percentages of coniferous and broadleaf wood (62% and 38% for Switzerland, respectively [Jungbluth et al., 2002]), but within our uncertainty range. However, assuming a much higher (theoretical) productivity than in reality is not in accordance to our general procedure to assess land occupation. And we do not consider preventing avalanches etc. as economic outputs of forestry, but as ‘environmental outputs’ of the forest. They are life support functions, to be assessed with impact assessment methods. The extent to which forestry allows for such functioning is ideally expressed in a better (lower) impact score. Therefore, we refrain from this allocation procedure and use the data presented in table 3.8, with the large uncertainty range.

[25] The 66% allocation is based on the forestry wood revenues, compared to the subsidies given to the forestry sector. Other studies are cited in [Jungbluth et al., 2002] suggesting only 5 to 10% of the total value of the forest to be due to the wood production (avalanche protection being about half of the total value in Switzerland, and allowing refuge to species is another, hard to quantify value).

Finally, we checked the above data from statistics with some case data ([1]: [Beetstra, 1998b], [2]: [Schweinle, 2000]). Nearly all cases gave results in the same range[26]:

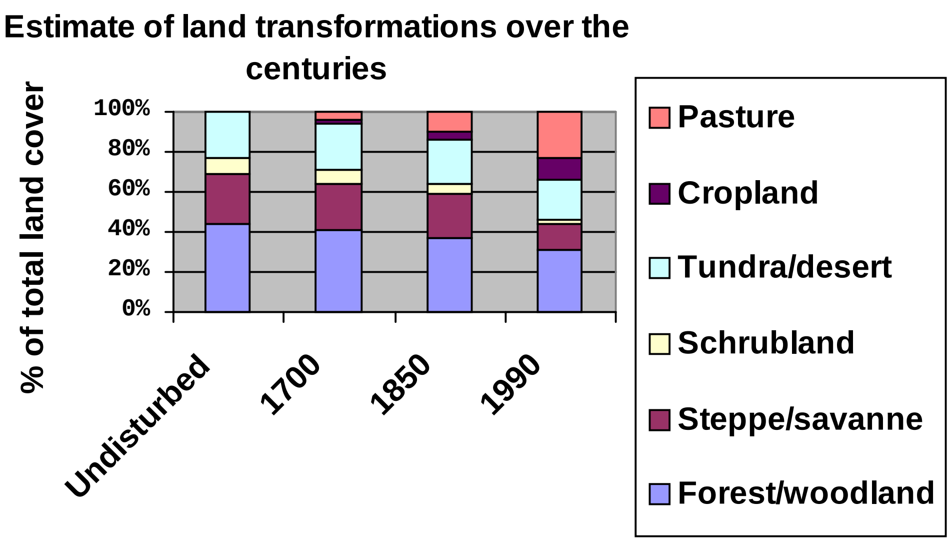
[1]	Douglas fir (wide stand) plantation; clearcut 75 years*	31'100 - 98'200	$m^2.y/t$
[1]	Selective clearcut cycle 5 years	25'900 - 81'800	$m^2.y/t$
[1]	Selective clearcut pine Norway in pristine forest	25'900 - 81'800	$m^2.y/t$
[1]	ibid, including renaturation of the pristine forest	2'616'000 - 8'262'000	$m^2.y/t$
[1]	Pine Sweden (FSC selective cut plantation)	135'000 - 427'000	$m^2.y/t$
[1]	Sustainable cut in pristine forest	556'000 - 1'750'000	$m^2.y/t$
[2]	Pinewood Germany*	9'710	$m^2.y/t$
[2]	Non-tropical hardwood Germany*	2'260	$m^2.y/t$
[1]	Sustainable cut pristine forest (CELOS-method Surinam)	855'000 - 1'280'000	$m^2.y/t$
[1]	Teak Brazil (FSC)	21'400 - 31'900	$m^2.y/t$

The cases from Beetstra are generally on the upper level of the range derived from FAO. The cases from Schweinle are even much higher than the FAO data. A review study to verify which data is better was not possible within the scope of this study, but since the TNO data are based on well-established statistics and not on single cases, we trust our lower land occupation values. The sustainable cuts generally occupy much more land, but the impact score is zero or very positive for the environment (see chapter 4).

3.3.2 Transformation for wood

Of the global land area (130 million km^2) 30% is forestry (25% closed forest, 3% open forest (low canopy coverage), 1% plantations and 1% forest fallow). This global forest area is decreasing with an annual speed of 94^e12 m^2/y over the period 1990-2000, in spite of an increasing area in Europe (9^e12 m^2/y) and in the United States (4^e12 m^2/y) [FAO, 2001]. On estimate, of the originally undisturbed 58.6 million km^2 forest and woodland, man had altered 4.2 million km^2 by 1700, 8.6 million km^2 by 1850 and 17.1 million km^2 by 1990. See figure 3.3, based on [Klein Goldewijk, 2001]. The in 1990 remaining 41.5 million km^2 forest and woodland in [Klein Goldewijk, 2001] seems to match the 38,7 million km^2 forest in 2000 according to [FAO, 2001].

Figure 3.3 Land transformations in the past, from [Klein Goldewijk, 2001].



Most land transformations in the past were due to pasture, followed by cropland, mainly in the 19th and 20th century. Most affected were grasslands, followed by forests/woodlands, tundra/desert and scrubland. Nowadays, land transformation in forests may have many different causes related to

[26] Except *, which are higher in [Beetstra, 1998b].

human activities [Bryant et al., 1997]. It is expected that until 2100 land transformations, especially in the (sub)tropical forested region, will be the main cause for loss of global biodiversity [Sala et al., 2000]. Therefore, assessing land transformation in forests is an important addition to the information of the static situation of land occupation (see above, 3.3.1).

For transformation attributable to forestry, in first instance only the increased forest area should be assessed (mainly in Europe and the US). This can be considered a **positive transformation** from an environmental point of view (see the impact assessment chapter 4). The data for annual land cover changes over the period 1990 to 2000 stem from the Forest Resource Assessement 2000 [FAO, 2001][27].

Of the increased forest area in Europe, unknown parts are caused by spontaneous land recovery from abandoned extensively used pasture, as in Switzerland, France and some other European countries [Müller-Wenk, 2001]. In Sweden, agriculture land is also lost to forests due to less farming. In the Netherlands, agriculture land is transformed into forestry due to national policy. Whether the transformations occur due to direct economic mechanisms or indirectly via policy, the 8.8^{e}12 m^2 yearly forest area increase in Europe can be attributed to the 483 e6 m^3 o.b. wood yearly extracted (output; functional unit: wood extracted yearly in Europe), resulting in an average value of 18 m^2/m^3 over bark. The range in this number is however from 1,6 (Finland) to 73 (Ireland). An average value therefore has no significance. Also for wood types imported to the Netherlands, the transformation data per country are very variable. Therefore, one representative country (that with the largest import share) per wood type is chosen. Based on European statistics [EEA, 1998] we can assume that about all of the transformation is from agriculture land.

For Northern America, Canada is most representative for coniferous wood, with no significant change in forest area. For non-coniferous wood, the USA is more representative, for which we can also assume agriculture to be the original state.

The calculation from over bark to tonne of fresh wood under bark follows the same steps as for occupation (see above)[28]. The results are shown in table 3.10.

Additionally, **negative transformations** from natural forests to commercially forested areas and plantations should be attributed to that commercial activity forestry. This mainly occurs in non-Western countries. Whether impacts due to such transformations can be measured will be discussed in chapter 4. Here an attempt is made to quantify such transformations.

An authorative study of FAO states that estimates of the contribution of the various causes of deforestation (agriculture, logging, road building, firewood extraction etc.) vary widely and can not be interpreted easily [FAO, 2000]. This makes it hard to perform an attribution of forest land transformations to the various following land uses, amongst which logging. However, in order to arrive at some results, we have taken the liberty of using one set of estimations, interpreting them conservatively[20].

Estimates for the contribution of the forestry sector to the decrease of natural forest area come from a World Resource Institute report [Bryant et al., 1997]. The estimated contribution of logging to the percent of threat to forest frontiers (large intact natural forest ecosystems) from their study is given in table 3.9, together with the contributions of land covers causing extensive loss of nature quality (mining, roads and other infrastructure).

[27] The land cover change data for Western countries from the UN/ECE over the period 1990-1995 [UN/ECE, 1997] show somewhat different results, varying per country from about half to double of the FAO values over 1990-2000, with extremely higher values for the main producers Russian Federation and Canada. As the UN/ECE website shows some other irregularities and is from 1997, the FAO data are taken to be authoritative.

[28] Here, no fraction exploitable forest is considered, as it is assumed that all forest area increase is left to forestry activities.

Table 3.9 Estimated contribution of logging and mining/infrastructure to threatened natural forest area (adapted from [Bryant et al., 1997][29]).

Region	% threatened from logging	% threatened from mining/infrastructure
Africa	61	9
Asia	30	6
North/Central America	24	8
Central	47	15
North	22	7
South America	37	29
Russia & Europe	15	10
Europe	80	0
Russia	16	10
Oceania incl. Australia	32	19
Global average	**28**	**15**

Using above figures and the FAO study on forest resource assessment, the global yearly loss of forest area could be attributed in part to forestry (- 94.000 miljoen m^2 x 0.28) if global wood extraction data were available. Due to many data gaps this is not possible, based on [FAO, 2001]. However, the factors in table 3.5 can also be applied to individual countries, as long as the uncertainty in these figures are acknowledged[30]. The results are given in table 3.10.

Table 3.10 Data on land transformation for wood types for building used in the Netherlands.

Wood type	**Transformation** * (fresh wood under bark)	
	[m^2/m^3]	[m^2/t]
Spruce (*Picea Abies*) Scandinavia	0,14	0,2 ± 0,08 (S)
Pine (*Pinus*) Scandinavia	0,14	0,18 (S)
Spruce Eastern Europe	8,4 - 8,9	12 ± 4,7 (USSR)
Spruce Central-Western Europe	0	0 (D)
Spruce Canada	0	0 (Can)
Coniferous wood for NL		***13 ± 40***
European oak	0	0 (D)
North-American oak	4,2	4,1 ± 0,6 (USA)
Non-tropical hardwood for NL		***5 ± 23***
Meranti S.E.Asia	141 - 255**	240 ± 100 (Indonesia)
Merbau S.E.Asia	108 - 195**	150 ± 58 (Malaysia)
Iroko Africa	556 - 830**	720 ± 200 (Cameroon)
Azobé W.Africa	14 - 28**	21 ± 9 (Gabon)
Amburana S.America	4240 -12'900**	8e3 ± 5e3 (Brazil)
Wane S.America	4240 -12'900**	12e3 ± 7e3 (Brazil)
Tropical hardwood for NL	**681 - 1'790****	**1'400 ± 400**

* Between brackets the country representative for transformation (highest import for that wood type in NL).

** Actually negative values (forest area is lost instead of gained).

[29] Due to overlap between the threat sources the total % in [Bryant et al., 1997] is >100%. We have chosen to decrease their original % given for all threat sources equally, by multiplying with their % of frontier forest under moderate or high threat.

[30] Due to this procedure (included in table 3.6), it is assumed that the uncertainty is a factor 1.5 higher.

3.4 Results background data: energy production

The background data for energy are based on the ETH database, [Frischknecht et al., 1996]. ETH modelled the land use data by assuming that all transformations are recovered, leaving only land occupation data. In these data, the renaturation time according to generic recovery times[31] for biodiversity were incorporated. In the present study, the renaturation part is disentangled from the (digital) database documention [Frischknecht et al., 1996]. The ETH data on land use are converted to both the occupation and transformation data according to our format. These new land use data and new nomenclature have been imported in the 1996 ETH database, and new aggregated processes for electricity, some fuels and transport were calculated. The selection of processes include the background data from ETH calculated with specific Dutch emission data [Eggels & van der Ven, 2000].

Data on renaturation is documented separately by us, as this renaturation is generally theoretical, and applied to result in zero transformation and only one unit for land use interventions (m^2.y). This renaturation data is therefore not included in the cases studies.

This conversion was performed for the most dominant processes for land use.

The results of a first rough dominance analysis on European electricity producers (UCPTE) fuel mixture are illustrated in figures 3.4 and 3.5. Figure 3.4 presents the land use impacts of the different fuel types per 1 TJ UCPTE electricity, whereas figure 3.5 presents the land used impacts of the different fuel types used in the production of the production of UCPTE electricity (the share of the fuels is not accounted for).

The dominant processes within each of the process chains in above figures were determined by performing a dominance analysis on the original ETH data for the following process chains:

- Natural gas;
- Hard coal;
- Oil;
- Nuclear energy;
- LKW transport 28t.

Through this procedure, process data on the most important energy carriers, materials transport and infrastructure are covered[32].

Specific Dutch data on land use for electricity production have been finally added to the aggregated process data sheets, to arrive at specifically Dutch background data.

In the Netherlands, natural gas and hard coal are the most important energy carriers used for the production of electricity [ECN, 2001]. Oil is only used either if the hard coal supply is ceased or the demand of natural gas is too high (severe winters). On basis of the lined up power by SEP[33] and electricity production in the year 1997 [ECN, 2001], the average load factor and efficiency for both hard coal and natural gas power plants in The Netherlands are calculated. The specific load factors for hard coal and natural gas are higher than the load factors given in [Frischknecht et al., 1996]. The load factors for hard coal and natural gas are 6073 and 3172 hours per year, respectively. In [Frischknecht et al., 1996] the load factors for hard coal and natural gas are 4000 and 1800 hours per year, respectively. In addition to the load factors, the average efficiencies of the Dutch power plants are also higher: natural gas, 47.3% (ETH: 38%) and hard coal, 38.8% (ETH: 38.3%)[34]. The land use impact is decreased by a higher load factor as well as a higher efficiency. To incorporate this significant influence, the ETH figures on land use have been adjusted to the Dutch situation.

[31] 50 years with exceptions for one nuclear process (80000 years) and onshore pipelines (5 years).
[32] About 50 main contributing different processes.
[33] SEP stands for the Dutch Electricity Generating Board. Nowadays this organisation is discontinued, since the electricity production is liberalized.
[34] Since the deviation of the average efficiency of the hard coal power plant is small, a correction for the efficiency is not accounted for.

Figure 3.4 **Dominant fuel types in UCPTE electricity for each land use class in the ETH database.**

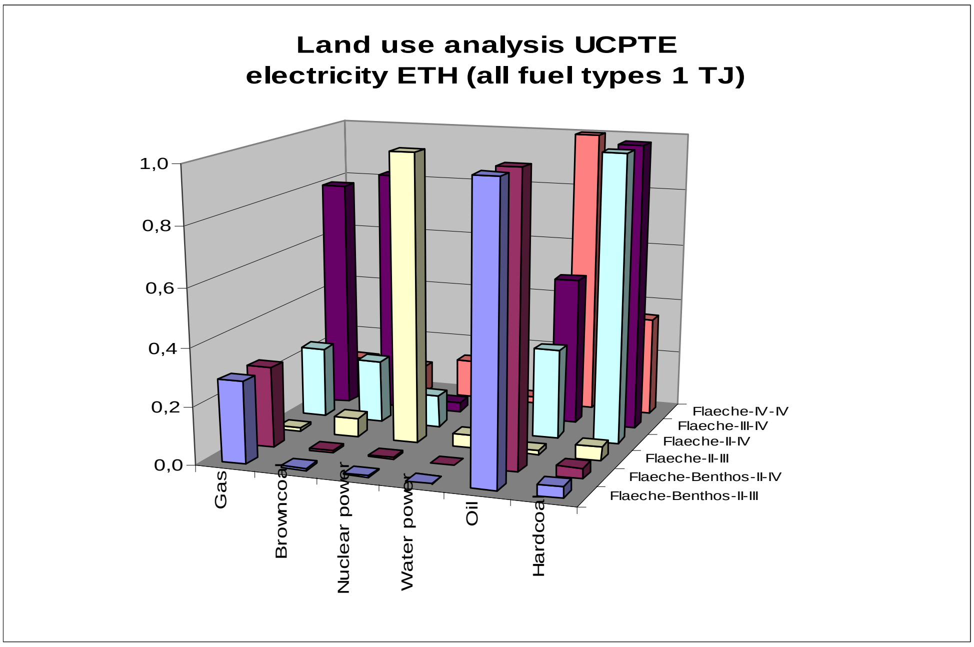
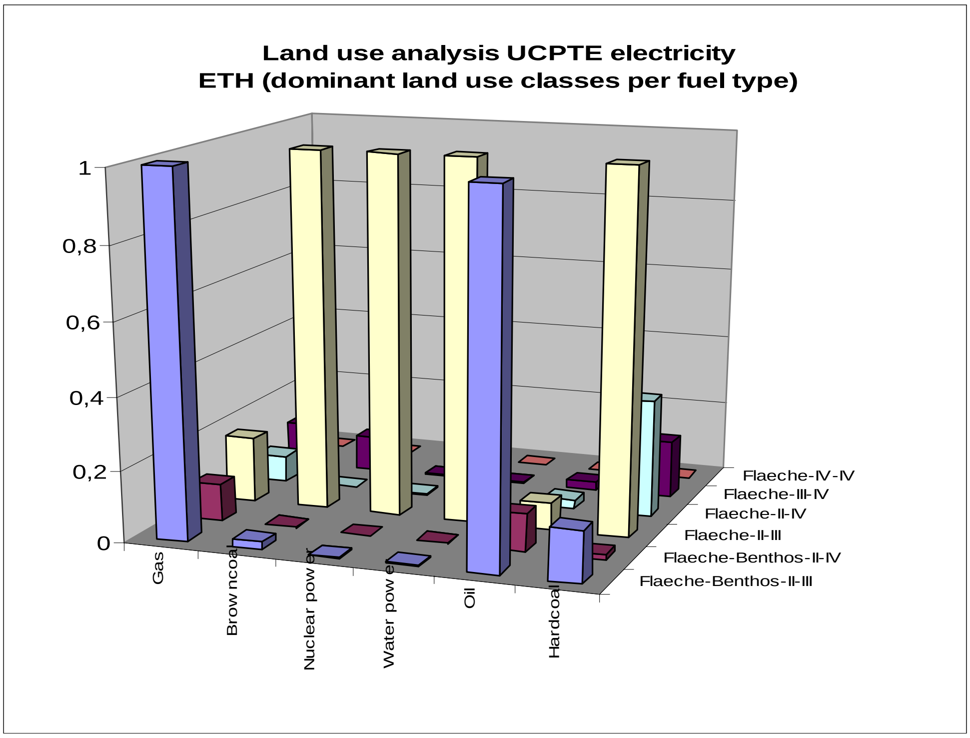


Figure 3.5 **Dominant land use classes for UCPTE electricity (assuming equal weights for all land use types).**



3.4.1 Occupation for energy production

Most processes in the ETH database consist of building, operation and demolition periods. The land use data in the ETH database is split by TNO into these 3 situations. The occupation impact for the building and recultivation periods is calculated by multiplying the building/recultivation time with the area, and then dividing by two[35]. In general, the building and demolition period is assumed in [Frischknecht et al., 1996] to take 5 years. In some cases, the land use for building and demolition is assumed to be 10% of the total ETH figure (e.g. power plants). In case of renaturation the ETH figure on land use is halved to acknowledge the average (half of total) quality during renaturation. The results of this conversion of ETH data to the TNO nomenclature and figures is given in annex 6. For two types of electricity production corrections were made for application in the Dutch situation, due to large differences in % of total time active per year (the so-called load), see below.

Specific Dutch natural gas power plant
ETH figure on land use is 20 m^2.y/TJ(in), efficiency is 38% and load is 1800 hours per year (land use including efficiency: 52.6 m^2.y/TJ). In the Netherlands the efficiency is 47.3% and load is 3172 hours per year. Correction for load and efficiency is taken into account. The land occupation (Occup ind built NL) for the Dutch situation is therefore 22.8 m^2.y/TJ instead of 53 m^2.y/TJ.

Specific Dutch hard coal power plant
The ETH figure on land occupation is 6.05 m^2.y/TJ(in), the efficiency is 38.3% (Dutch situtation according to ETH] and the load is 4000 hours per year. The efficiency of 38.3% is accounted for in a separate process data sheet of the ETH database. In the Netherlands the efficiency is 38.8% and load is 6073 hours per year. Deviation of the efficiency is small, so only a correction for the load is taken into account. The landuse occupation (Occup ind built NL) for the Dutch situation is 3.8 m^2.y/TJ instead of 5.7 m^2.y/TJ.

3.4.2 Transformation for energy production

In order to determine the transformation, the area occupied according to ETH is divided by the total period. For production processes, the periods consist of building, operation and recultivation. In general, the recultivation period takes 5 years. The building and operation period are process dependent. If no information is available, the life time is assumed to be 30 years. The renaturation processes are covered by the renaturation period. The renaturation period takes 50 years, with the exception of concentrated uranium (80000 years) and onshore pipeline for natural gas (5 years).

Specific Dutch natural gas power plant
Life time is assumed to be 30 years. The ETH land use is 20 m^2.y/TJ(in), efficiency is 38% and load is 1800 hours per year (land use including efficiency: 52.6 m^2.y/TJ). In the Netherlands the efficiency is 47.3% and load is 3172 hours per year. Correction for load and efficiency is taken into account. Landtransformation (Trans ind built II-IV NL) for the Dutch situation is 0.8 m^2/TJ instead of 1.75 m^2/TJ.

Specific Dutch hard coal power plant
Life time is assumed to be 30 years. The ETH land use is 6.05 m^2.y/TJ(in), efficiency is 38.3% (Dutch situtation according to ETH] and load is 4000 hours per year. The efficiency of 38.3% is accounted for in another process data sheet of the ETH database. In the Netherlands the efficiency is 38.8% and load is 6073 hours per year. Deviation of efficiency is small. For this reason, only a correction for the load is taken into account. Land transformation (Trans ind built II-IV NL) for the Dutch situation is 0.1 m^2/TJ instead of 0.2 m^2 /TJ.

[35] A linear increase c.q. decrease of land use is assumed during these periods, leading to an intervention of half of the area x time.

3.5 Results background data: transportation

Dutch data for direct land use (by the roads and railways themselves) is applied for the Dutch situation, with additional (indirect) land use for infrastructure and fuel from the ETH database (see also the introduction of section 3.4). For other countries the direct land use from the ETH data may still be valid, although this depends on the land use efficiency of the average power station per fuel type. At least for Switzerland the (here adapted) ETH data are valid. Therefore, both the unaltered and the Dutch data are given. Below, only the direct land use due to transportation is discussed. Here, an overview is given of both the direct and indirect land use for transportation:

Allocation procedure for land use by roads and railways

For freight transport (CBS categories: both truck and delivery van) the transport performance in tkm is the functional unit. However, for passenger traffic (CBS category: passenger cars & trains) there is no freight performance. Allocation of the road and rail traffic to freight on the basis of the freight performance is therefore not possible.

Instead, the number of vehicle kilometres is chosen as an allocation key for the land use due to roads. This is a proxy indicator for the area used on roads for cars, trucks and vans, during a certain time. In the end. These vehicle kilometres are presented per calendar year by CBS.

For railways the situation is more complex, because the number of vehicle km is not given, the vehicle length is different for freight and passenger trains and is not directly related to the performance. Therefore, we have chosen two different allocation keys: the transported weight and the turnover. As the turnover of the railways is not publicly available, we have taken the amount of governmental payment to the national railway company as a rough substitute. The different results will be shown as ranges in the occupation scores.

Finally, for transformation it should be determined which data should be used to attribute the change in area to. The average contribution of the freight transport to the traffic pressure (in driven km) is according to us the more relevant basis for this attribution, because the absolute traffic pressure over a certain period is what really steers decisions on road expansions. For roads this is for freight transport 14.0% during the period 1993-1996 and 13.2% during 1989-1996. For railways no significant trend could be determined, so no transformation attribution was necessary there.

In tables 3.11 and 3.12 the relevant CBS figures are given for the calculation of land use in the reference years 1988, 1992 and 1995. Table 12 is split to show the difference when using a different allocation procedure.

Table 3.11 Road transport data for the reference years 1988, 1992 and 1995 [CBS, 2000c].

Year	Passenger vehicle km [km]	Freight vehicle km (trucks and vans) [km]	Share Freight [%]
1988	7.90 E+10	1.04 E+10	11.6
1992	8.57 E+10	1.33 E+10	13.4
1995	90.0 E+10	1.49 E+10	14.2

Table 3.12a Railway transport data for the reference years 1988, 1992 and 1995, allocation on basis of kg transported weight (assuming persons to weigh on average 70 kg) [CBS, 1997b, 2001ab].

Year	Passenger weight [kt]	Freight weight [kt]	Share Freight [%]
1988	16100	5220	24
1992	23310	4870	17
1995	21350	4350	17

Table 3.12b Railway transport data for the reference years 1988, 1992 and 1995, allocation on basis of national government payments to the national railway company [CBS, 2001ab].

Year	Passenger transfer expenses [M€]	Freight vehicle Transfer expenses [M€]	Share Freight [%]
1988	633	10	1.6
1992	722	16	2.2
1995	751	52	2.9

3.5.1 Occupation for transport

Roads

The direct land use occupation by road freight transport is given in table 3.13 for the reference years 1988, 1992 and 1995 [CBS, 2000a]. This area is including embankments. These CBS land use figures include motorways, main or national roads, secondary or regional roads and other roads (paved, half- and unpaved roads). Since the freight performance (in tkm) is applied as functional unit, the land use is allocated on basis of the share of the freight transport (see tables 3.11 and 3.12).

Table 3.13 Land use occupation by the road freight transport [CBS, 2000a].

Year	Land use all traffic [m^2]	Allocated land use [m^2]	Occupation [m^2.y/tkm]
1988	1.17 E+9	1.36 E+8	**6.14 E-3**
1992	1.20 E+9	1.61 E+8	**6.28 E-3**
1995	1.21 E+9	1.72 E+8	**6.37 E-3**

Railways

The freight performance (in tkm) is the functional unit (see table 3.14). Only the active railway area (including its embankments) should be attributed to this functional unit. Therefore, the area for shunting-yard and unused railway is derived from the total land use for railways minus the area for active railway transport. The latter area is calculated from the length and width of the active railway net. The width of 30 m is derived from an earlier assessment on land use [Mak et al., 1996], based on a selection of 6% of all digital maps of railways in the Netherlands [Lengkeek, 1995].

Table 3.14 Railway transport data for the reference years 1988, 1992 and 1995 [CBS, 1997b, 2000a, 2001ab].

Year	Transport performance freight[1] [tkm/y]	Total land use railways[2] [mln m^2]	Length railway[3] [km]	Width railway[4] [m]	Land use rail performance[5] [mln m^2]	Land use shunting-yard / unused[5] [mln m^2]
1988	1049	101.3	2810	30 ± 0.4	84.3	17.0
1992	966	101.2	2791	30 ± 0.4	83.7	17.5
1995	721	99.2	2813	30 ± 0.4	84.4	14.9

1 [CBS, 1997b, 2001b].
2 [CBS, 2000a].
3 [CBS, 2001a].
4 [Lengkeek, 1995].
5 Calculated.

The part of the railway area to be allocated to freight transport is determined by using either the transported weight (table 3.16) or the transfer expenses (subsidies etc.) by the national government (table 3.15) as allocation factors (see also tables 3.12a and 3.12b) .

Table 3.15 Land use occupation by the railway freight transport, allocation on basis of transfer expenses by the national government.

Year	Land use [mln m^2]	Allocated total land use [mln m^2]	Occupation [m^2.y/tkm]
1988	84.3	1.31	**1.25 E-3**
1992	83.7	1.80	**1.86 E-3**
1995	84.4	5.44	**7.54 E-3**

Table 3.16 Land use occupation by the railway freight transport, allocation on basis of transported weight.

Year	Land use [mln m^2]	Allocated land use [mln m^2]	Occupation [m^2.y/tkm]
1988	84.3	20.7	**1.97 E-2**
1992	83.7	14.5	**1.50 E-2**
1995	84.4	14.3	**1.98 E-2**

The two equally possible allocation keys thus lead to a range in outcome of a factor 10. This model uncertainty has to remain.

Canals

Data are taken from [Mak et al., 1996]. The length of narrow Dutch waters used for freight was 3745 km in the period 1988-1996 [CBS, 2001b]. The average breadth according to a CBS land statistics expert (based on a 6% sample of digital maps with canals [Lengkeek, 1995]) is 24 ± 13 meter. This figures result in a range of land use for canals of 55-187 mln m^2.

The freight performance and land occupation by canals for the reference years 1988, 1992 and 1995 are given in table 3.17. Here no allocation is made to recreational vessels, as the investment to build canals was not done for them.

Table 3.17 Calculation of land occupation by canal transport [CBS, 1997b, 2001b].

Year	Freight performance [tkm]	Occupation minimum [m^2.y/tkm]	Occupation maximum [m^2.y/tkm]
1988	7478	**7.42 E-3**	**2.50 E-2**
1992	5926	**9.37 E-3**	**3.15 E-2**
1995	5986	**9.27 E-3**	**3.12 E-2**

3.5.2 Transformation for transport

Roads

For road transformation, the CBS figures on the change of land are given for January 1st of each year.

During 1993-1996 12 mln m^2 of land is added to roads [CBS, 2000a]. 45.1% of this land is withdrawn from agricultural land, 43.3% from building-sites for other developments (original mainly 90.5% agricultural land, 3.0% recreation), 2.3% forest, 2.3% building-sites for industrial sites (original mainly 79.3% agricultural land) and 1.9% recreation (various).

In summary, the withdrawal of the added land includes:

- 86.1% agricultural land (45.1% direct, 39.2% stock through building-sites for other developments and 1.8% stock through building-sites for industrial sites);
- 3.2% recreation (1.9% direct and 1.3% stock through building-sites for other developments);
- 3.1% forest (2.3% direct and 0.8% stock through building-sites for other developments);
- 7.6% other land use, unspecified in this report due to the low contribution of the various land types (less than 1%).

In the period 1989-1993 30.98 mln m^2 of land is added to this category [CBS, 1997a], resulting in a total addition of 42.98 mln m^2 during 1989-1996. The CBS figures on the addition of land are divided by the number of years in the concerned period (4 years in 1993-1996; 8 years in 1989-1996), resulting in a yearly addition of land. Subsequently, this yearly addition of land is allocated on basis of the average share of the freight transport in the involved period: 14.0% and 13.2% in 1993-1996 and 1989-1996, respectively. Furthermore, the gross transformations are calculated on basis of both the average freight performance during the involved period and the freight performance in the last year (1995). The results of the gross transformations for the periods 1993-1996 and 1989-1996 are given in table 3.18

Table 3.18 Gross transformations caused by road freight transport during 1993-1996 and 1989-1996 [CBS, 1997a, 2000a].

Period	Added land per year [m^2/y]	Average tkm per year [tkm/y]	Transformation (average performance) [m^2/tkm]	Transformation (performance in 1995) [m^2/tkm]
1993 - 1996	4.21 E+5	2.29 E+10	1.84 E-5	**1.56 E-5**
1989 - 1996	7.11 E+5	2.32 E+10	3.06 E-5	**2.63 E-5**

The impact on land use is lower using the freight performance in 1995. This is caused by the efficiency improvements which have been taken place in the course of years.

The above range of land transformation scores still need to be specified to the various previous states. Following the summarised data from CBS this is:
86.1 % agricultural land;
3.2 % recreation;
3.1 % forest.

Railways
The trend of land use for railway transportation has been zero or negative during the last 10 years in the Netherlands. Therefore, transformation data is at present zero. When the so-called Betuwelijn is built, actual data on the total performance until the full depreciation time, and on the area of the line can be used to calculate transformation data for this first Dutch only freight railway line.

Canals
There was no significant trend in narrow water area from 1980 to 1990, except one increase of 225 km in 1989. Therefore, as with railways, no significant transformation could be determined over this period.

3.6 Results background data: mining of other materials

3.6.1 Metals

Aluminium is the first metal for which a world-wide survey exists with quantitative environmental interventions and impacts [Beck, 2000]. Only with such surveys can land use be assessed easily.

From [Beck, 2000] the folllowing data are taken:

- 52,000 dry crude tonnes (dct)/ha in 1991 (47% Al) => 0.41 m^2.y/t Al
- 56,500 dct/ha in 1998 (45% Al) => 0.39 m^2.y/t Al
- initial land cover
  - 58.4 % hardwood forest
  - 14.5 % tropical rainforest
  - 14.5 % natural pasture
  - 3.5 % tilled agriculture
  - 2.6 % softwood forest
  - 1.8 % shrubland
  - 1.4 % commercial pasture
  - 0.4 % wetland vegetation

  initial climate:
  - 47.7 % tropical
  - 12.5 % subtropical
  - 38 % mediterranean
- final land cover:
  - 69.6 % native forest
  - 14.7 % livestock pasture
  - 3 % commercial forest
  - 2.4 % agriculture crops
  - 2.2 % urban development
  - 0.4 % industrial development
  - 0.2 % recreational area
  - 7 % other/unknown
- Two cases of transformation were mentioned:
  - 3,800,000 m^2 rehabilitated for 18.5 million dct => 0.46 m^2/t Al
  - 480,000 m^2 rehabilitated for 12.3 million dct => 0.09 m^2/t Al
- About 80% of the yearly mining land is wildlife habitat, and 95% of the remaining reserves are wildlife habitat. A third of that area (for 1/3 of the production) contain endangered species; all of this habitat is planned to be restored. The intention is to reforest 73% of the total mined area with indigenous species, give 17% to agriculture and keep 10% to infrastructure.

Another source [Sliwka & Bauer, 2000] report for the Trompetas case (Amazon region of Brazil) 0.18 m^2.y/t bauxite (0.40 m^2.y/t Al), and for an open pit mine in south-east Asia 0.74 m^2.y/t bauxite (1.6 m^2.y/t Al). In the Trompetas mining area (0.18^e6 m^2) 360 indigenous species have been planted in 20 years, with a density of 2500 plants/ha. Immediately after rehabilitation the area looks bare, but after 3 years the growth is up to 5 meters high.

The following general figures for land use by bauxite mining are concluded:

Occupation: **0.18 – 0.74 m^2.y/t bauxite** or **0.4 – 1.6 m^2.y/t Al**[36]

[36] The mean is near 0.4 m2.y/t.

Transformation: **0.2 – 1.0 m^2/t bauxite** or **0.1 – 0.5 m^2/t Al**[37]
The distribution of the initial and final states for transformation are estimated below, based on above figures:

64	% :	no transformation (forest => forest, agriculture => agriculture)
15	% :	from natural pasture to livestock pasture
7	% :	from tropical rainforest to native forest
7	% :	from tropical forest to other area (scrubland taken)
4	% :	from tropical rainforest to commercial forest
2	% :	from scrubland to urban area
1	% :	from agriculture to urban area
0.2	% :	from wetland to recreational area
0.2	% :	from wetland to livestock pasture

For future aluminium extraction from kaoline [de Vries, 1994] gives for occupation 22.5 m^2.y/t Al and for transformation 1.5 m^2/t Al (including overburden and tailings).

A third source [Wirtz & Schäfer, 1999] reports on the Bayer process (to produce aluminiumoxide from bauxite) that at the tropical Ewarton pit, Jamaica, each year 580,000 t Al_2O_3 (154,000 t Al) and 660,000 t red mud is produced. The red mud dump built 40 years ago extends over 100 ha, resulting in a land occupation of **0.16 m^2.y/t Al**, and a transformation of **0.13 m^2/t Al** (assuming 50 years total use). After use the dumps are recultivated into agriculture, with specially selected plant types. The transformation is thus from tropical forest to agriculture. These data should at least be included in the total land use data for aluminium.

For copper, another source is used [de Vries, 1994] and the data calculation from [Mak et al., 1996] is used for a mix of 85% open pit and 15% shaft mining, resulting in an occupation of **1.05-2.18 m^2.y/t copper ore** assuming 15 years of mining at one spot (with 0.7% Cu in ore 289 m^2.y/t Cu including 22% losses in the chain). In the ETH database [Frischknecht et al., 1994] the Bingham Canyon is mentioned, with 0,58 m^2.y/t ore based on the 128 years that the mine was used, but in the ETH database only 15 years were considered, leading to a much higher land use figure. This case is not considered due to the conflicting information given. The related transformation from ETH is **0.07 – 0.15 m^2/t copper ore** and 11-24 m^2/t Cu including 22% losses.

For iron, [Frischknecht et al., 1994] has given the following occupation data: **0.11 m^2.y/t iron ore**, and as transformation data **0.011 m^2/t iron ore**.
For manganese, we derive from the same source **4.2 m^2.y/t manganese ore**, and for transformation **0.42 m^2/t manganese ore**.

For other metals the copper data are used for the open pit situation (1,05-2,18 m^2.y/t for occupation and 0.07-0.15 m^2/t for transformation). The shaft mining case for copper in [de Vries, 1994] is taken to be exemplary for other shaft mining cases for metals. Transformation is then for shaft mining **0.05 – 0.1 m^2/t ore**. Occupation data is **0.7-1,5 m^2.y/t** for shaft mining using 15 years as an average, from [Frischknecht et al., 1994].

3.6.2 Other minerals
For kaoline [Frischknecht et al., 1994] gives an occupation value of **25-26 m^2.y/t** for temperate regions, assuming 15 years mining period. In [de Vries, 1994] the mining time is not mentioned, but assuming the same as in ETH, we arrive at the same value. For the tropics, this assumption results in a value of **0.7 m^2.y/t**. The corresponding transformation values are respectively: **1.6 – 1.7 m^2/t** and **0.047 m^2/t**.

[37] Both occupation and transformation with 45% Al in bauxite, including mining losses but excluding losses in the chain.

The data per m^2 layer in open pits calculated in [Mak et al., 1996] can be used for remaining surface resources: **0.03 – 0.8 m^2.y/t ore per m** layer thickness for occupation, and **0.0004 – 0.06 m^2/t ore per m** layer thickness for transformation.

Regarding generic recovery times, [de Vries, 1994] mentions for grassland 10-20 years based on a case in Illinois, USA (12 years), for tropical forest at least 100 years (when seedlings from the same forest type are transplanted from nearby, and when the top-soil is not removed, f.i. by bulldozers) and 200 to over 1000 years when these conditions are not met. On islands or in mountain areas the renaturation time is estimated as over 200 years based on a natural disaster case (a vulcano eruption over 100 years ago in Indonesia). In an Australian case of bauxite mining in a Eucalyptus forest renaturation was reached after some decennia. However, some ecosystems or species may be lost forever.

3.7 Results industrial production

As functional unit of industrial production, the Dutch production in ton is used. This production is derived from [CBS, 1999] and concerns the production years 1997 and 1998. On basis of the quantity-index [CBS, 2000d] the production is calculated back to the considered year.

Some industrial sales are not mentioned in [CBS, 1999]: they are unknown or confidential. Furthermore, some industrial sales are expressed in pieces, others in litres. In order to convert the pieces into ton, the conversion factor [economic value/amount] of the most available specific subcategory is applied[38]. This approach is checked with figures on the ratio [net turnover/total production] for some companies and seems to be reliable. The amount of milk, cream and beverage (in litres) is converted into ton by assuming a density for these liquids.

The CBS figures on industrial sales per sector for the reference years 1988, 1992 and 1995 are given in table 3.19.

Table 3.19 Industrial sales for the reference years 1988, 1992 and 1995 [CBS, 1999, 2000d].

Industrial products	Sales 1988 [ton]	Sales 1992 [ton]	Sales 1995 [ton]
Food products, beverages and tobacco products	3.97 E+7	4.27 E+7	4.57 E+7
Products of textile or leather	5.01 E+5	5.38 E+5	5.76 E+5
Products of paper; published or printed products	1.21 E+7	1.30 E+7	1.39 E+7
Chemical products and products of rubber or plastic	2.98 E+7	3.20 E+7	3.43 E+7
Metal and electrical products and transport equipment	2.20 E+7	2.36 E+7	2.53 E+7
Products of wood and other non-metalic mineral products, furniture and other goods	1.58 E+8	1.70 E+8	1.82 E+8
Total	2.62 E+8	2.82 E+8	3.02 E+8

3.7.1 Occupation for industrial production

The land use occupation by the industrial production is given in table 3.17 for the reference years 1988, 1992 and 1995 [CBS, 2000a]. The yearly land use is divided by the total production, taken from table 3.19.

[38] With the exception of subcategory sugar: that value is derived from the total of the category food products minus the total of the remaining subcategories.

Table 3.20 Land use occupation by industrial production [CBS, 2000a].

Year	Land use [m^2]	Occupation [m^2.y/ton]
1988	4.78 E+8	**1.82**
1992	5.26 E+8	**1.87**
1995	5.67 E+8	**1.88**

IVAM ER gave a figure of 8,6 m^2.y/t [Mak et al., 1996], but had excluded the production which could not be expressed in weight. Bulk chemicals are not included in above figures. Here, [Mak et al., 1996] used specific data from the ETH database [Frischknecht et al., 1994]: **0.008 m^2.y/t** (from a refinery), to which we adher here. For transhipment (also not included in above figures) ETH gives a value of **1 m^2.y/t** for transhipment, taken here as well.

3.7.2 Transformation for industrial production

Between 1993 and 1996 54,20 million m^2 of land has been added to industrial sites [CBS, 2000a]. 61.9% of this land is withdrawn from building-sites for industrial sites (originally 79.3% agricultural land, 5.5% natural sites, 5.5% water, 2.7% industrial sites, 2.4% forest, 1.9% recreation), 33.9% from agriculture and 1% from building-sites for other developments (originally 90,5% agricultural land, other land use do not influence the outcome).

In summary, the withdrawal of the added land includes:

- 83.9% agricultural land (33.9% direct, 49.1% stock through building-sites for industrial sites and 0.9% stock through building-sites for other developments);
- 3.5% natural sites (mainly through stock building-sites);
- 3.7% water (mainly through stock building-sites, in general other inland water, wider than 6 metres);
- 1.7% industrial sites (completely through stock building-sites);
- 2.2% forest (mainly through stock building-sites);
- 1.8% recreation (mainly through stock building-sites)
- 3.2% other land use, unspecified in this report due to the low contribution of the various land types (less than 1%).

In the period 1989-1993 53.42 mln m^2 of land is added to this category [CBS, 1997a], resulting in a total addition of 107.62 mln m^2 during 1989-1996.

The CBS figures on the addition of land are divided by the number of years in the concerned period (4 years in 1993-1996; 8 years in 1989-1996), resulting in a yearly addition of land. The gross transformations are calculated on basis of both the average industrial sales during the involved period and the industrial sales in the last year (1995, see table 3.21). The results of the gross transformations for the periods 1993-1996 and 1989-1996 are given in table 3.21.

Table 3.21 Gross transformations caused by the industrial production during 1993-1996 and 1989-1996 [CBS, 1997a, 2000a].

Period	Added land per year [m^2/y]	Average industrial sales per year [ton/y]	Transformation (average industrial sales) [m^2/ton]	Transformation (industrial sales in 1995) [m^2/ton]
1993 - 1996	1.36 E+7	2.90 E+8	**0.047**	**0.045**
1989 - 1996	1.35 E+7	2.84 E+8	**0.048**	**0.047**

The impact on land use is lower applying the last production year (1995). This is caused by the efficiency improvements which have taken place in the course of years.

Trend data are not available for bulk industry or transhipment.

3.8 Results living

The number of housing accomodations is chosen as functional unit. The CBS figures on the housing stock (CBS categories: both dwellings and housing units[39]) are based on the date set of 1 January. In table 3.22 the housing stock is given for the reference years 1988, 1992 and 1995 [CBS, 2000b].

Table 3.22 Housing stock for the reference years 1988, 1992 and 1995 [CBS, 2000b].

Year[1]	Dwellings [number]	Housing units [number]	Total [number]
1988	5.70 E+6	8.84 E+4	5.79 E+6
1992	6.04 E+6	8.98 E+4	6.13 E+6
1995	6.28 E+6	9.57 E+4	6.37 E+6

1 CBS figures on housing accommodations are based on the date set of 1 January. This implies that 1989 in [CBS, 2000b] represents 1/1/89 and refers to the calender year 1988.

Next to the direct land use for the house, we include from section 3.1 the raising sand often used to build Dutch houses on.

3.8.1 Occupation

The land use occupation by the housing stock is given in table 3.23 for the reference years 1988, 1992 and 1995. The number of residences is derived from table 3.22 [CBS, 2000a].

Table 3.23 Land use occupation by housing stock [CBS, 2000a].

Year	Land use [m^2]	Occupation [m^2.y/residence]
1988	2.10 E+9	**363**
1992	2.17 E+9	**354**
1995	2.24 E+9	**352**

Assuming a worst case average for the raising sand often applied in the Netherlands for building a house on, 250 m^3 (425 tonne) raising sand could be used per house. When used for only 50 years, this worst case relates to **13 m^2.y extra per residence** (4% of the occupation for the residence itself). This amount can thus be neglected at first.

3.8.2 Transformation

During 1993-1996 96.20 mln m^2 of land is added to the housing stock [CBS, 2000a]. 47.8% of this land is withdrawn from agricultural land, 46.9% from building-sites for other developments (original mainly 90,5% agricultural land, 3,0% recreation) and 2.8% from recreation (mainly sports ground).

[39] Since 1992 the CBS definition of housing units changed. The current definition exclude housing accomodation in homes for elderly. For this reason the CBS figures for the years 1988 up to 1992 are adjusted to the new definition by using the relative ratio of the balance increase housing units in the corresponding years according to the “old definition”.

In summary, the withdrawal of the added land includes:

- 90.2% agricultural land (47.8% direct and 42.4% stock through building-sites for other developments);
- 4.2% recreation (2.8% direct and 1.4% stock through building-sites for other developments);
- 5.6% other land use, unspecified in this report due to the low contribution of the various land types (less than 1%).

In the period 1989-1993 89.21 mln m^2 of land is added to this category [CBS, 1997a], resulting in a total addition of 185.41 mln m^2 during 1989-1996. The CBS figures on the addition of land are divided by the number of years in the concerned period (4 years in 1993-1996; 8 years in 1989-1996), resulting in a yearly addition of land. The gross transformations are calculated on the basis of both the average housing stock during the involved period and the housing stock in the last year (1995). The results of the gross transformations for the periods 1993-1996 and 1989-1996 are given in table 3.24.

Table 3.24 Gross transformations caused by the housing stock during 1993-1996 and 1989-1996 [CBS, 1997a, 2000a].

Period	Added land per year [m^2/y]	Average housing stock per year [number/y]	Transformation (average housing stock) [m^2/residences]	Transformation (housing stock in 1995) [m^2/residences]
1993 – 1996	2.41 E+7	6.25 E+6	3.86	3.78
1989 – 1996	2.32 E+7	6.09 E+6	3.81	3.64

The longer the period, the lower the gross transformation. This indicates that the pressure of the housing stock on land use has further increased the last few years. Using the housing stock in 1995 the impact on land use is reduced. This is caused by the increased housing stock in the course of years.

The transformation for the raising sand often necessary for building in the Netherlands is in a worst case (250 m^3, 425 tonne, 50 years): **0,34 m^2 extra per residence** (less than 10% of the land transformation for the residence itself in this worst case).

3.9 Results waste disposal

Data for waste disposal is taken from IVAM ER [Mak et al., 1996]. For burning chemical waste (136.5 million kg/year) 8 ha is used, excluding dumping of ashes. This results in **0.59 m^2.y/t**. For dumping, the occupation data is calculated from a maximum waste height of 30 metes, 25 years use of the dumpsite, and specific densities per waste type. For building waste around 1990 (3400^{e}3 t dumped, density 2000 kg/m^3 [CBS, 1991]) this results in **0.42 m^2.y/t**. For dumping waste in general a range is given of **1.5 – 2.3 m^2.y/t**.

3.10 Normalisation data

Normalisation data are data on the total environmental burden of a region. These data are used in LCA studies to relate case study data to this total burden (per impact category). This allows a comparison of the relative contribution of the case to the various types of impact in the region. In this study, these data on land use are collected for the Netherlands and for the 15 European Union countries of Europe (the EU-15).

The normalisation data for occupation consist of the areas of the various land use types within the region, multiplied with the impact score for that land use type. In this chapter we present only the physical normalisation data; the normalisation impact scores are given in chapter 4.

The normalisation data for transformation is simply the trend in land use change between the various land use types (and related quality) over the years.

The total land area[40] and land areal[41] of The Netherlands and the EU-15 are presented in table 3.25.

Table 3.25 Total land area and land areal of The Neterlands and EU-15.

County/Region	Total land area[1] [m^2]	Land areal[2] [m^2]
The Netherlands	3.73 E+10	3.39 E+10
EU-15	3.24 E+12	3.13 E+12

1 Including inland water bodies. Reference: [EAA, 1995].
2 Excluding inland water bodies. References: [EAA, 1995], [FAO, 2001].

Normalisation for the Netherlands
For the Netherlands, the normalisation data are mainly derived from the national statistics, CBS [CBS, 2000a]. For the various types of agriculture data from the Agricultural Economic Insitute are used [LEI, 2000]. An overview of the normalisation data for the occupation of land in the years 1989, 1993 and 1996 is given in table 3.26a at the end of this section. For transformation the data available is less detailed, and less detailed normalisation data are thus derived (table 3.26b).

Normalisation for the EU-15
The normalisation data for EU-15 are derived from [EAA, 1995, 1998]. In table 3.27a the normalisation data for occupation are given for the reference years 1970, 1980 and 1990. The transformation data are illustrated in table 3.27b. The total area of the illustrated categories corresponds with the land areal (see table 3.25, land area excluding inland water bodies). Hence, the built-up area corresponds with the categories living, mineral mining, building-sites for industry, industrial sites, parks and public gardens, sport grounds and other building area, defined for the Dutch situation. In order to determine the normalisation data for the reference years, in some cases extrapolations had to be carried out.
In [EAA, 1995] the total area of inland water bodies is given. However, it is not possible to determine the area for canals. Since the impact score for canals bodies is zero in the present operationalisation, normalisation of the canals is not taken into account.

[40] The national territories including inland waters. It includes agricultural land, forest land and other areas such as built-up land, inland waters (rivers, lakes, artificial waters, impoundments, coastal lagoons) but excludes "internal" waters such as estuaries and waters lying on the landward side of the "normal base line" along the coast [EAA, 1995].
[41] Excluding the area of the inland water bodies. The definition of inland water bodies generally includes major rivers and lakes [EAA, 1995].

Table 3.25a Normalisation data for the Netherlands (occupation per year).

Category	Land occupation in [m^2.y]			References[1]	CORINE classes[2]
	1989	1993	1996		
Agriculture	2.40E+10	2.38E+10	2.35E+10	1	2.1.1: 1-3 & 41-43; 2.2.1.2; 2.3.1.1-3; 2.4.5
Arable land, of which:	9.24E+09	9.58E+09	9.55E+09	3	2.1.1: 1-3 & 42
- Hemp			1.32E+07	3	2.1.1.42
Horticulture (outside)	1.44E+09	1.27E+09	1.31E+09	3	2.1.1.1-3
Fruit orchard	2.80E+08	2.92E+08	2.77E+08	3	2.2.1
Bulb growing	1.90E+08	2.00E+08	2.17E+08	3	
Plant and tree cultivation[4]	9.48E+07	1.12E+08	1.17E+08	3	
Greenhouse horticulture	1.56E+08	1.66E+08	1.72E+08	2	
Grassland	1.25E+10	1.17E+10	1.14E+10	3	2.3.1
Organic, of which:			2.30E+08	*4*	
- *Arable land*			8.60E+07	*4*	2.1.1.3
- *Horticulture (outside)*			1.31E+07	*4*	2.1.1.3
- *Fruit orchard*			5.54E+06	*4*	2.2.1.3
- *Grassland*			1.26E+08	*4*	2.3.1.3
Integrated fruit orchard			1.91E+07	*4*	2.1.1.2
Forest, of wich:	3.10E+09	3.11E+09	3.23E+09	1	3.1
- Pine wood			2.14E+09	5	3.1.2
- Leaf wood			1.06E+09	5	3.1.1
- Unexploitable			3.23E+07		
Living	2.10E+09	2.17E+09	2.24E+09	1	1.1: 1, 2 & 4
Mineral mining	6.46E+07	6.62E+07	5.09E+07	1	1.3.1
Building-sites for industry	1.19E+08	1.24E+08	1.18E+08	2	1.2.5
Building area (other)	9.52E+07	1.13E+08	1.17E+08	2	
Industrial sites	4.78E+08	5.26E+08	5.67E+08	1	1.2.1
Parks and public gardens	1.48E+08	1.54E+08	1.58E+08	2	1.4.1
Sports grounds	2.69E+08	2.94E+08	3.09E+08	2	1.4.2
Road transport, of which:	1.15E+09	1.20E+09	1.24E+09	1	1.5
- Road	8.05E+08	8.38E+08	8.65E+08		
- Embankment[5]	3.45E+08	3.59E+08	3.71E+08		
Railway, of which:	8.43E+07	8.37E+07	8.44E+07	1	
- Used railway	1.70E+07	1.75E+07	1.49E+07		1.2.2.4
- Shunt-yard/unused					1.2.2.2
Canals	1.21E+08	1.21E+08	1.21E+08	1	5.1.2.1

1 References: 1. [CBS, 2000a]; 2. [Visser, 2000]; 3. [LEI, 2000]; 4. [RIVM, 2000]; 5. [SBH, 1995].
2 For a list of the CORINE (CoORdination of INformation on the Environment [EEA, 2000]) classes see appendix 6 and 7.
3 According to references [2] and [3]. The value is corrected by a factor 1.2 to correspond with the CBS data.
4 Plantings and seedlings, excluding 1E+7 seed growing.
5 Embankment is 30% of road network.

Table 3.25b Normalisation data for the Netherlands (transformation per year).

Category	Land transformation in [m²]		References[1]	CORINE classes[2]
	1989-1996	1993-1996		
Agriculture	-6.04E+07	-6.18E+07	1	2.1.1: 1-3 & 41-43; 2.2.1.2; 2.3.1.1-3;
Arable land, of which:	3.90E+07	-6.00E+06	3	2.4.5
Horticulture (outside)	-1.65E+07	9.00E+06	3	2.1.1: 1-3 & 42
Fruit orchard	-3.00E+05	-3.60E+06	3	2.1.1.1-3
Bulb growing	3.45E+06	4.20E+06	3	2.2.1
Plant and tree cultivation[4]	1.35E+08	2.65E+08	3	
Greenhouse horticulture	1.95E+06	1.50E+06	2	
Grassland	-1.31E+08	-7.50E+07	3	2.3.1
Forest	1.69E+07	3.13E+07	1	3.1
Living	1.76E+07	1.85E+07	1	1.1: 1, 2 & 4
Mineral mining	-1.71E+06	-3.81E+06	1	1.3.1
Building-sites for industry	-1.25E+05	-1.50E+06	2	1.2.5
Building area (other)	2.73E+06	1.00E+06	2	
Industrial sites	1.12E+07	1.02E+07	1	1.2.1
Parks and public gardens	1.25E+06	1.00E+06	2	1.4.1
Sports grounds	5.00E+06	3.75E+06	2	1.4.2
Road transport, of which:	1.08E+07	9.67E+06	1	1.5
- Road	7.57E+06	6.77E+06		
- Embankment[5]	3.24E+06	2.90E+06		
Railway, of which:	-2.59E+05	-4.96E+05	1	
- Used railway	1.13E+04	1.65E+05		1.2.2.4
- Shunt-yard/unused	-2.70E+05	-6.61E+05		1.2.2.2
Canals	0.00E+00	0.00E+00	1	5.1.2.1

1 References: 1. [CBS, 2000a]; 2. [Visser, 2000]; 3. [LEI, 2000]; 4. [RIVM, 2000]; 5. [SBH, 1995] For a list of the CORINE (CoORdination of INformation on the Environment [EEA, 2000]) classes is referred to appendix 6.

3 According to references [2] and [3]. The value is corrected by a factor 1.2 to correspond with the CBS data.

4 Plantings and seedlings, excluding 1E+7 seed growing.

5 Embankment is 30% of road network.

Table 3.26a Normalisation data for the EU-15 (occupation per year).

Category	Land use in [m^2.y]			References[1]	CORINE classes[2]
	1970	1980	1990		
Agricultural land, of which:	1.59E+12	1.53E+12	1.48E+12	1	
- Arable land	7.49E+11	7.32E+11	7.22E+11	1,2	
- Permanent crops	1.95E+11	1.78E+11	1.74E+11	1,2	
- Permanent meadows and pastures	6.44E+11	6.20E+11	5.88E+11	1	
Wooded area (forest)	1.07E+12	1.11E+12	1.12E+12	1	3.1
- Pine wood	5.62E+11	5.77E+11	5.81E+11	1	3.1.2
- Leaf wood	2.27E+11	2.35E+11	2.39E+11	1	3.1.1
- Unexploitable	2.83E+11	2.97E+11	3.01E+11	1,3	
Built-up[3]	1.85E+11	1.97E+11	2.07E+11	1,2	
Road transport, of which:	2.99E+10	3.35E+10	3.81E+10	1	1.5
- Road	2.09E+10	2.34E+10	2.67E+10		
- Embankment	8.97E+09	1.00E+10	1.14E+10		
Railway, of which:	6.03E+09	5.85E+09	5.66E+09	1	
- Used railway	5.13E+09	4.97E+09	4.81E+09		1.2.2.4
- Shunt-yard/unused	9.05E+08	8.77E+08	8.48E+08		1.2.2.2

1 References: 1. [EAA, 1995]; 2. [EAA, 1998]; 3. [UN/ECE, 1997].
2 For a list of the CORINE (CoORdination of INformation on the Environment [EEA, 2000]) classes is referred to appendix 6.
3 Built-up area includes living, mineral mining, building-sites for industry, industrial sites, parks and public gardens, sport grounds and other building area.

Table 3.26b Normalisation data for the EU-15 (transformation per year).

Category	Land use in [m^2.y]		References[1]	CORINE classes[2]
	1970-1990	1980-1990		
Agricultural land, of which:	-5.22E+03	-4.68E+03	1	
- Arable land	-1.36E+03	-1.06E+03	1,2	
- Permanent crops	-1.05E+03	-3.91E+02	1,2	
- Permanent meadows and pastures	-2.81E+03	-3.23E+03	1	
Wooded area (forest)	2.46E+03	1.20E+03	1	3.1
- Pine wood	9.51E+02	4.22E+02	1	3.1.2
- Leaf wood	6.10E+02	4.36E+02	1	3.1.1
- Unexploitable	9.00E+02	3.46E+02	1,3	
Built-up[3]	1.12E+03	1.08E+03	1,2	
Road transport, of which:	4.12E+02	4.66E+02	1	1.5
- Road	2.88E+02	3.26E+02		
- Embankment	1.23E+02	1.40E+02		
Railway, of which:	-1.88E+01	-1.91E+01	1	
- Used railway	-1.60E+01	-1.62E+01		1.2.2.4
- Shunt-yard/unused	-2.82E+00	-2.87E+00		1.2.2.2

1 References: 1. [EAA, 1995]; 2. [EAA, 1998]; 3. [UN/ECE, 1997]
2 For a list of the CORINE (CoORdination of INformation on the Environment [EEA, 2000]) classes is referred to appendix 6.
3 Built-up area includes living, mineral mining, building-sites for industry, industrial sites, parks and public gardens, sport grounds and other building area.

For the land use by roads and railways additional estimations and calculations had to be performed in order to arrive at more specific normalisation data, to fit the impact assessment data available. These situations are described below.

<u>Roads</u>

For roads, the embankments have an environmental quality that is sometimes even higher than the land surrounding the road. Therefore, road embankments are treated separately. A case study on a Dutch highway [Beetstra, 1998a] states that the total highway width is 48m broad, of which the embankment is 16m broad. The embankment is thus 33% of the road area. For smaller roads, this percentage will be somewhat lower. Therefore an average percentage of 30% is assumed for road embankments.

The total land use by the Dutch road network (both paved and halfpaved as well as unpaved) is given in table 3.25 [CBS, 2000a]. The calculated average width is about 11 m (3m embankment). This figure includes motorways, main or national roads, secondary or regional roads and other roads. The figures on land use by both road and embankment are presented in table 3.29.

For the EU, the total road network length is available for the reference years 1970, 1980 and 1990[EAA, 1995].Using the average width of the Dutch road network (table 3.28) and embankment (table 3.29), the land use is calculated. These land use figures for the EU are given in table 3.30.

Table 3.28 Land use by the Dutch road network and length of this network [CBS, 2000a].

Year[1]	Area roads (hard) [m^2]	Area roads (halfhard and unhardened) [m^2]	Area total road network [m^2]	Length of road network [km]	Calculated average width [m]
1979	9.27 E+10	1.85 E+10	1.11 E+11	1.07 E+5 [2]	10.4
1981	9.55 E+10	1.84 E+10	1.14 E+11	1.07 E+5 [2]	10.7
1989	1.07 E+11	8.11 E+10	1.15 E+11	1.02 E+5	11.3
1993	1.10 E+11	1.01 E+10	1.20 E+11	1.07 E+5	11.2
1996	1.13 E+11	1.04 E+10	1.24 E+11	1.13 E+5	10.9

1 Date set is 1st of January.

2 Assumed to be equal to the figure for the Netherlands in the year 1980 in [EAA, 1995].

Table 3.29 Land use by road embankment, assuming 30% of road network.

Year	Average width of road [m]	Average width of embankment [m]	Land use by road [m^2]	Land use by embankment [m^2]
1979	7.3	3.1	4.77E+8	3.34E+08
1981	7.5	3.2	4.88E+8	3.41E+08
1989	7.9	3.4	4.93E+8	3.45E+08
1993	7.8	3.4	5.13E+8	3.59E+08
1996	7.6	3.3	5.30E+8	3.71E+08

Table 3.30 Length, width and land use of the road network and embankment in EU-15.

Year	Length of road network[1] [km]	Average width of road[2] [m]	Average width of embankment[2] [m]	Land use by road [m^2]	Land use by embankment [m^2]
1970	2.87^{E}+6	7.3	3.1	2.09^{E}+10	8.97^{E}+09
1980	3.18^{E}+6	7.4	3.2	2.34^{E}+10	1.00^{E}+10
1990	3.39^{E}+6	7.9	3.4	2.67^{E}+10	1.14^{E}+10

1 [EAA, 1995].
2 Derived from calculated average width of Dutch road and embankment (see table 3.26). The width in 1970 is assumed to be equal to the Dutch figure in 1979. The width in 1980 is an average of the Dutch figures in 1979 and 1981. The width in 1990 is calculated by extrapolation of the Dutch figure in 1989 to 1990 during the period 1989-1993.

<u>Railways</u>
The occupied land area for railways (including shunt-yard and unused railways) is given in [CBS, 2000a]. Shunt-yards and unused railways result in a very different impact score for land use, and are therefore distinguished for the normalisation data (see table 3.14 in paragraph 3.5.1, repeated below). The contribution of the shunt-yards and unused railways is 17% of the total land use in the Netherlands. For the EU-15, a contribution of 15% is assumed.

In table 3.31 the length of the railway network and land use by both this network and the shunt-yards and unused railways in the EU-15 is given. The land use by the railway network is calculated by using an average width of 30m (see table 3.14).

Table 3.14 Railway transport data for the reference years 1988, 1992, 1995 [CBS, 1997b, 2000a, 2001ab].

Year	Transport performance Freight[1] [tkm/y]	Total land use railways[2] [mln m^2]	Length railway[3] [km]	Width railway[4] [m]	Land use Rail Performance[5] [mln m^2]	Land use shunting-yard / unused [5] [mln m^2]
1988	1049	101.3	2810	30 ± 0.4	84.3	17.0
1992	966	101.2	2791	30 ± 0.4	83.7	17.5
1995	721	99.2	2813	30 ± 0.4	84.4	14.9

1 [CBS, 1997b, 2001b].
2 [CBS, 2000a].
3 [CBS, 2001a].
4 [Lengkeek, 1995].
5 Calculated.

Table 3.31 Length of railway network and land use by this network in EU-15.

Year	Length of railway network[1] [km]	Land use by railway network[2] [m^2]	Land use by shunt-yards and unused railways[3] [m^2]
1970	1.71^{E}+5	5.13^{E}+9	9.05^{E}+8
1980	1.66^{E}+5	4.97^{E}+9	8.77^{E}+8
1988	1.61^{E}+5	4.84^{E}+9	8.54^{E}+8
1990	1.60^{E}+5	4.81^{E}+9	8.48^{E}+5

1 [EAA, 1995].
2 Assuming a width of 30 m.
3 Assuming a contribution of 15% for the shunt-yards and unused railways to the total land use by railways.

3.11 Concluding the inventory data collection

From the above paragraphs, two tables with an overview of the results are extracted (3.32 and 3.33). It appeared possible to gather a broad set of data such as this, although it took about a man-year of time. It is clear that single LCA practitioners can not be expected to collect such data alone, for each case study. Therefore, these data should be incorporated in appropriate, publicly available databases.

Table 3.32 Land occupation for various processes.

Process type	Situation during occupation	Land occupation [m^2.y/t]	
		value	Range[42]
Sand extraction NL	Mud & water		
- **Raising sand**		**1.56**	**± 1.52**
- **Concrete/brick sand**		**0.89**	**± 0.56**
- **Limestonebricksand**		**0.22**	**± 0.14**
Gravel extraction NL	Mud & water	**0.73**	**± 0.45**
Clay extraction NL	Mud & water		
- **Forelands clay**		**0.13**	**± 0.12**
- **Polderclay**		**1.13**	**± 1.09**
Limestone extraction NL	Mud & water	**0.23**	
Kaoline extraction			
- Temperate	Mud & water	**26**	**± 1**
- Tropical	Mud & water	**0.7**	
Wood extraction			
- Spruce (Picea Abies) Scandinavia	Temperate / boreal forest	**4'300**	**± 1000**
- Pine (Pinus) Scand.	T/b forest	**3'900**	**± 1000**
- Spruce Eastern Europe	T/b forest	**22'000**	**±12'000**
- Spruce Central-Western Europe	Temp/mixed forest	***7'600***	***± 8'000***
- Spruce North America	T/b forest	**4'800**	**± 2'400**
Coniferous wood for NL	T/b forest	**9'200**	**± 4'400**
- Oak Europe	Temp/mixed f.	***1'800***	***± 2'000***
- Oak America	Temp/mixed f.	**980**	**± 600**
non-trop.hardwood for NL	Temp/mixed f.	**1'700**	**± 300**
- Meranti S.E. Asia	Tropical forest	**28'000**	**±16'000**
- Merbau S.E. Asia	Tropical forest	**22'000**	**± 4'000**
- Iroko Africa	Tropical forest	**22'000**	**± 9'600**
- Azobé Western Africa	Tropical forest	**3'900**	**± 2'400**
- Amburana S.America	Tropical forest	***17'000***	***±17'000***
- Wane S.America	Tropical forest	***24'000***	***± 24'000***
Tropical hardwood for NL	Tropical forest	**21'000**	**± 12'000**
Bauxite (Al ore) extraction	Mud & water	**0.2**	**0.18 – 0.72**
Iron ore extraction	Mud & water	**0.11**	
Manganese (Mn) ore extraction	Mud & water	**4.2**	
Copper ore extraction	Mud & water	**1.1-2.2**	
Open pit metals mining	Mud & water	**1.1-2.2**	
Shaft metals mining	Mud & water	**0.07**	

[42] If not given, taken to be 5 in last digit.

Table 3.32 Continued
Land occupation for various processes.

Process type	Situation during occupation	Land occupation [$m^2.y/t$]	
		value	Range[43]
Industrial production NL	Sealed surface & park	**1.9**	
Bulk production	Sealed surface	**0.008**	
Bayer process (Al-production)	Red mud reservoir	**0.16**	
Transhipment/stocks	Sealed surface	**1.0**	
Road transport NL - Direct land use - For fuel & cap. goods	 Sealed surface	[$m^2.y/tkm$] **0.0059**	
Railway transport NL - Direct land use - For fuel & cap. goods	Interrupted surface	[$m^2.y/tkm$] 0.008-0.020	
Canal transport NL - Direct land use - For fuel & cap. goods	 Water	[$m^2.y/tkm$] **0.020**	 ± **0.010**
Electricity production - from gas (EU) - from gas (NL) - from h.coal (EU) - from h.coal (NL) - from nuclear (EU) - from oil (EU)		[$m^2.y/TJ_{el}$] 53 23 5.7 3.8	
Living NL[44]	Sealed surface & park	**5.9**	
Waste disposal NL - burn chemical waste - dump building waste - dump general waste	 Sealed surface Dump site Dump site	 **0.59** **0.42** **1.9**	 ± **0.4**

[43] If not given, taken to be 5 in last digit.
[44] Average for NL: 352 m2.y/house, including living facilities in the district. Divided by 60 t (weight of an average house [Beetstra, 1998]) to compare with building materials.

Table 3.33 Land transformation for various processes.

Process type	Situation before transformation	Situation after transformation	Land transformation [m²/t]	
			Value	range[45]
Sand extraction NL				
- **raising sand**	Agriculture	Water (nature/recreation)	0.04	± 0.04
- **concrete/brick sand**	Agriculture	Water (nature/recreation)	0.05	± 0.03
- **limestonebricksand**	Agriculture	Water (nature/recreation)	0.02	± 0.015
Gravel extraction NL	Agriculture	Water (nature/recreation)	0.05	± 0.03
Clay extraction NL				
- **forelands clay**	Extensive pasture	Nature	0.126	
- **polderclay**	Intensive pasture	intensive pasture	0	
Limestone extraction NL	Nature	Extensive recreation	0.0083	
	Nature	Sealed surface	0.0013	
Kaoline extraction				
- temperate	Temperate forest	See under bauxite	1.7	1.6 – 1.7
- tropical	Tropical forest	extraction	0.047	
Other surface resources	See under kaoline extraction	See under kaoline extraction	0.03[46]	0.0004 = 0.06[47]
Wood extraction				
- spruce (Picea Abies) Scandinavia	Agriculture	Temperate / boreal forest	0.2	± 0.08
- pine (Pinus) Scand.	Agriculture	T/b forest	0.18	
- Spruce Eastern Europe	Native temp/b. forest	Degraded boreal forest	12	± 5
- Spruce Centr.-W Europe	Agriculture	Degraded temp/mixed f.	0	
- Spruce North America	Native boreal forest	Boreal plantation	0	
Coniferous wood for NL	75% Agriculture 25% Native bor. forest	T/b forest	*13*	*± 40*
- Leafwood Europe	Agriculture	Temperate/mixed forest	0	
- Leafwood America	Agriculture	Temperate/mixed forest	4	± 0.6
non-trop. hardwood for NL			*5*	*± 23*
- Meranti S.E. Asia	Native tropical forest	Degraded tropical f.	240	± 100
- Merbau S.E. Asia	Native tropical forest	Degraded tropical f.	150	± 60
- Iroko Africa	Native tropical forest	Degraded tropical f.	720	± 200
- Azobé Western Africa	Native tropical forest	Degraded tropical f.	21	± 17
- Amburana S.America	Native tropical forest	Degraded tropical f.	8,100	± 4.800
- Wane S.America	Native tropical forest	Degraded tropical f.	12,000	± 6.700
tropical hardwood for NL	Native tropical forest	Degr. trop. forest	1,400	± 400
Bauxite (Al ore) extraction	64%: no transformation 15% natural pasture 7% tropical rainforest 7% tropical rainforest 4% tropical rainforest 2% scrubland 1% agriculture 0.2% wetland 0.2% wetland	 15% livestock pasture 7% native forest 7% scrubland 4% commercial forest 2% urban area 1% urban area 0.2% recreational area 0.2% livestock pasture	0.6	0.2 – 1.0

45 If not given, taken to be 5 in last digit.
46 Per meter layer thickness.
47 Per meter layer thickness.

Table 3.33 Continued
Land transformation for various processes.

Process type	Situation before transformation	Situation after transformation	Land transformation [m²/t]	
			Value	range[48]
Iron ore extraction			0.11	
Manganese (Mn) ore extraction			0.42	
Copper ore extraction			0.11	0.07 – 0.15
Open pit metals mining			0.11	0.07 – 0.15
Shaft metals mining			0.08	0.05 – 0.1
Industrial production NL	2%: no transformation 87% agriculture 3.7% water 3.5% nature 2.2% forest 1.8% recreation	 Sealed surface & park Sealed surface & park Sealed surface & park Sealed surface & park Sealed surface & park	0.041	± 0.007
Bayer process (Al-production)	73% forest 15% natural pasture	Red mud reservoir; some to specific agriculture	0.13	± 0.01
Road transport NL - Direct land use - for fuel & cap. goods	95% agriculture 2.8% recreation 2.7% forest	 Sealed surface Sealed surface	0.00002	0.00002 – 0.00003
Railway transport NL - direct land use - for fuel & cap. goods	 Agriculture	 Interrupted surface Sealed surface	???	
Canal transport NL - direct land use - for fuel & cap. goods	 Agriculture	 Water Sealed surface	???	
Electricity production - from gas (EU) - from gas (NL) - from h.coal (EU) - from h.coal (NL) - from nuclear (EU) - from oil (EU)			[m²/TJel] 1.75 0.8 0.2 0.1	
Living NL[49]	96% agriculture 4% recreation	Sealed surface & park	0,063	0,060-0,065
Waste disposal NL - burn chemical waste - dump building waste - dump general waste	 Agriculture Agriculture Agriculture	 Sealed surface Dump site Dump site	 ??? negative negative	

The format to include these land use interventions is:

Occupation ([type], [further specification], [country code]);
Transformation ([type initial], [further specification]>[type final], [further specification], [country code])

See annex 7 for the format applied to all the land use types distingushed in this study. These will cover a vast amount of the land use types to be encountered in LCA.

[48] If not given, taken to be 5 in last digit.
[49] Average for NL: (3.6 - 3.9) m2.y/house, including living facilities in the district. Divided by 60 t (weight of an average house [Beetstra, 1998]) to compare with building materials.

4 Data collection: land use impacts

………………………………………………………………

In this chapter, the data required for the impact assessment is discussed and shown as an overview. The actual spatial differentiated data is compiled in a spreadsheet. See for an explanation of the methodology chapter 2 or annex 1.

4.1 General overview on the impact assessment of land use in LCA

For land use impact assessment, 4 types of biodiversity impact score may result, based on biome level and species density level factors (see chapter 2 and annex 1), depending on the type of intervention:

EO:	Ecosystem occupation	(x [m^2.y])*
ER:	Ecosystem renaturation	(x [m^2.y])*
ET:	Ecosystem transformation	(x [m^2]) *
ETR:	Ecosystem transformation due to renaturation	(x [m^2])*

* The impact factors themselves have no units; the unit after calculation of the impact scores is determined by the intervention type (m^2.y or m^2).

The impact scores EO and ER are in principle additive, resulting in a total occupation score. The same goes for ET and ETR. The reason not to add them up may be to show what happens if in the modelling, uncertain future renaturation is not included (excluding ER and ETR). Including or leaving out the renaturation data simplifies the presentation of the results (and its calculation), and is allowed when this choice is discribed, argumented and its consequences explained.

Within the EO and ER scores, there is also the distinction between using an average or a maximum reference. This requires the labels $_{av}$ and $_{max}$, and may result in two additional scores when both are shown. The scores based on a maximum reference relates to not including a background level and allows for a clear interpretation of occupation: preventing relaxation of land to a natural equilibrium. The score based on an average reference relates to comparison with the average land use quality in a region (every land quality higher than the –human induced- average is considered good for the environment). The maximum reference data seems subject to species density inventory artifacts, such as one peak in the whole region, or a better species mapping programme (allowing for more detailed assessments, resulting in higher maxima). Showing calculations with both references is of course preferable for giving insight and decision support.

For the life support indicator NPP, similar groups of impact scores are distinguished, depending on the inventory data format:

LO:	Life support occupation	([g/m^2.y] x [m^2.y])
LR:	Life support occupation during renaturation	([g/m^2.y] x [m^2.y])
LT:	Life support transformation	([g/m^2.y] x [m^2])
LTR:	Life support transformation due to renaturation	([g/m^2.y] x [m^2])

Of these, also LO and LR are additive, leading to total occupation scores, as are LT and LTR. For occupation (and renaturation) there is a reference, but here only an average reference is considered (as biomass data seem even more variable than species density data and average data are the most robust). The difference with using a maximum reference value will have similar implications as for the biodiversity scores. This implies however, that the biomass scores for occupation can only be consistently compared to the average biodiversity scores (within the same value system[50]).

[50] Remember that using a maximum reference implies assuming that all indicator scores below this maximum is ‘bad for the environment’. Using an average reference implies ‘doing better than the average in a region’.

A table with the resulting impact scores for the various processes is given in annex 7. The results of applying the impact assessment to case studies are discussed in chapter 5.

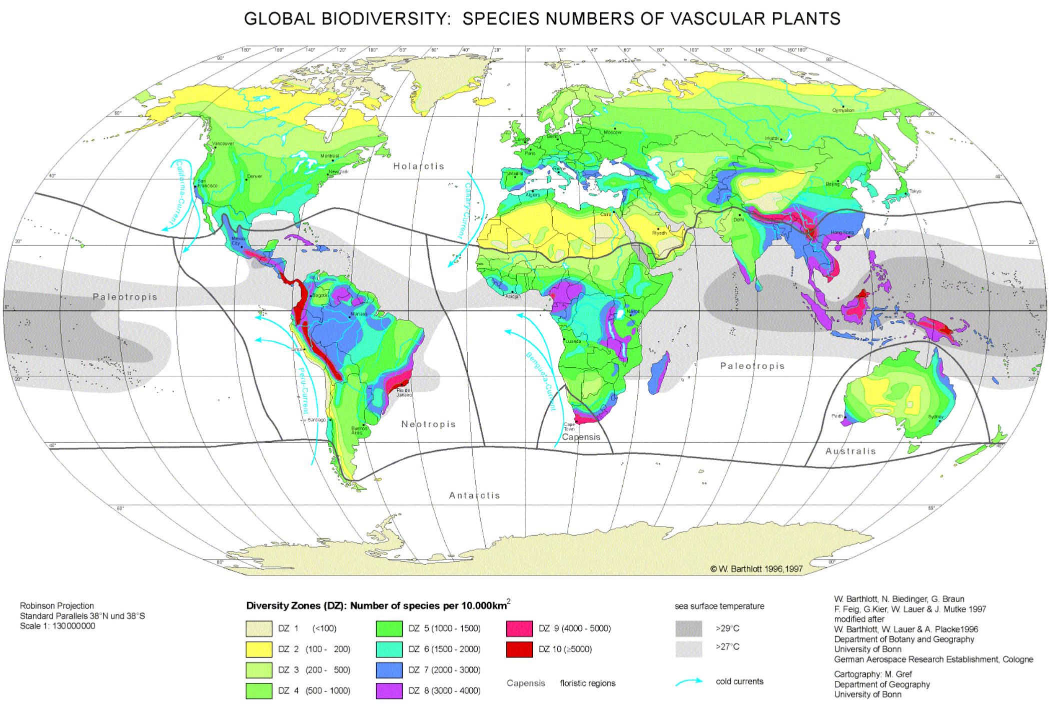
4.2 Data collection & treatment strategy

4.2.1 Approach – Biodiversity

Reference state

For the reference state of species diversity two sources of global species diversity are readily accessible: [Barthlott, 1997] and [Lindeijer et al., 1998]. The latter is expressed in terms of alpha (see annex 1 for the formula used), and can not easily be translated to the standardised area -0.01 ha- chosen in this project (see also annex 1 for the standardisation formula). Therefore, and because Barthlott is planning to publish new data based on the same standardisation formula as used by [Köllner, 2001], the first data source is the guideline for reference data on the species level in this research project. Data have been extracted manually from this map to arrive at regionally differentiated reference species densities. An overview of these species diversity data is illustrated in figure 4.1. For the standardisation procedure to 0.01 ha is referred to annex 1.

Figure 4.1 Species diversity on a global level (S/10.000 km^2), from [Barthlott, 1997].



On the ecosystem level there are various models available (see a discussion in [Weidema, 2001]), which give different results. As for calculationg relative impact scores consistency is more important than to have a 'true' model or absolute figures. For this reason, the IMAGE model 2.1 is chosen to assess areas of potential vegetation and actual land cover [Leemans et al., 1998], in accordance with [Weidema, 2001]. In addition, the advantage of using this model is that it includes consistent data on NPP as well. A thorough scientific review is performed for this large climate change model from RIVM. An overview of the biome data is given in the two maps below (figures 4.2 and 4.3).

The data behind these maps have been extracted from the CD-ROM of [Leemans et al., 1998] and have been used to derive the biome area data for the impact assessment. For a more detailed discussion and the quantitative backgrounds is referred to annex 1. These ecosystem-level data are the most important regional differentiation in our land use model.

Figure 4.2 Potential land cover of biomes in 2000 according to [Leemans et al., 1998].

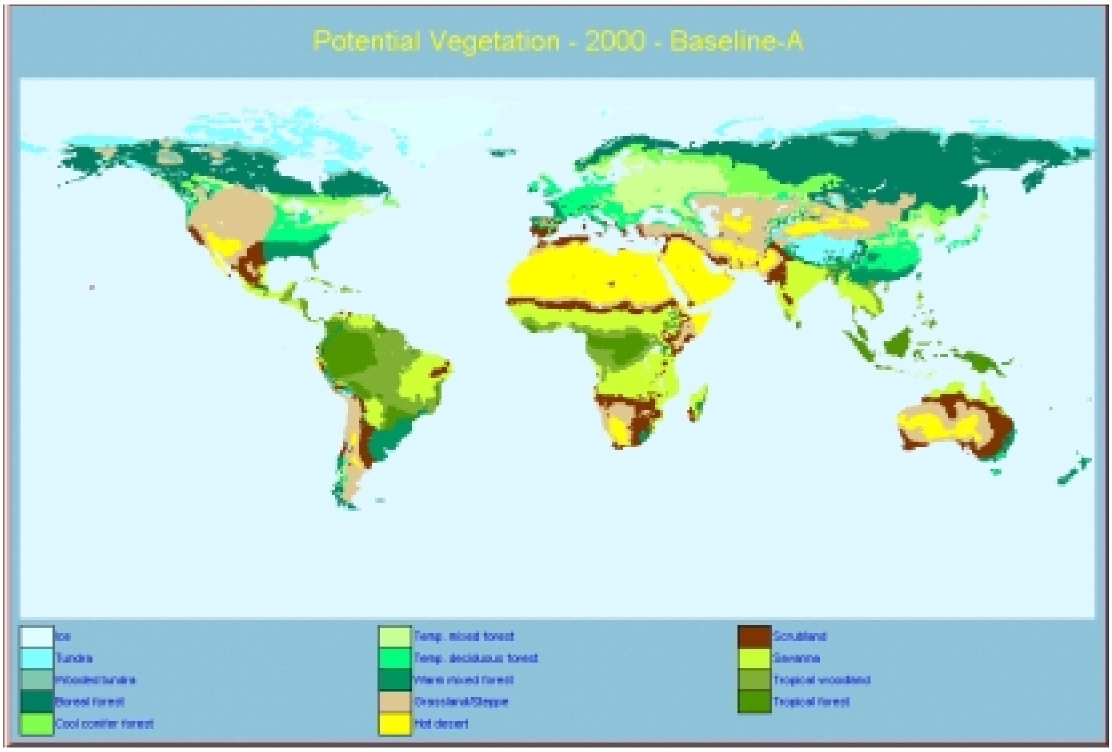
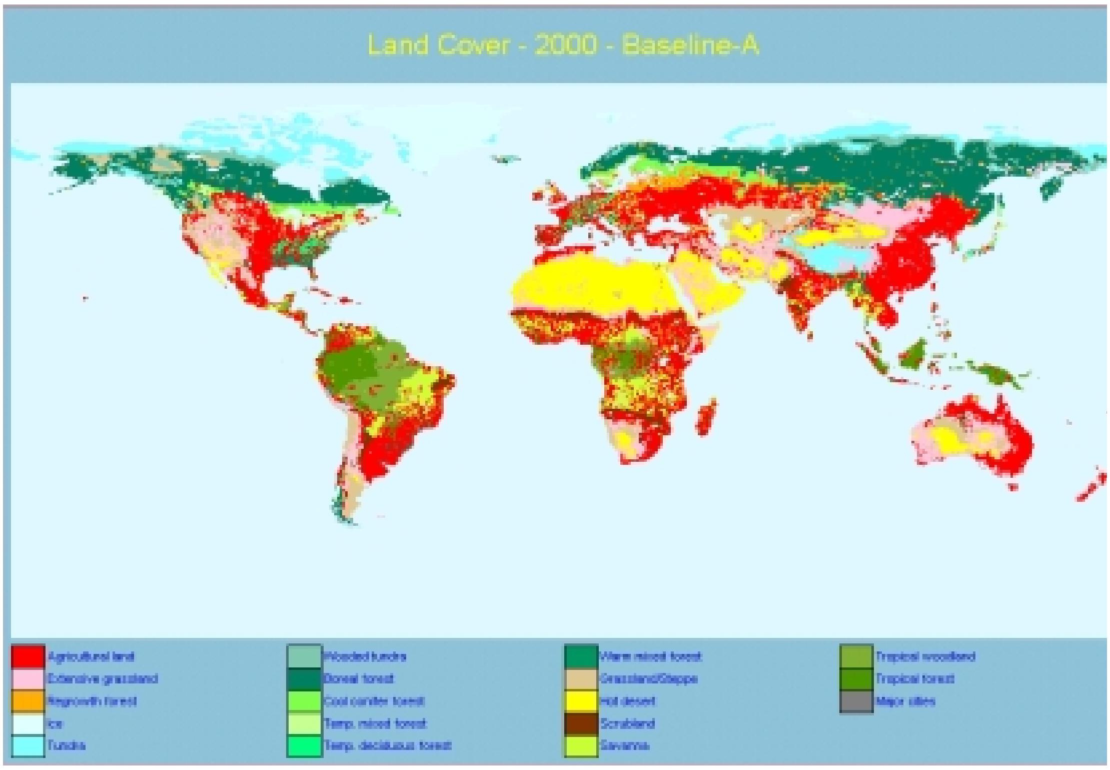


Figure 4.3 Actual land cover in 2000 according to [Leemans et al., 1998].



The data calculated for the reference state based on above maps are given in table 4.1a.

Table 4.1a Data on biome level ([Leemans et al., 1998], [Barthlott, 1997]).

Biome	Potential biome area [km²]	Actual area [km²]	Ecosys Scarcity	Ecosys Vulner. b = 0.2	Species Density [10'000 km⁻²]	Ecosys Quality	$S_{ref. av.}$ **[0.01ha]**	$S_{ref. max}$ **[0.01ha]**
Ice	6.66E+06	6.68E+06	**3.80**	**1.00**	1	**0**	**0**	**0**
Tundra	9.03E+06	9.54E+06	**2.80**	**0.96**	100 - 200	**1**	**4**	**5**
Wooded tundra	3.08E+06	3.77E+06	**8.22**	**0.85**	200 - 500	**2 - 2.5**	**9**	**13**
Tundra total	1.21E+07	1.33E+06	**4.19**	**0.93**	100 - 500	**1 - 2.5**	**5**	**7**
Boreal forest	2.53E+07	2.07E+07	**1.00**	**1.17**	200 - 1000	**2 - 5**	**15**	**25**
Cool conifer forest	4.72E+06	2.44E+06	**5.36**	**1.70**	500 - 1000	**5**	**19**	**25**
Temp. deciduous forest	6.20E+06	1.01E+06	**4.08**	**4.26**	1000 - 1500	**7.5 – 10**	**31**	**38**
Temp. mixed forest	7.69E+06	1.71E+06	**3.29**	**3.32**	500 - 1500	**5 – 7.5**	**25**	**38**
Warm mixed forest	4.76E+06	8.96E+05	**5.32**	**3.81**	1500 - 3000	**15**	**57**	**75**
Mixed forest total	1.24E+07	2.61E+06	**4.07**	**3.49**	500 - 3000	**5 – 15**	**36**	**51**
Grassland/Steppe	1.72E+07	9.02E+06	**1.47**	**1.67**	200 - 1500	**2 – 7.5**	**21**	**38**
Savanna	1.15E+07	5.39E+06	**2.19**	**1.84**	200 - 3000	**2 – 15**	**40**	**75**
Grassland total	2.87E+07	1.44E+07	**1.77**	**1.74**	200 - 3000	**2 – 15**	**28**	**52**
Hot desert	1.42E+07	1.16E+07	**1.78**	**1.18**	100 - 200	**1**	**4**	**5**
Scrubland	8.12E+06	2.24E+06	**3.12**	**2.80**	500 - 4000	**5 – 20**	**57**	**100**
Tropical woodland	6.79E+06	4.25E+06	**3.73**	**1.45**	1000 - 3000	**10 – 15**	**50**	**75**
Tropical forest	5.74E+06	3.78E+06	**4.41**	**1.40**	1500 - 9000	**15 – 45**	**132**	**226**
Agricultural land		3.15E+07						
Extensive grassland		1.23E+07						
Regrowth forest		3.31E+06						
Major cities		8.32E+05						
Total area	131E+06	131E+06						

In table 4.1b the three ecosystem factors are multiplied, and some weighted average values are given for major areas in the world. These weighted average values have been derived for over 90 countries and (parts of) continents based on an estimation of the % of total country area projected with the biome types according to figure 4.1b (see tables A1.2 and A1.3 in annex 1).

As can be seen from tables 4.1a and 4.1b, high ecosystem scores can generally be explained by a combination of high values for all ecosystem factors. Temperate and tropical forests and shrublands give the highest scores. This seems in accordance with the general opinion. However, the average score for tropical forests is not the highest biome value, which is contrast to general opinion about the value of tropical forests. Temperate deciduous forests and warm mixed forests have about equal to higher ecosystem scores, respectively, because the area left of these biomes is a factor 5 to 6 lower than their potential area. For tropical forests, the actual area is 'still' 66% of the potential area, according to these data. The fact that Western countries have intensively transformed their forest area in the past is thus included in these ecosystem level calculations, and for some biomes dominate the total score over the ecosystem quality in terms of species diversity.

Relative high scores in (parts of) continents are mainly caused by the occurence of biomes with a small potential and/or actual area.

For local scores for specific land use types, mainly the biodiversity data collected by [Köllner, 2000] are used (see tabel A1.1 in annex 1, and table 4.4). For transformations, biodiversity data on the land use type before and after the land use are subtracted from each other before dividing by the initial state (according to formula 14 in annex 1).

Table 4.1b Aggregated ecosystem level factors[1].

Biome	Ecosystem Scarcity	Ecosystem Vulnerability b = 0.2	Ecosys Quality	EsxEVxEQ	
				min	Max
Ice	3.80	1.00	0	0	0
Tundra	2.80	0.96	1	3	3
Wooded tundra	8.22	0.85	2 - 2.5	14	17
Tundra total	4.19	0.93	1 - 2.5	6	7
Boreal forest	1.00	1.17	2 - 5	2	6
Cool conifer forest	5.36	1.70	5	45	45
Temp. deciduous forest	4.08	4.26	7.5 – 10	174	220
Temp. mixed forest	3.29	3.32	5 – 7.5	55	82
Warm mixed forest	5.32	3.81	15	304	304
Mixed forest total	4.07	3.49	5 – 15	140	158
Grassland/Steppe	1.47	1.67	2 – 7.5	5	19
Savanna	2.19	1.84	2 – 15	8	61
Grassland total	1.77	1.74	2 – 15	6	34
Hot desert	1.78	1.18	1	2	2
Scrubland	3.12	2.80	5 – 20	44	175
Tropical woodland	3.73	1.45	10 – 15	54	81
Tropical forest	4.41	1.40	15 – 45	93	278
Europe				58	85
Northern Europe				27	36
Western Europe				151	187
Eastern Europe				56	82
Russian Federation				17	25
Middle-East				10	27
South-East Asia				114	205
Australia				41	93
Africa				32	78
South America				69	138
United States				81	110
Canada				15	21
Temperate zones				48	66
Tropics				41	113

[1] Totals per aggregated biome type are weighted by the relative actual area of each biome.

For road embankments -not included in [Köllner, 2001]- a separate study on different mowing regimes in the Netherlands has been used [van Schaik & van den Hengel, 1994]. For details see section 3.3. For specific Dutch land uses such as aggregate extractions, it was attempted to apply detailed biodiversity data from the Floron database [Groen, 2001], see annex 5 (in Dutch) and section 3.2. Also, expert estimates on numbers of species per 10 ha after aggregate extraction [Helmer, 2000] (Annex 4) have been used. More details are given in paragraph 4.2.

Total impact scores are then calculated by multiplying the biome level scores with the local species density scores, according to the LCIA formula's already given:

Approach – Life support

For the reference state for the biomass indicator a few review sources were available at the time of the literature study: [Lindeijer et al., 1998], [Weidema, 2001] (including references to NPP databases) and

[Leemans et al., 1998]. The latter IMAGE model (version 2.2: [IMAGE Team, 2001][51]) would best be used for the purpose, as it fits the biodiversity modelling from the same integrated assessment model (see figure 4.4a). However, it is preferred to use the data from [Lindeijer et al, 1998] (see figure 4.4b) because there biomass itself is expressed, giving a closer link to the broad scope of life support functions addressed here (see also appendix 1, section A1.4.3).

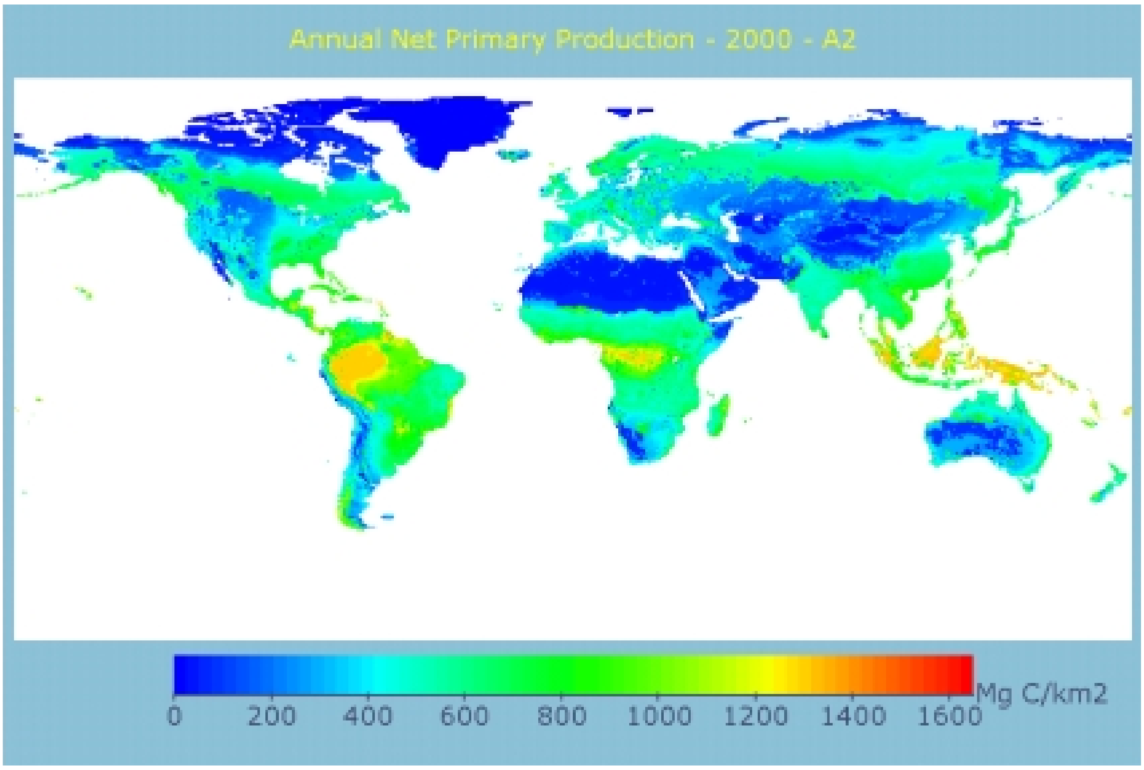


Figure 4.4a Global NPP values as tonnes C per km² per year from [IMAGE TEAM, 2001].

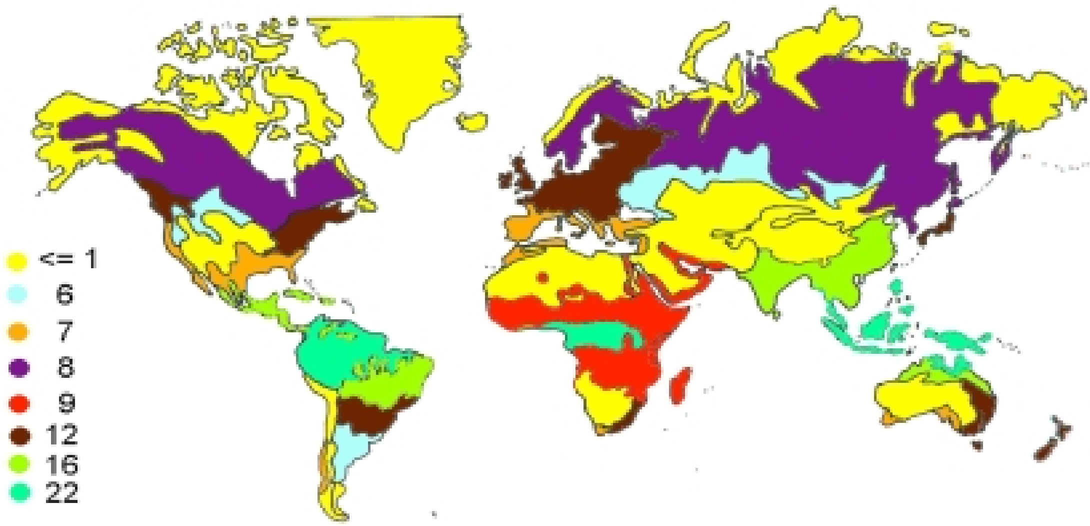


Figure 4.4b Global NPP values as tonnes biomass per hectare per year, from [Lindeijer et al., 1998].

[51] The model improvements compared to version 2.1 [Leemans et al., 1998] are for biomass much larger than for the biome data. Therefore, the biodiversity data were not updated to the 2.2 version. Due to the improvements in the biomass map in [IMAGE TEAM, 2001], the NPP map now fits better with the biomass map derived in [Lindeijer et al., 1998].

For practical reasons data on NPP values per biome type have been used, from [Ajtay et al, 1979] (cited in [Lindeijer et al., 1998]) to calculate the regional differentiation, to facilitate users in using these data (see table 4.2). Ecosystems not included in the regional differentiation are not in bold. See also the data comparison in section A1.4.3 of annex 1.

Table 4.2 Biome level biomass (NPP) data from [Ajtay et al., 1979].

Biome	NPP [g biomass/m^2.y]
Tundra/alpine total	140
Boreal forest	800
Temp. Deciduous forest	1200
Temp. Mixed forest	1200
Mixed forest total	1400
Grassland/Steppe	600
Savanna	900
Hot desert	90
Scrubland	700
Tropical woodland	1200
Tropical forest	2200
Wetlands	24
Swamps & marshes	30
Lakes/streams	4
Marine (seas)	2.5

For cases, data is scattered and sometimes inconsistent due to different definitions of NPP [Lindeijer et al., 1998]. Most case information are derived from a few studies, but data for agriculture and forestry products can be calculated based on productivity data. More details are given in the various following paragraphs.

4.3 Aggregate extraction (sand, gravel, clay)

4.3.1 Biodiversity impacts of aggregate extraction

During the actual extraction process the biodiversity is generally very low to zero. However, the extraction may end with (or be accompanied by) a renaturation process, in which the species density may recover or even render a higher level than the species density before the extraction began. Especially, this renaturation is important for transformation (the change from before to after a process). In addition, it is also important to determine the biodiversity score during the renaturation process, in as far as it is attributable to the extraction process. This means that biodiversity during renaturation is to be included in an occupation impact (see chapter 2).

For traditional extractions, a species density of 1 species per 0.01 ha is assumed during extraction, in accordance with the score for built up land according to [Köllner, 2001].

For modern aggregate extractions with renaturation, several succession stages can be included in the calculated renaturation[52]. How many are considered depends on whether all these development stages are attributed to the extraction itself. As the responsibilities for the full renaturation are not clearly defined (see chapter 2: sometimes the extractor keeps the land in property, and sometimes it is passed on to a separate nature development organisation), we have considered (1) the case that the whole renaturation until the climax state is attributed to the

[52] In fact most modern aggregate extractions in the Netherlands nowadays start with renaturation already during the extraction process.

extraction, and (2) the case that a single depreciation time is taken to delimit the attributed renaturation time. See table 4.3 for the species density estimates by experts from Stichting Ark (based on annex 4), using species density reference data from the Floron database (see annex 5). The resulting impact scores per tonne of aggregate are also given, based on the inventory data from tables 3.2 and 3.4.

Table 4.3 Species density estimates for modern aggregate extractions including renaturation, and resulting transformation (ETR) and occupation (ER)* renaturation impact scores (including the biome level).

Indyke sand/gravel extraction until clay layer; >> 2 m water, allowing for sedimentation & softwood climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	30 years depreciation ETR [/kt]	Average* maximum* ER [/m^2.kt]
Species density per 0.01 ha	Sini = 31 Sfin =19	8 /19		
Impact score	0.06 ± 0.04	**14 ± 6 26 ± 6**	**4.7**	**2.1 2.2**
Riverbed gravel extraction until water level; periodic flooding, pioneer, grassland (grazing), mixed climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	30 years depreciation ETR [/kt]	Average* maximum* ER [/m^2.kt]
Species density per 0.01 ha	Sini = 31 Sfin = 88	13 / 50 / 88		
Impact score	0.06 ± 0.04	**-25 ± 3 1.0 ± 2.4**	**4.5**	**-0.2 1.4**
Borders for riverbed sand/gravel extraction (dry); pioneer, grassland/bushes (grazing), mixed climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	30 years depreciation ETR [/kt]	Average* maximum* ER [/m^2.kt]
Species density per 0.01 ha	Sini = 31 Sfin = 50	4 / 15 / 50		
Impact score	0.04 ± 0.07	**4.2 ± 4.3 8.8 ± 2.6**	**4.7**	**1.4 2.0**
Clay extraction, undeep; until water level, pioneer, grazing after some years (+ willows), hardwood climax				
	ETR [/kt]	**Average* Maximum* ER [/kt]**	30 years depreciation ETR [/kt]	Average* maximum* ER [/m^2.kt]
Species density per 0.01 ha	Sini = 40 Sfin = 31	19 / 19 / 31		
Impact score	0.04 ± 0.03	**0.6 ± 3.4 10.0 ± 2.6**	**4.6**	**0.8 1.7**
Borders for undeep clay extraction (dry); pioneer, grazing after some years (+ bushes), hardwood climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	30 years depreciation ETR [/kt]	Average* maximum* ER [/m^2.kt]
Species density per 0.01 ha	Sini = 40 Sfin = 38	19 / 19 / 38		
Impact score	0.02 ± 0.05	**-1.5 ± 5.5 9.2 ± 3.4**	**6.0**	**0.8 1.7**

* For renaturation also an average or a maximum reference can be chosen

Table 4.3 Continued

Species density estimates for modern aggregate extractions including renaturation, and resulting transformation (ETR) and occupation (ER)* renaturation impact scores (including the biome level).

Lime pits (dry); pioneer, lime grasslands, mixed climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	*30 years depreciation ETR [/kt]*	*Average* maximum* ER [/m².kt]*
Species density per 0.01 ha	Sini = 39 Sfin = 82	31 / 50 / 82		
Impact score	-0.16 ± 0.05	**-0.5±1.9** **5.7 ± 1.5**	**4.5**	**-1.0** **1.1**
Pond in limepit; pioneer, lime marsh, climax				
	ETR [/kt]	**Average* maximum* ER [/kt]**	*30 years depreciation ETR [/kt]*	*Average* maximum* ER [/m².kt]*
Species density per 0.01 ha	Sini = 39 Sfin = 31	19 / 31 / 31		
Impact score	0.03 ± 0.03	**-8.2 ± 2.8** **1.7 ± 3.4**	**4.6**	**0.3** **1.6**

* For renaturation also an average or a maximum reference can be chosen

From table 4.3 it is visible that including much less of the total renaturation time and –impacts (ETR30y) implies a much higher transformation score (higher environmental impact). This is due to the fact that most of the recovery is not incorporated then. For occupation during renaturation, the situation is more complex. When using a maximum reference the score becomes lower when less renaturation time is included (because land is occupied during less time), but when using an average reference the score becomes higher –or less negative-, because improvement compared to an average reference is often not reached in the shorter renaturation time included.

These calculations illustrate the important value choice concealed in the choice of a reference state, and the necessity to include this in the interpretation of the impact scores.

1.1.2 Life support impacts of aggregate extraction

Like biodiversity, the biomass productivity during an extraction process will be very low to zero. However, the biomass productivity during and after renaturation may increase to a level comparable to or even higher than the original level. Data on standing biomass (terrestrial as well as aquatic) has been estimated by the same renaturation experts as for the data on species density (see annex 4 and section 3.2.3 above). From this data, the data relevant for our cases have been extracted and aggregated to table 4.4 below. In these NPP calculations, peak standing biomass as estimated by the experts is used as NPP value. This is not consistent with the reference NPP data, but is used due to lack of adequate NPP case data. Also, no attempt was made to calculate data when only 30 years of renaturation are included, as this is expected to give the same insight as obtained with biodiversiy (section 4.2.1 above, and chapter 5 and annex 8). As only average data have been determined, no uncertainty ranges could be given either. These will be of the same order of magnitude as the uncertainty in the impact scores for biodiversity.

Table 4.4 NPP estimates based on standing biomass*.

Indyke sand/gravel extraction until clay layer; >> 2 m water, allowing for sedimentation & softwood climax		
	Average LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	150	-66
Riverbed gravel extraction until water level; periodic flooding, pioneer, grassland (grazing), mixed climax		
	LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	-400	-345
Borders for riverbed sand/gravel extraction (dry); pioneer, grassland/bushes (grazing), mixed climax		
	LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	200	214
Clay extraction, undeep; until water level, pioneer, grazing after some years (+ willows), hardwood climax		
	LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	150	-464
Lime pits (dry); pioneer, lime grasslands, mixed climax		
	LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	-250	-128
Pond in limepit; pioneer, lime marsh, climax		
	LTR [m^2/t]	Average LR [m^2.y/t]
Impact score	-100	1187

* Conversion factors from wet to dry biomass used from annex 1 (section A1.4).

The poor data consistency due to lack of appropriate data shows the difficulty in applying this biomass indicator. Using standing biomass as an indicator for life support biomass would have been possible in a consistent manner here. Reference data for standing biomass is also available from [Amthor et al., 1998].

4.4 Impacts from other process types

In 4.4.1 and 4.4.2 details are discussed on the data used to arrive at the biodiversity and life support indicators, respectively. Next to these general data collection issues, for some specific land use types as distinguished in LCA additional choices need to be made. These are discussed in 4.4.3.

4.4.1 Biodiversity impacts of other process types

The main source of data for relative biodiversity scores to be used in LCA are derived from [Köllner, 2001]. Köllner has collected data on species numbers per unit area for 31 land use types (table 7-5 in [Köllner, 2001], see also table A1.1 in annex 1), based on in total 3700 cases from 16 sources in Switzerland and Germany. This data is standardised to a unit area of 0.01 ha according to the best fit following the Arrhenius relationship between species and area. The reference species density data from [Barthlott, 1997] are also standardised to this unit area, allowing for a globally consistent relative score for biodiversity. For details see annex 1. An overview of the data applied is shown in table 4.5.

Table 4.5 Biodiversity data for several land use types.

CORINE nr.	Land type	S 0.01 ha	st.dev	EOav	St.dev	EOmax	st.dev
	Individual types						
1.1.1	Continuous urban	**8**	**± 4**	**57**	±28	**60**	±30
1.1.2	Discontinuous urban	**22**	**± 9**	**32**	±13	**40**	±16
1.1.3	Urban fallow	**40**	**±13**	**0**	±0	**14**	±5
1.1.4	Rural settlement	**25**	**± 5**	**27**	±5	**36**	±7
1.2.1	Industrial area vegetation	**24**	**±13**	**28**	±15	**37**	±20
1.2.2.2	Rail fallow	**24**	**± 6**	**28**	±7	**37**	±9
1.2.2.4	Rail embankments	**32**	**±13**	**14**	±6	**26**	±10
1.2.5	Industrial fallow	**40**	**±11**	**-1**	±0	**14**	±4
1.3.4	Mining fallow	**38**	**± 6**	**3**	±0	**17**	±3
1.4.1	Green urban	**29**	**±15**	**19**	±10	**30**	±15
1.4.2	Sport facilities	**7**	**± 3**	**59**	±25	**61**	±26
1.5	Build-up land**)	**1**		**69**	±0	**70**	±0
2.1.1.1	Conventional arable	**10**	**± 6**	**53**	±32	**57**	±34
2.1.1.2	Integrated arable	**7**	**± 5**	**58**	±42	**61**	±44
2.1.1.3	Organic arable	**26**	**±10**	**25**	±10	**34**	±13
2.1.1.41	Fibre/energy crop kenaf	**10**	**± 5**	**53**	±27	**57**	±28
2.1.1.42	Fibre/energy crop hemp	**11**	**± 4**	**52**	±19	**56**	±20
2.1.1.43	Fibre/energy crop Chinese reed	**15**	**± 8**	**45**	±24	**50**	±27
2.2.1.2	Organic orchard	**23**	**± 7**	**29**	±9	**38**	±12
2.3.1.1	Intensive meadow	**17**	**± 9**	**41**	±22	**47**	±25
2.3.1.2	Less intensive meadow	**19**	**±11**	**36**	±21	**44**	±26
2.3.1.3	Organic meadow	**45**	**±20**	**-10**	±4	**7**	±3
2.4.5	Agriculture fallow + hedgerow	**53**	**± 6**	**-24**	±3	**-4**	±0
3.1.1.2	Broad-leafed forest (moist)[53]	**24**	**±10**	**29**	±12	**37**	±15
3.1.1.2	Broad-leafed forest (arid)[4]	**23**	**±10**	**31**	±13	**38**	±17
3.1.4	Forest edge	**48**	**±14**	**-14**	±4	**3**	±1
3.2.1	Natural grassland	**39**	**±15**	**1**	±1	**16**	±6
3.2.2	Heathland	**18**	**± 4**	**40**	±9	**46**	±10
3.2.4	Hedgerows	**44**	**±22**	**-9**	±4	**9**	±4
4.1.2	Peatbog	**19**	**±10**	**38**	±20	**44**	±23
	ETH database						
-*	Ecoinvent II			**29**	±12	**37**	±15
-*	Ecoinvent III (excluding green urban, rail embankments, rural settlements.)			**53**	±32	**57**	±34
-*	Ecoinvent IV			**69**	±0	**70**	±0

* Local biodiversity scores different from those of [Köllner, 2001] because the EDP-value for category II in [Köllner, 2001] was higher than that of category III (contrary to common sense) and the calculation prcedure to arrive at these values for categories II, III and IV were unclear.

These data may be applied in relative scores for most land use types in Europe, if more specific data is unavailable. Beyond this delimitation, the Köllner data can be used as a first approximation to the local impact of the various process types, if no other sources are available.

Road embankments

For roads, Köllner had not determined a vascular plant species density, although road embankments do show various plant species. The amount of plant species on highway embankments is measured over 10 years between 1982 and 1992 on 20 patches of 10 x 30 meters [van Schaik & van den

[53] Semi-natural forest

Hengel, 1994], starting from standardised management (mowing twice a year). Different mowing regimes were applied from 1982 on, of which 'No management' rendered the lowest species diversity (both weighted and unweighted). Unshaded areas showed the highest species density, and areas on clay showed a higher species density than on sandy soil. Species density weighted according to the Shannon-index (including population per specie [Pielou, 1996]) showed the same pattern. Selected results are shown in table 4.6 for both unweighted species density, and the standardised area (using the Arrhenius formula 6b, see annex 1).

Table 4.6 Unweighted species density on highway embankments in the Netherlands, based on [van Schaik & van den Hengel, 1994].

	Clay soil				Sandy soil			
	2x mowing/year		no management		2x mowing/year		no management	
	per 300 m^2	**per 0.01 ha**	**per 300 m^2**	**per 0.01 ha**	**per 300 m^2**	**per 0.01 ha**	**per 300 m^2**	**per 0.01 ha**
Unshaded	21±7	**17±6**	11±3	**9±2**	13±3	**10±2**	4±2	**3±2**
Shaded	17±5	**14±4**	10±2	**8±2**	8±5	**6±4**	7±3	**6±2**

Forests

For forests, there is little quantitative data on species density differences between types of forests. In [Köllner, 2001] only data for arid (>50% beech) and moist (alder, oak & birch) broadleaved semi-natural forests in is given (see table 4.4): 23 resp. 24 species per 0.01 ha[54]. This amounts to 58-60% of the average, and 46-48% of the maximum reference, resulting in local species density scores of 0.41-0.43 and 0.52-0.54, respectively (see table A1.1). These data are based on many field measurements: 223 for arid forests in the lowlands of Switzerland and 501 for moist forests in the north-west of Germany. Köllner used the Swiss reference also for the German data. When a German reference state is applied, the local species density scores drop to 0.24 and 0.36, respecively. We have used these adjusted scores for broadleaved forests in Europe.

In [Lindeijer et al., 1998] it was estimated by experts that in Scandinavia the species density in plantations would be 80% of the maximum, which would results in local species density scores of 0.2 using a maximum reference. In spite of attempts to obtain more local species density data for production forests, this was not succesful. Two recent other projects we know of dealing with cases in the tropics (from the University of Gent and from Pré Consultants Amersfoort) were also unsuccessful in this respect. In [Sas et al., 1998] an expert was cited stating that collecting such data consistently would require months of expert work, which was beyond the scope of this project.

Although the above scores are used in this report for broadleaved and coniferous forests in the temperate and boreal region when lacking more specific data, they can not be used in the tropics. There, the occupation impacts of selective cutting on the forest biodiversity are not quantified as far as we know of[55]. Clearcutting impacts are not expressed in occupation impacts, but in transformation impacts.Transformation impacts of clearcutting can be estimated when the percentage of the various resulting land uses is known. See section 3.3.2 for these percentages. Impact scores from Köllner can be applied for the various final states, and the biodiversity data from Barthlott can be used for the initial state.

Another proposal is to assess the percentage of the area under FSC (Forest Stewardship Council) certification, in a recently proposed land use method [Swan, 2001]. This certification ensures a sustainable long-term forest management. One could assume zero impact in FSC-certified area's, but still a rough estimate of the lower species density (or another –aggregated- biodiversity indicator) in the rest of the forest is required. The above estimates could be used for this. The certified areas are then used to correct these figures. However, from the Forest Resource Assessment 2000 of the FAO [FAO, 2001], also used for the intervention land use data for forestry (see chapter 3), it appeared that

[54] Standard deviation is 10 for both results [Köllner, 2001].

[55] Selective cutting implies a temporal, local decrease of species density, which is hard to quantify. The renaturation time to the situation before the selective cutting, and the area required, are included in the occupation data, as country-averaged output and land use data is used (including the time to leave large regions to recover to mature forest). See also section 3.3.

the average percentage of certified area is very low at present (0,45%) and has no significant influence on the impact assessment data.

4.4.2 Life support impacts of other processes

For most of the process types distinguished by [Köllner, 2001] we have found appropriate average NPP data, mainly from [Ajtay et al., 1977] and [Vitousek et al., 1986]. See table 4.7, and for details annex 1 (table A1.5).

Table 4.7 Net primary productivity data for several land use types, and life support impact assessment scores for Europe (average reference 1030) and the Netherlands (av. ref. 1200).

CORINE nr.	Land type	NPPact	LOnpp, EU	LOnpp, NL
	Individual types	[g/m^2.y]	[g/m^2.y]	[g/m^2.y]
1.1.1	Continuous urban	0	1.030	1200
1.1.2	Discontinuous urban	200	830	1000
1.1.3	Urban fallow	200	830	1000
1.1.4	Rural settlement	200	830	1000
1.2.1	Industrial area vegetation	200	830	1000
1.2.2.2	Rail fallow	200	830	1000
1.2.2.4	Rail embankments	200	830	1000
1.2.5	Industrial fallow	200	830	1000
1.3.4	Mining fallow	200	830	1000
1.4.1	Green urban	200	830	1000
1.4.2	Sport facilities	200	830	1000
1.5	Build-up land**)	0	1.030	1200
2.1.1.1	Conventional arable	650	380	550
2.1.1.2	Integrated arable	650	380	550
2.1.1.3	Organic arable	650	380	550
2.1.1.42	Fibre/energy crop hemp	1490	-460	-290
2.1.1.43	Fibre/energy crop Chinese reed	960	70	240
2.2.1.2	Organic orchard	650	380	550
2.3.1.1	Intensive meadow	900	130	300
2.3.1.2	Less intensive meadow	900	130	300
2.3.1.3	Organic meadow	900	130	300
3.1.1.2	Broad-leafed forest	1.200	-170	0
	ETH database			
-	Ecoinvent II	1.200	-170	0
-	Ecoinvent III (excluding green urban, rail embankments, rural settlements.)	650	380	550
-		200	830	1.000
-	Ecoinvent IV	0	1.030	1.200

Road embankments

For road embankments, the study from [v.Schaik & van den Hengel, 1994] gives standing biomass and biomass production data. Aboveground biomass was multiplied with a factor 1.5 to obtain total biomass (including belowground biomass) according to [Olson et al., 2001]. This is not yet NPP (yearly biomass productivity) data. As with the surface aggregate extraction cases, collecting adequate NPP data proved to be very difficult. The suggestion to use standing biomass as a life support biomass indicator instead is appropriate here too. The resulting data using standing biomass for NPP is given in table 4.8.

Table 4.8 Standing biomass (aboveground –Abiom.- and total) on highway embankments in the Netherlands, based on [van Schaik & van den Hengel, 1994].

	Clay soil				Sandy soil			
	2x mowing/year		no management		2x mowing/year		no management	
	Abiom./m²	total [g/m²]	Abiom./m²	total [g/m²]	Abiom./m²	total [g/m²]	Abiom./m²	total [g/m²]
Unshaded	1050	**1575**	520	**780**	395	**593**	180	**270**
Shaded	1090	**1635**	510	**780**	485	**728**	250	**375**

The largest uncertainty seems to be whether the soil is more sandy or more clay. As the distribution of soils under road embankements is not known, an equal contribution is assumed. Using an average NPP of 1200 $g/m^2.y$ for the Netherlands and 1030 $g/m^2.y$ for Europe (see table 4.9), this would result in the impact scores:

Occup road embankment NL -301 $g/m^2.y$
Occup road embankment EU -301 $g/m^2.y$
Trans III-III road embankment -860 $g/m^2.y$
Trans II-III road embankment -198 $g/m^2.y$

The inconsistency of these data with 'real' NPP data should however be acknowledged.

Forestry

For forests many case studies are available to assess NPP (see e.g. [Oak Ridge, 2001] and [Lindeijer et al., 1998]). However, many cases are natural forests, and data on the difference in NPP between natural and commercial forests is largely lacking. As far as known, only in [Lindeijer et al., 1998] cases were given of commercial forests explicitly compared to natural forests (used as reference states). In [Ajtay et al., 1979] generic data per biome type are given (see annex 1, section A1.3). Using these generic data for actual cases of commercial forestry would always result in an impact score of zero, because actually no real commercial forest data were applied. Therefore, the data from [Lindeijer et al., 1998] is used for occupation. For comparison, the IA factors that would arise when applying average biome values for NPP act and using the mixture of biomes in a region according to [IMAGE team, 2001] for NPPref (table 4.9, in light grey). Only the bold data are used in the case study. Averages are determined by weighted average of the areas of the forest types coniferous and broadleaved.

For transformation some data of [Ajtay et al., 1979] could be applied for the final states. See table 4.9 below.

Table 4.9 Data on NPP [gC/m^2y] for forestry as applied, from [Lindeijer et al., 1998], and compared to more generically derived figures from [Ajtay et al., 1979] (occupation)

Large IA uncertainty range !!! (not included)	Applied impact scores, mainly from [Lindeijer et al., 1998]			alternative set based on [Ajtay et al., 1979]		
Forestry type (occupation)	NPPref	NPPact	Impact score	NPPref	NPPact	Impact score
Commercial spruce forest (boreal)	**378**	**324**	**54**	800	360	440
Commercial spruce forest (cent.EU)	605	750	-145	1200	1500	-300
Commercial oak forest (central EU)	1060	1030	30	1200	1030	170
Average EU	1030	831	198	1030	1363	-333
Average NL	1200[1]	862	338	1200	1045	155
Eucalyptus plantation	280	432	-152	-	1080	-
Shorea Robusta (damar) India	657	448	209	900	1180	-280
Tectona Grandis (teak), Indonesia	1211	1008	203	2144	1440	704
Broadleaved forest average Asia	1580[1]	1008	572			
Broadleaved forest average USA	855[1]	1030	-176			

Table 4.9 Data on NPP [gC/m²y] for forestry as applied, from [Lindeijer et al., 1998], and compared to more generically derived figures from [Ajtay et al., 1979] (transformation)

Transformation	NPPini	NPPfin	Impact score			
Nat.broadl.forest to agriculture AFRICA	651[1]	650[1]	1			
Nat.broadl.forest to agriculture ASIA	1580[1]	650[1]	930			
Nat.broadl.forest to agricult. sAMERICA	1209[1]	650[1]	559			
Nat.broadl.forest to agriculture TROPICS	1030[1]	650[1]	380			
Nat.conif.forest to comm. forest nAMER	855[1]	360[1]	495			
Extensive meadow to conif.forest nEU	750[1]	360[1]	390			

[1] Actually based on [Ajtay et al., 1979] and [IMAGE team, 2001]

Considering the uncertainty ranges in NPP data, illustrated by the alternative IA set, these can be expected to be of the same order of magnitude as the renaturation data for biodiversity (at least half an order of magnitude).

4.4.3 Specific choices for various land use types

In this section an overview is given on some additional assumptions made to arrive at impact scores for various land use types as defined in LCA studies. An overview of these impact scores is given in annex 7. For land use type definitions these are:
- industrial area: 80% urban built, 20% urban vegetated;
- industrial built, roads & railways: 10% uncertainty in SD;
- renaturation ETH: 25% uncertainty in SD;
- renaturation aggregate extraction: 10% uncertainty
- road embankments: 2x mowing taken, 50% shaded, 50% unshaded
- mining metals & energy carriers: country contributions weighted according to [Lindeijer et al., 1998]

The Eco Indicator 99 scores calculated according to [Goedkoop&Spriensma, 1999], and applied renaturation times are:

* Occupation:

built, dump, railways & roads	1.15
int meadow, arable land	1.13
arable integrated, arable organic, urban vegetated	1.09
extraction	0.96
ind area, road embankments, railway embankments, urb green, sport	0.84
forest	0.11

* Occupation during renaturation:

II-III	0.51
II-IV	0.96
III-IV	0.96

* Renaturation times for transformation (trans x [y]):

III-IV	5 y
III-II	25 y
IV-II	50 y
III-Marsh:	50 y
III-conif forest	200 y
III-Tropical forest	1000 y

It should be realised that what is called transformation in the EI 99 report is actually about occupation during renaturation [Goedkoop, 2002], using the same calculation procedure as in the ETH database [Frischknecht et al., 1996].

4.5 Concluding remarks on the impact assessment data collection

Regionally differentiated data on land use biodiversity and life support via the indicators species density and NPP are globally available only on a very large grid. Manual interpretation of maps was still required for the most uncertain data: the global species density data. This makes the assessment of regionally differentiated ecosystem level impact factors a difficult job. The local species density impact factors per land use type are largely taken from [Köllner, 2001]. Generally, expert judgements are required to fill in data gaps, as for forestry and renaturation. The main range is however caused by the global species density data from [Barthlott, 1997]. Reduction of IA data uncertainty should therefore start by improving this ecosystem level quality factor.

For the life support indicator NPP there is a consistent global map available, but uncertainty ranges due to natural variability are equally large as for biodiversity. For cases, often NPP data is lacking, and only standing biomass measures or even estimates have to be used. This raises the question whether standing biomass could not be used as a life support indicator instead of NPP, simplifying data collection. It will depend on the extent to which standing biomass covers all aspects of life support covered by NPP. We have not further pursued this route.

5 Case studies

5.1 Aim of case studies

The aim of performing cases is to illustrate how the methodology and data collected in this study can be applied to exemplary building cases. This will also give an idea of the different possible resulting scores. For this reason, three different main building materials (concrete, brick and wood) in their application are considered, with for each material an aggregate extraction process including or excluding renaturation. For the wooden case two different wood sources are compared. The comparison with the Eco Indicator 99 land use scores for these cases are not presented in the main report, as they give no additional insight (see annex 8).

But first the case of normalisation data for the Netherlands is discussed, as this gives some general insight in the contribution of individual land use types to the total impact score of a country.

5.2 Contributions to the total normalisation scores of the Netherlands

The total normalisation scores for land use impacts of the Netherlands are based on the data in chapter 3, with an addition of the tropical hardwood consumption in the Netherlands according to the IVAM LCA database [IVAM, 2001] . The contribution of the various land use types to the Dutch normalsation is given in figure 5.1. A comparison with the results according to the Eco Indicator 99 scores for land use is given in figure 5.2.

From figure 5.1 it is clear that agriculture contributes by over 75% to the ecosystem occupation scores (EOav and Eomax). Using the average (av) or the maximum (max) reference makes no significant difference in the contributions.

The contribution to transformation (ETrel) is however very different from that of occupation. Forestry now determines over half of the total score, although the interpretation of this contribution is not trivial. Two-third of the transformation attributed to forests is due to the degradation of tropical forests due to Dutch use of tropical wood. One-third is actually *negative*, being the transformation of mainly agriculture ground to forestry in the Netherlands itself. In figure 5.1 only the absolute values could be shown. The transformation of tropical forests (6,7^{e}9 weighted m^2) is actually nearly equal to the total net transformation for the Netherlands in 1993 (7,6^{e}9 weighted m^2), taking the negative scores for inland forests, road embankments and urban green into account. Should this tropical wood consumption not be included (considering only the impacts caused on Dutch territory), then the transformation score for the Netherlands would have been negative.

Another difference in contribution between transformation and occupation is the large contribution of industrial area. This is partly a transformation from forests and extensive meadows (ETH category II), and partly from intensive agriculture (ETH category III).

Comparing the contributions according to the TNO land use method to that of the Eco Indicator 99, the resemblance between both is clear for occupation. This is obvious, because there was hardly any difference between using the average and a maximum reference in the TNO method (the only difference between both methods in the local impact score), and regional differentiation (on the biome level) does not make any difference for the Netherlands.

However, a major difference between both methods becomes clear when the transformation scores are compared. Here, the Eco Indicator can not distinguish between the initial state in a tropical forest and one in a temperate forest (in the Netherlands). Therefore, using wood from tropical forests is not well-expressed in the EI 99, whereas it is clearly explained using the TNO method. This illustrates the importance of regional differentiation using a reference state.

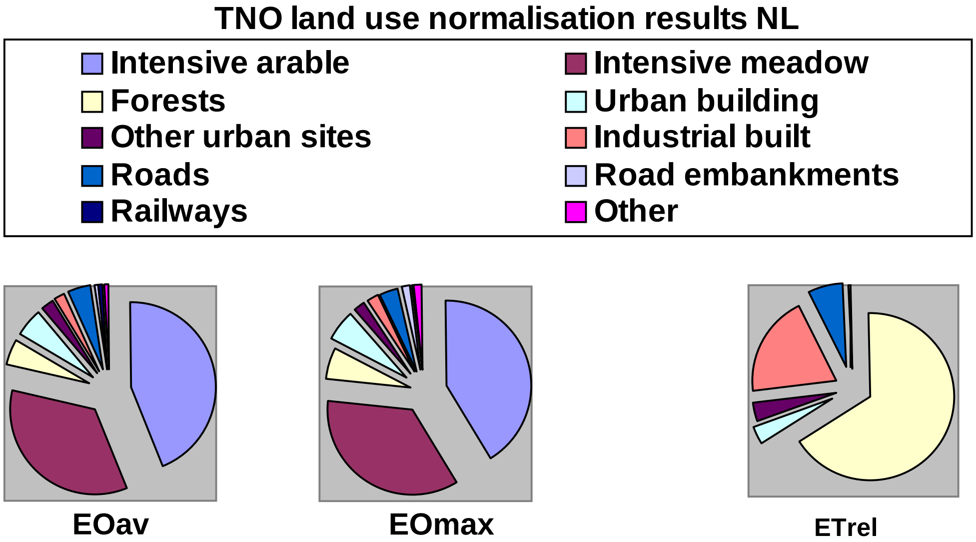


Figure 5.1 Contributions to the TNO biodiversity normalisation score for Dutch land use impacts.

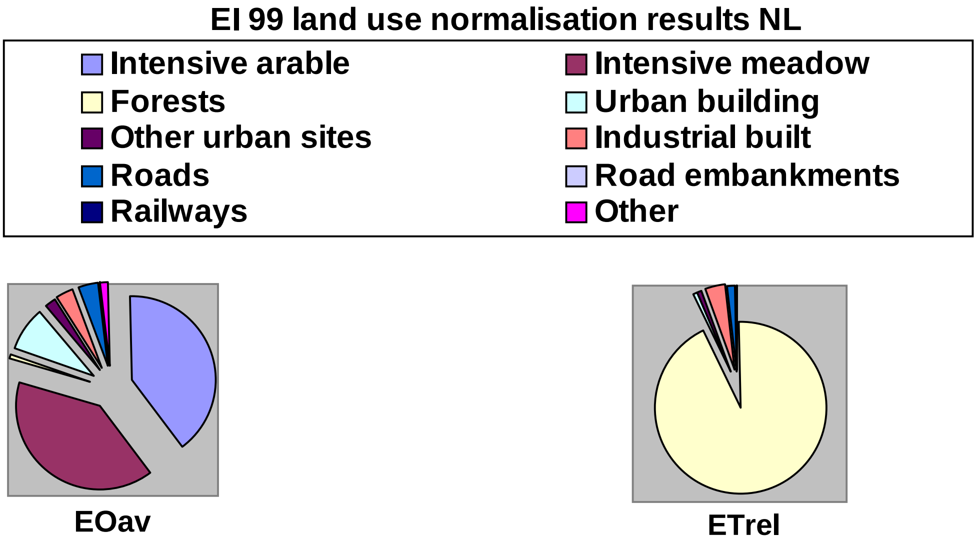


Figure 5.2 Normalisation contribution according to the Eco Indicator 99.

For the life support indicator NPP this contribution analysis is not repeated, as no additional insights are expected.

5.3 Selection of the building case details

The three main materials to which the data are applied are concrete, brick and wood. The choice for the first two materials is due to the commissioners main interest: aggregate extraction. Wood is added because this material has an extremely large land use per tonne compared to the other materials. Therefore, wood is a good material to test and judge the impact assessment method with, and to check its acceptance.

The choice for comparing above three materials limits very much the choice of cases, as all three materials should be applicable in the same case in real life. For this case, the outer wall is selected. For the materials brick and concrete the options traditional and modern extraction processes are taken into account. The traditional and modern processes pertain to the production year 1991 and 1998, respectively. For wood the options traditional and improved wood production/logging are considered. The rough material input from Eco-Quantum [Kortman et al., 2000] for the outer wall per m^2 (the functional unit) is given in table 5.1.

Table 5.1 Materials for the alternatives of the outer wall per functional unit.

Materials	Brick layment [kg/m^2]	Concrete (prefab) [kg/m^2]	Wood (prefab element)	
			Scandinavian [kg/m^2]	Incl. Canadian [kg/m^2]
Brick ($^1/^2$)	127.5			
Cement masonry ($^1/^2$)	75.6	0.882		
Concrete reinforced ($^1/^2$)		244.5		
PUR		0.05		
Wood profiled spruce			7.95	7.95
Wood profiled spruce preserved			10.12	
Wood red cedar NW America (12%)				8.14
LDPE			0.16	0.16
Paint alkyd			0.32	0.32
Total	**330.6**	**245.4**	**18.55**	**16.57**

1 Extraction processes with no renaturation.
2 Extraction processes including renaturation.

5.4 Results

Only the normalized scores for each alternative is shown, together with the contribution of the main process types to each impact score. For more detailed results (for occupation and renaturation separate) see annex 8.

5.4.1 Brick

Occupation in both an average and maximum reference system and transformation results are shown as normalized impact assessment scores for brick layments (figure 5.3). See annex A8 for separate scores for each impact category.

Using the maximum reference leads to higher scores for occupation than when using the average reference. Extraction processes without renaturation lead to lower occupation scores when using an average reference, but to higher scores when using a maximum reference. This is because when a maximum reference is used, every land occupation is considered negative for the environment (positive score). Using an average reference allows for negative total occupation scores, because renaturation may lead to higher indicator scores than what is the average in a country. Only when a short renaturation time is considered (by using a depreciation time of for instance 30 years) the difference in occupation score between using an average and a maximum reference disapppears (see the figures in annex 8).

The transformation score is slightly positive (negative to the environment) for extraction with renaturation because the average initial species density value is 36 per 0.01 ha, and the average final value is 32 per 0.01 ha[56]; in the case without renaturation the original agriculture ground is restored (no net transformation). Background processes such as energy production add to these scores.

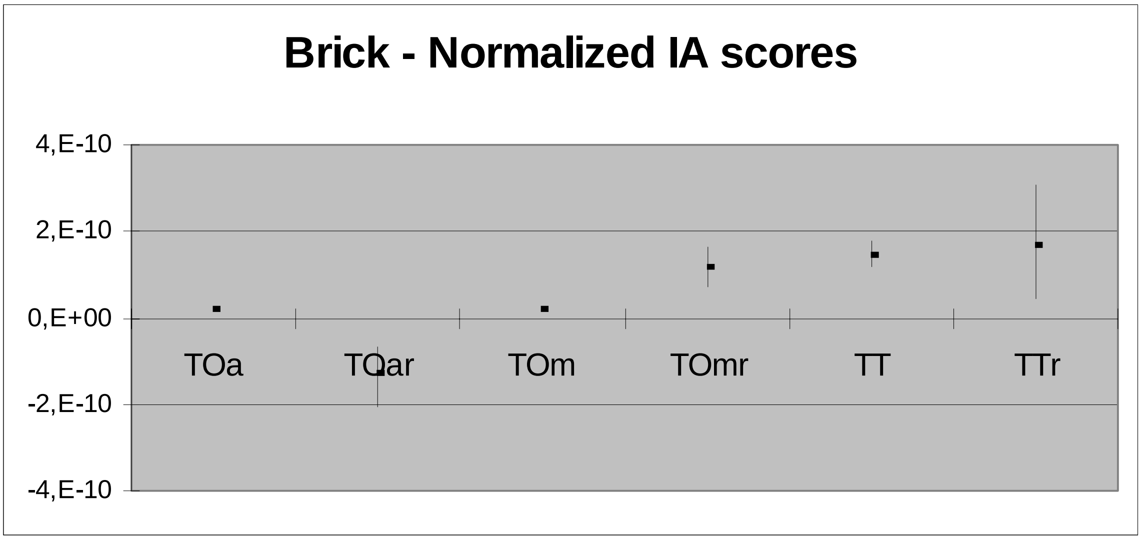


Figure 5.3 Normalized biodiversity impact scores for a FU of brick outer wall. Scores represent contribution to the Dutch normalisation scores per year (no units).

Legenda for all figures:
TO = total occupation score (occupation including renaturation)
TT = total transformation score (transformation including renaturation)
a = average reference, no renaturation
ar = average reference, including renaturation
m = maximum reference, no renaturation
mr = maximum reference, modern extraction
r = including renaturation
C = Canadian wood
S = Scandinavian wood

The normalized impact assessment scores give the impression that transformation is the most important impact category. However, this depends on the relative weighting of occupation to transformation, which is not performed here. The uncertainty ranges shown are only the uncertainty in the impact assessment data for the case; not the uncertainty in the normalisation data or in the inventory data. The uncertainty in the inventory data is shown in the results of the life support indicator (biomass).

The contribution of the background process types energy and industrial production are dominant in the case without renaturation. However, land use by the extraction processes in the life cycle is also an important factor, and is for occupation dominant when renaturation occurs. Other process types are hardly contributing to the total impact scores (see table 5.2).

[56] The uncertainty in the initial state is only 7% in this case. The uncertainty in the final state is much higher here, becausethese are only estimates of ranges.

Table 5.2 **Contribution of the most important processes to occupation in the average reference system (total occupation and transformation; including renaturation).**

Processes	Occupation		Transformation	
	No renat	Renaturation	No renat	Renaturation
Energy production	27%	3.5%	68%	69%
Extraction and production of minerals	37%	90%	14.5%	8%
Industrial production	36%	6.5%	17.5%	23%
Transport	0%	0%	0%	0%
Other (a.o. incineration/landfill)	0%	0%	0%	0%

The life support indicator results for the brick case are given in figure 5.4, also as normalised scores.

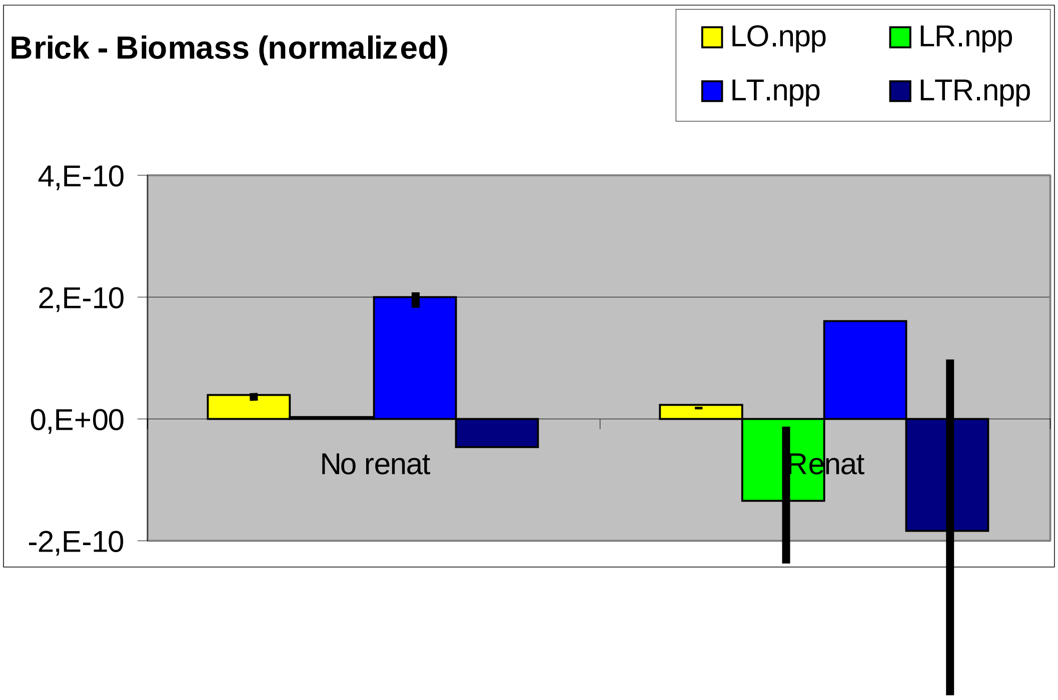


Figure 5.4 **Normalized life support impact scores for a FU of brick outer wall. The uncertainty range is only from inventory data, and large due to only one case considered. Scores represent contribution to the Dutch normalisation scores per year (no units).**

These results for life support impacts show a clear improvement when including renaturation, for occupation as well as for transformation.

The uncertainty in the case with renaturation seems very high, which may mislead its interpretation. The estimated uncertainty on the area required for extracting a tonne of clay results from a postulated formula: 1 – fraction of total amount extracted (see annex 3). For clay extraction data on only one case was available, resulting in a very high uncertainty score. In reality this score will be much lower; examining a large set of cases will surely lower this range.

5.4.2 Concrete

The biodiversty impact scores show a similar pattern as for brick (compare figure 5.5 with 5.3), except for transformation where the total transformation results for the case including renaturation are much lower than for brick. This is due to the very large improvement when renaturating a riverbed gravel extraction according to Stichting Ark (annex 4; initial state 31 species/0.01 ha; final state 75-100 species/0.01 ha).

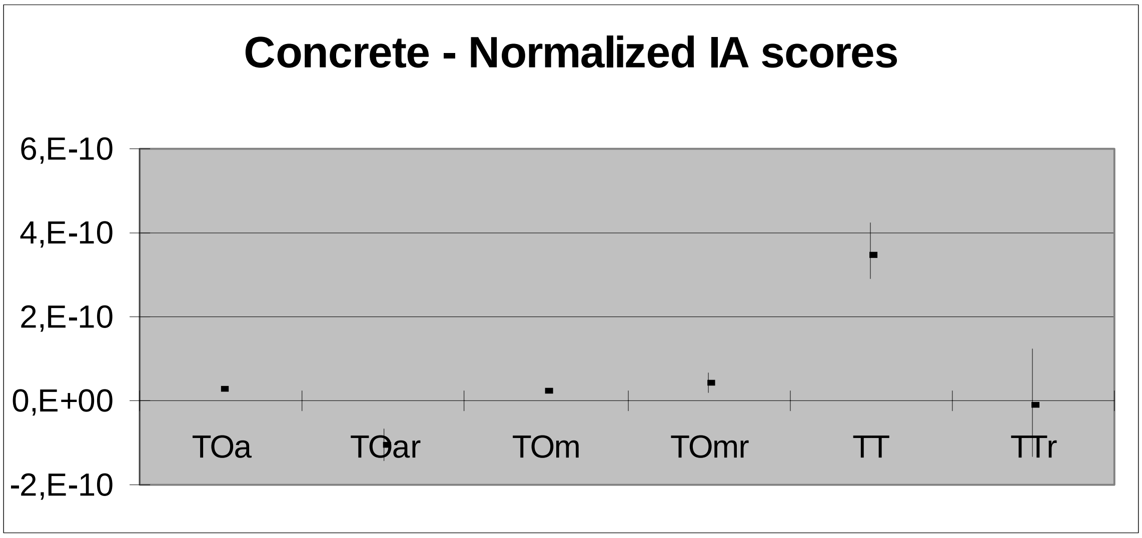


Figure 5.5 **Concrete prefab, normalized biodiversity impact assessment scores. Scores represent contribution to the Dutch normalisation scores per year (no units).**

The normalized life support indicator results are given in figure 5.6.

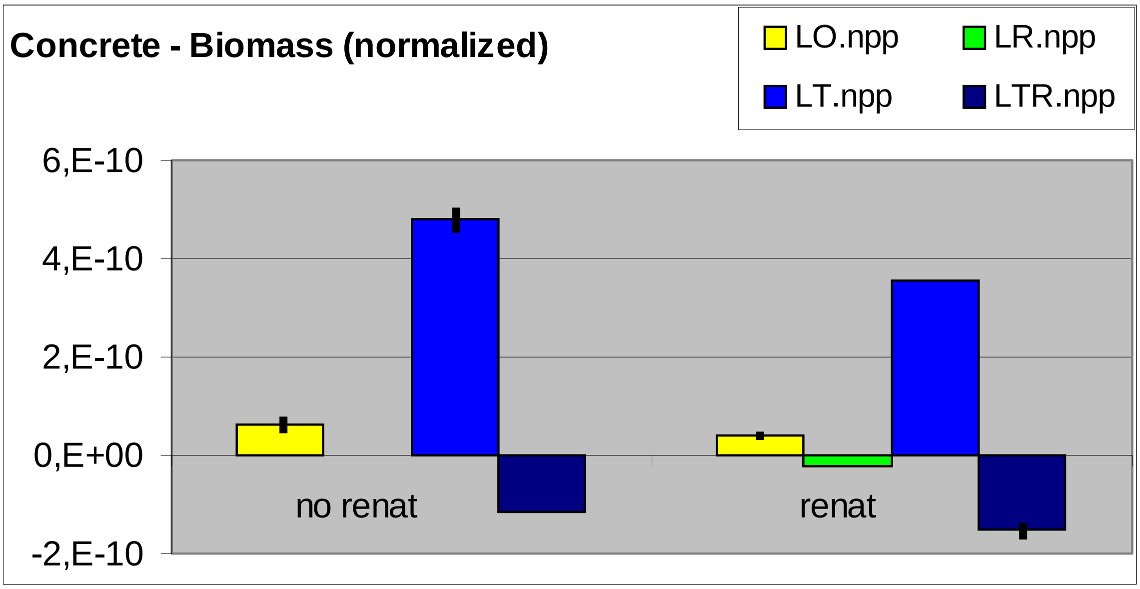


Figure 5.6 **Normalized life support indicator scores for a FU of concrete outer wall. Scores represent contribution to the Dutch normalisation scores per year (no units).**

Again, the trend when comparing a case without renaturation to a case including renaturation is an improvement on all scores, although these do not all appear significant. The uncertainty is here lower than in the brick case, because a much larger percentage of the total Dutch extractions was examined (see annex 3).

All results from both cases (brick and concrete) are in the same order of magnitude. From these cases, no clear preference can be given if a choice between the two outer wall types were to be made. When using average references for occupation (as in TOa for biodiversity and in LO and LR for life support) a clear improvement due to renaturation can be reproduced.

5.4.3 Wood

The results when comparing two wood based outer wall panels (one partly from Canadian coniferous wood and the other from only Scandinavian coniferous wood) are shown in figures 5.7 (biodiversity impacts) and 5.8 (life support impacts).

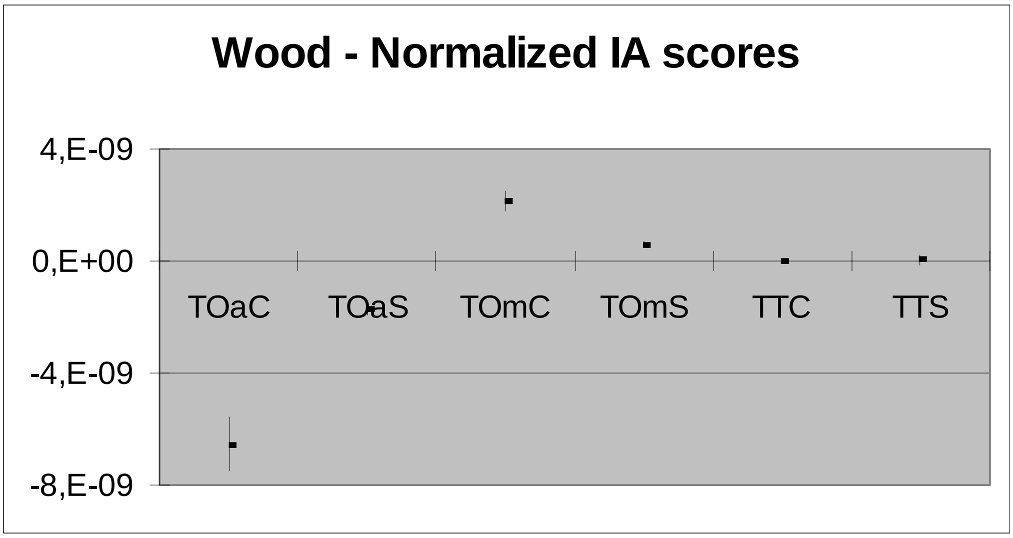


Figure 5.7 Normalized biodiversity impact scores for a FU of wood based panel outer wall, using partly Canadian wood (C) or only Scandinavian wood (S). Scores represent contribution to the Dutch normalisation scores per year (no units).

As seen from figure 5.7 there is a large difference in results depending on whether an average or a maximum reference is used. With an average reference (TOa) using only Scandinavian wood seems a disadvantage (less negative scores), whereas with a maximum reference (TOm) it seems an advantage. Here the implications of using a different reference are very clear. Because Canadian forestry is less intensive, more land is occupied for the same amount of wood output. When every land use with a more than average biodiversity is considered an environmental gain (TOa), this extensive forestry is an advantage because more land is used with an improved land quality. It can be considered as 'protecting land with a high biodiversity value against degrading towards built land'. When every human land use is considered an impact, it becomes an improvement when land is used more intensely, as in Scandinavian forests.

For life support, the impact scores are all positive (bad for the environment). This is because commercial boreal forests have lower NPP values than natural boreal forests (see [Oak Ridge, 2001] and [Lindeijer et al., 1998]). The occupation scores are lower for Scandinavian wood due to their higher land use intensity. This influences the inventory part of the score. The NPP values for both forests are considered on average the same[57]. The transformation scores on the contrary are higher for the case using only Scandinavian wood, because more wood is required, resulting in more energy required (the transformation scores are largely determined by energy production, see table A8.13b in annex 8).

[57] We do not know whether this would also be the same if standing biomass would have been chosen as an indicator (as suggested elsewhere) .

For the life support indicator NPP the following scores result (figure 5.8):

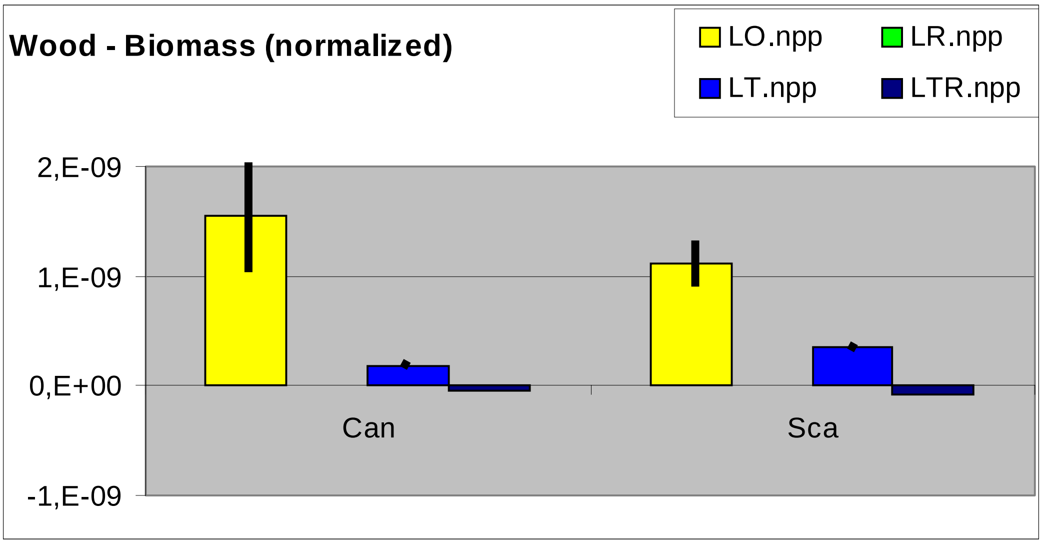


Figure 5.8 Normalized Life support impact scores for a FU of wood-based outer wall. Scores represent contribution to the Dutch normalisation scores per year (no units).

5.5 Conclusions on the case study

Cases can be calculated with the inventory and impact assessment data collected in this project. However, some time-consuming difficulties arise:

- Uncertainties are large and should not be left out of the assessment. However, the LCA software version used does not support calculation of uncertainties. Due to this, manual changes to all foreground inventory data had to be performed, with minimum, average and maximum values for all land use interventions. This range of LCI values was related only to the life support indicator. To include the uncertainty in the biodiversity different impact assessment 'methods' were defined for minimum, average and maximum values per type of IA score. As there are 6 IA score types for biodiversity, this leads to 24 'impact scores', which have to be combined to 6 scores with uncertainty ranges in a spreadsheet. Taking uncertainty into account is under the present software thus a tedious job.
- Dealing with the negative values which often result in IA of land use gives a problem in present LCA software. Even when manipulating the results in excel some difficulties arise, because negative contributions have to be converted to absolute values.
- The process of arriving at one intervention data set and one impact assessment data set for land use has been a task of over 1 personyear of worktime. Part of that work has been to allow automatic treatment of changes to these data sets, and to ensure faultless manipulation of these data in and out of the generic database in LCA software. Additionally, each change in either dataset results in a significant workload for recalculating the adjusted results. Thus, the establishment of such a data set can not be expected for the average LCA study, and the collected data should be incorporated in LCA databases for others to use.
- The interpretation of land use impacts is not straightforward due to the large uncertainties, the negative scores and the value choices inherent to the methodology. Only when some basic choices would be set on beforehand (adding up occupation and renaturation scores, and dealing

only with a maximum or an average reference), a rather simple picture with two scores per alternative would emerge. But an adequate interpretation would require to know where these scores came from (dominance analysis) and what the influence of a different reference would be.

The results give rise to the following methodological conclusions:

- Occupation during renaturation is always considered a negative impact to the environment, when a maximum reference is applied. This seems contrary to the intuitive, as renaturation is performed to improve the situation after an extraction. However, this improvement is shown by the negative transformation impact score (to be seen as positive for the environment). Both occupation and transformation score sshould therefore be shown. Whether the time and space used for this renaturation (the occupation during renaturation) should be considered positive or negative to the environment, depends on the viewpoint of the decision maker: is the situation better than the average in the area (=> average reference), or not as bad as the average but not yet nature (using a maximum reference).
- Taking renaturation into account has an important influence on the impact scores. Renaturation may reduce the land use scores when it results in negative figures, or it may enlarge the scores when positive figures arise. In these cases where aggregate extraction is a foreground process, the extent and quality or renaturation indeed determines the land use scores. In other cases, where background data dominate the land use scores, renaturation will not be so important, as renaturation is not dominant in the (adapted) ETH database we use for most background data.
- The inherent uncertainty due to limited spatial resolution causes a large uncertainty range in the impact scores. This may easily lead to insignificant differences between closely resembling alternatives, or between the choice of an average or a maximum reference. Only for wood a significant distinction between an average and a maximum reference could be determined in our case. This is so because the average reference values for wood extraction are negative, whereas the maximum reference values are positive, and because wood extraction dominates the whole land use picture by a factor 10 to 100 (in absolute figures, even though the material input is more than a factor 10 lower per functional unit). The here calculated uncertainty is however an overestimation. All minimum scores in the life cycle are added, and compared to total average and maximum scores. Averaging out of natural variations is thus prevented, whereas this will occur in reality. The actual uncertainty will therefore be lower than what we calculated.
- The present impact assessment method can assist to put environmental discussions in (global) perspective and ensure consistent reasoning all along the life cycle attached to a certain case study, because all details in the (differentiated) results can be explained and value choices can be controlled in the assessment.
- As the Eco Indicator 99 applies only one reference for the whole world, it does not allow for regional differentiation. It can thus not cope with a case like the use of tropical hardwood.

The focus of these studies was not to compare the different alternative materials. Nevertheless, a few conclusions can be drawn when comparing the results:

- Wood has much higher occupation scores than other materials, when choosing a maximum reference. When using an average reference, the score is much lower (i.e. strong negative). Both results are due to the low land use efficiency: for extracting wood much more land is required yearly than for minerals[58]. Methodological choices therefore will influence the results of land use impact assessment in LCA when silviculture (or agriculture) processes are a considerable part of the functional unit, unless these choices are made explicit.
- The transformation scores are less than a factor 10 higher for wood than for the other materials; due to the lower input per functional unit, the impact scores become of the same order of magnitude for all materials. This implies that for real life situations, the choice between for instance wood, concrete and brick as main materials is not obvious on beforehand according to

[58] The fact that forest land also has (other) life support functions such as landslide prevention is acknowledged by the impact assessment, and should to our opinion not be doublecounted by attributing only part of the forest area to the forestry output. Allocation of a land use intervention should only be done to economic outputs, not to natural functions (which are present also without the intervention).

the TNO land use methodology, when transformation is considered an important issue. When the use of potentially valuable land is considered more important, it becomes crucial how one perceives the extent –i.e. remaining nature quality- of land use by (in this case) forestry and renaturating aggregate extraction sites[59]. The present impact assessment method can assist as a kind of calibration device, to ensure that argumented deviations from calculated impact scores are considered equally along the whole life cycle. This is the core value of LCA: to put environmental discussions in (global) perspective and ensure consistent reasoning all along the life cycle attached to a certain case study.

- When comparing the occupation and transformation scores, the interpretation of these may become very important. Although the interpretation step was not included in this study, some considerations are given here.

  Occupation expresses the distance between the quality indicator level (biodiversity or land use) of a certain land use to a reference level (the natural equilibrium or regional average), and above occupation impact scores express the risk of contributing to regional species loss through stress due to low quality land occupation in that region.

  Transformation on the other hand expresses the change in land quality over time. These are not reversed changes, and contribute to the national trend in land use changes (as expressed by the normalization scores).

  Depending on the extent to which the national trend in land use changes is considered more damaging to biodiversity and life support functions than the present national mix of land uses, should normalized transformation scores be considered more important than land occupation scores (and receive a higher weight). In this consideration, the contribution by "imported land use" such as for tropical forests in regional normalization scores should be taken into account[60].

[59] Without devaluating the present method, it is clear that many land use related aspects relevant for decision making are not included, such as landscape degradation/differentiation, specific fauna habitat impacts and habitats containing endangered species. These need to be incorporated in such a decision making process, next to non-land use related ussues such as noise, emission impacts, and non-environmental aspects (economic, societal).

[60] For instance, the Dutch land transformation normalization scores would have been negative if tropical wood use in the Netherlands would not have been included. That would have implied that the land transformation trend in the Netherlands (more forests, mainly at the expense of agriculture) is good for the environment. Including the tropical wood consumption by the Netherlands turns the overall Dutch land use to "bad for the environment".

6 Conclusions

6.1 Overview of results

An adapted land use impact assessment method has been developed, with the following main characteristics:

- distinction between occupation of land, and land transformation;
- regional differentiation on the level of biomes (global ecosystem classification);
- incorporating ecosystem level and species level aspects of biodiversity;
- allowing comparison of two choices for the biodiversity reference level;
- allowing an upper boundary uncertainty analysis for the biodiversity impact assessment;
- using net biomass productivity as an indicator for life support impacts of land use;
- allowing a sensitivity analysis on the in- or exclusion of renaturation impacts.

Based on this method, land use intervention data has been collected for the following land use types:

- aggregate extraction in the Netherlands;
- forestry worldwide;
- various agriculture processes;
- various mineral extraction processes worldwide;
- the energy production sector in Europe[61];
- transport by roads, railways and canals in the Netherlands;
- industrial production in the Netherlands;
- housing in the Netherlands;
- waste disposal in the Netherlands.

These over 250 separate interventions are linked to the over 20 different impact categories (including minimum and maximum values and sensitivity analyses) in the TNO database and in a version of the IVAM Environmental Research database [IVAM , 2001].

Using the TNO database, the normalisation data for the Netherlands have been analysed and compared to the Eco Indicator 99 [Goedkoop & Spriensma, 1999]. Also case studies have been calculated, using bricks, concrete or wood as main material for a m^2 of outer wall. For each case two extraction processes have been compared, and a sensitivity analysis has been performed on the methodological choices (1) average or maximum reference, (2) in- or excluding renaturation and (3) attributing only part of the renaturation process to the extraction process itself. Also the results according to the biodiversity indicator have been compared to those of the life support indicator.

6.2 Conclusions on practical applicability, communicate-ability and scientific validity

Collecting land use inventory data is a time consuming job, and virtually impossible for a single LCA life cycle. It is therefore required that adequate land use data is included in all major databases, if land use is to be included in LCAs at all. Adequate means that at least regional differentiation is included, by stating the country code.

Also, at least 60 different land use types should be distinguished (15 per intervention type: occupation, renaturation, transformation and transformation due to renaturation) even without regional differentiation. By regional differentiation this easily expands to over 250 land use types, as shown in this project.

[61] Interpretation of the ETH database

Including transformation implies including a new kind of information in LCAs: the contribution of a case to the *trend* in the 'background level'. This is especially relevant for land use, where a relatively large part of the natural background is intervened by humans and changes still occur at a high speed. In interpreting land use impacts, this information will be as important as the contribution to the impacts of the total area occupied. One could state that *transformation expresses the trend in not-reversed land cover changes*, whereas *occupation expresses the relative distance of land use to an equilibrium*[62].

The choice of the equilibrium for occupation is crucial. When this is a maximum level, just about all land occupations will be 'bad for the environment', even during renaturation. When this is an average level, various land uses may render negative scores ('good for the environment'). For comparative studies this is not a problem, but the absolute scores do suggest a very big difference in interpretation. In fact, the choice of the reference level should be checked with the interpretation. **Due to our experience with mining and silviculture, we suggest that if not both options can be pursued and a decisionprocess involving all stakeholders is not feasible, using an average reference will probably give the most acceptable results.**

Major data ranges due to natural variations and definitions of categories will anyhow result in large uncertainty ranges. This implies that land use impacts will be assessed within a range of an order of magnitude.

Developing an impact assessment method for land use implies many choices, of which some can not be solved by the researcher and have to be presented to the user, including a proposed interpretation.

Thus, a general impression given by rough impact scores are the best results that can be obtained from any land use method, considering its regional differentiation and value choices involved. Land use impact scores will therefore only be indicative for possible problems and discussion points. The LCA user should be aware of this, and help the public or commissioner to interprete the results and to open discussions related to valuing land use.

Using relative species density as a local biodiversity indicator is questioned by experts. It has little botanical value, and although 'greenness' and diversity are important aspects of biodiversity as seen by the general public, for certain land use types important nuances can get lost with this indicator. However, the scheme allows for using a different indicator when this is more appropriate: only the relative biodiversity value (expressed in any terms) to a reference is required. Too many different indicators used consecutively will however give additional uncertainty in the interpretation, and too much detailing is anyway in contrast to the aim and ambition of this land use impact assessment method. Furthermore, **when the aim is performing global LCA's, vascular plant species diversity seems the most sophisticated indicator for which global data is available**.

Collecting data on species density for cases is a difficult task, where natural variability of the species density creates a large uncertainty range. A good example is forestry, where no case data is available at present. Expert judgements are now required for the local indicator score. Forest experts also propose a large set of indicators to be used. A weighting of these indicators to one score is not proposed. However, once the relative local scores are combined with the ecosystem level scores, details on determining local impact scores are lost in the uncertainty range and regional differentiation on the biome level. The large uncertainty due to using species density as a quality indicator on the biome level is therefore a more important problem to deal with.

Collecting data on NPP for the life support indicator is equally difficult. Here, **the question is raised whether standing biomass could also be applied as an indicator**. It is easier to collect data on standing biomass, and it may give similar results. This option was not pursued further.

On communicate-ability it is finally relvent to state the case studies showed that the results obtained give distinctive outcomes for different cases. The method is thus sufficiently discriminating. Also, the

[62] When using a maximum reference for biodiversity, the equilibrium is an optimal mix of succession stages, resulting in the highest nature value in the region. When using an average reference, the 'equilibrium' is what society has come up with as the average nature value.

results can be explained fairly easy, even though different methodological choices give different outcomes. **This makes this method readily applicable in situations where decisions have to be made among various stakeholders, which may hold different world views.**

6.3 Remaining conclusions related to the case studies

The choice of a reference level is likely to be decisice for the occupation results of an LCA study, especially when renaturation occurs and is fully attributed to the extractor. Whether or not renaturation occurs in practice is also very important.

In order to interpret normalized occupation and transformation scores, the relative importance of the present land use mix versus that of the trend in land use change in a region for biodiversity and life support should be assessed.

For the case of an outer wall, the choice between brick, concrete and wood as main material does not seem obvious from an LCA perspective. Especially as various other impact types (desiccation and noise impacts for instance) are not considered.

Energy and transport processes contribute up to over 50% of the total land use score for the cases with aggregate extraction, due to the extensive energy production process chain.

6.4 Remaining conclusions related to the aim of the project

The TNO method allows for rather detailed land use impact assessments, due to the regional differentiation, the possibility to include renaturation processes and the distinction between occupation and transformation. When detailed data are fed into the method, detailed conclusions may be drawn. However, performing such assessments in practice reveales that in many cases, detailed results have limited value, due to:
- ♦ the natural variation of background data;
- ♦ uncertainty about renaturation processes;
- ♦ value choices involved in interpreting for instance transformation results conflicting occupation results, or using an average or a maximum reference;
- ♦ relying on expert judgements due to lack of ‘hard’ case data.

These inherent limitations reduce the added value of a more refined method such as the present one. What is then the improvement over the methodology and data established in [Lindeijer et al., 1998]? The improvements can be pointed out as follows:
- ♦ Including the ecosystem level of the biodiversity impact assessment;
- ♦ Including an average reference for biodiversity, matching the value system of the NPP indicator (anthropocentric) as an alternative to the maximum reference and fNPP indicator (ecocentric);
- ♦ A local biodiversity factor matching that of the Eco Indicator 99 and allowing use of the extensive biodiversity data for LCA gathered by [Köllner, 2001];
- ♦ A more sensitive regional differentiation than in [Lindeijer et al., 1998] due to the biome level factors (absolute, with aggregated values in a range of a factor 100, whereas the relative indicator has a range of 0 to 1);
- ♦ A universally applicable format for land use interventions (independent of the impact asssessment method applied).

The improvements are thus: more interpretation options, more data available and more harmonised.

The Janus head in this is the risk of getting lost in the details and multitude of options. Simplification is also risky, but the possibility should be examined to:
- ✓ Leave out the upper and lower boundaries of the impact scores, and only use the average values;
- ✓ Add up occupation and renaturation to total occupation scores; also with transformation.

Concluding, testing of the methodology developed here has been performed in the case studies. This has illustrated several issues concerning the interpretation. Collecting the required data has also contributed to this testing, by showing which data can not been acquired easily, and may therefore limit further use of the method if not resolved. The inventory data collected can be used for all land use methods (to be) developed.

7 Recommendations

7.1 Recommendations for possible users

The recommendations for possible users of the methodology are kept short, as these issues were dropped from the core focus of the project due to time restrictions to deal with them adequately. Nevertheless, a distinction is made between general use guidance and tips for LCA practitioners.

General use guidance

- ✓ Considering the general importance of land use for global biodiversity, and its dominance over other environmental impact categories contributing to impacts on biodiversity [Sala et al., 2000], land use data should be gathered for processes to the same extent as emission data, and compiled in a similar manner;
- ✓ Acknowledge what the importance of land use impacts may be for the overall environmental picture of the decision at hand, and if relevant whether the relative importance of biodiversity and life support impacts of land use is to be determined;
- ✓ Determine whether LCA is the best tool to support your decision, and which level of detail and validity is required;[63]
- ✓ Make an informed trade-off between resources and requirements for the LCA or related tool;
- ✓ Have a screening LCA performed first, and allow for a good discussion on the results.

LCA practitioners

- ✓ Determine on beforehand the level of detail at which land use impacts should be assessed (including regional differentiation) and whether occupation and transformation should both be assessed;
- ✓ Determine the data limitation for the cases at hand (see the procedures of chapter 3, 4 and annex 1);
- ✓ Document land use data according to the format applied in annex 7;
- ✓ Apply the appropriate database including land use; take care for mixing with data from databases without land use;
- ✓ For inventory data to be added, use the proposed format (possibly separating transformations from and to other land use types, as proposed for the EcoInvent 2000) and specify further when required (see examples in annex 7);
- ✓ Make appropriate methodological choices (simplifications are also methodological choices);
- ✓ Determine a procedure to deal with uncertainties and negative values in the presentation of results.

7.2 Methodological improvement options

There are still quite some improvement options after this step of methodological improvement of land use impacts. TNO does not claim the intention or position to state relevance, feasibility or priorities in these improvement options at present. The aim is merely a short discussion of possible options. However, short comments are given on the percieved feasibility based on present knowledge.

Use other formulas to assess biodiversity

There are of course various formulas possible to assess biodiversity. For the local biodiversity score we have tried to remain as simple as possible, within the framework that a relative score should result (in

[63] For instance, in environmental impact assessments (EIA) a detailed LCA is not adequate, but a more quantified assessment could still use this IA format as one of the information carriers (see also Vogtländer et al., 2001]). It may be informative to include life cycle thinking and indicators for it whenever the production chain as a whole is involved in the decision, as with waste disposal. An initiative like this allows for making a cross-breeding between EIA and LCA that is sometimes applied in strategic EIA's.

accordance with general biodiversity policy and allowing the use of other indicators for specific processes in the LCA within the same IA framework). However, when one would abandon this requirement of a relative score, an even more simple formula would be possible, and merging the biome level quality level with the local score is then possible. Also, maybe also one of the remaining area-related biome level factors may be dropped if it would not add much to the distinctability or acceptability of the formula (which was not attempted in this study). Whether an absolute score would be acceptable is to be investigated. At least one major drawback of an absolute formula is that use of (relative) scores from a different indicator is no longer possible.

For transformation, there is the option to return to using the reference state instead of the initial state to arrive at a relative score (as in the IVAM formula). This would render the transformation formula more symmetrical (less in favour of renaturation compared to degrading processes). But also here the relative score could be dropped, at the expense of the option to use different indicator scores.

A very different way to express impacts is to use equivalency factors (as in [Guinée et al., 2002]): relate all impact sources (here land use types) to one reference (for instance tropical forest, or maize growing). Due to the regional differentiation amongst tropical forests, maize fields etc. we did not consider this a better solution. The main advantage that such a system would offer is that impacts are expressed relative to a well-known impact (when using 'the' tropical forest), which may assist in the interpretation. In fact, the present TNO scores could well be expressed as such. To keep the calculation scheme transparant, we did not perform this type of 'normalisation'.

Use other or additional biodiversity data for cases

For land use types where plain species diversity does not seem an adequate indicator, or when species density data is not available, one could use other indicators which express the change in local biodiversity. In fact, the only requirement on the species level indicator is that it can be assessed relative to the same reference (average and/or maximum) and that its score is determined in an objective manner. Independent expert judgements about the relative difference between the actual and the reference state (for occupation) and between the situation before and after a transformation could for instance also be used. Another option is to use the results of an accepted and weighted set of indicators. Expressing density of Red List species is an option too, inasfar as countries have specified them.

Include relaxation times in the impact assessment of land use

To include the extent of reversibility of the land use impact, one could include the natural recovery or relaxation time after leaving an area undisturbed as a next weighting factor. Stichting Ark [Helmer, 2001] has already estimated several relaxation times for the Dutch situation (see the table in section 3.2). [Lindeijer et al., 1998] and [Köllner, 2001] have also mentioned relaxation times for generic cases. Relaxation times for the various biomes should be determined to fit the present impact assessment method. In addition, different values may result for biodiversity or biomass (or other indicators).

Including relaxation times as an (addtitional) impact assessment factor has up till now been proposed by [Knoepfel, 1995], [Beetstra, 1998a] and [Mila de Canals, 2001][64]. Operationalisation of these methods used the assumption of full recovery to the original situation or the assumption of development to a climax stage. When full renaturation is assumed there is no net transformation, and renaturation is included as a type of occupation.

When renaturation does not occur or is not considered, multiplication of relaxation times with transformation scores (in m^2) may seem as if a type of occupation results (as the resulting units are the same). However, this is not the case: the time is an impact factor here, not an intervention factor, and occupation may be confused or doublecounted with it. Finally, when a higher nature value results than the situation before a transformation, a relaxation to the worse initial situation will not occur which may lead to an imbalanced method for positive versus negative transformations. Thus, it seems that only when disregarding renaturation could relaxation times be included as an additional impact factor. Really globally consistent relaxation data is not readily available and may include very large ranges due to natural variability, although some data is given by [Lindeijer et al., 1998], [Weidema,

[64] The method of this last author is limited to agricultural situations and life support functions.

2001] and [Köllner, 2001]. If applied, it can be seen as an improvement of the vulnerability factor for transformation.

Include fragmentation impacts

Fragmentation of an area by intersection leads to ecosystem isolation. This is especially relevant for (not flying) fauna, as they mainly use connected natural areas to migrate. Due to the high mobility and low relative density of fauna, this may be difficult to include based on statistical field data (as is carried out for flora). For flora, a proposal has recently been made to include intersection impacts in LCA [Vogtländer et al., 2001]. At ETH Zürich last year a student report is prepared on the subject with respect to railway impacts [Tobler, 2000]. A simplified approach was already suggested in [Lindeijer et al., 1998]. A more extensive elaboration would require a lot of statistical data interpretation. In [Weidema, 2001] it is argued that the amount and quality of habitats generally play a larger role for ecosystem connectivity than fragmentation. Thus, one could question whether it is feasible and important enough to operationalise this aspect for LCAs in general.

Include seabed and freshwater bed ecosystems and/or fauna impacts

Presently, only the waterplants in artificial lakes are included in the species level of the present impact assessment method. Waterbed organisms in seas, rivers and lakes are excluded, as are fauna impacts in general. The fluidity of waterbeds lead to a very dynamic species/ecosystem response to a human intervention, which somewhat resemble the large dynamics of fauna response on intersection and land use impacts. Therefore, problems are expected in assessing statistical data on impacts on fauna from land or waterbed interventions[65], due to the mobility of the species. The species level biodiversity indicator would therefore remain more rough than the present vascular plant species indicator. One exception could be the coral reefs, which are not mobile and species density (changes) could be assessed. These could actually be included easily in the present methodology.

On the ecosystem level, benthic and freshwater ecosystems could be included using the same methodology as presented in this report (assessing potential and actual area). The aggregation level is then still to be determined; distinguishing only a few aquatic biomes will lead to a complete alteration of the data set, as the area of these aquatic biomes will become $A_{pot, max}$.

Interventions in seabeds could become important especially for life support functions, as argued by Stive [Stive, 2001]. As an example, presently sand is supplied to the Dutch coast, amongst others to prevent damages due to flooding. This suppletion combined to the projected sea level rising can have unexpected impacts on the seabed, beach and dune structure. In the far past, such unexpected changes to the Dutch coast have also occurred. In the future, sand extraction before the coast could imply moving much higher volumes than present suppletion, with also higher possible impacts on the Dutch coast and its function to defend the land against the sea. A separate indicator would be required to take such impacts into account.

Improve the ecosystem level assessment

Two initiatives are relevant to mention related to assessing ecosystem quality globally.

The first is a feasibility study for the OECD to develop global biodiversity indicators, by the Dutch RIVM [ten Brink, 2000]. This study assesses the possibility of developing a National Capital Index for biodiversity, to measure the trend in nature (and its diversity) per country. The index is proposed to be a multiplication of the ecosystem *quantity* (in terms of % nature area) and the ecosystem *quality* (in terms of % of a baseline[66] –or reference- for any feasible number of indicators based on selected

[65] Data on species density are expected to be very scarce (generally they are only collected for certain regions, for birds, mammals, butterflies and reptiles, and may point in different directions, see for instance the maps in [EEA, 1995]). Data on populations and –characteristics (such as fish length and weight due to benthic trawler fishery, see [Lindeboom et al., 1998]) will be only available for selected species.

[66] The main argument for relating to a reference in [ten Brink, 2000] is increasing its relevance. The argumentation to select a pre-industrialized, low-impact level as reference include: the need for a theoretically optimal situation to be able to interprete trends as positive of negative, the recognition of the arbitrariness of any baseline due to the transitory nature of ecosystems,

species richness, species abundance and/or ecosystem structure), or if quality information is lacking, ecosystem *pressure* information (in terms of a score between 0 and 1000 –converted to %- for pressures such as climate change, acidification, fragmentation or socio-economic parameters). The ecosystem quality indicators of [ten Brink, 2000] are thus partly similar to the biodiversity factors used in this report, although species richness is considered to be least advantageous to assess trends within the pressure-state-impact-response framework for environmental policy.

Ecosystem quantity data was collected for 5 natural and 2 man-made habitat types. For forests and grasslands, bird species populations were used as an indicator, and for semi-desert and tundra biomes mammal (and bird) species populations were selected in case studies in [ten Brink, 2000]. Only for the Netherlands, also baseline data were given (for 350 plant species, 35 butterfly species, 90 bird species and 60 marine and river species in 1900-1950), but the World Conservation Monitoring Centre (WCMC, UK) considers it possible to collect baseline data for specific species in separate countries.

This initiative demonstrates the flexibility of using various species types as a basis for a relative biodiversity quality indicator. It also gives the expectation of a limitation to collecting more specific biodiversity data on all species present in a country. This suggests that for any case study, one could also use more specific species level data, when this is considered more relevant. By linking the species level factor to the generic biome level applied here, such data could be included consistently. If this National Capital Index approach would be accepted by the OECD, this would generate selected nationally relevant species and related data, to be used in future LCA cases when local data is also available.

The second initiative is a very ambitious one, on the scale of the IPCC: the Millennium Ecosystem Assessment (MA) [Reid, 2000]. From this article the following text is cited:
"It is proposed to be a four-year initiative to (1) use the findings of leading-edge natural and social science research on ecosystem goods and services to help make regional and global policy and management decisions, and (2) build capacity at all levels to undertake similar assessments and act on their findings. The MA would provide baseline information for the year 2000 on the geographic extent of different ecosystems (including terrestrial, freshwater, and marine environments) and the land or resource use patterns associated with them. Within the UN system, the MA will be conducted through a partnership arrangement among UNEP, UNDP, FAO, and the UN Educational, Scientific, and Cultural Organization. Finally, the assessment will be closely linked to a number of processes such as the International Geosphere Biosphere Program, the IPCC, the Global International Waters Assessment, and the UNEP Global Environmental Outlook process. The successful launch and completion of the MA will require political buy-in, financial support, and scientific engagement".

This project would thus generate at least detailed ecosystem area data on a global level, if pursued. The experience with the inclusion of the biome level data [Weidema, 2001] suggests that to include ecosystem data in a consistent manner is a very difficult exercise, to be performed by experts. This suggests using expert judgement and harmonisation by ecological experts, which could be provided in this process. Whether this process will proceed and give useful results for LCA is unclear at present.

For life support: include more indicators

Some authors have proposed a wide range of indicators on life support functions related to land use ([Baitz, 1998] and [Schweinle, 2001], see also [Lindeijer, 2000]). Data collection for all these indicators on a global scale has not been attempted yet. It is also not yet clear whether such a large set will give much additional information compared to using a more limited set. The assessment of three very different cases in [Schweinle, 2001] suggests that most indicators give correlated results. An assessment of which indicators would really give additional information should be the first step in this direction.

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no bias in favour of industrialised countries (f.i. when using a more recent reference), and impracticality of choosing a state before any human interference.

Include aesthetical (landscape) impacts

In [Lindeijer, 1998] it was suggested that the only probable indicator for landscape values with a more generic applicability would be landscape diversity. This can, however, only be assessed on a local scale, which is problematic in relation to LCA studies. In addition, the temporal aspects of the assessment are difficult, because most people tend to be at first instance adversive to changes, but may get used to them over time and may even later on appreciate the change they protested against earlier. Finally, aesthetic values relate to more societal values, which are generally excluded in LCAs. Like economic impacts, societal impacts are very different from the more physical environmental impacts. Including some societal impacts and excluding others will give a very unbalanced picture. Therefore, it is suggested to either exclude societal impacts in LCA at all, or take these impacts into account consistently.

Consider the land limitations within a country

It is possible to calculate the scarcity of land on a global level, as in resource depletion (see [Lindeijer et al., 2001]). Land occupation would then express the competition over the flow land use, and land transformation would express the (contribution to the) trend in loss of the resource high quality land (for agriculture or nature). But on a national level, assessing a large scale change in product(ion) systems –for instance related to organic agriculture or FSC forestry- may require the specification of how land use patterns will change within the country and abroad. This specification may include setting (national policy) contraints to the maximum amount of land available for a certain land use type per country. In this study, we have performed cases with marginal changes in land use and have not encountered this situation. Only when considerable changes are expected on a national level is such an assessment under constraints required.

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Annexes

Annex 1 Methodological choices for the Land use Impact Assessment

In this annex some recent methodological approaches to include land use in LCA are discussed. The basis for this discussion are three or four different approaches ([Lindeijer et al., 1998], [Goedkoop & Spriensma, 1999], [Weidema, 2001] and [Köllner, 2001]) and a methodological survey of these by CML (see annex 2). The main suggestions of this report are included in this chapter. Within the context of this Delft Cluster project (see section 1.2), the basis for testing land use methodologies was the method of IVAM ER.

A1.1 Methodology IVAM

In 1998 IVAM ER proposed a method to assess land use impacts on biodiversity and life support in LCA for RWS DWW [Lindeijer et al., 1998]. In this project we focussed on assessing biodiversity. The life support system was in [Lindeijer et al., 1998] assessed via biomass (see section A1.4.3 for a discussion on this issue). The main criteria used when developing the IVAM method were global data availability, fully quantitative and fitting the LCA framework. The formula's proposed for biodiversity were:

Ecosystem Occupation $EO = A \times t \times (\alpha_{ref} - \alpha_{act}) / \alpha$ (1)

Ecosystem Transformation $ET = A \times (\alpha_{ini} – \alpha_{fin}) / \alpha_{ref}$ (2)

where:
A = area occupied/transformed
t = occupation time,
ref = reference state (here: the present maximum in the area),
act = actual state during occupation,
ini = initial state
fin = final state
α is the biodiversity quality expressed in terms of vascular plant species

Interpretation of the reference state

The present maximum species diversity is chosed as a reference state. This is the highest present biodiversity state in the region[1]. This implies assessing only relative biodiversity differences in the region. Thus for occupation only the relative distance between the actual and the maximum present potential biodiversity is assessed:

"15% of potential biodiversity in the tundra is as bad as 15% of potential biodiversity in a tropical rainforest"

For transformation the difference between the loss of biodiversity and the same reference is assessed:

"2 species locally lost in an area with potentially 20 species is as bad as losing 20 species in an area with potentially 200 species"

Calculation of species density

The vascular plant species density is calculated from measurements using the Fischer relationship between amount of species S and area (see also figure A1.1):

$$S = \alpha \, LOG^{10}(A_{map}) \quad (3a)$$

or

$$\alpha = (S_{map} – 10) / LOG^{10}(A_{map}), \quad (3b)$$

[1] The region is in the IVAM approach a global physiotope; in the TNO approach it is the potential biome area (A1.4).

based on the postulation that for A_{map} = 1 m^2, S_{map} = 10 (one can find 10 species on 1 m^2, both in pioneer vegetations and on large tropical trees) [van Dobben, 1998],
with A_{map} = the unit mapping area of species diversity measurement, and S_{map} = the measured species diversity S from this data collection set.

Formulas similar to (1) and (2) were proposed for life support impacts in terms of free net primary biomass production.

This approach has been reviewed by CML (see annex 2). Comments were given on using a reference for land transformations within the economy, and on using the formula with α. Also doubts were raised on assessing land use transformations at all. The first two points were acknowledged and taken into account when adapting the method. As irreversible or not reversed land use transformations are assessed only with 2, we remain to assess these in LCA, using a novel way to attribute land use transformations to a functional unit (see section 3.1 of the main report) and a new way to interprete them (see A1.4 below)

A1.2 Methodology Köllner & Eco-Indicator 99

Köllner has done a PhD on land use impacts on biodiversity [Köllner, 2001]. In his thesis, he analysed Swiss vascular plant species richness data on a local and regional scale, and derived linear and non-linear local and regional equivalency factors for occupation and transformation valid for regions like the Swiss Alpes and planes. The linear formulas for local impacts used for this were:

For occupation:

Occupation damage on species diversity D_{occ} = A x t x EDP (4)

with Ecosystem damage potential (EDP) = 1 – ($S_{occ, type}$ / $S_{reference}$) (5)

where $S_{occ, type}$ = the species number in the occupied land type
$S_{reference}$ = the *average* species number in the region

For transformation:

Ecosystem damage potential D = A x t x (EDP_{ini} + EDP_{fin})/2 (6)

Only the so-called absolute transformation (an average of the situation before and the situation after a transformation, [Köllner, 2001] p. 57) is taken into account. In our point of view, this largely overlaps with the occupation results[2]. The net transformation (fin – occ) assessed by IVAM is not further elaborated in [Köllner, 2001], although it is suggested that for irreversible impacts very long renaturation times should be used.

Additional to these local impacts, regional impacts are taken into account in [Köllner, 2001]. According to Köllner the local impacts assess better ecosystem functioning, whereas the regional impacts assess better the conservation value of nature. The linear function for regional impacts on all vascular plant species is simply:

$EDP_{regional}$ = b (7)

Interpretation of the reference state

By using the average species number in the region as a reference for occupation, $S_{occ, type}$ may be larger than S_{region}, resulting in negative figures. We interprete this reference qualitatively as:

"When a land use occupation results in a higher species diversity than the average in the region, this has a positive contribution to ecosystem functioning because otherwise man would have used the land

[2] We interprete Köllners 'absolute transformation impacts' as "the average occupation impacts if the quality would change linearly from the initial to the final state".

more intensively than the average. Therefore this land use has a positive contribution to the environment"

For the linear transformation formula the same reference is used.

For the regional impacts (on the nature conservation value), the number of species around 1850 (when species diversity was at a high level) is taken as a reference. Interpretation:

"The extent of loss of species since ~1850 is a measure of the loss of nature value".

Although this reference does not show up in the formula, it does determine the calculated b values on the regional level.

Calculation of species density

S is calculated by [Köllner, 2001] with the Arrhenius species-area relationship (see also figure A1.1):

$$S = aA^b \qquad (8a)$$

with a = a constant (fitting parameter; S at A=0) and
b = the 'species accumulation factor' (fitting parameter, 0<b<1)

When b[3] and the number of species recorded according to a certain mapping scheme (S_{map}) are given for a certain region/ecosystem and type of land use, and the Arrhenius formula is appropriate, the number of species standardised to a certain area (S_{stand}) is calculated with:

$$\frac{S_{stand}}{S_{map}} = \frac{(A_{stand})^b}{(A_{map})^b} \qquad (8b)$$

S_{stand} (here in ha) allows comparisons of data from different areas of data collection.

Another form of (8a) is $\ln S = \ln a + b \ln A$ (8c)

Köllner gives for the Swiss or comparable situation (unknown significance):

$\ln S = 3.3 + 0.21 \ln A$ for high-intensity land use;
$3.8 + 0.09 \ln A$ for low-intensity land use;
$4.5 + 0.30 \ln A$ for non-use,

or for any land use type an average of

$4.1 + 0.2 \ln A$ (correlation coefficient $R^2 = 0.64$) (8d)

S_{stand} is given in [Köllner, 2001] for 30 land use types in Switzerland, for 5 Swiss regions and for 10 groups of land use types including the ETH Ecoinvent database types II, III and IV, using 0.01 ha as the standardized area [Köllner, 2001]. Not all data are equally significant. See table A1.1 for values of $S^{0.01\ ha}$ using b = 0.2. The values of S for 1 ha are added for illustration only; they are calculated according to formula 8b with b = 0,2.

With b = 0.2 and A_{stand} = 0.01 ha this gives for 8b: $0{,}398 S_{map}/(A_{map}\ [ha])^{0.2}$
or $2{,}512 S_{map}/(A_{map}\ [m^2])^{0{,}2}$ (8e)

In table A1.1 also the impact assessment factors according to formula (5) from [Köllner, 2001] are listed, for 2 different regions (Plateau and Alps). These regions represent the average and maximum reference state, respectively. The standard deviation in these figures are added by us, based on the uncertainty in $S^{0.01\ ha}$.

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[3] The species accumulation factor b determines the curve-shape of the species-area relationship, which is caused by the extent of exchange with the surroundings of the assessed ecosystem (see also figure A1.1). It is actually different for different ecosystems or regions, but an average value is applicable here (see [Köllner,, 2001]).

Table A1.1 Values for $S^{0.01\,ha}$, EDP_{av} and EDP_{max} from Köllner (b=0.2), and S for A_{map} = 1 ha

CORINE nr. *)	Land type	$S^{0.01\,ha}$	st.dev	$S^{1\,ha}$	EDP_{Pla} (av.)	st.dev	EDP_{Alps} (max)	st.dev
	Individual types							
1.1.1	Continuous urban	**8**	**± 4**	20	**0.80**	± 0.4	**0.84**	± 0.42
1.1.2	Discontinuous urban	**22**	**± 9**	55	**0.45**	± 0.18	**0.56**	± 0.23
1.1.3	Urban fallow	**40**	**±13**	100	**0**		**0.20**	± 0.07
1.1.4	Rural settlement	**25**	**± 5**	63	**0.38**	± 0.08	**0.50**	± 0.10
1.2.1	Industrial area vegetation	**24**	**±13**	60	**0.39**	± 0.21	**0.52**	± 0.28
1.2.2.2	Rail fallow	**24**	**± 6**	60	**0.40**	± 0.10	**0.52**	± 0.13
1.2.2.4	Rail embankments	**32**	**±13**	80	**0.20**	± 0.08	**0.36**	± 0.15
1.2.5	Industrial fallow	**40**	**±11**	100	**-0.01**	± .0002	**0.20**	± 0.06
1.3.4	Mining fallow	**38**	**± 6**	95	**0.04**	± 0.006	**0.24**	± 0.04
1.4.1	Green urban	**29**	**±15**	73	**0.27**	± 0.14	**0.42**	± 0.22
1.4.2	Sport facilities	**7**	**± 3**	18	**0.83**	± 0.36	**0.86**	± 0.37
1.5	Build-up land**)	**1**		3	**0.97**		**0.98**	
2.1.1.1	Conventional arable	**10**	**± 6**	25	**0.74**	± 0.44	**0.80**	± 0.48
2.1.1.2	Integrated arable	**7**	**± 5**	18	**0.82**	± 0.59	**0.86**	± 0.61
2.1.1.3	Organic arable	**26**	**±10**	65	**0.35**	± 0.13	**0.48**	± 0.18
2.1.1.41	Fibre/energy crop kenaf	**10**	**± 5**	25	**0.75**	± 0.38	**0.80**	± 0.40
2.1.1.42	Fibre/energy crop hemp	**11**	**± 4**	28	**0.73**	± 0.27	**0.78**	± 0.28
2.1.1.43	Fibre/energy crop Chinese reed	**15**	**± 8**	38	**0.63**	± 0.34	**0.70**	± 0.37
2.2.1.2	Organic orchard	**23**	**± 7**	58	**0.41**	± 0.12	**0.54**	± 0.16
2.3.1.1	Intensive meadow	**17**	**± 9**	43	**0.58**	± 0.31	**0.66**	± 0.35
2.3.1.2	Less intensive meadow	**19**	**±11**	48	**0.51**	± 0.30	**0.62**	± 0.36
2.3.1.3	Organic meadow	**45**	**±20**	113	**-0.14**	± 0.06	**0.10**	± 0.04
2.4.5	Agriculture fallow + hedgerow	**53**	**± 6**	133	**-0.34**	± 0.04	**-0.06**	± 0.01
3.1.1.2	Broad-leafed forest (moist)[4]	**24**	**±10**	60	**0.41**	± 0.17	**0.52**	± 0.22
3.1.1.2	Broad-leafed forest (arid)[4]	**23**	**±10**	58	**0.43**	± 0.19	**0.54**	± 0.23
3.1.4	Forest edge	**48**	**±14**	120	**-0.20**	± 0.06	**0.04**	± 0.01
3.2.1	Natural grassland	**39**	**±15**	98	**0.02**	± 0.008	**0.22**	± 0.08
3.2.2	Heathland	**18**	**± 4**	45	**0.56**	± 0.12	**0.64**	± 0.14
3.2.4	Hedgerows	**44**	**±22**	110	**-0.12**	± 0.06	**0.12**	± 0.06
4.1.2	Peatbog	**19**	**±10**	48	**0.53**	± 0.28	**0.62**	± 0.33
	ETH database							
-	Ecoinvent II	**27**	**±16**	68	***0.33***	± 0.20	**0.46**	± 0.27
-	Ecoinvent III (excluding ↓)	**32**	**±22**	80	**0.20**	± 0.14	**0.36**	± 0.25
-	Excl.from III (green urban, rail embankments, rural settlem.)	**29**	**±14**	73	***0.28***	± 0.14	**0.42**	± 0.20
-	Ecoinvent IV	**18**	**±11**	45	**0.54**	± 0.33	**0.64**	± 0.39
	Reference states							
S_{region}	Swiss plateau	**40**	**± 7**	100				
S_{region}	Jura	**44**	**± 7**	110				
S_{region}	Mountains above timberline	**27**	**± 8**	68				
S_{region}	Swiss alps	**50**	**± 8**	125				

Note: ? means that extrapolation is not possible or not considered valid (formula invalid or parameter b is unrealistic high), empty means no extrapolation performed and – means not applicable.

*) Numbers according to [EEA, 2000]

**) Without empirical basis

The standardisation area 0.01 ha was chosen because the mean measurement area of the data Köllner used was 0.016 ha. We conclude from table A1.1 that there is quite some uncertainty in the values of $S^{0.01\,ha}$. This is due to the natural variability in species density per use type. Finally, it should be

4 Semi-natural forest

remembered that above data was collected in Switzerland, with data from planes and from mountaneous areas.

Eco-Indicator 99

For the Eco-Indicator 99 a draft proposal of Köllner [Köllner, 2000] was applied, using the following formulas for local impacts:

for occupation: $EQ = A \times t \times PDF$ (9)

with $PDF = \{ (S_{reference} - S_{occ,\ type}) / S_{reference} \}$

This formula is the same as Köllner, except that the natural situation in the Swiss lowlands is used as a reference, instead of the average.

for transformation: $EQ = A \times t \times (PDF_{ini} - PDF_{fin})$ (10)

with $\Delta PDF = (S_{ini} - S_{occ,type})/S_{ini}$

This formula is also slightly different from [Köllner, 2000] because the values are not averaged, but the difference between initial and final state is used (as in [Lindeijer et al., 1998]). In the Eco-Indicator database, the transformation data is given the unit m^2 (excluding the t which is still mentioned in the report).

A1.3 Methodology LCAGAPS

The EU LCAGAPS project commissioned mainly by the Danish EPA included a topic on land use [Weidema, 2001]. For the biodiversity impact assessment of an adaptation of the IVAM approach was aimed at, adding ecosystem level impacts to the local species diversity indicator. In the end, the IVAM part was dropped to assess only transformations relative to the potential natural species diversity in the area, next to 3 ecosystem level factors:

Ecosystem occupation = $A \times SR_i \times ES_i \times EV_i \times S_{act,ori} / S_{ref,ori}$ (11)

with
- SR_i (relative Species Richness of biome i) $= S_i / S_{min}$ $(SR \geq 1)$
- ES_i (Ecosystem Scarcity of biome i) $= A_{pot,max} / A_{pot,i}$ $(ES \geq 1)$
- EV_i (Ecosystem Vulnerability of biome i) $= (A_{exi}/A_{pot,i})^{b-1}$ $(EV \geq 1)$

where
- S_I = the species density in biome i
- S_{min} = the species richness in the least species rich biome
- $A_{pot,I}$ = the potential (natural) area of biome i
- $A_{pot,max}$ = the largest value for A_{pot} (to render scores ≥ 1)
- A_{exi} = the existing ecosystem/biome area left
- S_{ori} = the number of indigenous species per unit of area
- S_{ref} = the species density in the reference state
- S_{act} = the actual species density

Data on SR, ES and EV were generated for the biomes[5] of the IMAGE 2.0 model [Leemans et al., 1998]. Also, some normalisation values for Europe were gathered.

Also a different life support indicator was chosen than the IVAM indicator (free net primary (biomass) productivity (fNPP)): net primary (biomass) productivity (NPP). Some data were gathered for that indicator too, including normalisation data.

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[5] Biomes are the major regional groupings of plants and animals discernable at a global scale [Woodward, 2000]

A1.4 Methodology TNO (this project)

For biodiversity, suggestions from the LCAGAPS project [Weidema, 2001] were used to include impacts on the ecosystem level.

A1.4.1 Biodiversity impacts from land occupation

Based on the discussion report from CML (annex 2), we follow the proposal made there for assessing land use for land occupation. This implies combining a local, relative biodiversity score with global relative ecosystem scores (based on [Weidema, 2001]). In this manner, the global perspective and the local details are both assessed:

EO (Ecosystem Occupation) = $A \times t \times SR_i \times ES_i \times EV_i \times SD$ (12)

with local Species Density factor (SD) = $(1 - S_{act,stand} / S_{ref,stand})$ $(SD \leq 1)$ (13)

and ecosystem level factors

SR_i (relative Species Richness of biome i) $= S_i / S_{min}$ $(SR \geq 1)$
ES_i (Ecosystem Scarcity of biome i) $= A_{pot,max} / A_{pot,i}$ $(ES \geq 1)$
EV_i (Ecosystem Vulnerability of biome i) $= (A_{exi}/A_{pot,i})^{b-1}$ $(EV \geq 1)$

where $S_{act, stand}$ = the actual species density, standardised to a certain area
$S_{ref,stand}$ = the species density in the reference state, standardised to a certain area
Si = the species density in biome i
S_{min} = the species richness in the least species rich biome
$A_{pot,Ii}$ = the potential (natural) area of biome i
$A_{pot,max}$ = the largest value for A_{pot} (to render scores ≥ 1)
A_{exi} = the existing ecosystem/biome area left

Choice of species density values for reference states

Formula (13) is determined in a consistent manner by translating data from different mapping schemes to one standardised area.

There are 2 basic formulas for computing area-species relationships: (3) according to Fisher and (8) according to Arrhenius. According to figure A1.1., the Arrhenius curves (using b as parameter) show a more linear relationship than the Fisher curves (using α). In principle, both proposed formulas could be used to translate species density data to a standard area unit, but both b and α are actually ecosystem-dependent, and b should be corrected for the fractal behaviour of (8), which is caused by botanist's inventory artifacts [Witte & Torfs, 2001]. A choice has to be made here on how to standardise the species density data to a unit area.

As we use data on biomes to include the ecosystem level, we should also use the reference species diversity data on this biome level. On the global biome level, some species density data per 10'000 km^2 is available [Barthlott, 1997], see figure 4.1 in the main report. When we standardise the range of species density scores (10 classes with S <100 to ≥5000) to the same standard area as Köllner has applied using formula (8b), the results seem acceptable (<2,5 to ≥126 species per 0.01 ha, compare table A1.1).

For middle-Europe Barthlott gives values of 1000-1500 vascular plant species/10'000 km^2 (25-38 species/0.01 ha), although parts of Switzerland are given values up to 2000 species/10.000 km^2 (50 species/0.01 ha). This is also in line with table A1.1.

Aggregated data for tropical rainforests shows species density numbers of about 400 species per 40.000 m^2 [Turner, 1996] (121 species/0.01 ha) up to 600 species per ha [Jacobs, 1981] (240 species/0.01 ha). In a part of the South American tropical rainforest there are about 390 tree species per km^2 assessed, to which an extrapolation of a factor 3 from woody species to total species can be applied [van Dobben, 1998]: 1170 vascular plant species per km^2 (185 species/0.01 ha). This all seems in reasonable accordance with [Barthlott, 1997]. The maximum value of Turner (about 9000 species per 10.000 km^2 according to formula 8b with b=0.2) is therefore also used as the upper limit of the species density in the reference data.

Figure A1.1a Linear species-area curves (A=1 => S=10)

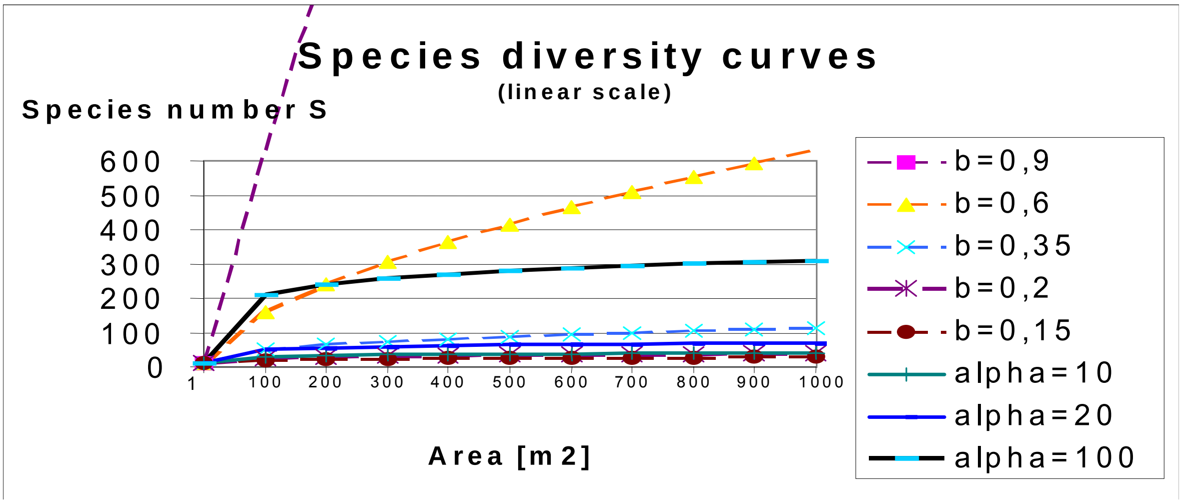
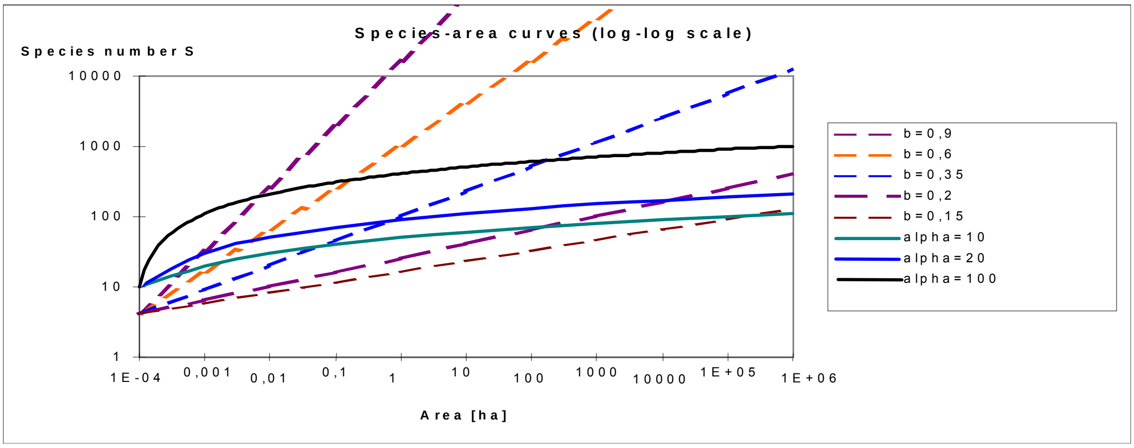


Figure A1.1b Log-log-scale species-area curves (A=1 => S=10)



Based on the above results, we choose to use mainly the data from Barthlott (table A1.2) for the reference state and those of Köllner (table A1.1) for generic cases, as these two sets seem to match fairly well using formula 8b with b = 0.2.

The Atlas Flora Europaeae (AFE, figure A1.3) give vascular plant species density data per 100 km^2 for Europe. It gives values up to 400 species per 100 km^2 (≅25 species/0.01 ha) for Switzerland, and up to 300 species per 100 km^2 (≅19 species/0.01 ha) for the Netherlands [AFE, 2001]. For Switzerland, the mean species density for the Plateau region is 40 species/0.01 ha, and the mean for the Central Alps region is 50 species/0.01 ha. The Plateau mean is more representative for other regions in Europe than the Alps. The maximum species density in Switzerland is about 100 species/0.01 ha (see table 7-5 in [Köllner, 2001]). For the Netherlands as a whole (40.000 km^2, 4e6 ha) there are about 1500 wild species [Witte, 1998], resulting in an average of 28 species/0.01 ha[6]. The maximum score for a Dutch grid cell is 504 species/km^2 (80 species/0.01 ha).

[6] By scaling down the average Barthlott data for the Netherlands (1250 species/10.000 km2 according to [Vogtländer et al., 2001]) to the standardised area 0.01 ha we arrive at 31 species/0.01 ha, which again shows the consistency of standardising the Barthlott data in the same way as Köllner does.

For the above we see that when extrapolating from the Flora Europaeae with formula 8b and b = 0.2 the scores for Europe seem much lower than what is determined on a 1 km^2 or lower grid base. We tentatively conclude that standardisation from this intermediate mapping area with the parameters chosen seems not permissible, and that we have to adapt these values in order to use them. The adaptation performed is to ‘standardise’ the AFE species density for the Netherlands and Switzerland to the average species density according to the more detailed assessments in these countries, and to use the resulting standardisation factor (4.0 for both countries according to above information) to get comparable average species density data for other countries in Europe. Maximum values to be used as a reference can unfortunately not be obtained in this way. For this, the data from Barthlott for Europe have to be used. This species level data is then matched to the biome level maps from [Leemans et al., 1998] to arrive at combined species-biome level impact scores for more detailed reference states in Europe.

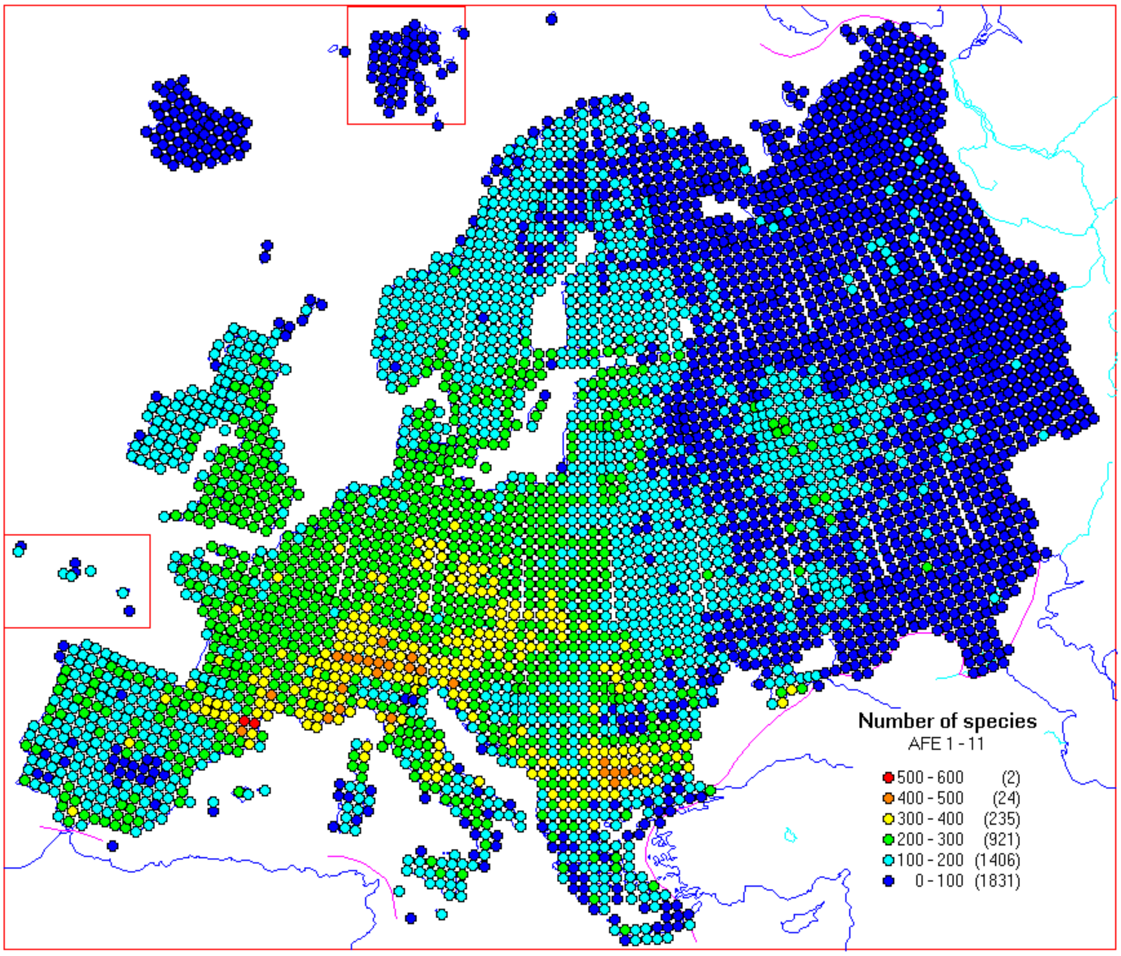


Figure A1.3 Species density map from the Atlas Flora Europaeae [AFE, 2001]

Choice of reference state

A remaining question relating to the reference state is whether to choose the average or the maximum species density in the region to refer to. When using an average value (as in [Köllner, 2001]), the species density due to an activity may be larger than the reference, resulting in a negative score for SD. This can be interpreted as “performing this activity is good for the environment”. When using the maximum in the region (as in [Lindeijer, 1998]) this can never happen, and all activities are more or less damaging to the environment. In fact, both references are possible and have their own reasonable argumentation:

Average reference

"Land use is not like emissions, which are always unwanted. We humans choose to use the land in competition with nature. A certain 'background level' is acceptable[7]. Occupation impacts are then interpreted as preventing land to be at the present average (background) quality level. Due to the high pressure on land, land area used in such a way that more than the average biodiversity level in a region remains is then rewarded in LCA. Therefore, when we do better than average, a negative impact score ('good for the environment') is given for occupation. This goes for every region in the world."

Maximum reference

" Land use may be unavoidable, but any impact on nature should be accounted for in a consistent way[8]. When other impacts are calculated in LCA without taking a threshold into account (zero background), and every intervention is considered, we should not do this differently for land use impacts. Therefore the same background level is chosen as for most other LCIA methods: the equilibrium situation to which nature tends to return. Occupation impacts are then interpreted as preventing land quality to recover to a natural equilibrium. This is the maximum existing[9] nature value in the region. This goes for every region in the world."

As the choice of average versus maximum reference state is very value-laden, we prefer to calculate results according to both reasonings. See table 4.4 in the main report for the calculated results.

For the ecosystem level, there is no difference between choosing an average or a maximum reference; the modelled area from [Leemans et al., 1998] is the same. However, when used consistently, this set of biome level data should be interpretable by valuing one exemplary figure out of the set. This will probably be the tropical forest. So actually the determination of the reference state for tropical forests *is* important. Its maximum is determined for tabel A1.2 using 600 species per ha (9000 species per 10.000 km^2) from [Jacobs, 1981] (in [Lindeijer et al., 1998]), as [Barthlott, 1997] does not give a maximum.

In order to be able to apply the ecosystem factors and reference states, one needs to know in which potential ecosystem a certain activity takes place. One way how to deal with this is to use the map of the potential areas from [Leemans et al., 1998], as was used to determine above species density scores (see figure 4.2 in the main report). However, it may be helpful to simplify the procedure for LCA practitioners. Therefore the potential biome per (part of the) country is determined, and the average/maximum species density per country (normalised to 0.01 ha). The average species density according to the map of [Barthlott et al., 1998], as interpreted using formula 6b with b=2 to standardise to 0.01 ha, is given in table A1.2. The maximum species density is given in tabel A1.3.

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[7] Note for LCA experts: this is an 'only above threshold' viewpoint [Potting & Haushild, 199???].

[8] This is a 'less is better' viewpoint.

[9] We could also have chosen the maximum in history, which would seem more fair between different regions, as some have long ago destroyed most of their nature value. However, this maximum can not be determined with any accuracy (for any land use impact indicator); see also [ten Brink, 2000] on biodiversity indicators for the OECD. By taking the regional historical loss of the original ecosystem area into account in EV, we take this issue into account on the ecosystem level.

Table A1.2 Average (reference) species density for 94 countries, per 10.000 km^2 and per 0.01ha (based on [Barthlott et al., 1997]).

Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01ha}$
Africa			**North&Central America**			**Asia (continued)**		
Algeria	**500**	13	canada	**300**	8	oman	**200**	5
Angola	**1500**	38	costa rica	**4500**	113	pakistan	**1000**	25
Botswana	**500**	13	cuba	**3500**	88	philippines	**3500**	88
Cameroon	**4000**	100	dominican rp.	**2500**	63	saudi arabia	**200**	5
Congo	**2500**	63	greenland	**50**	1	sri lanca	**2000**	50
Egypt	**200**	5	haiti	**2500**	63	thailand	**3000**	75
Ethiopia	**1500**	38	honduras	**3000**	75	turkey	**1500**	38
Gabon	**3000**	75	jamaica	**2500**	63	vietnam	**4000**	100
Ghana	**1000**	25	mexico	**2500**	63	**Europe**		
Guinea	**3500**	88	panama	**5000**	126	albania	**2000**	50
Kenya	**1500**	38	united states	**1000**	25	austria	**1500**	38
Liberia	**1750**	44	**South America**			bulgaria	**2000**	50
Libya	**200**	5	argentina	**1000**	25	czechoslov.	**1250**	31
Madagascar	**2750**	69	bolivia	**2000**	50	finland	**500**	13
Mauretania	**200**	5	brazil	**2000**	50	france	**1500**	38
Morocco	**1500**	38	chile	**1000**	25	germany	**1250**	31
Mozambique	**1500**	38	colombia	**2500**	63	greece	**2000**	50
Nambia	**1000**	25	french guiana	**1500**	38	ireland	**750**	19
Niger	**200**	5	guyana	**2500**	63	italy	**1750**	44
Nigeria	**1500**	38	peru	**3000**	75	netherlands	**1250**	31
sierra leone	**1500**	38	suriname	**2000**	50	norway	**500**	13
south africa	**2000**	50	venezuela	**2000**	50	poland	**1250**	31
Sudan	**500**	13	**Asia**			romania	**1500**	38
Tanzania	**1500**	38	china	**1500**	38	spain	**1500**	38
Tunesia	**1000**	25	cyprus	**1750**	44	sweden	**500**	13
Uganda	**2000**	50	india	**1000**	25	switzerland	**1600**	40
Zaire	**2000**	50	indonesia	**4000**	100	UK	**1000**	25
Zambia	**2000**	50	iran	**1000**	25	yugoslavia	**1500**	38
Zimbabwe	**1500**	38	iraq	**500**	13	USSR	**750**	19
			japan	**1500**	38	**Oceania**		
			north korea	**1500**	38	australia	**1000**	25
			south korea	**1750**	44	nw. caledonia	**2500**	63
			laos	**3000**	75	new zealand	**500**	13
			malaysia	**4000**	100	new guinea	**4000**	100
			mongolia	**400**	10	solomon isls.	**2500**	63

Table A1.3 Maximum (reference) species density for 94 countries, per 10.000 km^2 and per 0.01ha (based on [Barthlott, 1997]).

Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$
Africa			**N/C-America**			**Asia (contin.)**		
Algeria	**2000**	50	canada	**1000**	25	oman	**500**	13
Angola	**2000**	50	costa rica	**5000**	126	pakistan	**3000**	75
Botswana	**1000**	25	cuba	**4000**	100	philippines	**4000**	100
Cameroon	**5000**	126	dominican rp.	**3000**	75	saudi arabia	**1000**	25
Congo	**4000**	100	greenland	**100**	3	sri lanca	**3000**	75
Egypt	**500**	13	haiti	**3000**	75	thailand	**4000**	100
Ethiopia	**2000**	50	honduras	**4000**	100	turkey	**2000**	50
Gabon	**4000**	100	jamaica	**3000**	75	vietnam	**5000**	126
Ghana	**1500**	38	mexico	**5000**	126	**Europe**		
Guinea	**4000**	100	panama	**9000**	226	albania	**3000**	75
Kenya	**3000**	75	united states	**2000**	50	austria	**2000**	50
Liberia	**2000**	50	**S-America**			bulgaria	**3000**	75
Libya	**1500**	38	argentina	**2000**	50	czechoslov.	**1500**	38
Madagascar	**4000**	100	bolivia	**9000**	226	finland	**1000**	25
Mauretania	**500**	13	brazil	**9000**	226	france	**3000**	75
Morocco	**2000**	50	chile	**2000**	50	germany	**1500**	38
Mozambique	**3000**	75	colombia	**9000**	226	greece	**3000**	75
Nambia	**1500**	38	french guiana	**2000**	50	ireland	**1000**	25
Niger	**500**	13	guyana	**4000**	100	italy	**2000**	50
Nigeria	**2500**	63	peru	**9000**	226	netherlands	**1500**	38
sierra leone	**2000**	50	suriname	**4000**	100	norway	**1000**	25
south africa	**5000**	126	venezuela	**4000**	100	poland	**1500**	38
Sudan	**1000**	25	**Asia**			romania	**2000**	50
Tanzania	**4000**	100	china	**9000**	226	spain	**3000**	75
Tunesia	**1500**	38	cyprus	**2000**	50	sweden	**1000**	25
Uganda	**4000**	100	india	**4000**	100	switzerland	**2000**	50
Zaire	**4000**	100	indonesia	**9000**	226	UK	**1500**	38
Zambia	**4000**	100	iran	**2000**	50	yugoslavia	**3000**	75
Zimbabwe	**3000**	75	iraq	**1000**	25	USSR	**1500**	38
			japan	**2000**	50	**Oceania**		
			north korea	**2000**	50	australia	**4000**	100
			south korea	**2000**	50	nw caledonia	**3000**	75
			lao	**4000**	100	new zealand	**1000**	25
			malaysia	**9000**	226	new guinea	**9000**	226
			mongolia	**1000**	25	solomon isls	**3000**	75

Finally, the reference data at mining sites in the various countries have also been determined, in [Vogtländer, 2001] (table A1.4).

Table A1.4 Reference species density at mining locations in 94 countries, per 10.000 km^2 and per 0.01ha (taken from [Vogtländer et al., 2001] and based on [Barthlott, 1997]).

Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$	Country	$S_{10.000\ km2}$	$S_{0.01\ ha}$
Africa			**North&Central America**			**Asia (continued)**		
Algeria	1500	38	canada	1000	25	mongolia	1000	25
Angola	1500	38	costa rica	5000	126	oman	100	3
Botswana	500	13	cuba	3000	75	pakistan	500	13
Cameroon	5000	126	dominican rp.	2500	63	philippines	4000	100
Congo	4000	100	greenland	100	3	saudi arabia	100	3
Egypt	200	5	haiti	2500	63	sri lanca	2000	50
Ethiopia	1500	38	honduras	3000	75	thailand	3000	75
Gabon	4000	100	jamaica	2500	63	turkey	1500	38
Ghana	1500	38	mexico	3000	75	vietnam	4000	100
Guinea	1500	38	panama	5000	126	**Europe**		
Kenya	1500	38	united states	1500	38	albania	2000	50
Liberia	2000	50	**South America**			austria	1500	38
Libya	200	5	argentina	1000	25	bulgaria	2000	50
Madagascar	3000	75	bolivia	5000	126	czechoslov.	1500	38
Mauretania	100	3	brazil	2000	50	finland	500	13
Morocco	1500	38	chile	1500	38	france	1500	38
Mozambique	1500	38	colombia	4000	100	germany	1250	31
Nambia	500	13	french guiana	3000	75	greece	2000	50
Niger	100	3	guyana	3000	75	ireland	1000	25
Nigeria	2000	50	peru	5000	126	italy	2000	50
sierra leone	2000	50	suriname	3000	75	netherlands	1250	31
south africa	3000	75	venezuela	3000	75	norway	500	13
Sudan	100	3	**Asia**			poland	1000	25
Tanzania	3000	75	china	1500	38	romania	1500	38
Tunesia	1500	38	cyprus	2000	50	spain	1500	38
Uganda	2000	50	india	1500	38	sweden	500	13
Zaire	2000	50	indonesia	4000	100	UK	1000	25
Zambia	1500	38	iran	1000	25	yugoslavia	1500	38
Zimbabwe	1500	38	iraq	200	5	USSR	1000	25
			japan	1500	38	**Oceania**		
			north korea	1500	38	australia	1000	25
			south korea	1500	38	nw caledonia	3000	75
			lao	3000	75	new zealand	1000	25
			malaysia	3000	75	new guinea	4000	100
						solomon isls	3000	75

Calculating species densities for cases

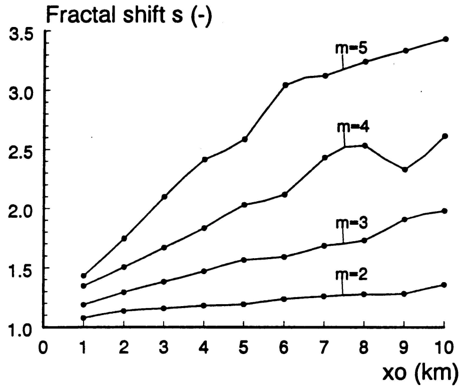
For specific LCA cases we divide by the baseline (reference state) for the same region where the case is in (formula 13). For Europe, the case data from Köllner have high representative value, although they are only collected in Switzerland. Because both the case data from Köllner (S_{act}) as the reference data (S_{ref}) are standardised in the same way, the relative impact factor (1 - S_{act}/S_{ref}) using the Swiss reference state is more generally applicable. Both an average reference ('plateau') and a maximum reference ('central-alps') in accordance to our system have been calculated in [Köllner, 2001]. This makes them applicable at least for most European cases.

Other case data would range from 10x10 m (0,01 ha) to about 1000x1000m (100 ha, 1 km^2) and may result from existing databases or be measured according to local mapping schemes. These mapping schemes influence the determined species density [Witte & Torfs, 2001], and may not be compared to the data from Köllner directly, as with the AFE data. By relating to the national average or maximum (using the same mapping scheme) this problem can be solved. The relative factor (1-

S_{act}/S_{ref}) can then be multiplied directly with the values for the ecosystem level valid for the region at stake.

When such national average or maximum is not known in the same mapping scheme, a different procedure could be followed. When the difference between case and national mapping scheme is not too big, a correction factor can be applied to align the mapping schemes. Instead of the Arrhenius formula, a more precise fractal shift factor may be derived from the figure below, from an article produced within the same Delft Cluster research programme as this project [Witte & Torfs, 2001]: The factors are stated to be valid for any occupancy measure.

Figure A1.4 Fractal shift factors when scaling down with cell width m times that of the original mapping scheme Xo (from [Witte & Torfs, 2001])



This figure can be extrapolated to smaller mapping schemes than 1, although large uncertainties may occur when going below 200 x 200 m. This figure can therefore only be used to scale down species density values in the range 0,4 to 50 km mapping cell width, up to a factor 5. For relating to the standardised area of Köllner (10 m cellwidth) this is not sufficient.

Finally, when the above two options are not feasible, case data can be scaled down to 0.01 ha using formula 6b with b=0.2, and then related to the reference according to table A1.2.

A1.4.2 Biodiversity impacts from land transformation

For land transformation, the same approach as for land occupation is chosen: using three ecosystem level factors and one species level factor. The species level factor is in accordance with the proposal from CML in annex 2 and with the Eco-Indicator 99. Here, use of a reference state for the species level is not necessary for the interpretation (as for occupation). For regional differentiation it is also not required. The argument of regional differentiation was used in [Lindeijer, 1998] to include S_{ref} in the transformation formula. As regional differentiation is now included on the ecosystem level, only a reference to the initial state is required. This results in a formula comparable to but still different from that for occupation[10]:

ET (Ecosystem transformation): $A \times SR_i \times ES_i \times EV_i \times (S_{ini,stand} - S_{fin,stand}) / S_{ini,stand}$ (14)

[10] Note that increasing the species density from 100 to 500 gives a transformation score of –4x, whereas a change from 500 to 100 gives a transformation score of 0.8x. This non-symmetrical result is unsatisfactory in the end; the former choice of IVAM ER to use the reference state to give a relative score seems superior in this respect. When for transformation the choice for a relative score would be abandoned, also a symmetrical result would occur. Both solutions have their specific drawbacks; further discussion and harmonisation are encouraged here.

For this formula, the same data from tabel A4.1 can be used, applying the appropriate land uses for the initial and final state.

Interpretation of transformation and arguments for the formula

Transformation is interpreted as the (contribution to a) trend. This is the trend in not reversed land changes. On a regional level (using normalisation data) this is the total sum (net) of all gross land changes together, during one year. Transformation impacts on a regional level then relate to the regional trend in land quality changes (in m^2 net changed quality per year). This gives additional information to the normalisation of occupation impacts (in m^2 prevented to recover to an average (economic) or maximum (natural) equilibrium for a year). Because transformations are especially important on a global level, the ecosystem level is included too.

Following the above reasoning, the formulas for occupation impacts and for transformation impacts allow for interpretations on two global aspects: (1) preventing recovery to an equilibrium, and (2) contributing to a trend in land quality change, respectively.

A1.4.3 Life support impacts from land occupation

The life support system of the earth consists of all essential life processes that determine the carrying capacity of ecosystems. These processes constitute the prerequisites for higher forms of biodiversity, and allow for use functions of the environment for humans. There are various types of impact related to the life support system, depending on how many different life support functions are distinguished and how many indicators are chosen. Examples of functions and their geographical scale are [Schouten et al., 1997]:

- CO_2-production and -assimilation with solar energy to oxygen — global
- Primary production of biomass — local to global
- Decomposition of organic carbon — local
- Ecological regulation — local to regional
- Other abiotic regulation functions — local to regional

According to [Lindeijer et al., 1998] biomass productivity is an adequate first indicator to assess impacts to life support functions not covered by other LCA indicators, as it is globally applicable. In the IMAGE model of RIVM [Leemans at al., 1998, IMAGE team, 2001] the net primary productivity (NPP) is used as one of the limited number of impact indicators in their global integrated impact assessment. In [Weidema, 2001] it is stated that biomass productivity is also a good indicator for the potential for agriculture (see the marked resemblance between the worldmap on the marginality of agriculture [Cassel-Gintz et al., 1997] and that of NPP [GLO-PEM, 2001], figures A1.5a and b), and is a more or less satisfying indicator for many biogeochemical substance and energy cycles. NPP provides the basis for maintenance, growth and reproduction of all heterotrophs (consumers and decomposers): it is the total food resource flow on earth [Vitousek et al., 1986]. In other words, NPP is the nutritional basis for the diversity of animals and other organisms which are not capable of converting solar energy into chemically stored energy [Bittermann & Haberl, 1998]. Organic matter plays a key role in topsoil formation, and vegetation cover determines topsoil conservation. According to the Oak Ridge NPP database text [Oak Ridge, 2001], NPP is a fundamental ecological variable because (1) it measures the energy input to the biosphere and terrestrial CO_2 assimilation, and (2) it indicates the condition of the land surface area and status of a wide variety of ecological processes. In [Lindeijer et al., 1998] the choice was made to assess only that part of the biomass that is left to nature after harvesting by humans (the so-called free net primary production fNPP, assessing the human appropriation of NPP [Bittermann & Haberl, 1998]). The formulas to assess life support impacts via biomass according to IVAM ER [Lindeijer et al., 1998] was:

Life Support occupation $(LO) = A \times t \times (fNPP_{ref} - fNPP_{act})$ (15), and

Life Support transformation $(LT) = A \times (fNPP_{ini} - fNPP_{fin})$ (16)

Weidema prefers to use the Net Primary Production (NPP) itself, as many life support functions also benefit from the biomass that is harvested later by humans. Calculating NPP is also easier than first subtracting the harvested part. See also annex 2 for a discussion on this issue by CML. The suggestion was made there to focus on the human-related life support functions, in accordance with Weidema. This proposal has been taken over here. Calculating both the free NPP and the NPP itself would allow for a comparison of the results, but interpretation of differences would be difficult. It depends on each specific function whether fNPP or NPP is a better indicator. Quantification of the relationship between each function and the (f)NPP indicator would be required to interprete a difference in detail. Natural variation also limits a sensible comparison of both, and experience from the EU project Biofit has shown that it is very hard to perform consistent measurements for fNPP [Weidema, 2000]. Therefore we adopt the NPP as main life support indicator, while recognizing its limitation.

Applying ecosystem level factors are not relevant here, because the contribution of biomass to the global life support system is considered to be our main focus, and this is more or less dependent of its location. On the local level the biomass difference between the situation before and after a land use process is measured by the transformation indicator. For a case of occupation only the diffference between the actual biomass level during the land use process and the reference biomass level of the various biomes is considered. This implies a moderate regional differentiation, comparable to that of the biodiversity indicator. The formulas to be applied are therefore similar to those from [Lindeijer et al., 1998], except for leaving out the f (for free):

LO (Life Support occupation) $= A \times t \times (NPP_{ref}-NPP_{act})$ (17), and

LT (Life Support transformation) $= A \times (NPP_{ini}-NPP_{fin})$ (18)

Important is how to calculate NPP. This point is addressed below in the next paragraph.

Choice of biomass productivity values for reference states

According to Nabuurs in [Lindeijer et al., 1998] (and ultmately in [Oak Ridge, 2001]) the Net Primary Productivity (NPP) is measured most adequately according to the ‘UNEP project method’:

NPP = change in biomass + change in dead matter + rel. rate of decomp. x total dead matter (19)

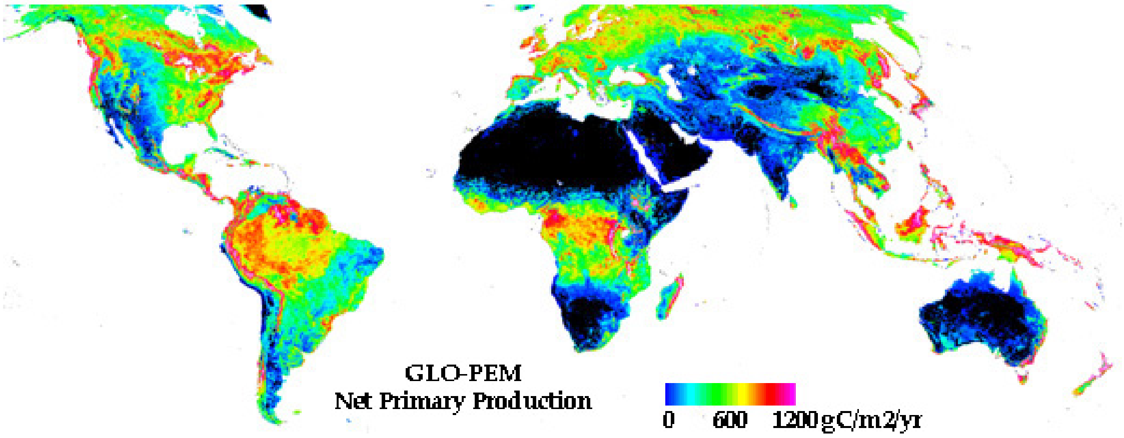
This applies to aboveground biomass and to belowground biomass. However, belowground biomass used to be ignored (especially data from the 1960’s and 1970’s [Oak Ridge, 2001][11]), or estimated with a so-called shoot-root ratio (default ratio between aboveground and belowground biomass). In practice, the situation is even worse as aboveground NPP (ANPP) is sometimes mistaken for aboveground biomass. ANPP is often determined in a more simple manner (according to [Oak Ridge, 2001]). The method to determine the NPP values may not be well-recorded, risking mis-use when comparing data from different sources. Therefore, we apply one source for reference data on NPP. There are three sets of global maps on NPP known to us: the one in [Lindeijer et al., 1998], the ones from the IMAGE model (versions 2.0 and 2.1), and the one from GLO-PEM (figure A1.5a below).

The maps from IMAGE and from GLO-PEM are expressed in weight of carbon per m^2 per year, whereas the map from [Lindeijer et al., 1998] is expressed in weight of biomass per ha per year. For the various regions the difference between weight of biomass and weight of carbon (C) in biomass varies between a factor 1.3 and a factor 2.4 (comparing the map in [Lindeijer et al., 1998] with the NPP map for the year 2000 from IMAGE [IMAGE team, 2001]). According to [Vitousek et al., 1986] this factor would have been between 1.96 and 2.7, taking 2.2 for their own calculations. Ideally, data on biomass should be used, because this seems to have a closer relationship with the impacts we try to assess, and no conversion factors have to be used (introducing further uncertainty). Therefore we compared the data from [Lindeijer et al., 1998] (using their table 4 in annex 1 from [Ajtay et al., 1977]) with that of [Amthor et al., 1998] (additional data from the Oak Ridge NPP database) and from [Olson et al., 2001] expressed in carbon. The IMAGE NPP map is included in the main report, to compare with the map from [Lindeijer et al, 1998]. The two maps are nearly identical, except for a different expression: $gC/m^2.y$ versus Mg biomass/ha.y. The average data from [Ajtay et al., 1977] is applied in our regionally differentiated impact assessment (fourth column of table A1.5).

[11] Using the term NPP should presenty imply including belowground biomass productivity according to [Oak Ridge, 2001].

Note that the areas of high agricultural marginality (with risk of erosion due to natural limitations) in figure A1.5b match on average the areas with low NPP from figure A1.5a. This illustrates one of the valuable types of information NPP gives (although standing biomass might have given a similar picture too).

a)



b)

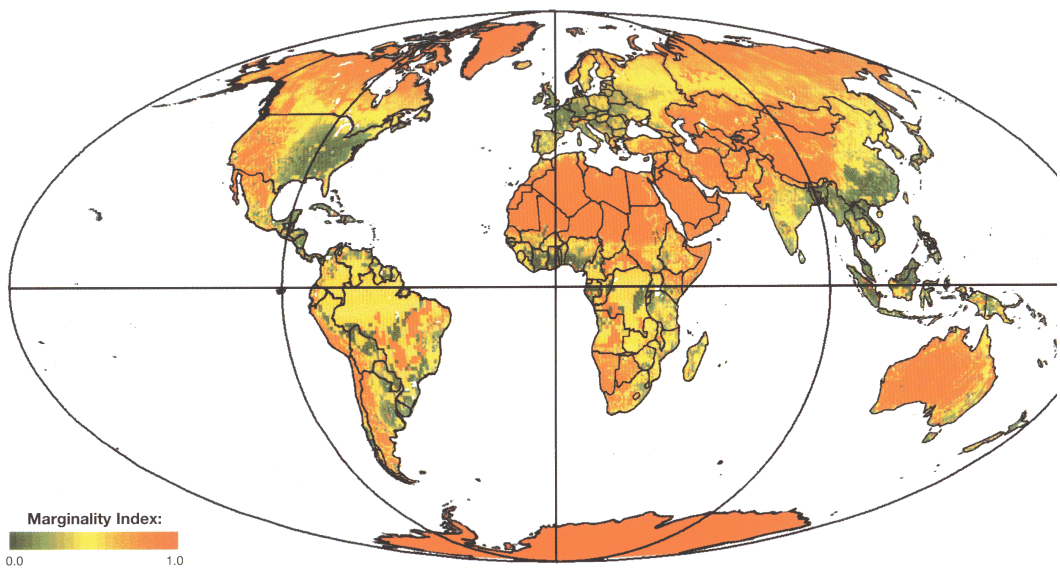


Figure A1.5 World maps of (a) global NPP figures [GLO-PEM, 2001] and (b) the extent to which agriculture is marginal due to natural limitations (according to [Cassel-Gintz et al., 1997])

Table A1.5 NPP data per biome, adapted from [Ajtay et al, 1977]* (average and range), [Amthor et al., 1998] and [Olson et al., 2001]*****

Biome	**NPP [tonnes biomass/ha.y]***	**NPP [tonnes biomass/ha.y]***	**NPP [g biomass /m^2.y]**	NPP [gC/m^2.y]**	NPP [gC/m^2.y]***
Tundra/alpine total	**1.4**	**0.1 – 4.0**	**10 – 400**	105	
Wooden tundra	**4.0[4]**		**400**		
Boreal forest	**8**	**4.0 – 20**	**400 – 2000**	355[2]	0 – 2446
Cool conifer forest	**10**		**1000**		
Temp. Deciduous forest	**12**	**6.0 – 25**	**600 – 2500**		
Temp. Mixed forest	**12**	**6.0 – 25**	**600 – 2500**	670	
Warm mixed forest	**14[1]**		**1400**		
Grassland/Steppe	**6**	**2.0 – 15**	**200 – 1500**	350	
Savanna	**9**	**2.0 – 20**	**200 – 2000**	790	67 – 603
Hot desert	**0.9**	**0.0 – 2.5**	**0 – 250**	11 – 67	0 – 315
Scrubland	**7**	**2.5 – 12**	**250 – 1200**	67 - 360	0 – 234
Tropical woodland	**12[5]**		**1200**	>700	
Tropical forest	**22**	**10 – 35**	**1000 – 3500**	925	(see above)
Agricultural land	**6.5**	**1.0 – 40**	**100 – 4000**	425	141 – 653
Fertilised meadow	**9**	**3.0 – 65**	**300 – 6500**		
Wetlands	**24[3]**			1180	
Swamps & marshes	**30**	**8.0 – 60**	**800 – 6000**		
Lakes/streams	**4**		**400[1]**	200	12
Marine (seas)	**2.5**		**250[1]**		
Human area	**2**		**200[1]**	100	7

[1] Average for total mixed forest calculated from [Vitousek et al., 1986] and applied to warm mixed forest

[2] 218 – 912 gC/m^2.y based on a dataset of 24 sites in Canada, China, Finland, Russia, Siberia, Sweden and Alaska [Oak Ridge, 2001]

[3] Calculated from gC/m^2.y by using a conversion factor of 2

[4] Taking the upper boundary of the data for tundra and alpine from [Ajtay et al., 1977]

[5] Taking the upper limit of the data for scrubland from [Ajtay et al., 1977]

Using above data and the biome model according to [Leemans et al., 1998] reference NPP data were calculated for 46 countries and regions. See table A1.6.

Table A1.6 Reference NPP data based on data from table A1.5 and the biome model from [Leemans et al., 1998]

Country	$S_{10.000\ km2}$	Country	$S_{10.000\ km2}$	Country	$S_{10.000\ km2}$
Africa	**650**	**N/C-America**		**Europe**	**1030**
algeria	**149**	canada	**542**	austria	**904**
angola	**945**	jamaica	**1800**	france	**1236**
egypt	**100**	mexico	**668**	germany	**1200**
gabon	**1540**	united states	**855**	italy	**1190**
guinea	**960**	**S-America**	**1210**	netherlands	**1200**
kenya	**765**	brazil	**1345**	norway	**785**
libya	**101**	chile	**892**	poland	**1200**
mauretania	**100**	peru	**1947**	switzerland	**983**
morocco	**197**	**Asia**	**1580**	USSR	**1101**
niger	**90**	china	**771**	GUS	**765**
south africa	**542**	india	**900**	ukraine	**1200**
Oceania		indonesia	**2144**	north-europe	**875**
australia	**647**	malaysia	**2200**	west-europe	**1220**
Climate ranges		saudi arabia	**105**	east-europe	**1170**
temperate	**1020**	kazachstan	**760**		
tropics	**1030**	middle-east	**390**		

Calculating biomass productivity for cases

For single cases, the NPP could be actually measured, although this requires a lot of sampling and drying the (wet) biomass samples to constant (dry) weight. For belowground biomass, the measurement is even more difficult [Oak Ridge, 2001].

When belowground NPP is not given, the ratio aboveground: belowground of 12 for the tropical forests and 3 for boreal forests can be used as a default. The ratio total biomass to total NPP is for boreal forests on average 16 (for aboveground only it is 19; for belowground only it is 12). These values are calculated from the NPP database (from [Clark et al., 2001] and [Gower et al., 2001], respectively). For grassland in Argentina, the formula Total NPP ($gC/m^2/yr$) = 1.54*ANPP + 111.7 was applied [Olson et al., 2001]. In the same reference, a ratio of 0.50 Belowground NPP:Total NPP for grasslands, deserts, and tundra and a ratio of 0.22 BNPP:TNPP for boreal, tropical and temperate forests was derived.

When data is collected in carbon, a default conversion factor of 2.2 to biomass can be used (45% C in dry organic matter). In [Barrett, 2001] the values 42% for herbaceous vegetation and 45% for woody vegetation are used. In [Olson et al., 2001] these vales are 45% and 50%, respectively. When data is collected in energy terms (kilocalories, kCal) the conversion factor is dividing by 5 [Vitousek et al., 1986].

Data may also be given in wet weight (without drying to constant weight). In such cases, conversion to dry weight have to be made. For aquatic ecosystems, a conversion factor from wet to dry weight of 0.17 to 0.32 is given in [Vitousek et al., 1986]. For benthic invertebrates this factor is in the range of 0.03 – 0.25 (table 3.6.26 in [Lindeboom & de Groot, 1998]). We suggest using a factor of 0.20 for water plants. For trees the factor of fresh weight versus that of dried wood can be taken as a default: 0.6 (0.4 – 0.8 calculated from [Wiselius, 2001]). For non-wooden terrestrial plants an intermediate value of about 0.4 is suggested as default.

Life support impacts from land transformation

For transformation, the difference between annual NPP before and after a transformation is calculated (see formula 18). NPP_{ini} and NPP_{fin} may be natural situations (see table A1.5) or any type of human land use.

Summary of methodological choices made for operationalisation of land use impact assessment

Developing an impact assessment method generally implies methodological choices. These may be more or less important, and some may be hard to acknowledge, leave alone to assess the model uncertainty resulting from making such choices. A different choice may also lead to a different associated data uncertainty. Calculating with sets of indicators based on different choices would be the best way to assess these. This was considered only feasible for a few choices in this project, as this project mainly focussed on data collection. However, it is important to express the choices involved and to show which alternatives could be chosen based on the same data availability.

For the operationalisation of the biodiversity indicator a large number of choices were implied, summarised in table A1.6. Most choices are well argumented but some are value choices which can not be made based on objective or objectified arguments alone. The main remaining value choice is the use of a maximum or an average reference level (cf. IVAM ER respectively Köllner). Both options are left open, with a preliminary preference for a maximum level, because it is most consistent with the 'less is better' paradigm under which most LCIA methods are developed[12].

[12] Note for experts: The 'less is better' paradigm implies that less intervention is always better, even when this is below the 'background level'. The reference state is thus for emissions generally a situation with zero emissions, comparable to the maximum biodiversity level.

Table A1.6 Methodological choices for the biodiversity indicator.

Subject	Choice made	Obvious alternative	Reason for the choice made
Levels included	Ecosystem level and species diversity level	Only species diversity level or only ecosystem level	Large model improvement due to covering 2 of the 3 biodiversity levels[13]
Species included	Only vascular plant species included; equal weighting of all species	Red list (threatened) species, only endemic species or only the original species	Case data only feasible for plants, and for threatened species only for a few countries available; fauna impacts less linearly related to land area; red list criteria inconsistent between countries; insufficient global data on endemic species. Additionally, in most land use types no original species are left, leading to disqualification of the (artificial and non-endemic) species present[14].
Species diversity factor	Score relative to a reference state	Absolute score; no relation to reference state	Species density differences are considered only regionally relevant. For occupation the relative score is imperative to the definition of occupation; for transformation the importance of the trend is also considered dependent on the region. Thus, for land use regionalisation is required.
Species diversity reference state	Maximum and average score for regional biome	-	-
Species diversity standardisation	To 0,01 ha via Arrhenius formula using a = 4,1 and b = 0,2	Via Fisher formula or other parameters	Alternatives would result in other absolute scores, but results would show the same conclusions for all cases; less data treatment required
Reference state data	Taken from Barthlott	Taken from IVAM ER	IVAM ER data standardisation via another route difficult; Barthlott expected to publish data using the Arrhenius formula soon, which is consistent with the standardisation
Factors included on ecosystem level	Scarcity, vulnerability & quality	Take only 1 or 2 factors, or adding natural recovery time as a factor for reversibility	Leaving out quality would lead to an overestimation of ice and desert. Adding the natural recovery time imples partly double-counting when actual renaturation times are included in occupation.
Biome definition for ecosystem level	Taken from Leemans (RIVM)	Taken from Barthlott or others	Adhered to choice made in Weidema & Lindeijer due to consistency and availability of actual and potential biome area[15]

[13] It is estimated that considering only the ecosystem level may cause the least value mismatch if only 1 level would be chosen.
[14] [Weidema e.a., 2001] have made the alternative choice here. One could say that looking at only the original species is the most biocentric, and taking all species is a very anthropocentric viewpoint. Uncertainty on what would be the original state and the corresponding increased data treatment requirement have contributed to choosing all species in this study.
[15] Different biome classifications may lead to considerable differences in scores. The model uncertainty is comparable to those of choices on the species level, but the consequences are probably larger, as the range in scores is a factor 10 higher than using the IVAM formula [Weidema e.a., 2001].

Considering the life support indicator, there are only a few methodological choices identified, in table A1.7:

Table A1.7 Methodological choices for the life support indicator

Subject	Choice made	Obvious alternative	Reason for the choice made
Basic indicator	Biomass	Soil quality, many other indicators possible	Many indicators possible due to many life support functions; in fact all of them should be included. Here the most obvious and easy one is chosen, conform IVAM ER.
Biomass indicator	NPP	fNPP, ANPP or Biomass/BPP (biological accumulation ratio)	NPP and ANPP (standing biomass) are more anthropogenic; fNPP and the biological accumulation ratio are more biocentric (relating to nature productivity and stability, respectively). fNPP could have been taken next to NPP but difficulties in explaining differences are expected, also due to data limitations.
Biomass factor	Absolute amount of biomass	Relative to a reference state	Biomass is not assessed relatively; a larger amount of biomass productivity is generally considered positive everywhere.

Concluding one can state that a lot of choices have to be made when operationalising indicators for biodiversity and life support. Often data availability is considered the decisive criterion for selecting an indicator. Two major value choices remain:

- a maximum or an average value as biodiversity reference state
- fNPP or NPP as life support indicator

Both are choices regarding a more biocentric or a more anthropocentric viewpoint, respectively. No final choice can be made here, and both values should in principle be calculated. However, for fNPP case data is limited and sometimes conflicting. Therefore, for biomass only NPP is calculated, in accordance with the suggestion in annex 2.

Annex 2 Land use in LCA (a report by E. van der Voet, CML)

Land use in LCA

Ester van der Voet
CML
final report July 13, 2001
CML working paper

Project “Toetsing land use methodiek”, DWW en Delft Cluster.

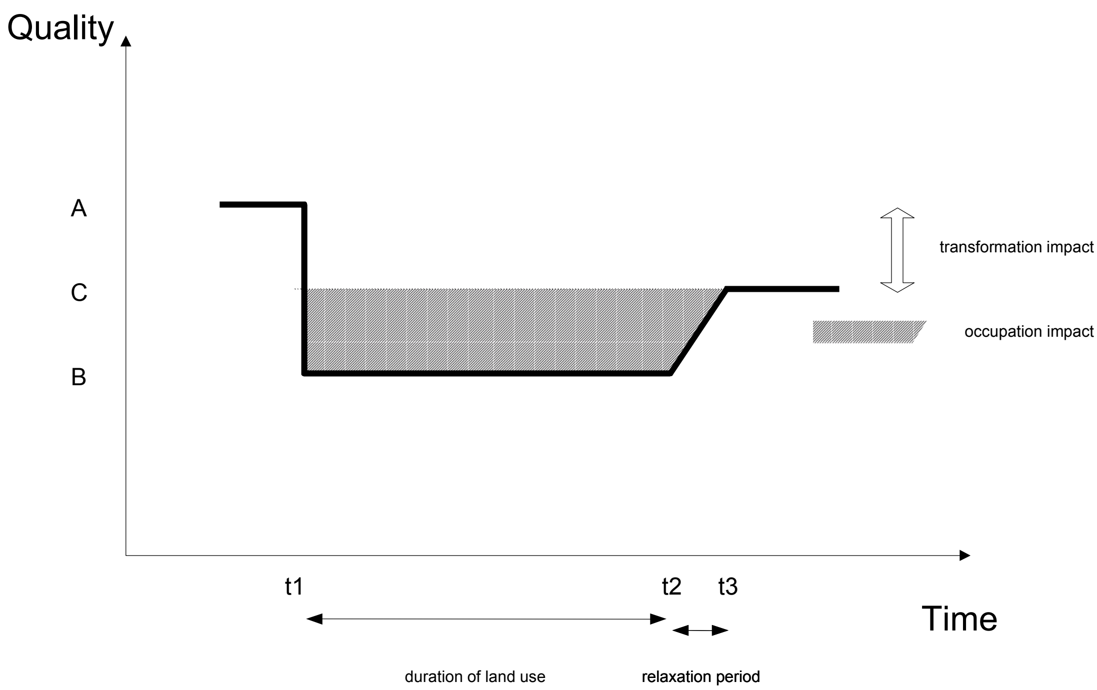
Table of Contents

1 Introduction

Land use has large impacts on the natural environment. Despite this, the attention for this type of impacts within LCA circles has been lagging behind. Recently, some methods have been developed or rather are still in development to include land use impacts in the LCA framework. Three of these are discussed in this paper:

- the IVAM method (Lindeijer et al., 1998) and this report as its succession)
- Köllner's method, as described in his PhD thesis and as included in the Eco-indicator 99 (Köllner, 2001 and Goedkoop & Spriensma, 2000)
- the LCAGAPS method of Weidema (Weidema, 2000).

The central figure used as a starting point for including land use impacts in LCA is the following (SETAC-WIA paper in preparation):



The figure is interpreted as follows: At time t1 land is being taken into use for a certain purpose. As a consequence, the quality of the area decreases from A to B. At time t2 the land use is terminated and recovery starts. At time t3 a new steady state has established itself which may differ from the original state; in this case the quality C is still lower than the original quality A.

It is not easy to operationalise this figure. This essentially dynamic picture must, as long as LCA still is an essentially static tool, be translated into static or steady state terms. This requires:

- a specification of the interventions in the LCA inventory
- the definition of indicators to measure "quality"
- the assessment of the impacts on the defined "quality" of certain types of land use
- the translation of these impacts into practical LCIA equivalency factors.

The methods treat these issues in different ways, as is discussed in the next sections.

2 The LCA inventory

In the Inventory, the process tree for the functional unit must be established. Two steps are required:

1. the definition of unit processes, to establish the interventions per unit of process output
2. the calculation of the required amounts of these unit processes for the functional unit.

If step 1 is made carefully, step 2 is relatively easy.

When dealing with land use, the interventions that must be linked to the unit processes will be in terms of the use of land area for certain purposes:

- extraction of raw materials (mining)
- production processes (agricultural land, area of industrial territory)
- transport (roads)
- use processes (residential area, recreational area)
- waste treatment processes (landfill sites, incineration plant area).

There is a difference between area already used by humans, or area newly extracted from nature. Generally, two types of interventions are distinguished:

- transformation: the conversion at the start from A to B
- occupation: the maintenance of quality B during a certain time period.

This implies that there are three possible situations:

1. Use of land area that already is being used for identical purposes
2. Use of land area that already is being used, but for other purposes
3. Use of land area that up to now was natural area.

A 4th situation could be: returning land that has been used up to now to natural area. In LCA studies, this will be an exceptional situation.
In the situations 2 en 3 there is transformation, a change in land use, that may be evaluated differently. It could even be argued that in situation 2 there is no transformation – the area was already added to the economic system and was no longer part of the environment. In all three situations the intervention occupation can be distinguished, that has a temporal as well as a spatial dimension (m^2.year).

Per unit process the following issues must be specified in the Inventory:

- the area transformed
- the nature of the transformation (from what into what), including irreversibility
- the area occupied
- the time of occupation
- the nature of the occupation.

When defining unit processes, we may encounter several problems:

- it can be difficult or even impossible to establish in practice what the previous use of the involved land area has been, and therefore whether or not transformation is an issue
- there may be problems in attributing a known transformation to the output of a unit process
- there may be allocation problems for occupation as well, in case of multi-functionality of land use.

The last problem is no different from the allocation required for emissions from multi-output processes. For this issue, rules have been established in ISO 14040 and a practical solution based on those rules may be found per case study.

The first and second problem related to unit process outputs are more difficult to handle. As in LCA we tend to abstract from time and location, we will find that it is impossible to link transformation to standardised, abstract processes. Transformation from nature (in figure 1 the jump for quality A to quality B) may have occurred in the distant past and may not be traceable. Even if we do know that in the real world the process of transformation is still going on, it is still not possible to attribute that to a unit process in a straightforward manner due to temporal problems: after transformation has taken place, the soil may be in use for many years to come and may deliver all kinds of outputs that are partly unknown and partly unknowable. In theory, there are two solutions for these problems:

1. Transformation could be attributed to the first activity after transformation, e.g. the first harvest or the first generation of houses. This is proposed by Guinée et al (LCA2 book) as a possible solution, equivalent to the solution for allocation of recycling of raw materials ("recycling of land": Heijungs et al., 1997). Guinée et al. do not elaborate this option. Problems arise especially in case of not directly production / consumption related land use, such as (rail)roads and other public facilities, where there is no apparent "first user" causing the transformation.
2. Transformation can be divided over the total output of the area involved, from transformation out of nature until the ending of human activity altogether. This involves the problem of the unknown past and unknowable future, as described above.

Both options are inconsistent with the LCA philosophy of abstraction from time and location, apart from practical data problems. The three methods discussed in this paper are remarkably silent on this issue. Lindeijer et al. (in prep.) are the only ones who enters this debate. They pose a solution by looking at the trend in land use changes over the latest years (decade?), and attributing the average yearly changes to the yearly outputs of various types of land use. They apply this solution to the Dutch situation. Although this method is applicable it has drawbacks:
The Dutch situation is rather irrelevant from the LCA perspective and cannot be extrapolated easily to other parts of the world.
The use of (net) trends in land use is irrelevant from an LCA perspective, and would lead to outcomes that are difficult to interpret. A negative trend, as is the case for agricultural soil in the Netherlands, would lead to a negative score for transformation into agricultural land, while a null trend would imply ignoring transformation.
The first problem could be evaded by using change on the global level, where there is no negative trend in any type of human land use at present. The second problem might be solved by using trends in changes instead of trends in land use. Not the yearly amount of square kilometres change in the total surface of agricultural soil, but the yearly amount of land being transferred from something else into agricultural soil then are the measure. The yearly amount of land transferred from agriculture into something else thus is out of the equation. It can be proven mathematically that this is identical to a generalisation of attribution to the first user:

A process delivering per unit of time a certain amount of products or services can be linked to transformation as follows:

transformation current process $T(P_t) = 0$
additional transformation at time t $T(\Delta P_t) = T_t$
with
T = transformation (units of transformation per unit product)
T = time t
P_t = production per unit of time
ΔP_t = increase in production per unit of time

In practice, it will hardly ever been known whether a unit product or service has actually caused any changes in land use, which means a generalisation must be made. If we want to generalise, a mix of newly transformed and already occupied area is in order. In equations:

$$T(P_t + \Delta P_t) = \frac{T(P_t)P_t + T(\Delta P_t)\Delta P_t}{P_t + \Delta P_t} = \frac{0 + T_t \Delta P_t}{P_t + \Delta P_t}$$

$$if \Delta P_t << P_t$$

$$T(P_t) = T_t \cdot \frac{\Delta P_t}{P_t}$$

This solution is workable and may stand up in further discussions.

The issue of allocation of transformation is clearly not resolved, and indeed has been given very little attention up to now. In this paper it will not be dealt with further. It is recommended that this issue is given more attention in circles of LCA methodology development such as SETAC and other platforms. Until then, a last possibility is to ignore the intervention of transformation, since at present it seems very difficult to deal with it adequately. This is not a very attractive option in view of the relevance of the issue, especially when dealing with transformation from nature to culture.

3 The LCA impact assessment

3.1 General structure

The general structure of the LCA impact assessment consists of classification, characterisation, normalisation and weighing.

Classification is a limited step, where interventions are being attributed to certain impact categories.

Characterisation means that the contribution per unit of intervention to each of the impact categories is established. The “unit of intervention” in this case refers to, for example, a square meter of a certain transformation, or a m^2.year of a certain occupation. The total contribution of a functional unit to a certain impact category is then calculated as follows:

$$\text{result}_{\text{impact category}} = \sum_{\text{unit process}} \text{characterisation factor}_{\text{impact category,type}} \times \text{intervention}_{\text{unit process,type}}$$

For each of the impact categories, indicators must be defined. The indicators then must be translated into characterisation factors that can be attributed to specific interventions. Two steps are required for this:

1. Per type of intervention, the contribution to certain impact categories must be made explicit. This involved some careful reasoning: as long as LCA is still primarily a static and non-site specific tool, time and location specific impacts must be abstracted into time and location independent terms.
2. Based on (1), characterisation or equivalency factors must be defined. When dealing with emissions, the approach is often used to define a reference emission and express the other emissions contribution to a certain impact category in relative terms (CO_2-equivalents, in the case of global warming). Whether or not to use such an approach for land use interventions as well is debatable. Strictly speaking there is no need for it. IVAM and Köllner do not use equivalency factors. LCAGAPS does by defining a reference ecosystem as unity and expressing its factors relative to this reference.

By multiplying interventions with characterisation or equivalency factors, the many different interventions connected to a functional unit can be added to a limited number of impacts.

The next step is normalisation by relating the result of the characterisation step somehow to the total present size of the impact category, or environmental problem. This ensures a certain harmonisation between the different problem categories: they are no longer expressed in their own units but relative to the total problem as-it-is.

Finally it is possible to assign weighing factors to the different impact categories to make a statement on their relative importance. This step is subjective by definition and is often not applied in LCA studies.

3.2 Models for estimating the impacts of land use

Land use has many different impacts. Starting from the “areas of protection” (WIA document), the following impacts can be distinguished:

1 availability of natural resources:

space as a natural resource: it cannot be depleted, but scarcity can occur, leading to competition over land

land use may cause a decrease of biodiversity, which in turn may decrease the availability of certain resources from biotic origins and of genetic information

2 life support services: certain types of land use may have adverse impacts on the regulatory functions of ecosystems

3 the intrinsic value of nature: land use may cause a decrease of biodiversity on the global or regional level (genes, species and ecosystems).

This document deals with the latter two impacts: the possible decrease of biodiversity (§ 3.2.1) and the possible impairment of life support services (§ 3.2.2) as a consequence of land use. The availability of natural resources is, although relevant, outside the scope of this project. The treatment of these subjects by the three different methods is discussed below.

3.2.1 Impacts on biodiversity

3.2.1.1 The possible impacts of land use on biodiversity

Decrease of biodiversity due to land use is caused by several mechanisms:

1. Impacts due to loss of natural area: by extracting land from nature, there is less space for natural ecosystems and the species dependent on those ecosystems. This may cause a loss of ecosystem, species and genetic diversity.
2. Indirect impacts:
    a. fragmentation: a loss of especially animal species due to the shrinking of the size of the undisturbed habitat, as well as the disturbance by human presence
    b. creating corridors: road shoulders and railway embankments can serve as refugia as well as connecting routes between natural areas, and thus may have a positive influence on the maintenance of species in cultural areas
    c. degradation of the landscape: eliminating structural elements in the landscape such as hedges, cops etc. implies a loss of habitat for smaller species of birds, insects and herbs
    d. change of the abiotic conditions and the microclimate (humidity, wind, nutrients, management regime) as a result of changed land use may lead to changes in the species composition, also of the microflora and -fauna.
3. Impacts of emanation:
    a. prevention of recovery: by occupying potential natural area, the natural situation cannot be reinstalled and the impact of space lost for natural ecosystems is extended
    b. knock-on effect: this occurs when, as is the case is Western Europe, large parts of the land are extracted from nature permanently. The remaining nature is stressed to the point of loss of resilience, resulting in a higher sensitivity for disasters and a lower potential for recovery.
4. Impacts due to changes of land use within the economy: it can be argued that such changes are irrelevant from an LCA point of view – the land already is and will remain under human dominance. On the other hand, by ignoring such changes it becomes impossible to distinguish various forms of land use on their potential for joint use of the land by nature. Specific impacts within this category are:
    a. impacts on agro- / silvi- and industrial ecosystems
    b. impacts on tame species and species adapted to the human habitat.

In order to be able to include this category of impacts, specific cultural ecosystems should be defined and specified.

It will not be possible to specify all impacts in such a manner that they fit within the LCA framework. After all, LCA aims at integrated evaluations of highly stylised product systems and not at providing an accurate picture of the impacts of certain types of land use on a specific location.

3.2.1.2 The impacts included in the various methods

In the IVAM-method the following biodiversity impacts are included:

- the impacts of loss of natural area on the vascular plant species diversity on the involved land area (1)
- the impacts of changes in land use within the economy on the vascular plant species diversity of the involved land area (4a)
- the impacts of prevention of recovery of the vascular plant species diversity on the involved land area by occupation of potential natural area (3a).

The impacts included by Köllner are rather similar. The details of the elaboration are different, see below. In addition, Köllners regional score is related to

- the knock-on effect,

but in a somewhat distant manner, treating cultural ecosystems in the same way as natural ones. In the derived Ecoindicator 99 this impact is included in a more direct manner (for details, see § 3.3).

Weidema &Lindeijer include:

- the impacts of prevention of recovery of the natural ecosystem diversity and the concurrent species diversity on the global level by occupation of potential natural area (3a)
- the knock-on effect (3b).

All methods are incomplete. IVAM and Köllner are more or less alternatives for the same impacts: differences in (vascular plant) species density dependent on the type of land use of the area involved. Weidema & Lindeijer indicate something different. On the one hand they are more encompassing: (1) they explicitly include ecosystems, (2) the impact assessment is related to the global problem of biodiversity loss. In some areas they are less developed: it is not possible to evaluate land use changes within the economy.

3.2.1.3 The use of vascular plant species diversity as a proxy for species diversity in general

All three methods use vascular plant species diversity as a proxy for species diversity in general. This seems a sensible choice, based on practical arguments of data availability. The question is whether vascular plant diversity is a good indicator for total species diversity. On the one hand the assumption is that more plant species means more species in general – this seems to be defendable. On the other hand the assumption is that impacts of land use on plant species diversity also have an impact on other species. This is clearly not always true (for example, fragmentation and disturbance have impacts especially on animals and much less on plants), but in many cases it is, if only because of the dependence of animals on plants. For the future the LCA community might consider expanding the indicator to total species diversity, or to compose indicators for plants, (different groups of) animals, and micro-organisms. At the moment, this does not seem possible. Using plant species diversity, it is clear that a distinction must be made between different types of ecosystems, natural as well as cultural, based on their typical species diversity. How many types are defined is a matter of debate. In view of the purpose – including land use impacts in LCA – it would seem advisable to define a limited number. The IVAM method (1998) distinguishes roughly 10 types of ecosystems (they call it "physiotopes") on the global level, based on climate zones and abiotic factors, each with their own characteristic average, estimated species density. In later publications more are added, especially cultural ecosystems for the Dutch situation (Lindeijer et al., in prep.). Köllner defines about 30 ecosystems, both cultural and natural, for the Swiss situation. Weidema & Lindeijer use "biomes" on the global level and distinguish roughly 15 ecosystems, all natural.

3.2.1.4 Species diversity and ecosystem diversity

All three methods contain a combination of species and ecosystem diversity. The IVAM method and the method of Köllner both work with species density per type of ecosystem. The LCAGAPS method bases its impact assessment on the global scarcity, the species richness and the vulnerability of ecosystems. One question is, whether to separate species from ecosystem diversity. These two variables are linked but indicate different aspects: the global pool of species wherever they may occur, and the global variety in ecosystems apart from the number of species included in them. An ecosystem based indicator has, from the point of view of including land use changes in LCA, some clear advantages. It connects smoothly to the inventory since ecosystems have spatial dimensions. Aspects such as irreversibility and recovery time belong to ecosystems rather than species and might be included there in a more straightforward manner. The LCAGAPS method is the only one of the three that includes aspects of (global) ecosystem diversity. Oddly enough, LCAGAPS is also the only method addressing – be it indirectly – global species diversity by weighing ecosystems based on their natural species richness. Both the IVAM method and Köllners method weigh local changes in species density relative to the species density of the (theoretical) natural or desirable state of the location. There is no link to either ecosystem scarcity or the global species pool. Köllner does include a possibility to deal with species diversity on the regional level by applying his method to rare species. Since these are rare Swiss species the relevance for the global level is limited.

All this would imply that the LCAGAPS proposal theoretically is the best candidate. Problems related to this methods are (1) the choice made within LCAGAPS not to deal with changes within cultured land, and (2) the limitation of LCAGAPS to the occupation intervention. Either IVAM or Köllner may be used to supplement LCAGAPS to be more complete, or the other way round, the weighing procedure of LCAGAPS may be used to increase the relevance of either IVAM or Köllner.

3.2.1.5 The ecological models used in the three methods

The crucial issue in the IVAM method is the species density under various land use regimes. The ecological model behind that is a function describing the relation between surface area and number of species, as inspired by the island theory: a larger area contains a larger number of species, but with decreasing returns. Lindeijer et al. choose a logarithmic function to describe this relation. The formula is $S = \alpha \log A$, with S the number of species and A the surface area. The differences between ecosystems (both natural and cultural) are expressed in the α constant, which is taken as the measure for defining characterisation factors (see § 3.3).

Köllner too uses a function describing the relation between the number of species and the surface area. The function is different from the one used in the IVAM methods: $S = aA^b$, or the logarithmic variant $\ln S = \ln a + b \ln A$, with S the number of species, A the surface area, a a constant representing the typical species richness and b the “species accumulation rate”. Köllner also discusses other functions but selected the one he uses as the best fit to his data. S is then specified for 30 ecosystems, for a standard surface of 100 m^2 and compared to a reference, being the Swiss situation in 1850. Species density for various types of land use is calculated as S_{occ} / S_{ref}. The inverse ($1 - S_{occ} / S_{ref}$) is called S_{lost}, or species potentially lost.

Stating a preference for either of the functions describing the relation between number of species and area is not possible without extensive ecological research. For the time being, Köllner seems to have dedicated most attention to this issue and put most effort in validating his equation. It seems therefore advisable, also in the light of harmonisation, to use the exponential function.

Weidema & Lindeijer propose a method to include impacts on ecosystems in LCA. At first sight this seems satisfactory: (1) it has a clear connection to the inventory because of the spatial dimension of ecosystems, which is lacking when dealing with species, and (2) ecosystems are more encompassing: they include species but also interspecies relations, structural aspects etc. In principle, a good ecosystem measure could make it superfluous to define a separate species indicator. Weidema & Lindeijer propose to weigh natural ecosystems according to the following characteristics:

- their biodiversity, expressed by the number of (plant) species they contain
- their inherent scarcity
- their vulnerability.

For the biodiversity characteristic, Weidema & Lindeijer also use a function describing the relation between area and number of species: $S = cA^z$, with S the number of species, A the surface and c and z “fitting parameters”. Essentially this is the same equation as Köllner uses.

3.2.2 Impacts on the life support function of the biosphere

3.2.2.1 The life support function of the biosphere

The biosphere provides society with goods (fish, wood etc., included in the LCA problem area Depletion of biotic resources), but also with services: a.o. climate regulation on the micro and macro level, regulation of water quantity as well as quality, soil fertility, prevention of erosion, regulation of pests and diseases etc.. These services are connected to different aspects of the biosphere:

- the presence and maintenance of natural biogeochemical cycles (C, N, P, S, O, H_2O). These cycles play an important role in climate regulation and other basic requirements for sustaining human (and other) life and are maintained by organisms. Micro-organisms, plants and animals all have their part in these cycles. The magnitude of these cycles depends on the amount of biomass available to process the substances in question.

- the structure of ecosystems (the presence of layers in the vegetation, soil covering, colour etc.). This structure is determined by the vegetation. It is very important especially for the regulation of water streams and the water storage capacity of the land.
- the presence of specific species, such as predators, reducers, pollinators etc. required to keep certain pests in check, keep soil processes going, or to take care of pollinating crops.

3.2.2.2 Life support for humans or for nature?

In the WIA discussion paper section 4.4.3 this issue is raised. Obviously, life support is provided for both human and other species. Long discussions on this issue have led to the conclusion that it is not possible to define a life support system for non human species in general. The reason for this is that every species has its own demands. We like a high humidity, but desert plants do not. We like an atmosphere with oxygen, but for sulphur bacteria this is different – they like hot water with a lot of S and do not care for oxygen. Also if specific life support functions are discussed, in practice this will often be from a human perspective – climate, erosion, soil fertility etc. In accordance with (Dutch) environmental policy (Ministry of the Environment, 1992) defining life support functions for humans would seem most appropriate.

3.2.2.3 Which life support functions are relevant?

In the WIA document WIA three life support functions are mentioned:

1. the closing of substance cycles
2. climate regulation
3. a properly functioning soil quality

These three issues do not seem to be covering the issue: incomplete, overlapping and partly irrelevant. A (still not complete but less incomplete) list of life support functions for humans, depending on the three aspects of the biosphere, is the following (from Van der Voet et al., 1997):

Maintenance of natural cycles:

- climate regulation (temperature and humidity)
- air purification (CO_2/O_2 management and O_3 / NO_x filtering)
- Soil fertility (for wood and low tech agriculture), a.o. by N fixation
- Soil buffering (by degradation of pollutants)
- Water purification

Structure of ecosystems

- climate regulation, by wind breaking and albedo
- air purification a.o. by dust filtering
- protection against flooding by regulating water streams (providing delays and buffers)
- protection against erosion, a.o. by evaporation, allowing for infiltration and reduction of runoff
- soil structure, by bioturbation

Specific species

- pollination of agricultural species by insects
- regulation of pests and diseases by natural predators and parasites.

3.2.2.4 The relation between biodiversity and life support functions – can life support be measured by a biodiversity indicator?

The relation between the functioning of the life support system and biodiversity is complex. The life support system functions when ecosystem processes function well. Ecosystems are composed of species which keep the processes going. As may be clear from the above, in most cases functioning is not dependant on the presence of individual species but on functional groups: photosynthesisers, high vegetation, nitrogen reducers, pollinators, the presence of all links in a food chain etc. This implies there is a great redundancy of species in most ecosystems, and the disappearance of one or even many may not make much of a difference in the performance of life supporting ecosystem processes. On the other hand, this redundancy is not without meaning: it makes ecosystems more resilient against threats and disasters and better able to cope with changing circumstances.

In all, it would not seem advisable to measure life support services by a biodiversity indicator, but rather by one or more separate life support indicators. A biodiversity indicator for life support, if wanted for whatever reason, should not be directed at species diversity per se but rather at diversity between and within functional groups, with special emphasis on micro-organisms.

3.2.2.5 The impact of land use on life support functions

Different types of land use may have impacts on the life support services of the biosphere:

- biomass impacts: by removing natural ecosystems and replacing them by human made ecosystems there may be a loss in biomass production. This is a gradual loss – in agriculture, biomass is produced as well, and even cities have a certain amounts of green zones, but generally biomass production will be lower in human dominated zones.
- structural impacts: by removing the vegetation and especially by covering the surface the water management of a region is seriously altered. This may lead to an increased occurrence of flooding and erosion and a decreased availability of water for both natural and cultural ecosystems. (NB In the Dutch situation the water management system is for a large part man made (drainage, pumps etc.) which means that the natural system can be replaced by a human system, in this case. However this is expensive and may be less effective, especially in future in view of the rising sea level. Official water management policy now goes in the direction of more freedom for rivers, creating overflow polders for high waters etc.)
- species impacts: problems have been known to occur when certain species disappear through human intervention. By making land use changes this may happen due to the loss of suitable habitats like hedges and cops. Enlarging the scale of the landscape may be the key process here.

3.2.2.6 Indicators for life support functions

WIA proposes to focus on substance cycles and soil quality as indicators. For substance cycles, biomass indicators are proposed: the net primary production (NPP) per unit of surface. An ongoing discussion refers to whether to use the "free NPP" (fNPP), biomass production by nature only, or to include the human-regulated biomass as well. Which one to choose depends on the point of view; when life support for humans is the subject, then obviously the all-including NPP is the most appropriate. Ecosystem functions for nature may be indicated better by the fNPP, although even in this case this is debatable – conversion of carbon, useful for both human and other species, is done just as well by agricultural crops. The NPP, as a measure for biomass, indicates the maintenance of the biogeochemical cycles and connected life support functions.
WIA further proposes that soil compaction and soil organic content should be translated into indicators. These indicators measure the quality of the soil and its ability to sustain crop production. They are also implicitly but obviously aimed at human life support – ecosystems sometimes have very poor soils, which is why the recovery may take such a long time, and still are able to maintain a large standing stock of biomass. A good example of such an ecosystem is tropical forests.

In addition, indicators could be added

- for ecosystem structure – one could think, perhaps, of (1) an indicator for metalling, and (2) an indicator for vegetation cover (0, 1, 2, 3 or more layers). This especially indicates the water retaining capacity of the area, but is also of relevance for albedo and air filtering.
- for life support species – perhaps an indicator aimed at micro-organisms or functional groups.

For the issue of life support, the developments have not progressed very far. For the time being it seems advisable to confine ourselves to biomass. For the future, a further operationalisation seems indicated. In this respect, the method proposed by Baitz et al. (1998) seems very interesting. They worked out a system for including the influence of land use on a number of life support functions (including erosion protection, wind protection, water storage capacity, soil buffering capacity and a number of others) in LCA studies. There is an indicator in development at RIVM aimed at measuring soil fertility / soil quality, which consists of an index based on the presence of functional groups of soil flora and fauna (micro-organisms, earth worms, mites, tiny insects etc.) (Schouten et al., 2000). This indicator does not refer to LCA but perhaps could be used as a starting point. These and possibly other proposals could be discussed and worked out further within the WIA or other LCA discussion platforms.

3.3 Using the models in the LCA framework: characterisation

In section 3.3.1, the characterisation procedure of the various methods is discussed. Since life support indicators are still in the first stages of development, the discussion is limited to the proposals on including loss of biodiversity. In 3.3.2, some issues of general importance as they emerge from the discussion surrounding the methods will be treated in more detail.

3.3.1 Characterisation procedures in the different methods

IVAM-method:
The characterisation factors are derived from the equation describing the relation between species density and area. Not the species density S but the constant α, the species accumulation rate, is used. The factors are calculated as follows:

- for transformation: $(\alpha_{ini} - \alpha_{fin}) / \alpha_{ref}$
- for occupation: $(\alpha_{ref} - \alpha_{occ}) / \alpha_{ref}$

For each case study therefore 4 α's must be established: (1) the reference situation α_{ref}, representing the potential natural ecosystem, (2) the situation before the transformation α_{ini}, (3) the situation during occupation α_{occ} and (4) the situation after termination of human activity α_{fin} which may or may not be equal to the reference situation, depending on the irreversibility of the damage done by human intervention. The α's are taken out of a database containing characteristics of a number of natural and cultural ecosystems. For the most part they still need to be determined. Lindeijer proposes to use the Floron database to determine α values for especially cultural ecosystems for the Dutch situation (Lindeijer et al., 2001). The result is possibly useful for the Netherlands but has a limited relevance for the rest of the world. Extrapolation of Dutch data to other areas of the world will be attempted later (oral communication, Lindeijer).

Köllners method
The measure used by Köllner for characterisation is called EDP – Ecosystem Damage Potential for species diversity. This measure is used in two ways: to assess local damage and to assess regional damage. The distinction Köllner makes between the two levels is not very clear to me, it is the same indicator: changes in species density on the area involved. The EDP values for local and regional impacts are derived from the S values (and not, as in the IVAM method, from the coefficient – which seems to make sense since S closer to what we're interested in than an abstract α). Four different formulae are presented: a linear and a non-linear one for local damage, and a linear and a non-linear one for regional damage. The local linear function bears most resemblance to the IVAM formula for occupation: $EDP_{local} = 1 - (S_{occ} / S_{region})$. The regional function includes the land use pattern already established in the region. The linear function is: $EDP_{region} = dS_{lost} / dL_i = b$ (a constant to be established empirically), L_i being the fraction of land already used for this purpose. The non linear functions are more precise but also include a value judgement: Köllner points out that a parking lot in an already species poor area comes out worse than the same parking lot in a species rich region. The choice is thus made to give a heavier weight to interventions in already stressed areas. The argument may also go to another direction: interventions in undisturbed sites might be considered more undesirable since the loss is larger than in already spoiled areas. When adopting this characterisation method, one must be aware of and agree with this choice.
Köllner then adds the local and regional scores to one total EDP score, based on equal weighting factors. EDP factors are used for both transformation and occupation. Köllner treats transformation as a specific type of occupation by using a "transformation time" to express it in m^2.year units. Next to that a "restoration time" is used for the land to get back to the original state after termination of the activity, also leading to a m^2.year occupation. NB the recovery is assumed to be complete, i.e. de situation before equals the recovered situation. For both transformation and restoration an EDP is used which is halfway between the original and occupied one. Thus, transformation and occupation are made into one intervention. Theoretically there is nothing against this, it even has certain advantages as discussed above. In this case however, Köllner ignores the allocation problem, which makes it highly questionable: the whole of the transformation as well as restoration are implicitly allocated to the functional unit under study.

Eco-indicator 99
Köllners method is, at least partly, integrated in the Eco-indicator 99. The area-species relationship from Köllner is used but not the EDPs. Instead, Goedkoop & Spriensma define PDFs (Potentially Damaged Fraction), as a counterpart of the PAFs they use for assessing damage to biodiversity by emissions. Köllners the distinction between the local and regional level is adopted, as is his expression of transformation in terms of m^2.years – only not a "transformation time" is used but a "restoration time". (NB It seems that in the SimaPro software, as opposed to the manual, the transformation score is expressed in m^2 only and cannot be added to the occupation score. The manual in this respect is not quite up to date (oral communication Lindeijer)). As a result, there are four characterisation factors. The general equation is: PDF = $(S_{ref} – S_{use}) / S_{ref}$. S_{ref} is then defined differently for the four situations. The local factor for transformation includes the species density of the "situation before" as a reference, the local occupation factor that of the natural situation. On the regional level, the impacts on the (remaining) natural area are the issue. The reference for transformation then is the species density in the natural are in the region before transformation, and S_{use} is the same after transformation. By transformation, natural area has been lost and the number of species thus has become lower. For regional occupation, S_{ref} is the potential natural situation in the region as a whole, while S_{use} is identical to the S_{use} for regional transformation. The surface area A also means different things. On the local level, A means area transformed or occupied, while on the regional level A is the area of natural territory within the region. By using different definitions for the same variable, Goedkoop & Spriensma succeed in giving the impression that they use just one formula, thereby preserving an image of simplicity. The complexity however is merely hidden, not eliminated, which is also a pity because Goedkoop & Spriensma offer a valuable contribution to the confusion around the issue of the reference situation (see below).

LCAGAPS
For species richness, S is used. The derivative of the equation, $S' = caA^{z-1}$, is used as an approximation of ecosystem vulnerability: the idea is that the steepness of the slope indicates the characteristic species loss per ecosystem for a (marginal) occupied unit of land. Ecosystem scarcity is measured as the inverse of the potential area per ecosystem type. The scores on the three criteria are multiplied to produce one indicator. This is the equivalency factor which is used to assess the impact of land occupation on natural ecosystems.
Although the idea is attractive there are still some details to worry about:
- the first characteristic could overlap with a species diversity indicator; if a species diversity indicator is used in addition to the ecosystem indicator, it requires some fine tuning
- the vulnerability of ecosystems is expressed as a function of the fraction that is already occupied: the more of the natural ecosystem area has disappeared, the more pressured and therefore vulnerable the remainder will be. This can be regarded as an operationalisation of the knock-on effect. Although this is relevant information, it seems incomplete. Vulnerability is also a characteristic of the ecosystem itself. An arctic, species poor ecosystem is more vulnerable than a temperate wetland: it is much more sensitive to interventions and recovers much more slowly. This could be a good argument to include the ecosystem specific natural recovery time in the vulnerability score, next to the already existing pressure on ecosystems.
- the method is limited to natural ecosystems and cannot be used for changes within cultured land.

3.3.2 Discussion of some issues of general importance

In summary, the IVAM method and Köllner's method attempt to achieve more or less the same: to specify (changes in) species density per type of ecosystem, related to a reference based on the (potential) natural ecosystem. There are some differences in the details of the approach. The LCAGAPS has a different starting point. Some issues causing confusion are discussed below:
1. the equation describing the relation between the number of species and the size of the area
2. the treatment of transformation and recovery
3. the difference between the local and regional level
4. the definition of transformation and occupation and the relation between the two
5. the use of references.

Area – species relationship
Two equations are proposed to describe the relation between area size and species density. Both are derived from literature. It is not possible to state a preference for either out of hand. However, Köllner dedicated an important part of his PhD to validating the different equations and selected his one as the best fit. This could be an argument to change to the $S = aA^b$ in the IVAM method as well (see § 3.2.1). Another difference is that both LCAGAPS and Köllner use S, the number of species, itself for deriving his EDP factors, while in the IVAM method the coefficient α is used. Either one seems valid but since S is closer to what we really are interested in it may be a good idea to conform to this proposal.

Transformation and recovery
In the IVAM method, transformation and occupation are strictly separated and are expressed in different units. The Köllner method integrates transformation, occupation and recovery into one intervention: transformation and recovery are multiplied by a transformation resp. recovery period, thus creating an occupation-like measure in m^2.year units which is directly added to the score for occupation. Although the idea is attractive, it is also questionable since Köllner ignores the required allocation or attribution of transformation as well as recovery. To attribute the whole of the transformation and recovery to the functional unit under study leads to double counting and an implicit (very) heavy weighing factor, and therefore seems inadvisable.

Scale levels
Köllner and concurrently also Goedkoop & Spriensma make a difference between land use impacts on the local level and on the regional level. The local level refers to the area involved in the functional unit; here the transformation and occupation is absolute. The regional level refers to Switzerland, here the marginal impact of transformation / occupation on the Swiss biodiversity is the issue. The IVAM method does not make this distinction. Its characterisation factors bear a great resemblance to Köllners factors for the local level.
An argument for including the regional level is that it allows for a marginal approach, in accordance with general LCA philosophy. This puts the intervention in perspective. Also it opens possibilities to deal with "already stressed areas". If we follow this line of thought it is then the question whether we are still interested in the local level – what does this add? Köllners reasoning is not very convincing here – he states, without much supporting argument or evidence, that the local level score indicates a proper ecosystem functioning, while the regional level indicates the nature conservation value. It is not clear why this should be so, nor why ecosystem functioning is added to a loss of biodiversity impact category.
On the other hand, the choice for any specific regional level is arbitrary. Köllner chooses Switzerland which is hardly relevant for LCA in general. Such a choice should somehow be derived from the global level: the world divided in a limited number of areas relevant for biodiversity issues, for example the biomes as defined by Weidema & Lindeijer.

Weidema & Lindeijer's method operates on the global level and defines its "areas" from that level. Comparable to Köllners regional level, it also deals with the present state of stress of the natural ecosystems. In that sense, the method is superior to both IVAM and Köllner: it allows for a marginal approach, without falling into the trap of having to define a regional level. However LCAGAPS has its limitations as well: it is not possible to evaluate changes within the area already in human use, which is an important part of both Köllner's method and the IVAM method. As stated before, it could be argued that changes within already colonised land are irrelevant compared to the original change from the natural situation. However in Western Europe there is hardly any natural area left. Nature has to survive in corners of the domesticated area, and differences between the possibility of co-use by nature are relevant indeed.

3.3.3 The definition of transformation and occupation
On the issue of occupation the methods agree. In all three, the occupation indicator measures the (continuing) discrepancy between the actual and the natural situation, and thus for the deviation from the potential optimum with regard to biodiversity.

Transformation however is treated differently in all three or rather four methods, as described below. Köllner treats it as a part of occupation. By doing that, transformation is implicitly limited to transformation from nature into culture only, since the starting point is always his reference of Switzerland in 1850. In the Eco-indicator the two are separated although expressed in identical units. Although transformation is defined as change, the transformation impact refers to the loss of (species within the) natural area only. In the IVAM method it is stressed that transformation is different from occupation and should be treated as a separate issue, and the units are also different. Here it is explicitly stated that transformation is about change, and that change within the already cultural area is included as well. In LCAGAPS transformation is not elaborated but it is described as the irreversible change caused by an intervention, thereby again implicitly limiting it to transformation from nature into culture.

Various issues need to be resolved to clarify what “transformation” is all about, and thus be able to decide which treatment it should get in LCA:

1. Does transformation refer to extraction of land from nature only, or does it include changes of land use within the economy as well?
2. Is recovery included, if so, why and how?
3. What is the difference, or the relation, between transformation and occupation?

Ad 1. The methods disagree on this issue, although often implicitly. In fact, this is a normative choice. If we choose to apply transformation only to conversion of land from nature to culture, as most methods do, we are blind for changes within the cultural area. This may be regarded as problematical, but on the other hand it could be argued that the changes within the economy are irrelevant compared to the conversion from nature, and that this last issue is really the relevant one. Another problem when assessing transformation from nature to culture only is the distinction with occupation impacts: occupation, too, is a measure for the discrepancy with the natural situation.

If the choice is made to include all changes in land use, changes within the cultural area can be evaluated as well, but the question then becomes what to compare it to. The use of the natural situation as a reference then is questionable: the change is from the situation now, not from nature (see below). The Ecoindicator method perhaps has the most subtle solution for this, by defining transformation on the regional level different from transformation on the local level. On the regional level, only conversion from nature into culture is evaluated while on the local level the changes as they occur, also within the cultural area, are the issue. If we choose to include changes within the cultural area, the problem of allocation as mentioned in section 2 becomes even more pressing. The problems mentioned referring to the allocation of transformation apply to the transformation from nature to culture. It seems irrelevant to allocate every transformation within the economy – a certain piece of land may have had dozens of destinations during its history, all of them irrelevant for the functional unit in question. The argument for allocating change only to the changer becomes even stronger.

Ad 2. Recovery is implied to mean: reinstatement of the original natural situation. Recovery is included by Köllner in the same way as transformation is: by integrating it together with occupation into the one intervention land use. Köllners assumption is that every transformation is completely reversible. In the Ecoindicator 99 method, derived from Köllner, a recovery period is used likewise for translating the regional transformation measure into m^2.year units, although no integration with occupation follows. In LCAGAPS recovery is not an issue, but the related concept irreversibility pops up in their definition of transformation. Transformation is defined as the irreversible part of the land use change as compared to the natural state. In the IVAM method the recovered situation is used to compare with the “situation before”, instead of the situation during use. In the IVAM method, too, the recovery may not be complete, i.e., there is irreversibility. LCAGAPS and IVAM are essentially identical in this respect. All four options are debatable. Köllner and the Ecoindicator 99 use recovery time as a unit conversion trick, while it remains unclear what exactly it is they express by it. Under their own starting point of including land use changes within the economy, the IVAM treatment of recovery seems only logical in exceptional cases, when recovery is so to speak part of the intervention. This may be so in the case of extraction of surface resources such as clay or sand, where the occupation is short and there is an agreement beforehand to re-arrange the area as a natural area. In cases where this is unclear or unlikely, using the recovered situation clouds the issue. In fact, the not-recovered situation is used in the IVAM method in most cases (Lindeijer et al., 2001) – this is not regrettable, but as it should be. Although reversibility certainly is relevant it may be better not to hide it in a transformation indicator but give it an explicit, specific place in the procedure.

Ad 3. The relation between transformation and occupation is, sometimes implicitly, made differently in the surveyed methods. When Figure 1 is taken as the starting point, it is as follows: A certain area is transformed, then occupied, then released and recovered. The total time line then can be broken down into occupation and transformation (or not, as is essentially Köllners solution). The difficulties and methodological tangles of translating this figure are discussed in Section 2 and will not be repeated here. Regarding the Impact assessment, the methods nearly agree on occupation. About transformation there are basically two schools. One sees transformation as change, i.e. the change from the original situation to a different one, brought about by the functional unit or at least by human influence. The other does not look at the change itself, but at the lasting impact of the change: it may be that a certain change cannot be reversed completely, even if human occupation ended forever, for example when land is converted to water due to sand extraction. The irreversible part of the conversion then is the transformation impact.

In all, the distinction of a separate measure for transformation as is done in the IVAM method certainly is defendable. It seems that some clarification is required in the definition of the impact of transformation. From the above, it could mean a number of different things:

- the transformation impact is the change in species density due to changes in land use caused by the functional unit, compared to the situation before
- the transformation impact is the change in biodiversity due to the conversion of natural into cultural land caused by the functional unit
- the transformation impact is the irreversible change in species density due to changes in land use caused by the functional unit
- the transformation impact is the irreversible change in biodiversity due to the conversion of natural into cultural land caused by the functional unit

Regarding the choice for the situation we want as a comparison, it seems most logical not to take the natural situation but the situation as it was before the conversion. In that case, there is the least possible overlap with occupation, which (also) is a measure for the difference with the natural situation. Regarding the other choice, i.e. defining transformation as either the change itself or the irreversible consequences of the change, the matter is less clear. Both provide relevant information. It seems simplest and most straightforward to go for the change itself, but in that case the reversibility of the intervention is a forgotten aspect. A possibility is to compose a transformation indicator out of the two aspects: the change itself, corrected for a factor for the reversibility of this change. On a geological time scale, everything is reversible so the time period involved in a complete recovery could be a measure. In that case, this factor will probably become completely dominant and we have to ask ourselves whether we want to attach so much weight to this aspect.

References in land use

The use of references is the cause of much confusion. To clarify, a number of different references, popping up at different places in the LCA framework, can be distinguished:

- In the Inventory, a reference for the intervention comes from the “less is better” LCA philosophy. Less land used is better, so the reference is zero square meters of land used. In the Impact assessment, the reference is the basis for comparison, the reference as a yardstick for the impact of the intervention. Since more than 1 impact is distinguished, more than 1 reference is required.
- For the impact category Occupation the reference, meaning the difference with the natural situation, can also be derived from “less is better”. Less in this case means less impact, the reference being zero impact, i.e. zero “species potentially lost” as Köllner puts it, therefore the natural situation. In all methods discussed here, the species density of the natural situation is the reference used to compare with the situation in use, except for Köllner who uses Switzerland 1850. Köllners choice therefore seems not to be in line with the above reasoning.
- The IA reference for transformation of course depends on its definition. In case transformation is regarded as change, the situation before the change is the reference. When transformation is seen only as change from nature to culture, then once again the natural situation is the reference. Also including change within the economy means that the situation before the change, be it nature or culture, is the reference. In case transformation is regarded as the irreversible part of the impact of

transformation, the natural situation automatically becomes the reference. Köllner solves the reference problem neatly by defining transformation (from nature to culture) as a part of the one intervention land use, combined with occupation and recovery. The natural situation thus is automatically the reference. In Ecoindicator 99, two references are used in accordance with the two types of transformation: the natural state for the regional level, and the situation before the intervention for the local level. In LCAGAPS, transformation is not elaborated. The IVAM method is most problematical in this respect. Two references appear in the equation for the transformation indicator: the situation before (ini) as well as the natural situation (ref). Dividing by α_{ref} or S_{ref} serves the purpose of enabling to add different interventions to one transformation score. This purpose could be served just as well through dividing by α_{ini} or S_{ini}, thereby enabling to evaluate changes within the economy as well and clearly making the choice for assessing changes within the economy as well.

- A final place for a reference to appear is when defining equivalency factors. When doing this for emissions, a reference substance is used (for example CO_2 for global warming) and the factors for the other substances are given relative to this reference (in CO_2 equivalents). There is no need to do this but it is possible, and is actually done by Weidema & Lindeijer. They use a reference ecosystem, the one with the lowest natural species density, to derive equivalency factors for the characterisation regarding other ecosystems. They prefer to call these “scaling factors” but they are essentially identical to the reference substances.

3.4 Normalisation and weighing

Not much is stated on these issues in either of the methods. Weidema & Lindeijer provide a table with global normalisation factors referring to their ecosystems. In the Ecoindicator 99 normalisation procedure is described, and normalisation factors for land use interventions are provided for woods, “urban”, and different types of agricultural land. Köllner and IVAM are silent on this matter.
On weighing there are some scattered remarks in all methods:

- Ecoindicator 99 uses a weighing factor of 10 to add PAFs to PDFs;
- Köllner proposes equal weighing between the regional and local level, implicit and (therefore?) quite complicated weighing between transformation and occupation (depends on allocation which he does not specify, also a factor 4.1 is mentioned);
- IVAM takes the point of view that no weighing between transformation and occupation should be attempted at present;
- LCAGAPS offers no statement on weighing, but weighing takes place within the equivalency factors – all three areas of interest are included with equal weight.

4 Conclusions and recommendations

As is clear from the above, the inclusion of land use impacts in LCA is by no means finished. The issue has come up only during the course of LCA development and relatively little attention has been paid to the issue compared to the impacts arising from emissions. Land use impacts differ from emission impacts in many ways. Therefore the lines of thought that have to be followed are also different. In recent years there has been more attention for land use related impacts, as a result of the pressure arising from the obvious importance of such impacts. This has led to the development of several methods, none of which is "finished". Some unfinished business is discussed in this paper in the previous sections.

A first conclusion is, that in all of the methods studied most attention has been paid to the Impact Assessment part of LCA. The LCA Inventory has had relatively little attention, despite the fact that a very important subject resides there: the rules for allocation. The allocation choices made in case studies have a large, sometimes even decisive influence on the outcomes. The combination of relative importance and relative neglect leads to a first, rather obvious conclusion that this issue is most unresolved and required most urgent attention.

For the LCA Impact Assessment there are various proposals which all have their merits and drawbacks. Below, the most important are discussed and some recommendations are made, both for the short term and for the long term.

LCA Inventory

Guinée et al. (in press) recommend – and make a start with – writing out neatly and clearly the whole Inventory procedure. This may clarify some issues and make apparent which discussions belong in what stage, what needs be resolved first and how to proceed from there.
The main problem in the Inventory is, as stated above, the Allocation, especially of the Transformation intervention. Solution of this subtle and complicated issue requires discussion within the broader LCA community. The SETAC group on LCA Inventory seems the appropriate platform, but there may be others as well. For the time being, there are several possibilities:

1. Exclude transformation as an intervention until this issue has been resolved
2. Include transformation only if it can be directly related to the functional unit. This may be the case for example when extracting clay or other raw materials for building. The transformation then can be attributed as a whole to the f.u.
3. The only concrete proposal available at present is Lindeijer's proposal to use trends. Despite the signalled problems (Section 2) this could be used until something better becomes available. A proposal is made in this paper to amend Lindeijer's proposal by not using the trend in land use itself, but in the changes therein (see Section 2). This can be regarded as a generalisation of the principle "transformation is attributed to the process causing the change".

LCA Impact Assessment

A first issue to be addressed is defining which of the impacts of land use can or should be included in LCA. All methods now available are incomplete in the impacts they consider. Partly, this could be amended, but partly, it will be beyond the scope of LCA. This issue cannot be resolved within the present project but could be subject of discussion in the SETAC-WIA taskforce on land use, or other such platforms.

Impacts considered in this paper are (1) impacts on biodiversity and (2) impacts on life support services. Some specific recommendations on these issues are made below.

<u>Impacts on biodiversity</u>

Biodiversity is linked to the two interventions transformation and occupation. All methods agree on these two interventions. They also more or less agree on the definition of occupation, which could be summarised as follows: *the impact category occupation refers to the difference with the no-impact,* *i.e. natural, situation of the land as caused by the intervention linked to the functional unit.* However there is a major difference between the methods regarding the treatment of transformation. They disagree on two main issues:

- the impact category transformation refers to the change in land use, vs. the irreversible part of the change in land use, caused by the functional unit
- transformation refers to a (irreversible) change from nature into culture, vs. refers to all (irreversible) changes in land use, caused by the functional unit.

In order to progress in the LCA Impact Assessment, a clear choice must be made, which must crystallise further in the LCA community. Such a choice also has its implications for the Inventory and the problems of allocation. For now, a proposal is made to define transformation as change, not the irreversible part of change, and to make it refer to changes in general, not just from nature into culture. The reasons are both practical (it is generally difficult to determine irreversibility) and to avoid double-counting (the difference with the natural situation is already indicated by the occupation impact category). This implies that the irreversibility of the impact is out of view, and that the natural situation cannot be used as a reference. In future some way must be found to deal with irreversibility.

The following recommendations are made regarding the applicability of *occupation* in this project:

- Use Köllners formula to describe the relation between area and number of species, since he has put in most effort in validating
- Use S and not α, since S is closer to what we are interested in
- Use LCAGAPS as a starting point, since this method fits best to general LCA framework requirements: an ecosystem indicator derived from the global level, no need for a further distinction between the local and regional level.
- Make additions to this method especially to be able to include the remaining species during occupation, by using the procedure out of the IVAM method instead of the LCAGAPS species richness factor nSR. The SR is identical to S_{ref}, and nSR is obtained through dividing by the species density of the ecosystem with the lowest species density S_{ref} / S_{min}. The nSR in LCAGAPS just weighs the intervention based on the natural species richness, implying that all gets lost; the IVAM procedure then must correct for the remaining biodiversity, S_{occ}. The higher S_{occ}, the lower the characterisation factor should be, since less species are "potentially lost". The equation for nSR then would be:
  $nSR = [(S_{ref} – S_{occ}) / S_{ref}] * [S_{ref} / S_{min}] = (S_{ref} – S_{occ}) / S_{min}$.
- It is also recommended to add recovery time to the Ecosystem Vulnerability factor. The recovery time in this case is an ecosystem characteristic, a general measure for how quickly a natural ecosystem recovers from disturbance, regardless of the exact nature of the intervention.

Further discussions to streamline this issue and harmonise the different methods, or to end up with a number of methods with different value choices, are indicated.

Recommendations regarding the application of *transformation* within this project are:

- Leave transformation out altogether until basic discussions have been solved
- If this is for some reason unacceptable, then use the Ecoindicator 99 PDFs but within the IVAM framework, i.e. do not add the recovery time to convert the score into m^2.year. When transformation is to include changes within the economy, the local PDFs are indicated.

For the future, the issue of transformation must be clarified profoundly. The different methods have different definitions, making them incomparable. Unless this is resolved, it hardly seems useful to elaborate on detailed Impact Assessment characterisation factors. When it is, it could be worthwhile to expand the LCAGAPS method with a transformation mode.

<u>Impacts on the life support function of ecosystems</u>:
Although some interesting proposals exist, there is even less agreement on this issue than on the treatment of biodiversity loss. Even the starting points are still under discussion. The one point of general agreement seems to be that biomass production is a valid indicator. For this project, it seems advisable to limit characterisation of life support functions to biomass, either NPP or fNPP. It also seems advisable to link this to Occupation, in view of the time dimension of biomass production. For the future, it is recommended:

- to decide whether LSF for humans or for other species are indicated; defining LSF for humans is politically accepted as well as more straightforward

- to define a more complete list of life support functions and decide which ones are relevant to include in LCA
- to make the link between land use, occupation and transformation, and damage to life support functions
- to elaborate characterisation factors for the different (categories of) life support functions.

<u>Normalisation en weighing</u>:
There is little attention for these issues in the currently available methods. Since it is important that land use impacts can be evaluated in the same framework as the other interventions, it seems that no method is really applicable without such a list.
Weighing is a subject that is no different for land use than for the other impact categories. There seems no need for special attention here.

5 Acknowledgements

The issue of land use in LCA is a very complicated one. This paper may hopefully serve to clarify some of the issues giving rise to debate and controversy. Significant contributions were made by Joris Broers for his suggestions regarding the Allocation procedure, by Sangwon Suh and Arjan de Koning regarding the use of references, by Reinout Heijungs and Gjalt Huppes on the general LCA framework, and by Erwin Lindeijer making available his knowledge and experience in the area of Land use in LCA.

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Annex 3 Land use by aggregate extraction in the Netherlands (in Dutch)

Overzicht gegevens landgebruik zand- grind-, klei- en mergelwinning

Erwin Lindeijer, TNO Industrie
21 oktober 2001

Voorgelegd aan leden van de FODI d.d. 8 november 2001
(zie voor het verslag het eind van deze bijlage)

Achtergrond

Rijkswaterstaat (Dienst Weg- en Waterbouwkunde, DWW) heeft opdracht gegeven om gegevens te achterhalen over landgebruik door oppervlaktedelfstoffenwinning, voor toepassing in LCA's (milieukundige productbeoordelingen). Het uiteindelijk doel van de studie is om ordegrootte effecten van landgebruik te kunnen beoordelen, en bijvoorbeeld grofweg verschillen in landgebruikseffecten van winning oppervlaktedelfstoffen en winning van (tropisch) hout te signaleren. Ook voor hout worden namelijk dergelijke gegevens verzameld. Dit uiteindelijk doel is nog ver weg, omdat de milieubeoordelingsmethode nog onvolledig is. Dit stuk is echter bedoeld om de fysieke basisgegevens alvast te bepalen, zodat te zijner tijd een beoordeling kan plaatsvinden.

Aanpak bepaling landgebruik winningen

Er is een onderscheid gemaakt tussen *bezet houden* van land en *verandering* van landgebruik. Tijdens een ontgronding wordt een gebied tijdelijk *bezet* gedurende de ontgrondingsperiode. Dit bezet houden wordt uitgedrukt in oppervlak maal jaren per hoeveelheid gewonnen grondstof [m^2.y/t].

Daarnaast wordt –vaak landbouw-grond meestal definitief *veranderd* in een waterplas, al dan niet als natuurgebied. Ook bij andere ontgrondingen vindt meestal een netto verandering plaats. Zo wordt bij kleiwinning tegenwoordig een stuk natuurgebied gecreëerd. Het gaat hierbij om een verandering in kwaliteit in een bepaald oppervlak land. In dit stuk kijken we vooral naar het oppervlak[16]. De eenheid waarin we deze landgebruiksverandering uitdrukken is oppervlak per hoeveelheid gewonnen grondstof [m^2/t].
Tijdens een eventuele aktieve herstelperiode (bijvoorbeeld tot recreatieplas of natuurgebied) is ook sprake van bezet houden en verandering van landgebruik. Zolang de ontgronder nog economisch gebonden is aan het gebied rekenen we dit landgebruik aan die ontgronder toe. Dus zolang de ontgronder bezig is met herstelwerkzaamheden wordt hiervoor ook land bezet. De uiteindelijke kwaliteit na herstel wordt dan ook vergeleken met de situatie voor de ontgronding. Als de

[16] De kwaliteit wordt voorlopig bepaald aan de hand van botanische gegevens. Die kwaliteit kan na de ontgronding zowel hoger als lager zijn dan ervoor, of hetzelfde blijven. Uiteindelijk zouden hier ook gegevens over diersoorten aan toegevoegd moeten worden. De waardering van veranderingen in het landschap worden ook niet meegenomen. Of daar een milieukundige beoordeling voor op te stellen is, valt te betwijfelen [Lindeijer et al., 1998].

ontgronding eindigt, en het herstel wordt aan de gemeenschap overgelaten, dan wordt aan de ontgronding niet het herstel toegerekend, en dus ook niet een eventuele milieuwinst door herstel.

Als tijdens een ontgronding meerdere grondstoffen gewonnen worden, rekenen we het landgebruik aan die soorten grondstoffen toe, op basis van de hoeveelheid en de economische waarde (opbrengst) ervan.

Basisgegevens landgbruik NL:

Hieronder hebben we een overzicht gemaakt van alle ons beschikbare gegevens over bezet houden van land en oppervlak land veranderd tijdens ontgrondingen. Het doel hiervan is om algemeen toepasbare landgebruiksgegevens vast te stellen. Gezien de natuurlijke variatie in situaties moet hierbij gebruik gemaakt worden van gewogen gemiddelden. De onzekerheid in dit gewogen gemiddelde wordt dan voornamelijk bepaald door onzekerheden (zoals in dichtheden, de putgrootte en de jaarlijkse onttrekking) van de belangrijkste winlocaties, en de representativiteit van de groep winningen voor heel Nederland.

Hierbij is gebruik gemaakt van veel achtergrondgegevens. Deze worden achterin dit stuk kort samengevat. Wellicht zijn deze niet voor iedereen even toegankelijk, maar voor de volledigheid en transparantie worden ze toch vermeld. De bij de berekeningen gebruikte aannames over hersteltijden worden t.z.t. nog aangepast naar aanleiding van een rapportage die Stichting Ark voor ons schrijft [Helmer, 2001]. In deze achtergrondgegevens worden berekeningen uitgevoerd en kentallen toegepast, waar we een aantal vragen over willen stellen.

Vragen aan de FODI naar aanleiding van onderstaande landgebruiksgegevens:

- Is de berekeningswijze terecht, en zijn de gewogen gemiddelden (zie figuren 1 en 2) op een acceptabele manier afgeleid?
- Zijn de kentallen ten aanzien van de dichtheden (1,6-1,7 voor zand/grind; 1,8-2,6 voor klei) gerechtvaardigd?
- Zijn de kentallen over de economische waarden voor grind en beton/metselzand(BMZ; € 4,54/ton), ophoogzand (OHZ; € 1,36/ton) en klei (€ 5,45/ton) uit [Tukker et al., 1999] toepasbaar als grove indicatoren voor de economische toerekening van het oppervlak?
- De gegevens over bezet houden van DWW hebben betrekking op winningen in Noord-Oost Nederland, en zijn structureel wat hoger dan die van Beetstra, die betrekking hebben op midden- en zuidoost Nederland. Is dit structureel verschil verklaarbaar op basis van het verschil in winningsregio?
- De onzekerheid in brongegevens is bij het gebruik van 2 bronnen voor grindwinning Stevol bepaald op 3%. In het algemeen kan hier 5% worden aangehouden voor variaties in opgegeven waarden van oppervlakte en gewonnen hoeveelheid. De onzekerheid door dichtheidsvariaties is voor zand en grind 6% en voor klei 36%. De variatie in economische waarde is niet goed bekend, maar is vermoedelijk van ondergeschikt belang[17]. Gezamenlijk wordt dit 8% respectievelijk 37%, volgens de kleinste kwadratenmethode. Zijn deze waarden redelijk?
- Naast bovenstaande onzekerheden is de representativiteit van de cases een grote bron van onzekerheid. De gegevens voor bezet houden betreffen voor BMZ 38% van de totale (jaarlijkse) winning, voor grind 39% van de totale jaarlijkse winning, voor OHZ 3% en voor klei 10% van de totale jaarlijkse winning (zie de achtergrondgegevens). Op basis van de percentages voor representativiteit kan een inschatting worden gemaakt van de onzekerheid door gebrek aan representativiteit. Dit wordt hier met de volgende vuistregel bepaald: onzekerheid = 1 –

[17] Variatie in economische waarde laat slechts een beperkte variatie in toegerekend landgebruik zien. Deze is minder relevant dan de onzekerheid in de representativiteit. Als er meer precieze gegevens over de variatie in economische waarde worden aangeleverd, kan hier wel een rekenvoorbeeld aan worden gewijd.

representativiteit[18], dus voor bezet houden: voor BMZ 62%, voor grind 61%, voor ophoogzand 97% en voor klei 90%. De gegevens voor verandering betreffen voor BMZ en grind samen 50% van de totale (jaarlijkse) winning, voor OHZ 15%, voor kalkzandsteenzand 30% en voor klei 9% van de totale jaarlijkse winning (zie de achtergrondgegevens). Dit levert als aanvullende onzekerheid door representativiteit op: BMZ/grind 50%, OHZ 85%, 70% KZSZ en voor klei 91%. Is deze bepaling van de onzekerheid acceptabel?

- De uit bovenstaande bepalingsmethode via kleinste kwadratenmethode bepaalde totale onzekerheid is als volgt:

	bezet houden	verandering
beton/metselzand (BMZ)	63%	51%
grind	62%	51%
ophoogzand (OHZ)	97%	85%
kalkzandsteenzand (KZSZ)	niet bepaald; gelijk aan BMZ	71%
klei	97%	98%

Zijn deze onzekerheidspercentages een acceptabele richtlijn?
(zie voor een bespreking van deze vragen het eind van deze bijlage)

Resultaten landgebruiksgegevens

Het resultaat van de twee voornaamste bronnen (DWW zelf en onderzoek van Beetstra) en de daaruit bepaalde algemene waarden inclusief onzekerheidsintervallen worden hier eerst grafisch weergegeven.

	BMZ Beetstra	BMZ DWW	BMZ algemeen	OHZ Beetstra	OHZ DWW	OHZ algemeen	Grind algemeen	Klei uiterwaarden	Komklei	Kalkzandsteenzand	Mergel
hoog	1084	3374	1452	372	3487	3079	1184	248	2220	36	
laag	246	766	330	6	53	47	278	4	34	8	
gewogen gemiddelde	665	2070	891	189	1770	1563	731	126	1127	22	230

Figuur 1: Bezet houden van land door oppervlaktedelfstoffen in Nederland. Exclusief herstelperioden.

[18] De resultaten voor de onzekerheid komen met deze vuistregel in dezelfde ordegrootte als de hoogste en laagste waarden in de cases, en in dezelfde ordegrootte als wanneer een ongewogen standaarddeviatie bepaald zou zijn. Gebruik maken van een ongewogen standaarddeviatie is overigens niet terecht bij een gewogen gemiddelde.

	BMZ/grind Beetstra	BMZ DWW	BMZ/grind algemeen	OHZ	Kalkzandsteenzand	Klei uiterwaarden	Komklei	Mergel
hoog	94	54	75,5	74	34	249	49,5	
laag	30	18	24,5	6	6	3	0,5	
gewogen gemiddelde	62	36	50	40	20	126	25	9,6

Figuur 2: Verandering van land door oppervlaktedelfstoffen in Nederland. Exclusief de bijbehorende kwaliteitsveranderingen; deze kunnen zowel positief als negatief zijn!

Voor beton en metselzand en grind is de verdeling van kwaliteitsverandering als volgt (op basis van onderstaande tabel 2):

Van landbouw naar recreatieplas	30%
Van landbouw naar natuur	12%
Van extensieve weiden naar wegen	7%
Van extensieve weiden naar recreatieplas	9%
Van extensieve weiden naar recreatieterrein	6%
Van extensieve weiden naar bebouwing	6%
Van extensieve weiden naar natuur	30%

Voor uiterwaardenklei is de verandering 100% van extensieve weiden naar natuur, en voor komklei is het 53% van intensieve weiden naar waterplas, en 47% geen verandering. Voor mergel is de verandering voor 11% van natuur naar bebouwing/infrastructuur en voor 74% van natuur naar waterplas met extensieve recreatie. De overige 15% kent netto geen verandering. Voor kalkzandsteenzand is slechts 1 waarde gevonden; hier wordt uitgegaan van dezelfde situaties als voor BMZ.

Voor industriële productie op basis van deze grondstoffen worden overigens niet de gemiddelde waarden voor landgebruik door industriële productie overgenomen (1900 m^2.y/kt bezet houden en 41 m^2/kt verandering), omdat dit bulkproducten betreft. De waarde voor olieraffinaderijen is 8 m^2.y/kt bezet houden (geen gegevens over verandering beschikbaar), maar we verkiezen specifieke gegevens te gebruiken. Deze zijn als volgt (op basis van gegevens van de industrie en [Beetstra, 1998b]:

	Bezet houden [m^2.y/kt]	Verandering [m^2/kt]
Cementfabriek	75	0.8
Betonmortelcentrale	463	15
Baksteenfabriek	1320	18

De basis van deze fabrieksgegevens wordt beschreven aan het eind van elk stuk voor de winning.

Achtergrondgegevens landgebruik

1. Achtergrond basisgegevens *bezet houden* van land

Er komen veel cases uit het werk van Ferdinand Beetstra. Een deel van zijn werk vergde echter herberekeningen en correcties. Die zijn hieronder meegenomen.

Uit het proefschrift van Ferdinand Beetstra [Beetstra, 1998a] en een aanvullend rapport [Beetstra, 1998b] komen een tiental voorbeelden van zand-, grind- en kleiwinning, op basis van werkelijke situaties (bronnen: vergunningen en brochures van ontgronders):

- Case 1: zand- en grindwinning Azewijnse Broek Gelderland

Oppervlak: 15 ha winput, plus 33% extra bezet (voor toegangswegen etc.) komt neer op een totaal oppervlak van 200.000 m^2.
Productie: Er vindt gedurende 16 jaar winning van grind en zand plaats: 300 kiloton/j beton/metselzand (**BMZ**), 80 kiloton/jr ophoogzand (**OHZ**) en 380 kiloton/jr grindwinning => 12.160 kt totaal (grind en zand samen).
Berekening bezet houden: 200.000 m^2 x 16 jr / 12.160 kt = 263 m^2.y/kt zand/grind.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4.538,-/kton, OHZ € 1.361,-/kton en grind € 4.538,-/kton is de totale opbrengst respectievelijk 4.538x300 + 1.361x80 + 4.538x380 = 3.194.610 Dfl. Toerekening aan BMZ: 1,36/3,19 x 100% = 43% van het totale oppervlak, aan OHZ: 0,11/3,19 x 100% = 3%, aan grind 1,72/3,19 x 100% = 54% van het totale oppervlak. Bezet houden wordt dan voor deze case **287 m^2.y/kt BMZ**, **75 m^2.y/kt OHZ**, **284 m^2.y/kt grind**.
Herstelperiode: Gedurende de 16 jaar winning in agrarisch gebied (weiland en landbouw) worden de randen heringericht tot recreatiegebied. De randen betreffen 50.000 m^2; het bezet houden is volgens deze redenering reeds meegenomen in de berekening.

- Case 2: zandwinning Hogeweg Gelderland

Oppervlak: 12 ha winput, 33% extra bezet (voor toegangswegen etc.) komt neer op 150.000 m^2.
Productie: 5 jaar winning met 100.000 t/jaar zandproductie geeft 500 kiloton zand totaal.
Berekening bezet houden: 150.000 m^2 x 5 jr / 500 kt = **150 m^2.y/kt zand**.
Herstelperiode: Gedurende de winning in agrarisch gebied (weiland en landbouw) worden de randen heringericht tot recreatiegebied. De randen betreffen 30.000 m^2; het bezet houden is volgens deze redenering reeds meegenomen in de berekening.

- Case 3: grindwinning Maas midden Limburg

Oppervlak: 15 ha, 33% extra bezet (voor toegangswegen etc.) => 200.000 m^2.
Productie: 15 jaar winning, 20 m diep x 150.000 m^2 x dichtheid 1,8 m^3/t x 95% bruikbaar => 5.130 kt totale grindproductie.
Berekening bezet houden: 200.000 m^2 x 15 jr / 5.130 kt = **585 m^2.y/kt grind**.
Herstelperiode: de vermelde 16 jaar hersteltijd van weide- en landbouwgebied tot volwaardige recreatieplas wordt niet overgenomen. Aangenomen wordt dat herstel rondom de plas reeds plaatsvindt tijdens de winning en dat de botanische kwaliteit aan het eind van de winning reeds maximaal is.

- Case 4: zand- en grindwinning Beers Noord-Brabant

Oppervlak: 155 ha, 30% extra bezet (voor toegangswegen etc.) => 2.010.000 m2.
Productie: 7 jaar winning, 23.000 kt BMZproductie en 5.750 kt grindproductie.
Berekening bezet houden: 2.010.000 m^2 x 7 jr / 28.750 kt = **489 m^2.y/kt zand/grind**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ fl. 10,-/ton en grind fl. 10,-/ton hoeft er geen verdeling te worden gemaakt tussen beide.
Herstelperiode: Van oorspronkelijk extensief weidegebied wordt 20 ha wordt bebouwd, 55 ha wordt recreatie(plas), 37 ha wordt recreatie- en opslagterrein en 43 ha wordt natuurbouw. Voor het stuk dat natuur wordt is de (door Beetstra gemodelleerde) 55 jaar hersteltijd genomen, waarbij wordt aangenomen dat over de 55 jaar genomen de helft al natuur is. Dit betekent nog 55/2 x 430.000 / 28.750 = **411 m^2.y/kt zand/grind extra**.

- Case 5: zand- en grindwinning Beers-Oost Noord-Brabant

Oppervlak: 113 ha, 54 ha oevers => 1.670.000 m^2.

Productie: 9 jaar winning, 10.500.000 m^3= 17.000 kton BMZproductie, 2.000.000 m^3 = 3.260 kton OHZ en 2.600.000 m^3 = 4.240 kton grindproductie (beide met dichtheid 1,63 t/m^3). Totale productie 24.500 kton.
Berekening bezet houden: 1.670.000 m^2 x 9 jr / 24.500 kt = **613 m^2.y/kt zand/grind**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4.538,-/kton, OHZ € 1.361,-/kton en grind € 4.538,-/kton is de totale opbrengst respectievelijk 4.538x17.000 + 1.361x3.260 + 4.538x4.240 = € 100.739.000. Toerekening aan BMZ: 77/101 x 100% = 77% van het totale oppervlak, aan OHZ: 4,4/101 x 100% = 4%, aan grind 19,2/101 x 100% = 19% van het totale oppervlak. Bezet houden wordt dan voor deze case **681 m^2.y/kt BMZ**, **184 m^2.y/kt OHZ**, **674 m^2.y/kt grind**.
Herstelperiode: 54 ha oever wordt ingericht tot natuurgebied. 27 ha bodem van bestaande plas wordt verdiept. Voor het stuk dat natuur wordt is de (door Beetstra gemodelleerde) hersteltijd van 55 jaar meegenomen, waarbij wordt aangenomen dat over de 55 jaar genomen de helft al natuur is. Dit betekent nog 55/2 x 550.000 / 24.500 = **617 m^2.y/kt zand/grind extra**.

- Case 6: zand-, grind- en kleiwinning Heeswijkse Kampen Noord-Brabant

Oppervlak: 57 ha, 33 ha omgeving => 900.000 m^2.
Productie: 7 jaar winning, 4.000 kton BMZproductie, 3.200 kton OHZ, 4.000 kton grindproductie, 250.000 m^3 klei (met dichtheid 1,8-2,6: 2,2 t/m^3) = 550 kton klei. Totale productie 11.750 kton.
Berekening bezet houden: 900.000 m^2 x 7 jr / 11.750 kt = **536 m^2.y/kt zand/grind/klei**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4.538,-/kton, OHZ € 1.361,-/kton, grind € 4.538,-/kton en klei € 5.445,-/kton is de totale opbrengst respectievelijk 4.538x4.000 + 1.361x3.200 + 4.538x4.000 + 5.445x550 = € 43.653.700. Toerekening aan BMZ: 18,2/43,7 x 100% = 41,5% van het totale oppervlak, aan OHZ: 4,36/43,7 x 100% = 10%, aan grind 18,2/43,7 x 100% = 41,5% en aan klei 3,00/43,7 x 100% = 7% van het totale oppervlak. Bezet houden wordt dan voor deze case **654 m^2.y/kt BMZ**, **197 m^2.y/kt OHZ**, **654 m^2.y/kt grind** en **802 m^2.y/kt klei**.
Herstelperiode: 33 ha omgeving wordt ingericht tot woongebied, sportvelden en 2,1 ha bos. Voor de herinrichting tot uitbreidingswijk wordt geen herstelperiode gerekend (is voor rekening van woningbouw). Voor het stuk dat bos wordt is de (door Beetstra gemodelleerde) hersteltijd van 55 jaar meegenomen, waarbij wordt aangenomen dat over de 55 jaar genomen gemiddeld de helft al bos is. Dit betekent nog 55/2 x 21.000 / 11.750 = **49 m^2.y/kt zand/grind/klei extra**.

- Case 7: zand- en grindwinning Neer Limburg

Oppervlak: 40 ha, 12 ha omgeving => 520.000 m^2.
Productie: 15 jaar winning, 3.260 kton industriezandproductie en 1.400 kton grindproductie. Totale productie 4.650 kton.
Berekening bezet houden: 520.000 m2 x 15 jr / 4.650 kt = **1677 m^2.y/kt zand/grind**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4,54/ton en grind € 4,54,-/ton hoeft er geen verdeling te worden gemaakt tussen beide.
Herstelperiode: Het oorspronkelijk agrarisch gebied wordt na herinrichting voor 2/3 natuur met extensief recreatief medegebruik en 1/3 natuur. In totaal wordt door ons voor 2/3 herstel tot natuur in 55 jaar meegerekend Dit betekent nog 55/2 x 347.000 m^2 / 4.650 = **2050 m^2.y/kt zand/grind extra**.

- Case 8: grind- en zandwinning Stevol Limburg

Oppervlak: 184 ha, 26 ha omgeving => 2.100.000 m^2.
Productie: 10 jaar winning, 6.000 kton industriezandproductie en 20.000 kton grindproductie. Totale productie 26.000 kton.
Berekening bezet houden: 2.100.000 m^2 x 10 jr / 26.000 kt = **808 m^2.y/kt zand/grind**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4,54/ton en grind € 4,54,-/ton hoeft er geen verdeling te worden gemaakt tussen beide.
Herstelperiode: Het oorspronkelijk agrarisch gebied wordt na herinrichting 135 ha plassen, aangevuld met natuurgebieden met extensieve recreatie tot natuur. Alleen voor dat laatste is de 55 jaar van Beetstra meegerekend. Dit betekent nog 55/2 x 347.000 m^2 / 4.650 = **2050 m^2.y/kt zand/grind extra**.
Andere bron: Op basis van het grondstoffenplan provincie Limburg [Provincie Limburg, 1999] komen wij tot de volgende berekening: 209 ha, 10 jaar productie, totale productie 25.000 kton => 2.090.000 m^2 x 10/25.000 = **836 m^2.y/kt grind** exclusief herstelperiode, hetgeen goed overeen komt met bovenstaande berekening maar tevens een indruk geeft van de marge in gegevens voor 1 case.

- Case 9: zand- en kleiwinning Uitbreiding de Boterhoek Gelderland

Oppervlak: 15 ha, 3 ha extra voor toegangswegen e.d. => 180.000 m^2.
Productie: 5 jaar winning, 880 kton BMZ en 220.000 m^3 klei (met dichtheid 1,8-2,6: 2,2 t/m^3) = 484 kton klei. Totale productie 1.364 kton.
Berekening bezet houden: 180.000 m^2 x 5 jr / 1.364 kt = **660 m^2.y/kt zand/klei**.
Toerekening van het oppervlak op basis van economische waarde: met BMZ € 4.538,-/kton en klei € 5.445,-/kton is de totale opbrengst respectievelijk 4.538x880 + 5.445x484 = € 6.628.820.
Toerekening aan BMZ: 4,0/6,6 x 100% = 60% van het totale oppervlak en aan klei 2,6/6,6 x 100% = 40% van het totale oppervlak. Bezet houden wordt dan voor deze case **614 m^2.y/kt BMZ** en **744 m^2.y/kt klei**.
Herstelperiode: 5 ha van het oorspronkelijk intensief weidegebied wordt waterplas; de rest wordt 2 meter lager afgeleverd, als intensief cultuurlandschap. Er wordt een (door Beetstra gemodelleerde) herstelperiode van 4 jaar aangenomen voor de overige 12 ha, waarvan gemiddeld de helft al tijdens herstel in de eindtoestand wordt aangenomen. Dit komt neer op 2 x 12.000 / 1.364 = **18 m^2.y/kt zand/klei extra**.

♦ Case 10: zand- en grindwinning Stein Limburg

Oppervlak: 1,3 ha => 13.000 m^2.
Productie: 2 jaar winning, 41.000 m^3, waarvan 70% zand (46 kton) en 30% grind (20 kton). Totale productie 66 kton.
Berekening bezet houden: 13.000 m^2 x 2 jr / 66 kt = **394 m^2.y/kt zand/grind**.
Toerekening van het oppervlak op basis van economische waarde: zand € 4,54/ton en grind € 4,54,-/ton hoeft er geen verdeling te worden gemaakt tussen beide.
Herstelperiode: Het oorspronkelijk braakliggend bedrijventerrein wordt gebruikt industrieterrein. Herinrichting duurt 2 jaar totdat bebouwd kan worden. Dit betekent nog 2 x 13.000 m^2 / 66 = **394 m^2.y/kt zand/grind extra**.

♦ Case 11: kleiwinning uiterwaarden (buitendijks) Gelderland

Oppervlak: 2 ha per jaar, incl. toegangswegen 24.000 m^2
Productie: 4 meter diep w.v. 90% klei => 190 kton/jr (met dichtheid 2,2 i.p.v. de 1,6 van Beetstra).
Berekening bezet houden: 24.000 m^2.y/190 kton = **126 m^2.y/kt uiterwaardenklei**
Herstelperiode: er wordt een herstelperiode van 50 jaar aangehouden op basis van [Helmer, 2001]; er ontstaat een bebost natuurgebied met water. Dit betekent 24.000 x 50 / 190 = **6316 m^2.y/kt uiterwaardenklei extra**, waarbij een hoge natuurwaarde ontstaat in de loop van deze 50 jaar. Een langere ontwikkelingsperiode is ook mogelijk; dan ontstaat er een nog hogere natuurwaarde, tot een climax stadium na maximaal 300 jaar.

♦ Case 12: kleiwinning de Boterhoek (binnendijks) Gelderland

Oppervlak: 9 ha (90.000 m^2)
Productie: in 5 jaar wordt 223.100 m^3 klei gewonnen (met dichtheid 2,2: 491 kton)
Berekening bezet houden: 90.000 x 5 / 491 = **917 m^2.y/kton komklei**
Herstelperiode: van intensief weidegebied wordt 2,5 ha waterplas en de rest weer weidegebied. Herstelperiode volgens Beetstra 4 jaar, waarvan gemiddeld 2 jaar reeds in definitieve toestand. Dit betekent: 2 x 65.000 / 491 = **265 m^2.y/kt komklei extra**

♦ Case 13: klei- en zandwinning Glijmenshof (binnendijks) Gelderland

Oppervlak: 26 ha = 260.000 m^2
Productie: in 5 jaar 240.000 m^3 klei (met dichtheid 2,2: 528 kton) en 50.000 m^3 zand (85 kton). Totale productie 613 kton)
Berekening bezet houden: 260.000 x 5 / 613 = 2121 m^2.y/kton klei/zand
Toerekening: op economische waarde (klei € 5.445,-/kton, zand € 4.538,-/kton): totale opbrengst 5.445 x 528 + 4.538 x 85 = € 3.260.860,-. Toerekening aan komklei: 2,9/3,3 = 88% naar klei geeft 0,88 x 260.000 x 5 / 528 = **2167 m^2.y/kt komklei** en 0,12 x 260.000 x 5 / 85 = **1835 m^2.y/kt zand**.
Herstelperiode: gebied wordt weer in oorspronkelijke toestand (intensief weidegebied) teruggebracht; volgens Beetstra vergt dit 4 jaar, waarvan weer 2 jaar in rekening moet worden gebracht: 2 x 260.000 / 613 = **848 m^2.y/kton klei/zand extra**.

♦ Case 14: kleiwinning Lobith (binnendijks) Gelderland

Oppervlak: 10 ha (100.000 m^2)
Productie: in 5-8 jaar wordt 150.000 m^3 klei gewonnen. Met dichtheid 2,2 wordt dit 330 kton.
Berekening bezet houden: 100.000 x (5 tot 8) / 330 = **242-1515 m^2.y/kton klei**.

Herstelperiode: gebied wordt weer in oorspronkelijke toestand (intensief weidegebied) teruggebracht; volgens Beetstra vergt dit 4 jaar, waarvan weer 2 jaar in rekening moet worden gebracht: 2 x 100.000 / 330 = **606 m^2.y/kton klei extra.**

Uit bovenstaande cases van het werk van Beetstra zijn de volgende gewogen gemiddelden bepaald voor bezet houden van land:

Tabel 1 Overzicht cases **bezet houden** (**vet**), gewogen gemiddelden [m^2.y/kton] (**voorlaatste kolom**) met **standaarddeviatie** (laatste kolom) en totale productie cases [kton] (prod., niet vet)

Case	1	2	3	4	5	6	7	8	9	10	11	12	13	14	Gew. gem.	Stand. dev.
BMZ	**287**	**150**		**489**	**681**	**654**	**1677**	**808**	**614**	**394**			**1835**		**665**	562
prod.	300	500		23000	17000	4000	3260	6000	880	46			85		55.071	
OHZ	**75**				**184**	**197**									**189**	67
prod.	80				3260	3200									6540	
grind	**284**		**585**	**489**	**674**	**654**	**1677**	**808**		**394**					**731**	430
prod.	380		5130	5750	4240	4000	1400	20000		20					40.920	
uiter.	**klei**										**126**				**126**	
prod.											190				190	
kom	**klei**					**802**			**744**			**917**	**2167**	**879**	**1127**	599
prod.						550			484			491	528	330	2383	

Daarnaast is voor **zand/grind/kalkzandsteenzand** gebruik gemaakt van een CD-ROM van RWS DWW, samengesteld door bureau Nieuwland [Nieuwland, 2001]. Hierin zijn van ruim 400 winlocaties zo veel mogelijk gegevens verzameld (vooral zandwinningen). Helaas ontbreken productie-hoeveelheden en de tijdsduur van de winningen. Voor 7 locaties in Noord-Oost Nederland zijn door DWW tijdsduur en productie bepaald [Broers, 1997], waardoor voor deze gevallen van BMZ en OHZ bezet houden bepaald kon worden.

De gegevens van de CD-ROM zijn als volgt bewerkt: het oppervlak (30-75 ha gemiddeld) en de diepte (meestal 20 tot 40 m) zijn per put gecombineerd tot volumegegevens van de putten, waarbij een correctie is uitgevoerd via gegevens over het talud (het gecorrigeerde putvolume bleek gemiddeld 70% kleiner dan oppervlak x diepte). Dit volume is vermenigvuldigd met een gemiddelde dichtheid van 1,7 t/m^3 om tot productiegegevens te komen. In 7 gevallen zijn deze berekeningen vervangen door gegevens uit vergunningen, beleidsplannen of brochures van de ontgronder, wat enigszins hogere danwel lagere waarden opleverden. Tenslotte zijn oppervlak en tijdsduur voor de 7 winputten gedeeld door de productiegegevens per put voor het bezet houden van land.

De resultaten (gewogen gemiddelden) voor bezet houden zijn:

- Voor beton/metselzand putten waar ophoogzand bijgewonnen wordt is het bezet houden **2070 m^2.y/kt BMZ** (3 cases; totale productie 10.300 kt).
- Voor putten waar alleen ophoogzand wordt gewonnen is het bezet houden **1770 m^2.y/kt OHZ** (4 cases; totale productie 22.645 kt).

Als de jaarlijkse productie voor de cases van Beetstra en DWW bij elkaar worden opgeteld en vergeleken met de nationale jaarlijkse productie, ontstaat het volgende beeld voor de regionale representativiteit:

	Jaarproductie cases [kt]	Nationale jaarproductie [kt]	Representativiteit
BMZ	7238	19.000	38%
OHZ	2014	75.000	3%
grind	4333	11.000	39%
klei	620	6.160	10%

Voor kalkzandsteenzand is gebruik gemaakt van gegevens van de industrie [Schuur, 1996]: 22 m^2.y/kt kalkzandsteenzand. ***Deze konden niet geverifieerd worden.***

Voor **mergel** is ook gebruik gemaakt van gegevens van de industrie [Lanser, 2001], geverifieerd met die van [Beetstra, 1998b]:

♦ Case mergelwinning Pietersberg

Oppervlakte: 30 ha per jaar in gebruik
Productie: 1.574.000 ton (1999), maar gemiddelde ontgrondingsvergunning is 1,3 miljoen ton/jaar
Berekening bezet houden: 300.000 m^2 / 1300 kt = **230 m^2.y/kt mergel**.
Hersteltijd: Voor het herstel van het gehele gebied (136 ha, totale winning ± 100 jaar) tot natuurgebied (inclusief waterplas) wordt door Beetstra 55 jaar hersteltijd gerekend [Beetstra, 1998a]
Hiermee wordt dit 1.360.000 m^2 x 55 / 1,3 miljoen kton x 100 = **0,58 m^2.y/kt mergel extra**.

Voor de **cementproductie** is eveneens gebruik gemaakt van gegevens van de industrie, die zijn vergeleken met die van [Beetstra, 1998b]:

Fabriek	Productie [kton/j]	Fabrieks-oppervlakte [ha]	Bezet land [m^2.y/kt]
Maastricht	2000	15	75
Rozenburg (maalinrichting)	500	8,4 (40% verhard)	67
IJmuiden (maalinrichting)	1000	4	80
Gewogen gemiddelde			**75**

Voor **betonmortelcentrales** komen uit [Beetstra, 1998b] 2 cases:

Fabriek	Productie [kton/j]	Fabrieksoppervlakte [ha]	Bezet land [m^2.y/kt]
Hallbeton (A'dam-Oost)	96	6	625
Betoncentrale Eindhoven	120	4	333
Gewogen gemiddelde			**463**

Voor **baksteenfabrieken** zijn in [Beetstra, 1998b] 11 cases beschreven. Hieruit is een gewogen gemiddelde bepaald: **1320 m^2.y/kt baksteen**.

2. Achtergrond basisgegevens *verandering* van landgebruik:

Op basis van dezelfde bronnen en gegevens ([Beetstra, 1998a], [Beetstra, 1998b] en [Nieuwland, 2001]) zijn voor verandering van landgebruik de volgende waarden bepaald. Daarbij is in de meeste gevallen de economische toerekening niet getoond, omdat deze niet wezenlijk de resultaten verandert; daar waar dat wel het geval is, is de toerekening beschreven op basis van de percentages in de achtergrondgegevens hierboven.

- Case 1: 200.000 m^2 / 12.160 kt = **16 m^2/kt zand/grind** veranderd van landbouwgrond in recreatieplas
- Case 2: 150.000 m^2 / 500 kt = **300 m^2/kt zand** veranderd van landbouwgrond in recreatieplas
- Case 3: 200.000 m^2 / 5.130 kt = **39 m^2/kt grind** veranderd van weiland/landbouwgrond in recreatieplas
- Case 4: 2.010.000 m^2 / 28.750 kt = **70 m^2/kt zand/grind** veranderd van extensief gebruikt uiterwaardegebied in 23% infrastructuur, 27% recreatieplas, 18% recreatie- en opslagterrein, 10% woningbouw en 21% natuur
- Case 5: 1.670.000 m^2 / 24.500 kt = **68 m^2/kt zand/grind** veranderd van extensief gebruikt weiland in natuurgebied (32% oever, 68% plas)
- Case 6: 900.000 m^2 / 11.750 kt = **77 m^2/kt zand/grind** veranderd van kleinschalige weilanden tot 34% bebouwing, 2,5% bos en 63,5% recreatieplas met natuurwaarde
- Case 7: 520.000 m^2 / 4.650 kt = **112 m^2/kt zand/grind** veranderd van landbouwgebied in 67% extensief gebruikte recreatieplas en 33% natuur

- Case 8: 2.100.000 m^2 / 26.000 kt = **81 m^2/kt grind** veranderd van akkerbouw/veeteelt/fruitteelt in 64% natuurplas met extensieve recreatie en 36% natuurgebied met extensieve recreatie
- Case 9: 0,6 x 180.000 m^2 / 880 kt = **123 m^2/kt zand** en 0,4 x 180.0000 m^2 / 484 kt = **149 m^2/kt klei voor 28% veranderd** van intensief weidegebied in waterplas; rest blijft intensief weidegebied
- Case 10: 13.000 m^2 / 66 kt = 197 m^2/kt zand/grind, maar niet wezenlijk veranderd (van braakliggend industrieterrein in industrieterrein in aanbouw), dus eigenlijk **0 m^2/kt zand/grind** veranderd
- Case 11: 24.000 m^2 / 190 kt = **126 m^2/kt uiterwaardenklei** veranderd van extensief weidegebied in natuurbouw met water, meestal in open verbinding met de rivier
- Case 12: 90.000 m^2 / 491 kt = **183 m^2/kt komklei voor 28% veranderd** van intensief weidegebied in een waterplas; de rest blijft intensief weidegebied
- Case 13: 0,88 x 260.000 m^2 / 528 kt = 433 m^2/kt komklei en 0,12 x 260.000 m^2 / 85 = 367 m^2/kt zand, maar niet wezenlijk veranderd (het gebied wordt weer voor intensieve beweiding gebruikt, dus eigenlijk **0 m^2/kt klei/grind** veranderd
- Case 14: 100.000 m^2 / 330 kt = 303 m^2/kt komklei**,** maar niet wezenlijk veranderd (het gebied wordt weer voor intensieve beweiding gebruikt, dus eigenlijk **0 m^2/kt klei/grind** veranderd

Uit bovenstaande cases van het werk van Beetstra zijn de volgende gewogen gemiddelden bepaald voor verandering van landgebruik, en de grootte van de steekproef (tabel 2):

Tabel 2 Overzicht totale productie per soort overgang en gewogen gemiddelden voor verandering landgebruik (cases Beetstra), met daaronder de berekening van de grootte van de steekproef in % van de totale winning in NL

	Totale productie [kt]	Verandering [m^2/kt]
BMZ/grind van landbouw naar recreatieplas	27436	77
BMZ/grind van extensieve weiden naar wegen	6613	16
BMZ/grind van extensieve weiden naar recreatieplas	7763	19
BMZ/grind van extensieve weiden naar recreatieterrein	5175	13
BMZ/grind van extensieve weiden naar bebouwing	5595	73
BMZ/grind van extensieve weiden naar natuur	27478	69
BMZ/grind van landbouw naar natuur	10898	85
Uiterwaardenklei van extensieve weiden naar natuur	190	126
Komklei van intensief weidegebied naar waterplas	1833	25

	Jaarproductie cases [kt]	Nat. jaarprod. [kt]	Deel van totale winning
Totaal BMZ/grind	11231	30000	37%
Totaal klei	541	6160	9 %

De nationale jaarproducties komen uit het Tweede Structuurschema Oppervlaktedelfstoffen (deel 1, toelichting, concept 27/10/2000).

Uit de CD-ROM van DWW volgen de volgende gemiddelden voor verandering landgebruik:

Soort grondstof	Totale productie cases	Verandering landgebruik
Ophoogzand	283.000 kt	**43 m^2/kt OHZ**
Beton/metselzand	80.000 kt	**36 m^2/kt BMZ**[19]

[19] Helaas is er voor beton- en metselzand (BMZ) alleen geen waarde gevonden. Voor mixen met ophoogzand zijn wel gevonden: 28 m2/kt BMZ/OHZ en 32 m2/kt OHZ/BMZ. Samen met de gegevens voor OHZ suggeert dit dat BMZ een lager landgebruik zal hebben dan OHZ. Bij toerekening van alle drie de gegevens op economische waarde (uitgaande van een gelijke

Kalkzandsteenzand 19.000 kt **20 m^2/kt kalkzandsteenzand**

Hiervoor zijn voor OHZ, BMZ en kalkzandsteenzand op basis van respectievelijk 18, 16 en 1 winputten de verandering van landgebruik bepaald. Voor deze gegevens zijn weinig gegevens bekend over de jaarlijkse productie. Uitgaande van gemiddeld 20 jaar winning zou de totale productie uit de DWW cases extra bijdragen: 13% voor BMZ, met 97.000 kt jaarlijkse nationale winning 15% voor OHZ en 30% voor kalkzandsteenzand. De geschatte bijdrage van alle cases aan de nationale winningen wordt daarmee:

BMZ/grind:	**50%**
OHZ:	**15%**
KZSZ:	**30%**
klei:	**9%**

Voor mergel [Lanser, 2001] is een waarde van **1,3 m^2/kt** gevonden voor de verandering van natuur (kalkgrasland) naar bedrijfsterrein (15 ha veranderd voor 120.000 kton; 11% van het totale gebied). Daarnaast ontstaat het ENCI bergmeer van ongeveer 100 ha [Beetstra, 1998a] (74% van het totale gebied), dat als extensieve recreatieplas kan worden geclassificeerd. Hiervan uitgaande is deze aanvullende verandering 1.000.000 m^2 / 120.000 kt = **8,3 m^2/kt** zijn van kalkgrasland naar extensieve recreatieplas. De rest van het gebied wordt weer natuur.

Als algemene waarden voor Nederlandse oppervlaktedelfstofwinningen hebben wij hieruit gehaald:

40 m^2/kt ophoogzand
50 m^2/kt beton- en metselzand en grind[20]
20 m^2/kt kalkzandsteenzand
10 m^2/kt mergel[21]
126 m^2/kt uiterwaardenklei
25 m^2/kt komklei

Hierbij is dus niet inbegrepen de kwaliteitsverandering. Deze kan positief of negatief zijn, of gedeeltelijk beide. De verdeling van de verschillende situaties voor zand en grind is alleen tentatief af te lezen uit de cases van Beetstra (zie tabel 2). Bij gebrek aan betere statistieken zal de verdeling van situaties uit die cases geëxtrapoleerd worden voor heel Nederland.

Voor de **cementproductie** is gebruik gemaakt van gegevens van de industrie, die zijn vergeleken met die van [Beetstra, 1998b]:

Fabriek	Meegenomen productiejaren	Totale productie [kton]	Fabrieksoppervlakte [ha]	Verandering [m^2/kt]
Maastricht	100	200000	15	0,75
Rozenburg (maalinrichting)	75	37500	8,4 (40% verhard)	0,90
IJmuiden (maalinrichting)	75	75000	4	0,53
Gewogen gemiddelde				**0,83**

Voor **betonmortelcentrales** komen uit [Beetstra, 1998b] 2 cases:

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productie BMZ en OHZ voor de cases BMZ/OHZ en OHZ/BMZ gezamenlijk) komen we uit op 36 m2/kt.

[20] Op basis van een gewogen gemiddelde uit de cases van Beetstra (62 m2/kt BMZ/grind) bij een totale productie van 91.000 kt, en het gewogen gemiddelde uit de cases van DWW (36 m2/kt BMZ) bij een totale productie van 80.000 kt.

[21] Exclusief de verandering van de vorm van de Pietersberg zelf (53-65e6 m3 afgegraven => 0,4 m3/t)

Fabriek	Totale productie [kton]	Fabrieksoppervlakte [ha]	Verandering [m^2/kt]
Hallbeton (A'dam-Oost)	2880	6	21
Betoncentrale Eindhoven	3600	4	11
Gewogen gemiddelde			**15**

Voor **baksteenfabrieken** zijn in [Beetstra, 1998b] 11 cases beschreven. Hieruit is een gewogen gemiddelde bepaald: **18 m^2/kt baksteen**.

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Meeting report notitie ontgrondingen d.d. 8 november 2001

From
Drs Erwin Lindeijer

Subject
Toetsing data fysieke ingrepen door ontgrondingen voor landgebruik in LCA

Our number

Present
J. Broers, RWS DWW
E. Lindeijer, TNO Industrie
H. Maessen, St. Grind, Panheel Groep
K. van Mourik, Terca (baksteen)
J. Rademakers, SIGHT
G. Sigmond, FODI

Absent
G. van Solkema, Dekker
J. van de Kamp, K3
H. Huydts, ENCI

Onderwerpen:
- Inleiding over het waarom en wat van Landgebruik in LCA (kader) door J. Broers
- Discussie over het kader (milieubeoordeling en toerekening milieuwinst)
- Bespreking fysieke ingrepen (modelberekeningen E. Lindeijer) a.d.h.v. de voorliggende notitie

(de vragen in de notitie zijn niet direct besproken; wel zijn onderstaande modelberekeningen voorgelegd en besproken, als illustratieve invulling van de bepaalde getalswaarden voor de diverse ontgrondingen)

Modelberekeningen representatieve cases ontgrondingen

	OHZ	KZSZ	BMZ	grind	uiterw.klei	komklei	mergel
netto oppervlak put [ha]	**25**	**40**	**30**	**185**	**2**	**20**	**25**
diepte [m]	**20**	**40**	**20**	**10**	**4**	**4**	**4**
volume [m3]	4000000	12800000	4800000	14800000	64000	640000	800000
dichtheid [t/m3]	**1,7**	**1,7**	**1,7**	**1,7**	**2,2**	**2,2**	**1,7**
totale opbrengst [kt]	6800	21760	8160	25160	141	1408	1360
duur winning [jaren]	**35**	**10**	**20**	**10**	**1**	**5**	**1**
bruto oppervlak [ha]	**30**	**50**	**37**	**210**	**2,4**	**25**	**30**
netto/bruto oppervlak	0,8	0,8	0,8	0,9	1,0	1,0	1,0
bezet houden [m2jaren/kt]	1544	**230**	907	835	170	888	221
landelijk gemiddelde	1560	22	890	730	126	1130	230
verandering [m2/kt]	44	23	45	83	170	**178**	**221**
landelijk gemiddelde	40	20	50	50	126	25	10

De gegevens in deze tabel zijn aangepast naar het commentaar van de FODI-leden.

Samenvatting inleiding
Het beoordelingskader van de huidige LCA's is beperkt tot milieu in nauwe zin. Landschappelijke effecten, duurzaamheid en maatschappelijke acceptatie in brede zin zitten daar niet in. En tot op heden zijn zelfs biodiversiteitseffecten en life support effecten (invloed op natuurlijke draagkrachtfuncties) geen standaard onderdeel van LCA-berekeningen. De LCA-methode, die ingezet is om de kwalitatieve DuBo voorkeurslijstjes te vervangen door een meer kwantitatief/geobjectiveerd/genuanceerd oordeel over bouwproducten, is hiermee nog erg gebrekkig. Middels onderhavig project wil DWW hier stapsgewijs verbeteringen in doorvoeren, en de toepasbaarheid . Er is een methode ontwikkeld om biodiversiteitseffecten op (planten)soortsniveau en op ecosysteemniveau mee te nemen, en om biomassa-productie als maat voor life support mee te nemen, gekoppeld aan de oppervlakte en de tijd die een activiteit duurt. Er wordt gekeken naar het in beslag nemen van land (bezet houden) en netto veranderingen.(voor-na). Na afloop van dit project (in 2002) wordt de uitkomst geëvalueerd, waarna eventueel implementatie volgt.

8.4 Samenvatting discussie

Voor ontgrondingen wordt in de methode rekening gehouden met een mogelijk hogere natuurwaarde na de ontgronding. Voor ontgrondingen in NL is er tegenwoordig meestal tot altijd sprake van een verbetering van de kwaliteit, waarbij ontgronders ofwel zelf het gebied onder natuurontwikkeling in bezit houden, ofwel dit overdragen aan natuurbeheerders (in de Wet Ontgrondingen is opgenomen dat bij vergunningverlening nu eeuwigdurend onderhoud geldt, met de mogelijkheid tot afkoop). Er wordt gesteld dat de verandering die optreedt door de ontgronding eeuwenlang zichtbaar blijft in het landschap. Als deze verandering als positief wordt gewaardeerd, moet dat dan ook in de LCA tot uitdrukking worden gebracht. Het afbreken van de effecten na een bepaalde tijd op basis van de economische verantwoordelijkheid voor b.v. een natuurontwikkelingsproces (b.v. bij overdracht naar het Wereld Natuur Fonds) wordt door de ontgronders afgewezen, hoewel dit wel gangbaar is in LCA.

Omdat het niet mogelijk is om in LCA te rekenen met een eeuwigdurend effect is het voorstel gedaan om uit te gaan van de tijdsduur tot het ontstaan van een climax-stadium (van 50 tot 200 jaar volgens de rapportage van Stichting Ark).

Tegenwoordig zijn bijna alle winningen secundair (ontgronden alleen voor de grondstof – primaire winningen- wordt niet meer maatschappelijk geaccepteerd). De ontgronders zijn bezorgd of het verschil tussen een primaire en een secundaire winning wel duidelijk uit de verf komt. Aanleidingen zoals het verhogen van de overstromingsveiligheid zijn soms de primaire reden waardoor de ontgronding plaats kan vinden. Lindeijer geeft aan dat waar de effecten niet in de milieubeoordeling kunnen worden meegenomen, een toerekening moet plaatsvinden waardoor de ontgrondingen niet alle effecten (positief danwel negatief) toegewezen worden. Gangbaar is om dit op basis van relatieve economische inspanningen te doen. Positieve verdrogingseffecten door een winning (zoals aangegeven door Maessen) zouden eventueel niet via economische toerekening meegenomen hoeven te worden, omdat verdroging in de nabije toekomst wel meegenomen zouden kunnen worden in LCA.

Voor de toerekening van een gecombineerde winning vindt allocatie op basis van de relatieve economische waarde van de outputs plaats. Dit principe wordt uitgelegd aan de hand van een aantal voorbeelden. Volgens Lindeijer is het binnen de range van uitkomsten van de cases minder belangrijk welke economische waarden er gekozen wordt. Wel biedt hij nog aan om voor 1 van de beschreven cases een gevoeligheidsanalyse in het rapport op te nemen.

In de plaatjes van Lindeijer wordt gesuggereerd dat er pas na afloop van de ontgronding met herstelwerkzaamheden wordt begonnen. Volgens de aanwezige ontgronders is dit tegenwoordig niet meer zo. Soms mag men na een kleiwinning zelfs na een paar maanden al niet meer op de ontgonnen plek komen, om herstel toe te laten. Erwin neemt hiervoor een apart plaatje op in de rapportage, en zal dit aspect meenemen in de berekening van de effecten door ontgrondingen. Overigens zal naar verwachting geen significant verschil in biodiversiteit aan te tonen zijn op basis van de NL plantensoorten database Florbase. Volgens Stichting Ark ontstaat die hogere biodiversiteit wel na verloop van tijd. Maessen overhandigt een folder over een moderne ontgronding, en geeft aan dat er een case is waar de natuurwaarde regelmatig gemonitoord wordt. Indien dergelijke gegevensverzamelingen gangbaar worden, kunnen er inderdaad recente modern-modal procesgegevens worden gebruikt in LCA's met ontgrondingen.

De ontgronders zijn bezorgd over wat er met de resultaten van deze studie gebeurt, omdat men in de markt elk verschil in milieubeoordeling (hoe onsignificant ook) zal willen gebruiken om een bepaald materiaal/product er beter uit te laten komen. Volgens Lindeijer zal hem dat vooral in de beoordelingsstap zitten. Daar worden een paar belangrijke methodische keuzen gemaakt, die hij expliciet wil maken, door resultaten volgens beide keuzen te tonen zonder een eindoordeel te vellen. En de LCA methodiek heeft natuurlijk de beperkingen van een smal blikveld op duurzaamheid.

Ook zijn de ontgronders bang dat er, door het toepassen van een gewogen gemiddelde in de markt, buitenlandse 'vervuilers' de goede scores van de NL ontgrondingen omlaag halen bij toepassing van generieke gegevens. Broers merkt op dat er twee mogelijkheden zijn om hier goed mee om te gaan. Je kunt architecten/projectontwikkelaars specifieke gegevens laten gebruiken voor bouwen met NL materiaal (al dan niet met een keurmerk), of er zou een zogenaamd marginaal proces moeten worden toegepast: op welk proces heeft een ontwerper invloed? Voor ontgrondingen is dat altijd een buitenlands proces (we moeten importeren om te voldoen aan de te hoge vraag in NL). Dat betekent niet dat oppervlaktedelfstoffen slecht voor het milieu zijn, maar dat voor oppervlaktedelfstoffen we de meeste winst behalen als we iets aan dat marginale (buitenlandse) proces doen. In hoeverre het mogelijk is om deze zaken mee te nemen binnen het MMG-traject (voor het bepalen van de milieubelasting van een woongebouw) kan niet aangegeven worden, en gaat voorbij dit project.

8.5 Samenvatting bespreking fysieke ingrepen

Er wordt geconcludeerd dat de ontgrondingscases in de notitie van Lindeijer veelal niet over modern-modale processen gaat. Er is sprake van een groot verschil in de effecten van historische cases en die van de ontgrondingen die nu plaatsvinden. Lindeijer geeft aan dat de fysieke gegevens over de ontgrondingen veel robuuster zijn dan het verhaal over de beoordeling ervan. Immers, de hoeveelheid die je uit een put kunt halen en de tijd die dat vergt verandert niet zoveel. Om die reden stelt hij voor om toch de modelberekeningen die hij heeft uitgevoerd te bespreken. De modelberekeningen zijn gemaakt om de berekende gemiddelde waarden voor de fysieke ingrepen door ontgrondingen in NL te illustreren met fictieve cases die wel realistisch zijn. Zie bijgevoegde tabel. Hierover worden de volgende opmerkingen gemaakt:

Rademakers: de correctiefactor 0,8 voor het berekenen van het werkelijk volume is voor kleiwinning niet terecht, omdat er erg ondiep wordt gewonnen. Lindeijer zal het aanpassen.

van Mourik: de gebruikte dichtheid is erg hoog. Op basis van droge stof zou die 1,3 – 1,6 moeten zijn, met 1,55 als gebruikte waarde. Lindeijer geeft aan dat hier het natte klei genomen moet worden, omdat de klei pas in het volgend proces (baksteen bakken) gedroogd wordt. Daar gaat dan in kg veel meer klei in dan er uitkomt; het water verdampt. Hij heeft uit een literatuurbron een range van dichtheden gehaald, en daar het gemiddelde 2,2 uit gehaald.

Omdat deze modelberekeningen pas ter vergadering zijn uitgedeeld, kunnen de ontgronders (ook degenen die er niet waren) hier deze week nog op reageren.

Annex 4 Renaturation after aggregate extractions in NL (in Dutch) (A report by W. Helmer, Stichting Ark)

Ontwikkelingstijden van ecosystemen na ontgronding

Algemeen

De situatie voor en na ontgronding is doorgaans zo verschillend dat na ontgronding niet zozeer sprake is van ecologisch herstel (met hersteltijden e.d.), maar eerder van nieuwe omstandigheden waaronder zich ecosystemen kunnen ontwikkelen, die er daarvoor niet waren. De invloed van ontgrondingen geldt dan ook vooral de ecologische potenties. Die invloed kan:

1) *negatief* zijn: na ontgronding zijn de ecologische potenties van een plek afgenomen.
   Bijvoorbeeld een maïsakker op een potentieel rivierduin, die door zandwinning verandert in een diepe put met steile oevers
2) *positief* zijn: na ontgronding zijn de ecologische potenties toegenomen.
   Bijvoorbeeld een sterk ontwaterd en intensief bemest weiland, dat na kleiwinning verandert in een schrale zandvlakte vlak boven de grondwaterspiegel
3) *'veranderlijk'* zijn: na ontgronding zijn de ecologische potenties van een plek sterk veranderd ten opzichte van de uitgangssituatie, zonder dat sprake is van een duidelijke toe- of afname.
   Bijvoorbeeld een beboste leemhelling, die na diepe mergelwinning verandert in steile krijtwanden met uittredend kwelwater.

Wat precies de (nieuwe) ecologische potenties na ontgronding zijn, hangt in sterke mate af van de wijze waarop de ontgronding wordt uitgevoerd en opgeleverd:

- de opleveringsdiepte: is die boven (schijn)grondwater-invloeden, staat die onder invloed van grondwaterschommelingen, wordt het ondiep water met licht tot op de bodem of blijft er een diepe plas achter
- het overblijvende reliëf en de steilte van de buitentaluds
- de aard van het substraat: is dat oorspronkelijk of is het aangevuld, is het mineraal of humeus.

Tenslotte is het van belang of na oplevering een natuurlijke successie wordt toegelaten of dat er een veel meer door mensen gestuurde ontwikkeling (recreatiegebied, hercultivering tot landbouwgrond, nastreven van bepaalde vastomlijnde natuurdoelen) plaatsvindt.
Wat dit laatste betreft: in het onderstaande schema is uitgegaan van een spontane successie op basis van natuurlijke processen zoals overstromingsinvloeden, erosie en sedimentatie, natuurlijke begrazing, grondwaterstandsschommelingen e.d. Antropogeen gestuurde vegetatietypen, zoals hooilanden, heidevelden of blauwgraslanden zijn vaak afgeleiden van deze natuurlijke successie c.q. gefixeerde stadia uit deze successie.
In deze context van natuurlijke processen krijgt ook het begrip climax een nieuwe invulling: de climax is niet een statisch eindstadium van een vegetatie-ontwikkeling, maar een dynamische evenwichtssituatie waarin opbouw- en afbraakfases, met alle tussenliggende successiestadia, in een ruimtelijk mozaïek naast elkaar voorkomen.

Hierna worden de verschillende soorten ontwikkelingen na ontgronding uitgewerkt voor de meest voorkomende situaties in Nederland. Daarbij is een hoofdverdeling gemaakt op basis van de vijf belangrijkste substraattypen : klei, zand, grind, krijt en veen. Vervolgens is er een onderverdeling gemaakt op grond van verschillende opleveringsdieptes ten opzichte van het gemiddelde waterniveau. Zoveel mogelijk is geprobeerd de tijdsduur na beëindiging van de ontgronding (t, in jaren), de biodiversiteit (s, aantal soorten vaatplanten/10 ha) en de biomassa-productie (m, in tonnen nat-gewicht per ha) van de verschillende ontwikkelingsstadia te kwantificeren. Waar dit relevant leek zijn zijwegen bewandeld in de richting van een meer recreatieve of agrarische oplevering van de ontgrondingslocatie.

De getallen zijn met flinke marges aangegeven, omdat de omstandigheden van plek tot plek aanzienlijk kunnen verschillen. Vooral de aanvang en tijdsduur van verschillende successiestadia is aan grote variaties onderhevig, afhankelijk van factoren als:
- de troebeling en/of kwaliteit van het water
- de aanvoer van sediment
- de aanwezigheid van zaadbronnen
- de min of meer toevallige volgorde waarin planten zich vestigen
- de begrazingsintensiteit
- de invloed van golfslag
- de toevallige omstandigheden (weertype, waterstand) op het moment van oplevering

Het tijdvak t wordt steeds begrenst door twee cijfers, die resp.:
- de kortste periode aangeven waarin een bepaald successiestadium zich na ontgronding kan vestigen (1e cijfer)
- de periode waarin een successiestadium tot volle wasdom komt c.q. overgaat in een volgend stadium (2e cijfer)

Stichting Ark en ontgrondingen

Veel van de hierna gepresenteerde gegevens zijn afkomstig uit de voorbeeldgebieden van Stichting Ark. Stichting Ark zet zich in voor natuur- en landschapsontwikkeling in kust-, rivier- en beekdalsystemen. Hiervoor heeft de stichting de afgelopen 10 jaar ca. 60 voorbeeldgebieden opgezet, waarvan een groot deel inmiddels weer is overgedragen aan andere terreinbeherende organisaties.
Basisprincipes in het werk van Ark zijn:
1. natuurontwikkeling op basis van natuurlijke processen (rivierdynamiek, nat. begrazing e.d.)
2. creëren van grote betrokkenheid van lokale bevolking voor duurzaam draagvlak natuurbescherming
3. koppeling met economische functies zoals waterwinning, veiligheid, recreatie en ontgrondingen

Wat de samenwerking met ontgronders betreft: veel voorbeeldgebieden van Stichting Ark zijn voormalige ontgrondingen:
- *enkele tientallen kleiwinningen in de Gelderse Poort, verder stroomafwaarts langs Waal, Nederrijn en IJssel en langs de Zandmaas*
- *een tiental zandputten + oevers in diezelfde regio's*
- *een tiental oude grindplassen + oevers langs de Grensmaas en in het Maasplassengebied*
- *een mergelgroeve in het Beneden-Geuldal*

Een aantal grote projecten (o.a. Gelderse Poort, Grensmaas) heeft Stichting Ark in opdracht van het Wereld Natuur Fonds opgezet en uitgevoerd. Stichting Ark werkt daarnaast samen met Rijkswaterstaat, diverse provinciebesturen, waterschappen, waterwinbedrijven, landelijke en provinciale natuurbeschermingsorganisaties en particuliere grondbezitters.
Stichting Ark maakt onderdeel uit van het Netwerk Natuurontwikkeling waarin een grote groep mensen en bedrijven (o.a. Stroming-planning, Rombus-natuurfilms, Meander-natuurarrangementen) ieder op hun eigen manier, samenwerkt bij de verwezenlijking van bovengenoemde idealen.

1. Ontwikkeling op kleibodems

1.1 In diep water
Diepe putten met een kleibodem komen vooral voor in het IJsselmeergebied en in de uiterwaarden van de grote rivieren. Het zijn veelal voormalige zandwinningen die tijdens overstromingen of door zijdelingse inspoeling een slibbodem krijgen. Het water boven de slibbodem is dermate troebel dat er in de diepe plas zelf geen groei van vaatplanten mogelijk is. Als onderdeel van een groter watersysteem met ondieptes kunnen dergelijke diepe putten echter wel als zinkput voor slib fungeren, waardoor het water boven de ondieptes helderder wordt dan in de situatie zonder zo'n slibvang (zie verder ondiepe klei en zandplassen). Deze eigenschap maakt dat ook zandgaten in uiterwaarden, ondanks voortdurende aanvoer van slib, langdurig geschikt kunnen blijven voor dagrecreatie.
Pas als de opslibbing de bodem op minder dan 1,5-2 meter diepte brengt wordt de ontwikkeling van een vegetatie mogelijk (zie verder ondiepe kleiput). Dit proces kan langs de rivieren, bij een opslibbingssnelheid van hooguit enkele dm/jaar, vele 10-tallen tot enkele 100-den jaren duren. In IJsselmeer en Markermeer nog langer.

1.2 In ondiep water
Het betreft hier met name kleiputten, die niet tot op de zandige, grindige of venige ondergrond zijn uitgegraven, maar het kunnen ook ontzandingen/ontgrindingen in de uiterwaarden zijn, die door opslibbing of aanvulling met slib nog maar enkele meters diep zijn.
pionierfase
in het water o.a. fonteinkruiden, nymphaeïden (gele plomp, watergentiaan) en langs de oever vele tientallen moerasplanten. In de uiterwaarden kan, met name door zomeroverstromingen, de verdere successie steeds weer geblokkeerd worden, waardoor pionierfasen decennia lang stand houden. Ook plantenetende watervogels en bodemwoelende vissen kunnen de pionierfase verder oprekken.
t: 1-50 s: 10-50 m: 5-10
meest voorkomende successie
er is een groot verschil tussen kleiputten, waarin voortdurend nieuwe sedimentatie van slib plaatsvindt (uiterwaarden) en kleiputten waar die slibsedimentatie ontbreekt en de verlanding bepaald wordt door ophopend organisch materiaal (laagveenvorming). In het eerste, meer dynamische geval is de soortenrijkdom aan vaatplanten doorgaans minder groot (vaak liesgras-dominantie) en komt de bodem relatief snel omhoog waardoor zich eerder houtige gewassen (met name struikwilgen) vestigen. In het tweede geval kan een breed scala aan water- en moerasplanten ertoe bijdragen dat de kleibodem met een dikke laag organisch materiaal wordt bedekt, waardoor zich geleidelijk aan een laagveenmoeras ontwikkelt
laagveenmoeras
t: 10-200 s: 50-100 m: 10-15
wilgenstruweel
t: 2-50 s: 25-50 m: 15-25
'climax'
Beide successiereeksen ontwikkelingen zich uiteindelijk in de richting van een zeer bosrijk landschap, waarin grote herbivoren (bevers!) slechts een beperkte invloed hebben. Op organische bodem ontwikkelen zich doorgaans broekbossen met veel elzen. In opslibbingsgebieden kan de ontwikkeling via zachthoutooibos uiteindelijk zelfs richting een productief hardhoutooibos gaan
broekbos
t: 100-300 s: 50-100 m: 10-15
zachthoutooibos
t: 15-100 s: 50-100 m: 15-20

1.3 Regelmatig overstromend
In uiterwaarden of in kwelgebieden, waarbij de bodem van de kleiput net boven het gemiddelde grondwaterniveau ligt
pionierfase
Deze bestaat zowel uit vele 10-tallen soorten kruiden (slijkgroen, vlooienkruid, goudzuring e.d.) als uit kiemplanten van wilgen, populieren en elzen (de laatsten vooral in geïsoleerde laagtes met fluctuerend kwelwater)
t: 0-5 s: 50-100 m: 5

meest voorkomende successie
Afhankelijk van de begrazingsdruk tijdens de opgroeifase van de jonge boompjes zal zich grasland of een meer of minder gesloten zachthoutooibos ontwikkelen. In het verleden startte de begrazing vaak pas enkele jaren na oplevering van de ontgronding, wat leidde tot een dicht ooibos. De groeikracht van het jonge wilgenbos is namelijk enorm, met een jaarlijkse lengtegroei van bijna twee meter. Tegenwoordig (ook onder invloed van RWS die niet teveel stromingsweerstand wenst) start de begrazing vrijwel gelijktijdig met de ontgronding, hetgeen een veel opener bos of zelfs grazige laagtes met kruipende boterbloem, hondsdraf, vossenstaart, zilverschoon e.d. oplevert. In perifere laagtes met fluctuerende grondwaterstanden gaat de successie meer richting elzenbroek.
'zilverschoon-grasland'
t: 2-20 s: 50-100 m: 10-15
zachthoutooibos
t: 2-50 s: 50-100 m: 20-30
elzenbroek
t: 5-50 s: 50-100 m: 10-20
'climax'
Na de pionierfase treedt er nog maar sporadisch kieming van jonge wilgen op (bijv. op plekken waar de bodem opnieuw verstoord wordt door mollen, zwijnen e.d.). Wilgen kunnen wel weer vegetatief uitlopen. Onder een natuurlijk begrazingsregime vestigen zich, zowel in de graslanden als in de wilgenbossen jonge meidoorns, essen en eiken, waarvan de laatste door kunnen groeien als de groeiplaats niet langer dan 3 maanden overstroomt en 's zomers liever helemaal niet. Deze hardhoutsoorten zullen geleidelijk aan de wilgen van het eerste uur verdringen. Het resultaat is uiteindelijk een mozaïek van vochtige graslanden, brandnetelruigtes en wilgenbosjes met een toenemend aandeel aan meidoorns, essen, eiken e.d.
t: 50-200 s: 100-150 m: 15-20

1.4 Zelden overstromend
Omdat de overstromingsfrekwentie veelal gerelateerd is aan de hoogteligging, en de hoogstgelegen delen van het rivioerengebied vaak zandig zijn, komt dit type weinig voor. Een uitzondering vormen ontkleiingen in hoog-bekade uiterwaarden of in het beneden-rivierengebied, die boven het grondwater-niveau met een kleibodem worden opgeleverd.
pionierfase
Deze kan rijk zijn aan kruiden en kiemplanten van houtige gewassen, waarvan de zaden ook met top-hoogwaters worden aangevoerd.
t: 0-2 s: 100-150 m: 5
meest voorkomende successie
Min of meer vergelijkbaar als met de regelmatig overstromende kleibodems, zij het dat de terreinen makkelijker toegankelijk zijn voor grazers waardoor het aandeel open vegetatie over het algemeen groter is. Deze open, grazige vegetaties kunnen zich, zeker wanneer de bodem wat zavelig is, ontwikkelen tot bloemrijke glanshaver-graslanden. Veel hooilandbeheer richt zich met name op deze plantengemeenschap.
'glanshaver-grasland'
t: 2-50 s: 100-150 m: 5-10
'climax'
Hier zal de ontwikkeling naar een (open) hardhoutooibos zich sneller doorzetten dan op de regelmatig overstromende kleibodems. De soortenrijkdom is er nog groter
t: 50-100 s: 150-200 m: 15-20

1.5 Droog, nooit overstromend
Komt alleen voor op drooggelegde, binnendijkse winningslocaties, bij afgravingen van tertiaire klei of op de bodem van mergelgroeves wanneer deze kunstmatig of na solifluctie met leem wordt afgedekt.
pionierfase
Door het ontbreken van zaadaanvoer via de rivieren, zijn deze locaties vaak armer aan soorten. Door de slechte doorlatendheid van de bodem kunnen zich wel langdurig ondiepe plassen met regenwater in het terrein bevinden met een geheel eigen pionierwereld van russen e.d.
t: 0-5 s: 50-100 m: 5
meest voorkomende successie

Nog toegankelijker voor grazers, waardoor bosontwikkeling langer geremd kan worden. Wilgen en populieren, die hier kiemen zijn vaak minder vitaal dan in de overstromingsgebieden. Boomsoorten als berk, ratelpopulier en een breed assortiment aan hardhoutbomen en struiken nemen het snel van de wilgen over.
In hoger gelegen klei- of leemgroeves kunnen jonge berken en bramenruigtes snel de overhand nemen.
t: 5-50 s: 50-100 m: 5-10
'climax'
Grazig, sorrtenrijk bostype
t: 50-200 s: 100-200 m: 10-15

2. Ontwikkeling vanuit zandig substraat

2.1 In diep water
Diepe zandplassen houden alleen een zandige bodem als ze niet overstromen of anderszins in contact met slibrijk rivierwater staan. Dit is dus vooral het geval op binnendijkse locaties. Dergelijke plassen worden bovendien vaak gevoed door kwelwater, waardoor de waterkwaliteit en het doorzicht zeer goed zijn. Voedselrijk (rivier)water leidt tot algenbloei waardoor er voor waterplanten vaak te weinig licht op de bodem doordringt. In helder water kan plantengroei tot op een diepte van 8 meter voorkomen. Veel hangt daar af van de oplevering van de plas, want hoe steiler de oevers en hoe dieper de plas, hoe minder planten (en andere organismen) zullen profiteren van de goede waterkwaliteit. In potentie zijn dit ideale recreatieplassen. Het type zand is ook nog van invloed op de plantenontwikkeling: bestaat de onderwaterbodem uit vrij grof (rivier)zand of uit grof zand met een slibfractie dan groeit er veel meer dan op instabieler fijne stuifzandfracties, zoals bijv. bij Panheel.
pionierfase
kranswieren, fonteinkruiden en andere waterplanten.
Zolang er zand gewonnen wordt in de plas is deze vaak nog zo troebel dat de omstandigheden nog weinig geschikt zijn, m.u.v. de uithoeken van de plas
t: vanaf 0-5 s: 10-20 m: <5
meest voorkomende successie
Er treedt nauwelijks successie op, al kunnen er jaarlijks soorten verdwijnen of bijkomen, bijv. door zaadaanvoer via watervogels. Grote groepen plantenetende vogels zoals meerkoeten kunnen periodiek een enorme aanslag op de waterplanten plegen
'climax'
Wanneer de structuur en soortenrijkdom van de watervegetatie hun optimum bereiken
t: vanaf 10-50 s: 25-40 m: < 5

2.2 In ondiep water
Dit kunnen de ondiepe oeverzones van een diepe zandplas zijn, zelfs wanneer deze langs de rivier ligt, omdat het aangevoerd slib dan voornamelijk naar de diepere delen zal afzinken. Verder kunnen dit zandbodems zijn nadat de bovenliggende veen- of kleilaag is afgegraven. Kleiwinningen met een zandige bodem langs de rivier kunnen zowel geïsoleerd van de rivier liggen als in open, stromende verbinding (nevengeulen). In het laatste geval zal de opslibbing minder snel plaatsvinden doordat het stromende water het slib meevoert.
pionierfase
In een mesotroof milieu kan zich een onderwatervegetatie met oeverkruid e.d. vestigen. Op voedselrijkere zandbodems vestigen zich waterranonkels, kranswieren, fonteinkruiden e.d. Door de relatieve voedselarmoede kan deze pionierfase langer aanhouden dan op kleibodems.
t: 0-10 s: 20-30 m: <5
In stromende nevengeulen kan de pionierfase zelfs permanent zijn totdat de geul geïsoleerd raakt van de rivier, maar dan zal zich vaak direct slib gaan afzetten (zie kleiput)
meest voorkomende successie
In situaties waar opslibbing van de zandbodem plaatsvindt: zie ontwikkeling ondiepe kleiput.
Vindt geen opslibbing plaats dan zal de successie over het algemeen een stuk langzamer verlopen door de relatieve voedselarmoede van de zandgrond. Er vindt door afstervende water- en oeverplanten een geleidelijke ophoping van organisch materiaal plaats, die op den duur een verlandingsvegetatie mogelijk maakt die we ook van veenputten kennen (zie daar).

t: 10-50 s: 30-100 m: 5-10
Specifiek voor geïsoleerde plassen met een zandige bodem en uittredend kwelwater is de trilveen-verlanding via een waterplantenvegetatie met o.a. holpijp, waterviolier, diverse zeggen, russen en orchideeën.
t: 50-100 s: 50-150 m: 5-10
'climax'
voor aanzandende zandplassen, zie hierna
voor opslibbende zandplasen: zie bij kleiput
voor organisch verlandende zandplassen: zie bij veenbodem, waarbij de ontwikkeling naar hoogveen vanuit een trilveen via veenmostrilveen gaat.

2.3 Regelmatig overstromend
Ook hier geldt op de meeste plaatsen dat regelmatige overstromingen gepaard gaan met slibafzettingen (zie verder kleibodems). Alleen op zeer dynamische plaatsen, waar tijdens overstromingen de stroomsnelheden te hoog zijn voor slibsedimentatie blijft de bodem zandig. Dit is bijvoorbeeld het geval met de zandige oevers van nevengeulen of andere uitgegraven laagtes, die in open verbinding met de rivier staan of zijn komen te staan na verlaging of verwijdering van zomerkades. Ook kunnen ontgrondingen in open verbinding met de rivier ertoe leiden dat de rivier er vers zand afzet.
pionierfase
Doordat zandig substraat vaak in de stroombaan van de rivier ligt, zijn dit meestal plekken waar relatief veel zaad wordt aangevoerd. Het gaat over het algemeen dan ook om soortenrijke pionierstadia, waaronder kiemplanten van veel verschillende bomen en struiken. Op plekken met regelmatig nieuwe overzandingen kan de pionierfase een aantal malen terugkeren
t: 0-5 s: 150-250 m: < 5
meest voorkomende successie
Afhankelijk van de begrazingsdruk ontwikkelt zich vanuit de pionierfase een overwegend grazige ruigte of een min of meer gesloten ooibos, waarin ten opzichte van kleibodems een groter aandeel aan zwarte populieren te vinden is. Ook treedt eerder een ontwikkeling naar hardhoutsoorten op. De biomassaproductie is wat lager dan op kleibodems. Op plekken met forse aanzandingen kan een successie plaatsvinden in de richting van zelden overstroomde zandgronden (zie daar).
t:5-20 (grasland) of 5-50 (zachthoutooibos) s: 200-300 m: 15-20
'climax'
Er zal op den duur, onder invloed van begrazing een mozaïek van vochtige graslanden, kruidenrijke ruigtes en soortenrijke struwelen en bosjes ontstaan waarbij geleidelijk aan het aandeel hardhoutsoorten zal toenemen. Een groot verschil met de kleiige bodems is dat de zandige standplaatsen veel dynamischer zijn. Erosie en sedimentatie hebben hier een veel grotere rol, al dan niet in combinatie met boom-ontwortelingen, waardoor er voortdurend weer nieuwe pioniersituaties ontstaan en de successie weer van voren af aan begint.
t: 50-200 s: 200-300 m: 15-20

2.4 Zelden overstromend
In combinatie met ontgrondingen komen dergelijke standplaatsen weinig voor. Op een aantal plaatsen in het rivierengebied (o.a. Kekerdom, Ooij, Gendt, Weurt, Leeuwen, Hurwenen) zijn echter diepe zandgaten direct naast de rivier gegraven. Deze zandgaten trekken bij hoogwater de rivier 'naar zich toe'. Op de smalle landstrook tussen rivier en zandgat kunnen dan grote hoeveelheden zand neerslaan, die in de loop der jaren kunnen opwaaien tot jonge rivierduinen, die uiteindelijk tot aan of zelfs boven het hoogwaterniveau kunnen uitgroeien.
Overstromingsgevoelige soorten kunnen op zandgrond langduriger overstromingen verdragen dan op kleibodem, omdat zandbodems na overstroming weer sneller 'ademen'.
pionierfase
Het extreme milieu, periodieke overstromingen in combinatie met stuivend zand en een snel uitdrogende bodem trekt een geheel eigen wereld van pioniers aan, veelal afkomstig uit drogere en warmere streken in Midden- en Zuid-Europa. Door de aanzandingen en verstuivingen kan dit pioniermilieu decennia lang aanhouden
t: 0-30 s: 50-100 m: < 5
meest voorkomende successie

Er is minder zaadaanvoer dan op de frequent overstromende zandgronden, maar soorten die slecht tegen overstromingen bestand zijn, kunnen zich hier wel langer handhaven. Onder invloed van natuurlijke begrazing kunnen zich zeer soortenrijke stroomdalvegetaties ontwikkelen. Van de struiken zijn het vooral vlier en meidoorn die de weg bereiden naar het hardhoutooibos.
stroomdalgrasland
t: 10-50 s: 150-250 m: 5-10

jong hardhoutooibos
t: 10-50 s: 150-250 m: 10-15
'climax'
In dit milieu ontstaat uiteindelijk de soortenrijkste levensgemeenschap van onze klimaatzone: een open hardhoutooibos met een grote afwisseling aan kruidenrijke graslanden, ruigtes, struwelen en een zeer structuurrijk bostype met veel lianen. Periodiek optredende hoogwaters zorgen voortdurend voor nieuwe zaadaanvoer en nieuw kaal substraat waardoor zich lokaal steeds weer opnieuw pioniermilieus zullen voordoen.
t: 50-200 s: 300-400 m: 15-20

2.5 Droog, nooit overstromend
Hoewel er tegenwoordig weinig zand meer op de hogere gronden wordt gewonnen, zijn het vooral deze groeves, die in deze categorie thuishoren.
pionierfase
Ten opzicht van de situatie langs de rivieren is er op de hogere gronden veel minder aanvoer van zaad. De pionierfase bestaat dan ook vooral uit windverspreiders (bijv. berk, grove den, wilgenroosje, viltkruid, kamille- en kruiskruidsoorten) en soorten die zich in de zaadbank ter plekke hebben kunnen handhaven (struikhei, brem, pilzegge e.d.)
t: 0-5 s: 10-20 m: < 5
meest voorkomende successie
Afhankelijk van de begrazingsdruk ontstaan soortenrijke schraalgraslanden en heides die lange tijd open kunnen blijven of juist snel (bij weinig grazers) dichtgroeien met bramen, lijsterbes, jonge berken en dennen. Via eiken-berkenbos kan zich uiteindelijk een eiken-beukenbos ontwikkelen.
heide/schraalgrasland
t: 5-50 s: 50-100 m: 5-10
jong berken-dennenbos
t: 5-50 s: 20-50 m: 5-10
eiken-berkenbos
t: 20-100 s: 20-50 m: 10-15
eiken-beukenbos
t: 50-200 s: 20-50 m: 10-15
'climax'
Een mozaïek van bovengenoemde successiestadia, waarbij storm, bosbrand en ziektes periodiek en lokaal voor terugzetting van de successie zorgen, waarna begrazing weer de remmende factor richting bos is.
t: > 200 s: 50-150 m: 10-15

3. Ontwikkeling vanuit grindig substraat

Ontgrondingen waarvan de bodem na afloop uit grind bestaat komen in Nederland vrijwel alleen voor in het Limburgse Maasdal ten zuiden van de Peelrandbreuk (bij Neer). Het betrof tot dusver vooral diepe winningen, maar als de Grensmaasplannen doorgaan zullen er tussen Maastricht en Roosteren vele honderden hectares ondiepe grindwinningen bijkomen. Voor de vegetatie-ontwikkeling na ontgronding is het van belang of de bodem bestaat uit teruggestort grind (vaak een grove fractie) of uit het moedermateriaal met een oorspronkelijke pakking van verschillende grind- en zandfracties. In de laatste situatie vinden veel meer soorten houvast. Ongesorteerd bodemmateriaal biedt ook een betere uitgangssituatie in situaties met veel rivierdynamiek doordat er nieuwe zand- en grindbanken uit kunnen ontstaan.
3.1 In diep water

Hier geldt min of meer hetzelfde als voor de diepe zandplassen. De plassen die het verst van de rivier liggen zijn over het algemeen het best van kwaliteit vanwege de beperkte opslibbing en het aantrekken van relatief schoon kwelwater (omgekeerd hebben deze plassen vaak ook de grootste gevolgen voor het grondwaterpeil in de omgeving). Voor plassen die dichter langs de rivier liggen is het belangrijk of en hoe ze op die rivier zijn aangetakt. Hoe verder stroomafwaarts de instroomopening en hoe langer de geul ernaartoe, des te geringer de opslibbing.
In de helderste plassen ontwikkelt zich boven ca. 8 meter diepte een vegetatie, maar door de huidige oplevering van de grindplassen, met relatief steile oevers, is deze zone in de meeste plassen zeer smal. De relatief grote plasoppervlakte (grote strijklengte), in combinatie met steile oevers, maakt de oevervegetatie kwetsbaar voor golfslag. Een vlakkere oeverafwerking en onderwaterriffen zouden de natuurlijke potenties van grindplassen sterk vergroten (zie ook RIZA/Stroming-rapport, 1992, waterplanten in de Maasplassen). Een centrale 'zinkput' blijkt ook in grindplassen bij te dragen aan slibvrije oeverzones waarop zich typische grindvegetaties kunnen vestigen.
pionierfase
kranswieren, fonteinkruiden (schedefonteinkruid !) en andere waterplanten zoals aarvederkruid. Bestaat het bodemmateriaal uit teruggestort grof grind zonder pakking dan vestigen zich vaak alleen maar zoetwatersponsen en nauwelijks vaatplanten.
Zolang er grind gewonnen wordt in de plas is deze vaak nog zo troebel dat de omstandigheden nog weinig geschikt zijn, m.u.v. de uithoeken van de plas
t: vanaf 0-5 s: 10-20 m: <5
meest voorkomende successie
Er treedt nauwelijks successie op, al kunnen er jaarlijks soorten verdwijnen of bijkomen, bijv. door zaadaanvoer via watervogels. Grote groepen plantenetende vogels zoals meerkoeten kunnen periodiek een enorme aanslag op de waterplanten plegen
'climax'
Wanneer de structuur en soortenrijkdom van de watervegetatie hun optimum bereiken
t: vanaf 10-50 s: 25-40 m: < 5

3.2 In ondiep water
Ondiepe wateren met een grindbodem, die binnen het overstromingsbereik van de Maas liggen, raken meestal snel toegedekt met zandig of slibrijk sediment. Uitzonderingen zijn de oevers van een diepe grindplas (slib bezinkt in de diepste delen) of snelstromende grindgeulen, zoals de uitgediepte Grensmaas zelf of nevengeulen van deze rivier (o.a. bij Meers)
pionierfase
Stromende grindgeulen verkeren permanent in de pionierfase (met enkele soorten waterplanten waaronder de vlottende waterranonkel als meest bekende), totdat de rivier of beek zich verlegt waarna er sedimentatie van zand en slib zal optreden met een geheel eigen successie (zie daar).
In de ondiepe oeverzone van grote grindplassen bestaat de pionierfase uit kranswieren, fonteinkruiden en andere waterplanten.
t: vanaf 0 s: 10-20 m: < 5
meest voorkomende successie
In oeverzones met veel golfslag kan de pionierfase eeuwig voortduren. Op meer luwe plaatsen kunnen er vanaf het land moerasplanten steeds verder het water ingroeien. Wortels, afstervend plantenmateriaal en slib dat tussen de wortels wordt ingevangen vormen een nieuwe onderwaterbodem bovenop het grind.
t: 5-20 s: 10-50 m: 5-10
'climax'
Een oeverbos met elzen, wilgen en een ondergroei van tientallen soorten moerasplanten. Plaatselijk opengeknaagd door bevers en open gehouden door de activiteit van grote grazers.
t: 20-100 s: 50-100 m: 10-15

3.3 Regelmatig overstromend
In de huidige situatie zijn dit vooral grindbanken in de uitgediepte Grensmaas en op plekken waar deze rivier momenteel verbreed wordt (Meers).
pionierfase
Soortenrijke vegetatie met veel adventieven uit het bovenstroomse Maasdal. Veel opslag ook van jonge wilgen en ook de eerste zwarte populieren. Op de meest dynamische plekken, vlak langs de

rivier, krijgt de vegetatie geen kans om verder uit te groeien en kan de pionierfase zich decennia lang handhaven.
t: vanaf 0 s: 100-200 m: < 5
meest voorkomende successie
Wordt de vegetatie niet telkens weggespoeld dan kunnen zich vanuit de pioniervegetatie zowel dichte ruigtes ontwikkelen als jonge zachthoutooibossen. Onder invloed van begrazing kunnen dergelijke ruigtes en bossen meer of minder open en grazig zijn. Vanaf het moment dat de vegetatie zich weet te handhaven zal ze bij iedere overstroming ook sediment invangen, waardoor er naast grind een steeds groter deel van de bodem met zand en zelfs slib zal worden afgedekt.
t: 5-50 s: 100-200 m: 5-15
'climax'
Een ruig type ooibos waarin zo nu en dan gaten vallen onder invloed van extreme hoogwaters, ijsgang en/of bevervraat. Gaten die langdurig opengehouden kunnen worden door andere grazers. Het aandeel zachthoutooibos zal geleidelijk aan afnemen ten gunste van hardhoutsoorten, zeker wanneer de vegetatie grote hoeveelheden zandig sediment gaat invangen (zie daar).
t: 50-200 s: 200-300 m: 10-15

3.4 Zelden overstromend
Dit zijn vooral de hogere oevers van grindwinningen in de periferie van het Maasdal. Het grove grindige substraat laat na overstroming sneller lucht door dan een zandige bodem, waardoor we op grindbodems met een vergelijkbaar overstromingsregime vaak meer overstromingsgevoelige soorten tegenkomen. Daar staat tegenover dat voedingsstoffen sneller wegspoelen. Ten opzichte van zand en zeker ten opzichte van klei betekent dit dat meststoffen uit de landbouw sneller verdwijnen, maar ook dat er voor veel plantensoorten te weinig voedsel overblijft om te overleven. Het netto resultaat lijkt te zijn dat er op puur grindig substraat toch wat minder soorten voorkomen dan op zand.
pionierfase
Een periodiek overstromende kale grindvlakte direct na ontgronding is aanvankelijk een nog extremer milieu dan een kale zandvlakte. Door het achterwege blijven van verstuivingen en het geleidelijk neerslaan van slib- en humusdeeltjes in de grindholtes vestigen zich toch gestaag een permanente vegetatie en duurt de pionierfase minder lang dan in zandige milieus. De ligging langs de Maas, met een minder groot en soortenrijk achterland dan bijv. de Rijn
t: 0-10 s: 25-75 m: < 5
meest voorkomende successie
Er is minder zaadaanvoer dan op de frequent overstromende grindbodems, maar soorten die slecht tegen overstromingen bestand zijn, kunnen zich hier wel langer handhaven. Onder invloed van natuurlijke begrazing kunnen zich zeer soortenrijke stroomdalvegetaties ontwikkelen. Van de struiken zijn het vooral vlier en meidoorn die de weg bereiden naar het hardhoutooibos.
stroomdalgrasland
t: 10-50 s: 150-250 m: 5-10
jong hardhoutooibos
t: 10-50 s: 150-250 m: 10-15
'climax'
Een open hardhoutooibos met een grote afwisseling aan kruidenrijke graslanden, ruigtes, struwelen en een zeer structuurrijk bostype met veel lianen. Periodiek optredende hoogwaters zorgen voortdurend voor nieuwe zaadaanvoer en nieuw kaal substraat waardoor zich lokaal steeds weer opnieuw pioniermilieus zullen voordoen.
t: 50-200 s: 300-400 m: 15-20

3.5 Droog, nooit overstromend
Hier zijn maar enkele voorbeelden van bekend aan de Vlaamse zijde van de Grensmaas, waar in de jaren vijftig enkele kleinere oppervlakkige grindgroeves zijn achtergelaten. Verder komt dit milieu op zeer kleine schaal voor bij erosiegeulen in kalksteengroeves.
pionierfase
Doordat er, in tegenstelling tot het overstromingsgebied, geen aanvoer van fijner sediment plaatsvindt, kan de pionierfase vele decennia duren, omdat maar weinig planten houvast vinden op het grove grind. De vegetatie raakt bovendien snel los getrapt waarna het verhaal weer van voren af

aan begint. Tot de vegetatie behoren wel zeldzame planten van rots- en rolsteenhellingen in Midden-Europa zoals smalle raai, diverse vetkruiden en leeuwebeksoorten.
t: 0-30 s: 10-20 m: <1
meest voorkomende successie
Heeft zich eenmaal een ijl plantendek gevestigd en/of blijven er in de herfst bladeren op de grindvlakte liggen, dan kan er op deze dunnen organische laag een verdere vegetatieontwikkeling plaatsvinden. Dit is meestal een ijle grazige vegetatie met zeer verspreide opslag van doornige struiken, die als enige kunnen standhouden tegen begrazing hoe extensief die meestal ook is. Door de open structuur van het grasland kunnen pioniers zich nog relatief lang hierin handhaven.
t: 20-100 s: 20-100 m: 5-10
'climax'
Zeer open landschap met kruidenrijke graslanden en hardhoutsoorten in doornstruwelen.
t: 100-300 s: 100-300 m: 5-10

4. Ontwikkeling vanuit kalkbodems
Ontgrondingen met na oplevering een kalkbodem komen in Nederland alleen in Zuid-Limburg voor (Pietersberg, Julianagroeve, Groeve 't Rooth, Curfsgroeve).

4.1 In diep water
Is mogelijk het toekomstperspectief voor de ENCI-groeve in de Pietersberg, wanneer deze tot 40 meter onder Maaspeil wordt uitgegraven en er na de afgraving gestopt wordt met pompen..
De verwachting is niet dat zich in een diepe kalkplas snel plantenleven zal kunnen ontwikkelen, omdat zwevende kalkdeeltjes het water lange tijd zeer troebel zullen maken. Vanuit de randen kunnen mogelijk wel water- en moerasplanten de plas ingroeien. Zie verder : in ondiep water

4.2 In ondiep water
Onderin kalksteengroeves kunnen zich ondiepe plassen vormen, die zowel gevoed kunnen worden door regenwater als door grondwater dat uit de wanden van de groeve sijpelt. Met het water kan ook veel leem van de hellingen spoelen, waardoor de plas meer een lemige bodem krijgt dan een kalkbodem. In dat laatste geval: zie ontwikkeling ondiepe kleiput (buiten overstromingsbereik).
pionierfase
In en rond kalkplassen kan zich snel een vrij productieve en (afhankelijk van zaadbronnen in de omgeving) soortenrijke pioniervegetatie ontwikkelen met o.a. lisdodde, waterweegbree, wolfspoot en verschillende soorten russen.
t: 0-5 s: 50-100 m: 5-10
meest voorkomende successie
Het afstervend plantenmateriaal dat onder water terecht komt vormt de basis voor de vorming van kalkmoerassen met een soortenrijke zeggevegetatie, tientallen andere kruiden en struikwilgen en elzen langs de randen
t: 5-50 s: 100-150 m: 5-10
'climax'
Een kalkmoerasbos dat zich geleidelijk aan boven het waterniveau verheft en dat zich, onder invloed van begrazing ontwikkelt tot een mozaïek van wilgen- en elzenstruwelen temidden van een kruidenrijk zeggemoeras
t: 30-100 s: 100-150 m: 10-15

4.3 Droog, nooit overstromend
Dit is het overgrote deel van de mergelgroeves na stopzetting van de winning. Spectaculair zijn de loodrechte kalkwanden die, afhankelijk van hun ligging op het zuiden of noorden bloedheet of vochtig en bemost kunnen zijn met ieder hun geheel eigen flora en fauna, waaronder rotsbewoners, muurplanten en lianen. Helaas zijn in het verleden, om veiligheidsredenen veel van deze steile kliffen toegedekt en afgevlakt met ander bodemmateriaal. In de toekomst zal dit hopelijk achterwege blijven (zie rapport 'Verborgen Valleien' van Stichting Ark, 1999, dat inmiddels min of meer als richtlijn voor het inrichten van mergelgroeves geldt). Er kan overigens ook een 'natuurlijke' afdekking met zandig en lemig materiaal optreden, wanneer tijdens hevige regenval solifluctie optreedt van hoger gelegen zand- en leemlagen, die over de kalkhellingen afschuiven.

De geïsoleerde ligging van de meeste kalksteengroeves bemoeilijkt een snelle kolonisatie van veel kalkminnende planten. Rondtrekkende grazers kunnen dit kolonisatieproces bespoedigen.
pionierfase
Soortenrijke, open kruidenvegetatie van kalkminners
t: 0-5 s: 100-150 m: 5-10
meest voorkomende successie
Zoals hier boven al aangegeven ontwikkelt zich op de steilwanden een geheel eigen milieu, voor een deel ook omdat de grazers hier niet kunnen komen. Dat laatste is wel het geval in de vlakkere delen van de groeve en afhankelijk van de begrazingsdruk ontwikkelt zich hier een grazige vegetatie van soortenrijke kalkgraslanden of een ruigere vegetatie met opslag van berken, boswilgen, acacia's, vlinderstruiken, doornstruiken en een ontwikkeling naar soortenrijk eiken-haagbeukenbos. Kenmerkend voor deze struwelen en bossen zijn een rijke voorjaarsflora en de uitbundige sluiers van bosrank.
kalkgrasland
t: 5-50 s: 150-250 m: 5-10

jong eiken-haagbeukenbos
t: 5-50 s: 150-200 m: 10-20
'climax'
Mozaïek van kalkgraslanden, doornstruwelen en eiken-haagbeukenbos in alle stadia van opbouw en afbraak Door solifluctie lokaal en periodiek terugkeer naar pionierfase. Naarmate het bosklimaat zich verder ontwikkelt vestigen zich steeds meer zeldzame bosplanten zoals eenbes, heelkruid en christoffelkruid.
t: 50-200 s: 250-400 m: 10-20

5. Ontwikkeling vanuit venig substraat

5.2 In ondiep water

Deze situatie komt niet veel voor en is beperkt tot die plekken waar ondiep turf wordt gewonnen (in NL bijna nergens meer), voormalige petgaten weer worden uitgediept of waar klei wordt afgegraven tot op de venige ondergrond
pionierfase
verschillende soorten fonteinkruiden en oeverpioniers
t: 0-10 s: 5-20 m: <5
meest voorkomende successie
verlanding door ophoping van organisch materiaal in het water (fonteinkruiden, nymphaeïden, pijlkruid en later krabbescheer) en door een rijke oevervegetatie (lis, harig wilgeroosje, kleine lisdodde, riet, mattenbeis en watereppes) die het water ingroeit. Laagveenvorming o.a. via vorming van soortenrijke drijftillen. Afhankelijk van de diepte en de kwaliteit van het water en de omvang van de plas komt de organische laag binnen één of enkele decennia boven de waterspiegel uit (in kleine, ondiepe, heldere plassen het eerst). Vervolgens kan de successie via vestiging van grauwe wilg en els verder gaan in de richting van een broekbos, maar dit is ook het moment waarop rietsnijders de vegetatie naar hun hand kunnen zetten (door te maaien bevorderen ze riet ten opzichte van bosontwikkeling). Met uitzondering van reeën en elanden, maar die laatste ontbreken (nog) in Nederland, is dit milieu weinig geschikt voor grote grazers vanwege de slappe bodem.
laagveen
t: 10-50 s: 100-150 m: 5-15
broekbos
t: 10-50 s: 50-100 m: 5-15
'climax'
In grondwater beïnvloede situaties is dit een broekbos waarbij bomen, vanwege de slappe bodem, snel ontworteld raken en weer gaten in het veen trekken, waardoor zich hier weer water- en oeverplanten vestigen.
Groeit de organische laag boven de grondwater-invloed uit en raakt ze meer onder invloed komt van het zure regenwater, dan gaat de successie verder richting zure berken- en gagelbossen of (in open terrein) via veenmosrietland met veel (veen)mossen, varens en zegges, naar hoogveen.
t: 50-200 s: 20-50 m: 5-10

5.3 Regelmatig overstromend

Op veraarde veengrond die regelmatig overstroomt zal een voortdurende mineralisatie en nauwelijks ophoping van organisch materiaal plaatsvinden. Liggen dergelijke veenafgravingen binnen de invloedsfeer van rivieren dan kan er slib op het veen worden afgezet (zie daar)

pionierfase
Zaailingen van tal van moerasplanten, waaronder ook kiemplanten van veel bomen en struiken
t: 0-1 s: 25-75 m: <5
meest voorkomende successie
Rietruigtes en andere ruigtes van moerasplanten. Afhankelijk van de begrazingsdruk zal het aandeel grasland (vgl. veenweide) of juist het aandeel zachthoutooibos een groter aandeel in de vegetatie nemen.
t: 1-50 s: 50-100 m: 10-20
'climax'
Een mozaïek van ooibos, (riet)ruigtes en natte graslanden
t: 50-100 s: 100-150 m: 10-15

5.4 Zelden overstromend

De ontwikkeling van veengrond, die zo nu en dan overstroomt vertoont grote overeenkomsten met veengrond die nooit overstroomt, waarbij de drooglegging van de grond (zie hierna), maar ook de kwaliteit van het inundatiewater voor een belangrijk deel bepaalt in welke richting de successie gaat. Is het gebiedseigen water dat door hevige regenval tot boven maaiveld wordt opgestuwd dan is een successie richting hoogveen nog steeds mogelijk. Gaat het om relatief voedselrijk beek- of rivierwater, waarmee de veengrond overstroomd raakt, dan blokkeert de hoogveensuccessie en zal zich, zeker in een overigens drooggelegde situatie een ontwikkeling naar broekbos voordoen, zoals die hieronder voor veraard veen is beschreven.

5.5 Niet overstromend

pionierfase
Komt er na ontgronding een goed ontwaterde veenbodem aan de oppervlakte te liggen dan zal het veen oxideren,'veraarden' en op den duur vergaan. De bodem klinkt in met één tot enkele cm/jaar, afhankelijk van de drooglegging. Blijft de veenbodem na ontgronding nat dan kunnen zich hier direct hoogveen-pioniers zoals zonnedauw en bepaalde veenmossoorten vestigen. De soortenrijkdom wordt in beide gevallen voor een belangrijk deel bepaald door wat er nog in de zaadbak aanwezig is en wat er aan windverspreiders in de buurt staat. Vooral op veraard veen zullen zich ook snel zaailingen van berk en els vestigen.
veraard veen
t: 0-1 s: 10-50 m: <5
nat veen
t: 0-1 s: 10-25 m: < 1
meest voorkomende successie
Op veraard veen zal zich afhankelijk van de begrazingsdruk een grasland (vgl. veenweide) ontwikkelen of een ruigte met riet(gras), wilgenroosjes, bramen en jonge bomen.
t: 1-50 s: 25-75 m: 5-15
Op natte veenbodems kan de successie naar hoogveen verder gaan, waarbij zich een dunne veenmoslaag vormt met een eigen, door regen gevoede waterhuishouding. Bomen krijgen het hier steeds moeilijker in.
t: 1-50 s: 10-25 m: 5-10
'climax'
Op veraard veen ontwikkelt zich een begraasd mozaïek van veenweiden en broekbossen
t: 50-100 s: 50-100 m: 10-15

Een volwassen hoogveen ziet er doorgaans uit als een vrijwel boomloze vlakte met een microreliëf van bulten (dophei, pijpenstrootje, veenbes, lavendelheide) en slenken (snavelbies, wollegras e.d.)
t: 50-200 s: 25-50 m: 5-10

Annex 5 Analysing the impact of aggregate extractions with the Floron database (in Dutch)

Effecten van oppervlakte-delfstoffenwinning op de flora; analyse van de landelijke floradatabank FlorBase

2001

Rapport 2001.18

Stichting FLORON

Titel: *Effecten van oppervlakte-delfstoffenwinning op de flora; analyse van de landelijke floradatabank FlorBase*

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Inhoud

5.1. Inleiding

Door TNO-Industrie wordt onderzoek uitgevoerd naar de milieu-effecten van de productie van goederen. Daarbij wordt de gehele levenscyclus, van het vergaren van de grondstoffen tot het verwerken van het uiteindelijke afval, in beschouwing genomen. Een dergelijke levenscyclusanalyse (LCA) maakt het mogelijk verschillende producten te vergelijken op hun totale milieu-effecten.
In een van de deelprojecten wordt onderzocht in hoeverre het mogelijk is om de natuureffecten van het winnen van delfstoffen in de LCA te betrekken. Daarbij wordt gefocust op de effecten van delfstoffenwinning in Nederland. Het gaat daarbij om de oppervlaktewinning van delfstoffen als kalksteen, klei, grind en zand. Het onderzoek naar de natuureffecten spitst zich in eerste instantie toe op de effecten op de flora.
Er heeft in Nederland geen monitoring plaatsgevonden van de natuureffecten van delfstoffenwinning. Het opzetten van een dergelijke monitoring is in essentie eenvoudig, maar het zou een flink aantal jaren duren voordat er bruikbare resultaten uit zouden komen. Omdat er op korte termijn behoefte is aan inzicht in de orde van grootte van de omvang van de effecten, heeft TNO-Industrie de Stichting Floristisch Onderzoek Nederland (FLORON) gevraagd een analyse uit te voeren op basis van de floragegevens in de landelijke floradatabank FlorBase, die door FLORON en het Nationaal Herbarium Nederland (NHN) wordt beheerd. Een analyse van veranderingen in de tijd op de locaties van de delfstoffenwinningen is niet mogelijk, omdat tijdreeksen ontbreken. Er is daarom verzocht om een ruimtelijke analyse, waarin locaties van delfstoffenwinning worden vergeleken met referentielocaties. FLORON heeft zich bereid verklaard de gewenste analyse uit te voeren. Daarbij is de opdrachtgever er nadrukkelijk op gewezen dat de ruimtelijke schaal waarop informatie in FlorBase is opgenomen, eigenlijk te grof is voor een analyse van de effecten van delfstoffenwinning, i.c. ondiepe of diepe ontgronding. In de Discussie komen wij hierop uitgebreid terug. Voor de opdrachtgever zijn het feit dat de analyse op korte termijn kan geschieden, en het feit dat betere basisgegevens op dit moment nauwelijks voorhanden zijn, redenen geweest om de analyse toch te wensen.

5.2. Methode

5.2.1. Analysemethode

Ontgrondingen vinden plaats in een bepaalde periode. De voorkomende flora op de locatie in die periode wordt vergeleken met de flora in min of meer dezelfde periode op twee referentielocaties[22]. Vanwege de aard van de floradatabank hebben alle locaties de vorm en de grootte van een kilometerhok, zoals die op de Topografische Kaart van Nederland worden weergegeven. De flora wordt vergeleken op een aantal somparameters, waarmee verschillende aspecten van de botanische natuurwaarde eenvoudig tot uitdrukking kunnen worden gebracht:

- Totaal aantal voorkomende soorten.
- Aantal aandachtsoorten: internationaal of nationaal beschermde soorten, Rode-Lijstsoorten of Doelsoorten voor het Nederlands natuurbeleid.
- Aantal Rode-Lijstsoorten.
- Aantal Doelsoorten.
- Aantal zeldzame soorten.
- Botanische natuurwaarde van voorkomende ecotoopgroepen.
- Botanische natuurwaarde van grondwatergebonden, relatief waardevolle ecotoopgroepen.

Per type ontgronding worden voor elke ontgrondingslocatie twee referentiehokken geselecteerd, die aan bepaalde voorwaarden moeten voldoen om vergelijkbaar te worden geacht. De gemiddelde waarden van de bovengenoemde parameters kunnen per type ongronding worden bepaald. Ook kan er een statistische analyse plaatsvinden van die waarden, waarbij wordt uitgegaan van de hypothese dat er geen onderscheid is tussen de waarden van de ontgrondingslocaties en de referentielocaties.

5.2.2. Gebruikte bestanden

Locaties van ontgrondingen
TNO-Industrie heeft een Excelbestand aangeleverd, waarin de locatie van 362 ontgrondingen is vermeld in de vorm van de hectometercoördinaten van het middelpunt van de winning. Voor enkele typen winning waren ook de referentiehokken al aangegeven, maar voor de meeste locaties nog niet. Het kmhok, waarin de hectometercoördinaten liggen, is gekozen als de ontgrondingslocatie. In de gevallen dat er sprake was van meer dan één winningslocatie binnen een kmhok, is het kmhok slechts eenmaal geselecteerd. Van elke ontgrondingslocatie is ook het type delfstoffenwinning aangegeven (tabel 1).

Tabel 1. Type delfstoffenwinning.

type	omschrijving
1	Grindwinning
2	Winning Beton- en metselzand actief
3	Winning Beton- en metselzand in ontwikkeling
4	Kleiwinning actief vanaf 1995
5	Kleiwinning actief vanaf 1999
6	Kleiwinning oud
7	Kalksteenwinning
8	Ophoogzand tijdens winning
9	Ophoogzand na winning

Ecodistricten
Voor het selecteren van referentiehokken is het van belang kmhokken te kiezen die in zo veel mogelijk ecologisch relevante eigenschappen vergelijkbaar zijn met de ontgrondingslocatie. De indeling van

[22] Voor de referentielocaties begint de referentieperiode 2 jaar eerder en eindigt 2 jaar later; dit is gedaan om voldoende referentielocaties ter beschikking te hebben.

Klijn (1997) van Nederland in 38 ecodistricten is daarvoor een bruikbare indeling. FLORON beschikt over een bestand waarin elk kmhok is toegedeeld aan één ecodistrict[23] en dat is gebruikt.

Florbase
De landelijke floradatabank FlorBase bevat waarnemingen van alle in het wild voorkomende plantensoorten op het niveau van kilometerhokken. De waarnemingen in versie 2G zijn afkomstig uit de periode 1975-2000. De gegevens zijn verzameld door vrijwilligers, provincies, terreinbeherende organisaties en onderzoeksinstituten, waarbij verschillende inventarisatietechnieken zijn gehanteerd. Bij de analyse van FlorBase is alleen gebruik gemaakt van de presentie van plantensoorten, niet van hun abundantie, aangezien die in veel gevallen onbekend is.

5.2.3. Geschiktheid van de opgegeven locaties voor analyse

Na het verwijderen van de dubbele records zijn 357 ontgrondingslocaties beschikbaar voor analyse. Voor elke ontgronding is aangegeven uit welke periode er flora-gegevens gewenst zijn. Alleen kilometerhokken waarvan binnen de genoemde periode voldoende floragegevens beschikbaar zijn, zijn dus geschikt. Om te bepalen hoeveel soorten er binnen een kilometerhok ten minste bekend moeten zijn is als benedengrens gehanteerd het gemiddelde aantal soorten binnen het ecodistrict, min de standaarddeviatie (tabel 2).

Van de 357 ontgrondingslocaties hebben er 5 een wateroppervlakte van 50 hectare of meer. Omdat het soortenaantal hierdoor negatief kan worden beïnvloed zijn deze kilometerhokken apart bekeken. Twee kilometerhokken met 75 hectare water en één hok met 50 hectare water konden wel in de berekening worden meegenomen; het aantal bekende soorten lag hier vlak onder de drempelwaarde. Twee andere locaties vielen af (80 hectare water, 12 soorten en 50 hectare water, 10 soorten) vanwege hun onvolledige inventarisatie.

Op basis van het bovenstaande zijn 85 van de 357 kmhokken bruikbaar voor analyse. Er zijn *geen ontgrondingen* die afvallen voor analyse vanwege het ontbreken van goede referentiehokken.

5.2.4. Keuze van de referentiehokken

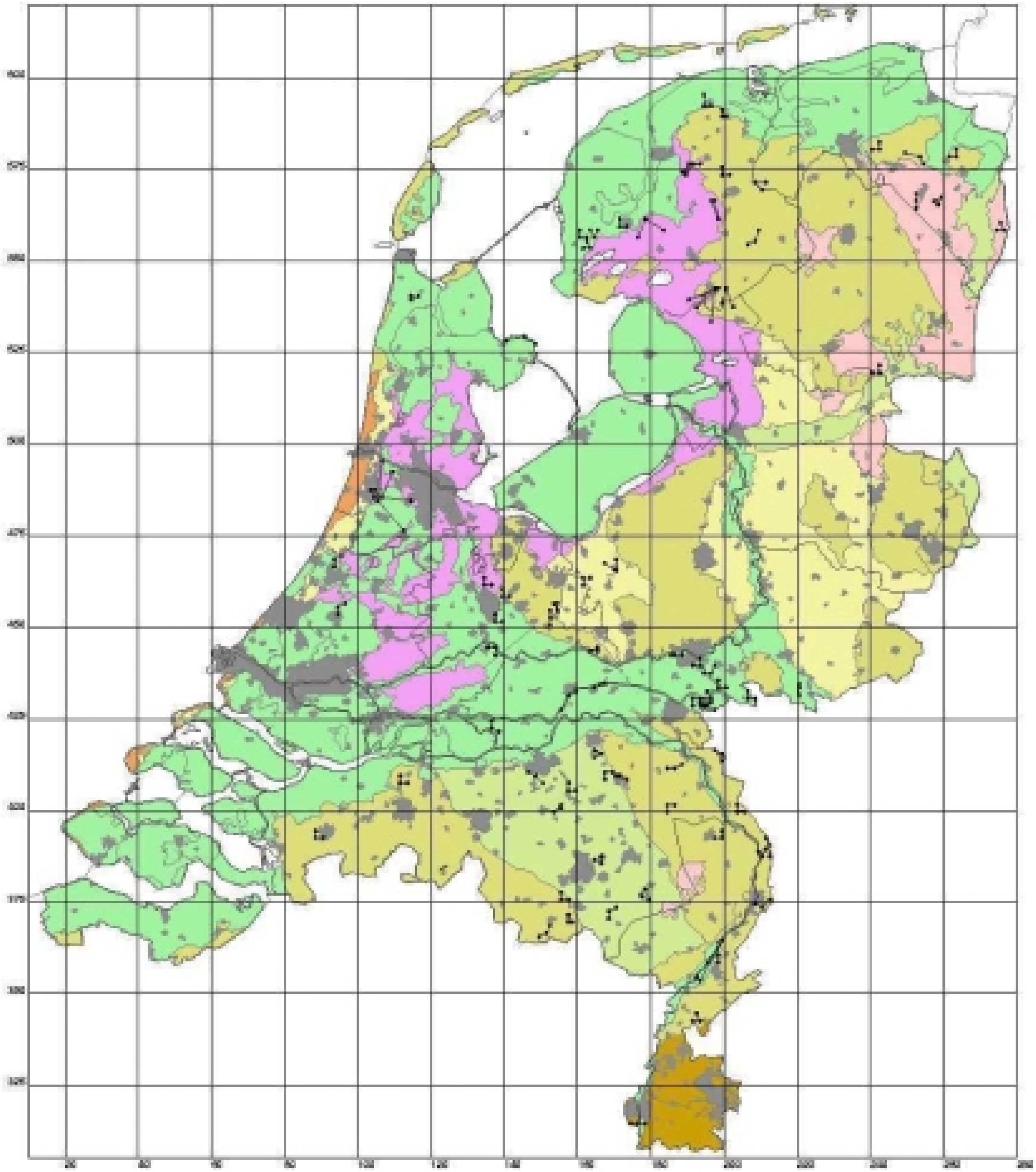
Bij elk kilometerhok met een ontgronding zijn twee referentiehokken gezocht. Voor een deel van de hokken met een ontgronding zijn al referentiehokken voorgesteld door TNO-Industrie. Die referentiehokken zijn gekozen op basis van het kaartbeeld op de topografische kaart. De voorgestelde referentiehokken zijn echter alleen geschikt als er uit de gewenste periode voldoende floragegevens beschikbaar zijn, en als ze in het goede ecodistrict zijn gelegen. Dat is bepaald met een klein in Basic geschreven programma, dat op basis van ecodistricttype en aantal aangetroffen plantensoorten in de goede periode de twee dichtstbijzijnde hokken bepaalt, die aan de criteria voldoen. Ook hier is als minimum het gemiddelde aantal soorten binnen het ecodistrict min de standaarddeviatie genomen. Op grond van deze criteria zijn 12 van de voorgestelde referentiehokken geschikt.
In de overige gevallen zijn referentiehokken gezocht, die op een zo kort mogelijke afstand liggen, in hetzelfde ecodistrict (figuur 1). De referentiehokken mogen echter niet direct grenzen aan het kmhok waarin de ontgronding is gelegen, omdat de ontgronding zich over de hokgrens kan uitstrekken.

Tabel 2. Ecodistricten (ed, edtyp, omschrijving) en het gemiddeld aantal aangetroffen soorten per kilometerhok (gemid) en de standaarddeviatie (stdev).

ed	gemid	stdev	edtyp	omschrijving
1	188	80	L1	Krijtlandschap
2	153	69	L2	Lössgebied
3	177	83	P1	Midden-Nederlands stuwwalcomplex
4	194	75	P2	Geïsoleerde stuwwal
5	175	55	P3	Geïsoleerd keileemplateau

[23] Langs de randen van ecodistricten zijn de kmhokken toegedeeld aan het ecodistrict met de grootste oppervlakte binnen het hok.

6	177	68	P4	Pleistocene opduiking
7	191	72	P5	Overig keileemgebied
8	171	59	P6	Horst
9	162	71	P7	Oud rivierterrassenlandschap
10	176	63	P8	Zuidwest-Nederlands rivierzandgebied
11	196	68	P9	Oost-Nederlands dekzandgebied
12	170	77	P10	Glaciaal bekken
13	197	64	P11	Puinwaaierlandschap
14	156	43	P12	Hoogveen(ontginnings)landschap
15	187	69	P13	Beekdalcomplex
16	201	74	P14	Centrale-slenkgebied
17	306	75	D1	Kalkrijke duinen
18	201	90	D2	Kalkarme duinen
19	289	79	H1	Strandwallengebied
20	189	80	H2	Rivierengebied
21	148	50	H3	Jonge indijking
22	145	51	H4	Zeeklei-inversielandschap
23	190	63	H5	Laagveengebied
24	152	56	H6	Droogmakerij
25	127	57	H7	Polder
26	158	53	H8	Deltagebied
27	137	52	W1	Sedimentatiebekken
28	131	50	W2	Grote verzoete binnenzee
29	189	71	W3	Randmeer
30	135	55	W4	Verzoet estuarium
31	127	56	W5	Verzoete zeearm
32	136	39	W6	Brak meer
33	103	37	Z1	Estuarium
34	128	71	Z2	Zout meer
35	111	40	Z3	Zeearm
36	143	67	Z4	Kustzone Noordzee
37	128	55	Z5	Waddenzee
38	206	82	S1	Stedelijk gebied



Figuur 1. Locatie van de kmhokken met ontgrondingen, waarvoor floragegevens beschikbaar zijn; ook de referentiehokken zijn aangegeven.

5.2.5. Waarderingsparameters

De verschillende waarderingsparameters zijn niet onderling onafhankelijk, omdat de combinaties van soorten op basis waarvan een waarde wordt berekend, elkaar gedeeltelijk overlappen. Er zijn binnen de in paragraaf 2.1 genoemde parameters drie groepen te onderscheiden.

- Het aantal voorkomende soorten is de beste benadering van de biodiversiteitsmaat, die in internationaal onderzoek vaak wordt gebruikt om de waarde van een gebied aan te geven. In Nederland wordt het soortenaantal, zonder weging, niet gezien als een belangrijke of goede biodiversiteitsparameter, omdat hoge soortenaantallen in veel landschappen eerder op verstoring en menselijke activiteiten wijzen dan op botanische waarde.
- Het aantal voorkomende aandachtsoorten, Rode-Lijstsoorten, Doelsoorten of zeldzame soorten benadrukt het voorkomen van soorten waar het Nederlandse natuurbeleid extra waarde aan hecht. Aandachtsoorten zijn het meest omvattend, Rode-Lijstsoorten en Doelsoorten het meest

overeenkomend (Van der Meijden et al., 2000) en zeldzame soorten omvatten ook relatief veel nieuwkomers in de Nederlandse flora die nog bezig zijn hun areaal uit te breiden.

- De derde groep betreft de gesommeerde botanische natuurwaarde van ecotoopgroepen. Deze parameter hecht geen belang aan de zeldzaamheid of bedreigdheid van afzonderlijke soorten, maar aan de volledigheid van voorkomende clusters van soorten die worden aangetroffen bij bepaalde combinaties van biotische en abiotische standplaatsfactoren (vegetatiestructuur, saliniteit, vochttoestand, voedselrijkdom en zuurgraad). Naast de volledigheid van elke ecotoopgroep is ook de relatieve zeldzaamheid van belang.

Aantal soorten per kilometerhok
Het aantal soorten dat binnen een kilometerhok kan worden aangetroffen is van veel variabelen afhankelijk, waarbij bodem, hydrologie, morfologie, grondgebruik, mate van isolatie, en gebeurtenissen in het verleden vaak een belangrijke rol spelen. In het algemeen geldt: hoe meer variatie in de genoemde factoren, des te meer soorten. Het overzicht van het gemiddelde soortenaantal per ecodistrict in tabel 2 geeft weer dat de variatie in deze factoren in Nederland tot aanzienlijke verschillen in soortenaantal leidt.

Aantal Rode Lijst-soorten
Voor het bepalen van het aantal Rode Lijst soorten is uitgegaan van de lijst van ‘Bedreigde en kwetsbare vaatplanten in Nederland’ (Van der Meijden et al., 2000) die het basisrapport vormt voor een officiële Rode Lijst Vaatplanten. 499 van de bijna 1500 in Nederland in het wild voorkomende plantensoorten zijn op die lijst vermeld.

Aantal Doelsoorten
Ook de lijst met doelsoorten is in bovengenoemd rapport vermeld. De lijst is gelijk aan de Rode Lijst, aangevuld met soorten die in Nederland zeldzaam zijn of een neergaande trend vertonen, én waarvan Nederland centraal in het Europese areaal is gelegen. De lijst bevat 544 soorten.

Aantal Aandachtsoorten
Naast de lijsten van Doelsoorten en Rode-Lijstsoorten zijn in Nederland de wettelijk beschermde soorten en de internationaal[24] beschermde soorten van belang voor het natuurbeleid. Tezamen vormen deze lijsten de lijst van Aandachtsoorten, waarbij ook nog de soorten zijn gevoegd die op de eerdere Rode Lijst (Weeda et al., 1990) waren vermeld, maar niet meer op de nieuwe. In totaal zijn er 660 aandachtsoorten.

Aantal zeldzame soorten
Meegeteld zijn alle soorten met een ‘uurhokfrequentie’ kleiner dan, dat wil zeggen soorten waarvan het aantal uurhokken in Nederland in 1990 lager werd geschat dan 190 van de 1670 6 (Mennema et al., 1980). De lijst met zeldzame soorten komt voor een aanzienlijk deel overeen met de Rode. De lijst omvat echter ook zeldzame soorten die zich nog aan het uitbreiden zijn, en zeldzame soorten die stabiel zijn in hun voorkomen.

Gesommeerde botanische natuurwaarde
De gesommeerde natuurwaarde van de voorkomende ecotoopgroepen is gebaseerd op het voorkomen en de mate van ontwikkeling van afzonderlijke groepen van soorten die bij bepaalde abiotische milieucondities kunnen worden aangetroffen. Rekening houdend met hun relatieve zeldzaamheid kunnen de waarden van de afzonderlijke ecotoopgroepen per kmhok worden gesommeerd (Witte en Van der Meijden, 1995; Witte, 1998). De waarderingsmethode is speciaal ontwikkeld om floristische gegevens in de floradatabank te analyseren en te hanteren in ecologische voorspellingsmodellen. Elke voorkomende plantensoort heeft een zekere indicatiewaarde voor de aanwezigheid van een of meer ecotoopgroepen, en via drempelwaarden wordt afgeleid of een bepaalde ecotoopgroep daadwerkelijk voorkomt. Er worden 39 ecotoopgroepen onderscheiden.

[24] Het betreft soorten die in de bijlagen 2, 4 of 5 van de Habitatrichtlijn worden genoemd, en de soorten die beschermd zijn volgens de Conventie van Bern.

Gesommeerde botanische natuurwaarde grondwaterafhankelijke ecotoopgroepen
Deze parameter is gelijk aan de bovenstaande, maar wordt op basis van 28 ecotoopgroepen berekend. Het betreft alleen de aquatische en natte ecotoopgroepen, en de voedselarme vochtige ecotoopgroepen die vaak op grondwaterafhankelijke bodem voorkomen.

5.3. Resultaten per ontgrondingstype

De gesommeerde resultaten worden weergegeven per type ontgronding. Linksboven in de tabellen staat het aantal bruikbare hokken van een type ontgronding aangegeven; het aantal referentiehokken is steeds het dubbele hiervan. Per parameter wordt het gemiddelde gegeven voor de ontgrondingshokken (met standaarddeviatie), het gemiddelde voor de referentiehokken (idem), en het verschil. Positieve getallen in de onderste rij duiden op hogere waarden in de hokken met een ontgronding.
In geen enkel geval is er sprake van een significant verschil in toestand tussen de ontgrondingslocaties en de referenties. Alle hieronder aangegeven verschillen zijn dus ten hoogste voorzichtige tendenties.

Grindwinning
Grindwinning vindt plaats op de terrassen langs de Maas. Behalve het totale aantal soorten hebben alle berekende waarden de neiging om in kilometerhokken met een ontgronidng iets lager uit te vallen dan in de referentiehokken. Alle verschillen vallen echter ruimschoots binnen de standaarddeviatie.

7 hokken	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	168 ± 60	3.0 ± 1.8	5.3 ± 2.7	3.8 ± 1.8	8.6 ± 4.7	16.4 ± 16.3	3.6 ± 5.5
Referentie	168 ± 37	3.5 ± 3.0	6.3 ± 5.7	3.9 ± 3.3	7.6 ± 4.1	23.0 ± 18.0	6.0 ± 5.8
Verschil	0 ± 59	-0.5 ± 3.9	-1.1 ± 5.5	-0.2 ± 4.1			

Winning bouw- en metselzand (actief)
Alle ontgrondingen t.b.v. bouw- en metselzand liggen binnen het rivierengebied. De ontgrondingshokken laten voor alle parameters een negatief beeld zien; dit valt echter nog steeds binnen de standaarddeviatie. In deze groep zitten twee kilometerhokken met een grote oppervlakte aan open water (75 hectare resp. 50 hectare).

5 hokken	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	163 ± 68	2.0 ± 1.8	4.2 ± 4.0	2.1 ± 2.0	9.0 ± 5.3	16.0 ± 20.4	9.1 ± 15.1
Referentie	217 ± 34	3.5 ± 1.9	7.4 ± 3.3	4.3 ± 2.0	12.3 ± 7.3	35.8 ± 21.5	17.3 ± 10.2
Verschil	-54 ± 70	-1.5 ± 2.4	-3.2 ± 4.3	-2.2 ± 2.7			

Kleiwinning
Kleiwinning vindt in alle gevallen plaats in het rivierengebied. De kilometerhokken met kleiwinning lijken gelijk of iets gunstiger te scoren dan de referentiehokken. In alle gevallen ligt het verschil binnen de standaarddeviatie.

9 hokken	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	271 ± 79	5.3 ± 2.8	13.1 ± 6.9	7.0 ± 4.3	24.6 ± 17.5	61.2 ± 44.9	29.9 ± 25.8
Referentie	244 ± 39	4.8 ± 3.6	11.1 ± 5.9	6.5 ± 4.3	23.8 ± 9.7	47.1 ± 27.8	17.6 ± 13.5
Verschil	26 ± 66	0.6 ± 3.1	2.1 ± 6.2	0.5 ± 4.7	0.7 ± 13.9	14.2 ± 26.7	12.3 ± 15.3

8.5.1 Kalksteenwinning
In het onderzoek is slechts één kalksteenwinning betrokken. Het betreft de mergelgroeve in de Sint-Pietersberg bij Maastricht. De Sint-Pietersberg is floristisch gezien van oudsher een zeer bijzonder gebied, dat niet goed met kilometerhokken in de omgeving is te vergelijken. Het grootste deel van de bijzondere soorten bevindt zich overigens op het onvergraven deel van de berg (Vreeken, 1999), dat ten dele ook binnen het kmhok van de winningslocatie is gelegen.

1 hok	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	430	40	73	46	87	379	63
Referentie	243	12.5	23.5	15	29.5	171	25.5

8.5.2 Ophoogzand tijdens winning

Ophoogzand wordt in allerlei delen van Nederland gewonnen. Er wordt onderscheid gemaakt tussen locaties waarin de winning nog plaatsvindt en locaties waarin de winning is afgerond (zie volgende groep). De natuurwaarde op de winningslocaties lijkt gemiddeld iets hoger dan in de referentiehokken.

14 hokken	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	199 ± 42	2.4 ± 2.2	4.3 ± 3.8	2.6 ± 2.2	5.6 ± 5.1	20.5 ± 25.2	13.5 ± 23.4
Referentie	177 ± 53	1.5 ± 2.5	3.3 ± 4.8	1.6 ± 2.8	5.3 ± 7.0	13.0 ± 25.4	4.5 ± 10.5
Verschil	22 ± 39.5	0.9 ± 2.1	1.0 ± 2.7	0.9 ± 2.2	0.4 ± 3.7	7.5 ± 19.9	9.0 ± 20.6

8.5.3 Ophoogzand na de winning

Locaties waar de winning van ophoogzand is afgerond zijn als aparte groep beschouwd. Hoewel dit de grootste groep van ontgrondingshokken is, zijn ook hier geen significante verschillen waarneembaar. Opvallend is dat alle parameters in deze groep voor zowel de ontgrondingen als voor de referenties hoger zijn dan in de vorige groep, *ophoogzand tijdens winning*. We hebben daarvoor geen verklaring.

44 hokken	Nsoort	Nrodelijst	Naandacht	Ndoel	Nzeldzaam	NWtotaal	NWnat
Ontgrond	210 ± 57	3.3 ± 3.2	6.7 ± 5.2	3.9 ± 3.5	8.4 ± 8.7	29.3 ± 33.1	15.5 ± 23.8
Referentie	194 ± 60	2.5 ± 3.1	5.2 ± 5.0	3.0 ± 3.4	7.6 ± 8.7	30.6 ± 36.2	18.3 ± 29.1
Verschil	15 ± 59.3	0.9 ± 3.6	1.4 ± 5.3	0.9 ± 4.0	0.8 ± 6.2	-1.3 ± 39.4	-2.8 ± 31.1

5.4. Discussie

Binnen de randvoorwaarden waarbinnen de analyse moest worden uitgevoerd, lijken er weinig belangwekkende veranderingen in de flora op te treden als gevolg van delfstoffenwinning. Het is de vraag in hoeverre dat een gevolg is van de analysemethode of de beschikbare gegevens, en in hoeverre dit de "werkelijkheid" weergeeft.
Stel dat we de beschikking zouden hebben over floragegevens per hectare in plaats van per kmhok. In dat geval zouden we twee ontwikkelingen als gevolg van een ontgronding zichtbaar maken en goed kunnen kwantificeren:

- Een deel van de oppervlakte van de ontgrondingslocatie wordt permanent verhard of wordt permanent een diepe plas. Op zulke locaties ontbreken plantensoorten bijna geheel, zodat het verlies aan botanische waarde de complete soortensamenstelling van vóór de ingreep betreft. Dit is het effect van *biotoopverlies*.
- Een ander deel van de locatie zou geschikt blijven of weer worden voor plantengroei. Afhankelijk van de nieuwe omstandigheden zou er sprake kunnen zijn van een toename of een afname van de botanische waarde ten opzichte van de situatie, voordat de ontgronding een aanvang nam. Ook de indirecte effecten van een ontgronding op de omgeving, bijvoorbeeld als gevolg van veranderingen in de hydrologie, zou op zo'n wijze geanalyseerd kunnen worden. Dit is het effect *van verandering van standplaatscondities, grondgebruik of beheer*.

Het totaaleffect van de ontgronding op de botanische waarde van een locatie zou op basis van basisgegevens met een ruimtelijk detail van 1 hectare nauwkeurig kunnen worden berekend, aangezien de meeste ontgrondingen enkele tientallen hectaren beslaan en soms nog vele tientallen tot honderden hectaren indirect beïnvloeden. Nu de analyse op basis van kmhokgegevens moet worden gemaakt, zijn beide effecten niet te scheiden. Bovendien bevat een kmhok vaak delen, waarin geen ontgronding plaatsvindt en waar dus nog lang de "oude soortensamenstelling" wordt bewaard. Een en ander kan er eenvoudig toe leiden dat er binnen het kmhok een – tijdelijke – toename van het aantal verschillende biotopen en daardoor soorten optreedt, waardoor de effecten van biotoopverlies worden gemaskeerd. In de referentiehokken zal de toename van het aantal nieuwe biotopen meestal geringer zijn, er zal vaak een intensiever landgebruik plaatsvinden en de ontwikkelingen in de flora zullen vaker aansluiten bij de autonome ontwikkelingen die in grote delen van het ecodistrict optreden als gevolg van geleidelijke veranderingen in gebruiksintensiteit, bemesting, zuurdepositie, ontwatering, en dergelijke.
Door de floragegevens uit de periode van ontgronding te gebruiken, wordt een beeld geschetst van de botanische waarden van ontgrondingen die niet het eindplaatje behoeven te zijn. Als de ontgronding eenmaal is afgelopen, zal er een zekere successie plaatsvinden naar een nieuwe evenwichtssituatie. Daarbij verdwijnen vaak allerlei pionierbiotopen, die vaak relatief soortenrijk en waardevol zijn; of de evenwichtssituatie dat ook is, is onder meer afhankelijk van abiotische standplaatscondities, het beheer en de inrichting van het gebied. Dit is nu niet geanalyseerd, terwijl het eigenlijk het meest essentieel is bij de analyse van de effecten van delfstoffenwinning: hoe verhoudt een nieuwe, min of meer stabiele situatie, zich tot de situatie vóór ontgronding.
Alleen voor het ontgrondingstype ophoogzand zijn er vrij veel locaties geschikt om met behulp van de beschikbare gegevens te analyseren. Voor de andere typen winning zijn er te weinig geschikte locaties om te verwachten dat er statistisch significante uitspraken kunnen worden afgeleid. Zelfs bij ophoogzandlocaties zijn er echter geen duidelijke aanwijzingen voor een bepaalde trend.

Het is derhalve de vraag of het toevoegen van floragegevens voor meer ontgrondings-locaties tot een betere analyse zouden leiden. Dat zou onder meer vrij snel en betrekkelijk goedkoop kunnen gebeuren door een floristische inventarisatie van de in bedrijf zijnde ontgrondingslocaties uit te voeren.

5.5. Conclusies en aanbevelingen

5.5.1. Conclusies

Er zijn bruikbare floragegevens in 85 van de 357 kilometerhokken waar ontgrondingen t.b.v. delfstoffenwinning plaatsvinden.
Er zijn geen duidelijke effecten op de floristische samenstelling van de vegetatie in kmhokken waar een ontgrondingslocatie is gesitueerd, in vergelijking met referentiehokken in de nabijheid.
De ruimtelijke schaal waarop floragegevens in de landelijke floradatabank FlorBase opgeslagen zijn, is te grof voor een goede analyse.
De grootte en de ligging van de ontgrondingslocaties is onvoldoende duidelijk voor een goede analyse.
Voor verschillende typen ontgrondingen is het aantal geschikte locaties voor analyse gering.
De effecten van het ontgronden zelf en de effecten van het veranderde landgebruik zijn niet te scheiden, zodat onduidelijk is welk deel van de effecten “definitief” is en welk deel beïnvloedbaar door mitigerende en beheersmaatregelen.
De invloed van een ontgronding op de omgeving door bijvoorbeeld veranderde hydrologische condities zijn niet in beschouwing genomen.

5.5.2. Aanbevelingen

Voor een goede analyse van de effecten van ontgronding is het verstandig een aantal locaties te monitoren, waarbij het om de volgende aspecten gaat:

- Wat is de floristische uitgangssituatie, voordat de voorbereidingen van de ontgronding een aanvang nemen.
- Welke oppervlakte gaat verloren voor flora en vegetatie door verharding of aanleg van diep uitgegraven wateren.
- Wat is de floristische samenstelling van de vegetatie (per zone of deelgebied, afhankelijk van inrichting en beheer) na de ontgronding (1, 2, 4, 7, 12 jaar na de ingreep).
- Welke rol spelen mitigerende en beheersmaatregelen in het behouden of uitbreiden van botanische waarden.

Door een goede keuze van onderzoekslocaties en een niet al te gedetailleerde ruimtelijke indeling van onderzoeksgebieden kan zo’n monitoringprogramma tegen aanvaardbare kosten in 10-15 jaar leiden tot kwantitatief en kwalitatief goede gegevens over de effecten van delfstoffenwinning op de flora.
De looptijd van een dergelijk onderzoek kan wellicht aanzienlijk worden bekort door bijna beëindigde ontgrondingen te gaan monitoren op de veranderingen in de flora vanaf het moment van beëindiging, en de uitgangssituatie te schatten uit historische gegevens van direct vóór de winning, of uit de omgeving.

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Annex 6 ETH process changes

Occupation

Processes derived from Natural gas chain	TNO name	Unit	Value
Bohrmeter für Exploration und Produktion offshore [per m]	Occup benthos EU	m^2.y/m	693
Infra Pipeline offshore [piece]	Occup benthos EU	m^2.y/piece	0,000069
Infra Pipeline onshore [piece]	Occup ind veg EU	m^2.y/piece	0,000046
Infra Brenngas Kraftwerk	Occup ind built EU	m^2.y/TJ	50
Infra Erdgas HD-abnehmer D	Occup ind built D	m^2.y/TJ	10,2
Infra Erdgas HD-abnehmer D	Occup ind built D	m^2.y/TJ	92,1
Infra Erdgas HD-abnehmer I	Occup ind built I	m^2.y/TJ	9,4
Infra Erdgas HD-abnehmer I	Occup ind built I	m^2.y/TJ	84,5
Infra Erdgas HD-abnehmer A	Occup ind built A	m^2.y/TJ	6,4
Infra Erdgas HD-abnehmer A	Occup ind built A	m^2.y/TJ	57,5
Infra Erdgas HD-abnehmer NL	Occup ind built NL	m^2.y/TJ	3,7
Infra Erdgas HD-abnehmer NL	Occup ind built NL	m^2.y/TJ	33,5
Infra Erdgas HD-abnehmer B	Occup ind built B	m^2.y/TJ	8,1
Infra Erdgas HD-abnehmer B	Occup ind built B	m^2.y/TJ	72,8
Infra Erdgas HD-abnehmer F	Occup ind built F	m^2.y/TJ	12,7
Infra Erdgas HD-abnehmer F	Occup ind built F	m^2.y/TJ	114,4
Rohgas (Erdgas) - Offshore	Occup ind built EU	m^2.y/m^3	0,00002
Rohgas (Erdgas) - Onshore	Occup ind built EU	m^2.y/m^3	0,0003
Rohgas (Erdgas) - Onshore	Occup ind built EU	m^2.y/m^3	0,00001
Infra produziertes Erdgas, NL, GUS, D, N and Alg.	Occup ind built EU	m^2.y/m^3	0,0001
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Occup ind built EU	m^2.y/tkm	0,000035
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Occup ind built EU	m^2.y/tkm	0,000316
Infra Transport Erdgas-Pipeline GUS	Occup ind built GUS	m^2.y/tkm	0,000028
Infra Transport Erdgas-Pipeline GUS	Occup ind built GUS	m^2.y/tkm	0,000253
Infra Strom Hochspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	0,95
Infra Strom Hochspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	0,95
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	1,615
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	1,615
Infra Strom Niederspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	2,47
Infra Strom Niederspannung - Bezug in UCPTE, CH	Occup ind built EU	m^2.y/TJ	2,47
Sand fuer Bau	Occup extr sand EU	m^2.y/kg	0,0024
Processes derived from hard coal chain			
Infra Schiene	Occup rail EU	m^2.y/tkm	0,001045
Infra Schiene	Occup rail EU	m^2.y/tkm	0,00247
Infra Strasse LKW, lieferwagen	Occup road EU	m^2.y/tkm	0,0038
Infra Strasse LKW, lieferwagen	Occup road EU	m^2.y/tkm	0,00874
Infra Strasse PKW Westeuropa	Occup road EU	m^2.y/km	0,0019
Infra Strasse PKW Westeuropa	Occup road EU	m^2.y/km	0,00418
Infra Binnenfrachter	Occup ind built EU	m^2.y/tkm	0,0038
Infra Binnentankschiff	Occup ind built EU	m^2.y/tkm	0,0038
Infra Transport Europa	Occup road EU	m^2.y/t	0,95
Infra Transport Uebersee	Occup ind built	m^2.y/t	1,9
Infra Stk Kraftwerk	Occup ind built EU	m^2.y/TJ	5,7
Infra Brk Kraftwerk	Occup ind built EU	m^2.y/TJ	3,6
Infra Steinkohle Untertagbau ab Bergwerk	Occup extr coal h mine	m^2.y/t	0,48
Infra Steinkohle Untertagbau ab Bergwerk	Occup extr coal h mine	m^2.y/t	0,48
Steinkohle aus Tagbau ab Bergwerk	Occup extr coal h pit	m^2.y/t	2,25
Steinkohle aus Tagbau ab Bergwerk	Occup extr coal h pit	m^2.y/t	0,97
Stk aus Untertagbau ab Australien-Bergwerk	Occup extr coal h mine AUS	m^2.y/t	0,18
Stk aus Untertagbau ab Nord-Amerika-Bergwerk	Occup extr coal h mine USA	m^2.y/t	0,18
Stk aus Untertagbau ab Ost-Europa-Bergwerk	Occup extr coal h mine oEU	m^2.y/t	0,18
Stk aus Untertagbau ab Sued-Afrika-Bergwerk	Occup extr coal h mine sAF	m^2.y/t	0,18

Stk aus Untertagbau ab UCPTE-Bergwerk	Occup extr coal h mine UCP	m^2.y/t	0,18
Rohbraunkohle ab Bergwerk UCPTE	Occup extr coal b pit UCP	m^2.y/t	0,15
Rohbraunkohle ab Bergwerk UCPTE	Occup extr coal b pit UCP	m^2.y/t	0,35
Processes derived from Transport LKW 28t chain			
Betonkies	Occup extr bentonite EU	m^2.y/kg	0,0024
Other processes related to Oil chain			
Infra Regionalverteilung CH, Euro	Occup ind built CH	m^2.y/t	0,080
Infra Kraftwerk oelthermisch CH, UCPTE	Occup ind built CH	m^2.y/TJ(in)	5,23
Infra Kraftwerk oelthermisch Laender	Occup ind built EU	m^2.y/TJ(out)	13,8
Infra Raffinerie [piece]	Occup ind built EU	m^2.y/piece	0,2
Infra Foerderung Erdoel Offshore	Occup ind built EU	m^2.y/piece	0,0225
Infra Foerderung Erdoel Onshore	Occup ind built EU	m^2.y/piece	0,0792
Infra Ferntransport	Occup road EU	m^2.y/piece	0,95
Infra Ferntransport	Occup road EU	m^2.y/piece	0,95
Nuclear energy			
Uran in Uranerz aus Tagebau-Mine	Occup extr uran pit	m^2.y/kg	3,82
Uran in Uranerz aus Tiefbau-Mine	Occup extr uran mine	m^2.y/kg	0,09
Uran natuerlich in Urankonzentrat	Occup ind built Uconc	m^2.y/kg	0,92
Infra Anreicherung Diffusion	Occup ind built Udiff	m^2.y/piece	0,30
Infra Anreicherung Zentrifuge	Occup ind built Ucentr	m^2.y/piece	0,15
Infra Brennelementfertigung	Occup ind built Uelem	m^2.y/piece	0,22
Infra Endlagerung B	Occup ind built U dump B	m^2.y/piece	14,04
Infra Endlagerung C	Occup ind built U dump C	m^2.y/piece	299
Infra Kernkraftwerk DWR	Occup ind built DWR	m^2.y/piece	7,30
Infra Kernkraftwerk DWR UCPTE	Occup ind built DWR UCP	m^2.y/piece	8,65
Infra Kernkraftwerk SWR	Occup ind built SWR	m^2.y/piece	8,98
Infra Kernkraftwerk SWR UCPTE	Occup ind built SWR UCP	m^2.y/piece	10,49
Infra Konversion	Occup ind built Uconv	m^2.y/piece	0,04
Infra Wiederaufarbeitung	Occup ind built Urecov	m^2.y/piece	1,01
Infra Zwischenlagerung zu Lager B	Occup ind built EU	m^2.y/piece	32,3
Infra Zwischenlagerung zu Lager C	Occup ind built EU	m^2.y/piece	161,6
Schwach radioaktive Abfaelle	Occup dump w radact EU	m^2.y/m^3	36,0

Renaturation	TNO name	Unit	Value
Processes derived from Natural gas chain			
Bohrmeter für Exploration und Produktion offshore [per m]	Renat benthos III-II 50y EU	m^2.y/m	6500
Infra Pipeline Onshore [piece]	Renat ind veg III-II 50y EU	m^2.y/piece	0,00015
Infra Erdgas HD-abnehmer D	Renat ind built III-II 50y D	m^2.y/TJ	41,6
Infra Erdgas HD-abnehmer I	Renat ind built III-II 50y I	m^2.y/TJ	38,1
Infra Erdgas HD-abnehmer A	Renat ind built III-II 50y A	m^2.y/TJ	26,0
Infra Erdgas HD-abnehmer NL	Renat ind built III-II 50y NL	m^2.y/TJ	15,1
Infra Erdgas HD-abnehmer B	Renat ind built III-II 50y B	m^2.y/TJ	32,9
Infra Erdgas HD-abnehmer F	Renat ind built III-II 50y F	m^2.y/TJ	51,5
Rohgas (Erdgas) - Onshore	Renat extr gas III-II 50y EU	m^2.y/m^3	0,000005
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Renat ind built III-II 50y EU	m^2.y/tkm	0,00018
Infra Transport Erdgas-Pipeline GUS	Renat ind built III-II 50 GUS	m^2.y/tkm	0,00014
Infra Strom Hochspannung - Bezug in UCPTE, CH	Renat ind built III-II 50y EU	m^2.y/TJ	0,4
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Renat ind built III-II 50y EU	m^2.y/TJ	0,5
Infra Strom Niederspannung - Bezug in UCPTE, CH	Renat ind built III-II 50y EU	m^2.y/TJ	0,6
Sand für Bau	Renat extr sand III-II 50y EU	m^2.y/kg	0,0010
Processes derived from hard coal chain			
Infra Schiene	Renat rail III-II 50y EU	m^2.y/tkm	0,0004
Infra Strasse LKW, lieferwagen	Renat road III-II 50y EU	m^2.y/tkm	0,0004
Infra Strasse PKW Westeuropa	Renat road III-II 50y EU	m^2.y/km	0,0002
Infra Steinkohle Untertagbau ab Bergwerk	Renat coal h mine III-II 50y	m^2.y/t	0,4

Steinkohle aus Tagbau ab Bergwerk	Renat coal h pit III-II 50y	m^2.y/t	5,9
Stk aus Untertagbau ab Australien-Bergwerk	Renat coal mine III-II 50 AUS	m^2.y/t	2,4
Stk aus Untertagbau ab Nord-Amerika-Bergwerk	Renat coal mine III-II 50 USA	m^2.y/t	2,4
Stk aus Untertagbau ab Ost-Europa-Bergwerk	Renat coal mine III-II 50 oEU	m^2.y/t	2,4
Stk aus Untertagbau ab Sued-Afrika-Bergwerk	Renat coal mine III-II 50 sAF	m^2.y/t	2,4
Stk aus Untertagbau ab UCPTE-Bergwerk	Renat coal mine III-II 50 UCP	m^2.y/t	2,4
Rohbraunkohle ab Bergwerk UCPTE	Renat coal b pit III-II 50 UCP	m^2.y/t	1,8
Processes derived from Transport LKW 28t chain			
Betonkies	Renat bentonite III-II 50 EU	m^2.y/kg	0,0010
Other processes related to Oil chain			
Infra Raffinerie [piece]	Renat ind built III-II 50 EU	m^2.y/piece	0,0389
Infra Foerderung Erdoel Onshore		m^2.y/piece	0,0375
Nuclear energy			
Uran in Uranerz aus Tagebau-Mine	Renat uran pit III-II 50y	m^2.y/kg	5,0
Uran in Uranerz aus Tiefbau-Mine	Renat uran mine III-II 50y	m^2.y/kg	0,2
Uran natuerlich in Urankonzentrat	Renat Uconc III-II 80000y	m^2.y/kg	1950
Infra Anreicherung Diffusion	Renat Udiff III-II 50y	m^2.y/piece	0,22
Infra Anreicherung Zentrifuge	Renat Ucentr III-II 50y	m^2.y/piece	0,13
Infra Brennelementfertigung	Renat Uelem III-II 50y	m^2.y/piece	0,17
Infra Endlagerung B	Renat Udump B III-II 50y	m^2.y/piece	12,5
Infra Endlagerung C	Renat Udump C III-II 50y	m^2.y/piece	190,0
Infra Kernkraftwerk DWR	Renat ind DWR III-II 50y	m^2.y/piece	3,9
Infra Kernkraftwerk DWR UCPTE	Renat ind DWR III-II 50y UCP	m^2.y/piece	4,6
Infra Kernkraftwerk SWR	Renat ind SWR III-II 50y	m^2.y/piece	4,6
Infra Kernkraftwerk SWR UCPTE	Renat ind SWR III-II 50y UCP	m^2.y/piece	5,4
Infra Konversion	Renat Uconv III-II 50y	m^2.y/piece	0,024
Infra Wiederaufarbeitung	Renat Urecov III-II 50y	m^2.y/piece	0,7
Infra Zwischenlagerung zu Lager B	Renat ind built EU III-II 50y	m^2.y/piece	14
Infra Zwischenlagerung zu Lager C	Renat ind built EU III-II 50y	m^2.y/piece	70
Schwach radioaktive Abfaelle	Renat dump w radact III-II 50	m^2.y/m^3	50

Correction in production processes: conversion to Dutch situation for coal and gas power plants

Occupation	Name	Unit	Value
Infra Brenngas Kraftwerk NL	Occup ind built NL	m^2.y/TJ	22,8
Infra Stk Kraftwerk NL	Occup ind built NL	m^2.y/TJ	3,8

<u>Occupation</u>	Name	Unit	Value
Processes derived from Natural gas chain			
Bohrmeter für Exploration und Produktion offshore [per m]	Trans benthos II-IV EU	m^2/m	260
Infra Pipeline offshore [piece]	Trans benthos II-IV EU	m^2/piece	0,0000021
Infra Pipeline onshore [piece]	Trans ind veg II-IV EU	m^2/piece	0,0000014
Infra Brenngas Kraftwerk	Trans ind built II-IV EU	m^2/TJ	1,75
Infra Erdgas HD-abnehmer D	Trans ind built II-IV D	m^2/TJ	0,2
Infra Erdgas HD-abnehmer D	Trans ind built III-IV D	m^2/TJ	1,8
Infra Erdgas HD-abnehmer I	Trans ind built II-IV I	m^2/TJ	0,2
Infra Erdgas HD-abnehmer I	Trans ind built III-IV I	m^2/TJ	1,6
Infra Erdgas HD-abnehmer A	Trans ind built II-IV A	m^2/TJ	0,1
Infra Erdgas HD-abnehmer A	Trans ind built III-IV A	m^2/TJ	1,1
Infra Erdgas HD-abnehmer NL	Trans ind built II-IV NL	m^2/TJ	0,1
Infra Erdgas HD-abnehmer NL	Trans ind built III-IV NL	m^2/TJ	0,6
Infra Erdgas HD-abnehmer B	Trans ind built II-IV B	m^2/TJ	0,2
Infra Erdgas HD-abnehmer B	Trans ind built III-IV B	m^2/TJ	1,4

Infra Erdgas HD-abnehmer F	Trans ind built II-IV F	m^2/TJ	0,2
Infra Erdgas HD-abnehmer F	Trans ind built III-IV F	m^2/TJ	2,2
Rohgas (Erdgas) - Offshore	Trans ind built III-IV EU	m^2/m^3	0,000002
Rohgas (Erdgas) - Onshore	Trans ind built II-IV EU	m^2/m^3	0,00003
Rohgas (Erdgas) - Onshore	Trans ind built III-IV EU	m^2/m^3	0,000001
Infra produziertes Erdgas, NL, GUS, D, N and Alg.	Trans ind built II-IV EU	m^2/m^3	0,00001
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Trans ind built II-IV EU	m^2/tkm	0,00000067
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Trans ind built III-IV EU	m^2/tkm	0,00000602
Infra Transport Erdgas-Pipeline GUS	Trans ind built II-IV GUS	m^2/tkm	0,00000053
Infra Transport Erdgas-Pipeline GUS	Trans ind built III-IV GUS	m^2/tkm	0,00000480
Infra Strom Hochspannung - Bezug in UCPTE, CH	Trans ind built II-IV EU	m^2/TJ	0,0333
Infra Strom Hochspannung - Bezug in UCPTE, CH	Trans ind built III-IV EU	m^2/TJ	0,0333
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Trans ind built II-IV EU	m^2/TJ	0,0567
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Trans ind built III-IV EU	m^2/TJ	0,0567
Infra Strom Niederspannung - Bezug in UCPTE, CH	Trans ind built II-IV EU	m^2/TJ	0,0867
Infra Strom Niederspannung - Bezug in UCPTE, CH	Trans ind built III-IV EU	m^2/TJ	0,0867
Sand fuer Bau	Trans extr sand III-IV EU	m^2/kg	2,9E-05
Processes derived from hard coal chain			
Infra Schiene	Trans rail II-IV EU	m^2/tkm	0,000022
Infra Schiene	Trans rail III-IV EU	m^2/tkm	0,000052
Infra Strasse LKW, lieferwagen	Trans road II-IV EU	m^2/tkm	0,00008
Infra Strasse LKW, lieferwagen	Trans road III-IV EU	m^2/tkm	0,000184
Infra Strasse PKW Westeuropa	Trans road II-IV EU	m^2/km	0,000040
Infra Strasse PKW Westeuropa	Trans road III-IV EU	m^2/km	0,000088
Infra Binnenfrachter	Trans ind built III-IV EU	m^2/tkm	0,00013
Infra Binnentankschiff	Trans ind built III-IV EU	m^2/tkm	0,00013
Infra Transport Europa	Trans road II-IV EU	m^2/t	0,033
Infra Transport Uebersee	Trans ind built II-IV EU	m^2/t	0,067
Infra Stk Kraftwerk	Trans ind built II-IV EU	m^2/TJ	0,2
Infra Brk Kraftwerk	Trans ind built II-IV EU	m^2/TJ	0,1
Infra Steinkohle Untertagbau ab Bergwerk	Trans coal mine II-IV	m^2/t	0,02
Infra Steinkohle Untertagbau ab Bergwerk	Trans coal mine III-IV	m^2/t	0,02
Steinkohle aus Tagbau ab Bergwerk	Trans coal h pit II-IV	m^2/t	0,18
Steinkohle aus Tagbau ab Bergwerk	Trans coal h pit III-IV	m^2/t	0,08
Stk aus Untertagbau ab Australien-Bergwerk	Trans coal mine II-IV AUS	m^2/t	0,01
Stk aus Untertagbau ab Nord-Amerika-Bergwerk	Trans coal mine II-IV USA	m^2/t	0,01
Stk aus Untertagbau ab Ost-Europa-Bergwerk	Trans coal mine II-IV oEU	m^2/t	0,01
Stk aus Untertagbau ab Sued-Afrika-Bergwerk	Trans coal mine II-IV sAF	m^2/t	0,01
Stk aus Untertagbau ab UCPTE-Bergwerk	Trans coal mine II-IV UCP	m^2/t	0,01
Rohbraunkohle ab Bergwerk UCPTE	Trans coal b pit II-IV UCP	m^2/t	0,01
Rohbraunkohle ab Bergwerk UCPTE	Trans coal b pit II-IV UCP	m^2/t	0,03
Processes derived from Transport LKW 28t chain			
Betonkies	Trans bentonite III-IV EU	m^2/kg	2,9E-05
Other processes related to Oil chain			
Infra Regionalverteilung CH, Euro	Trans ind built II-IV CH	m^2/t	0,003
Infra Kraftwerk oelthermisch CH, UCPTE	Trans ind built II-IV CH	$m^2/TJ(in)$	0,183
Infra Kraftwerk oelthermisch Laender	Trans ind built II-IV EU	$m^2/TJ(out)$	0,484
Infra Raffinerie [piece]	Trans ind built II-IV EU	$m^2/piece$	0,007
Infra Foerderung Erdoel Offshore	Trans ind built III-IV EU	$m^2/piece$	0,0050
Infra Foerderung Erdoel Onshore	Trans ind built II-IV EU	$m^2/piece$	0,0015
Infra Ferntransport	Trans road II-IV EU	$m^2/piece$	0,033
Infra Ferntransport	Trans road III-IV EU	$m^2/piece$	0,033
Nuclear energy			
Uran in Uranerz aus Tagebau-Mine	Trans uran pit II-IV	m^2/kg	0,18
Uran in Uranerz aus Tiefbau-Mine	Trans uran mine II-IV	m^2/kg	0,004
Uran natuerlich in Urankonzentrat	Trans ind built Ucon II-IV	m^2/kg	0,042

Infra Anreicherung Diffusion	Trans ind built Udif II-IV	m^2/piece	0,009
Infra Anreicherung Zentrifuge	Trans ind built Ucen II-IV	m^2/piece	0,006
Infra Brennelementfertigung	Trans ind built Uele II-IV	m^2/piece	0,007
Infra Endlagerung B	Trans ind built II-IV	m^2/piece	0,201
Infra Endlagerung C	Trans ind built II-IV	m^2/piece	4,337
Infra Kernkraftwerk DWR	Trans ind built II-IV	m^2/piece	0,15
Infra Kernkraftwerk DWR UCPTE	Trans ind built UCP II-IV	m^2/piece	0,18
Infra Kernkraftwerk SWR	Trans ind built II-IV	m^2/piece	0,18
Infra Kernkraftwerk SWR UCPTE	Trans ind built II-IV UCP	m^2/piece	0,21
Infra Konversion	Trans ind built II-IV	m^2/piece	0,0017
Infra Wiederaufarbeitung	Trans ind built II-IV	m^2/piece	0,021
Infra Zwischenlagerung zu Lager B	Trans ind built II-IV EU	m^2/piece	0,56
Infra Zwischenlagerung zu Lager C	Trans ind built II-IV EU	m^2/piece	2,79
Schwach radioaktive Abfaelle	Trans dump radact II-IV EU	m^2/m^3	1,60

Process - Trans-renaturation		Unit	Landuse
Processes derived from Natural gas chain	**Actually from IV to II, because recultivation from IV to III is not considered in Trans !**		
Bohrmeter für Exploration und Produktion offshore [per m]	Trare benthos III-II EU	m^2/m	260
Infra Pipeline onshore [piece]	Trare ind veg III-II EU	m^2/piece	0,0000060
Infra Erdgas HD-abnehmer D	Trare ind built III-II D	m^2/TJ	1,7
Infra Erdgas HD-abnehmer I	Trare ind built III-II I	m^2/TJ	1,5
Infra Erdgas HD-abnehmer A	Trare ind built III-II A	m^2/TJ	1,0
Infra Erdgas HD-abnehmer NL	Trare ind built III-II NL	m^2/TJ	0,6
Infra Erdgas HD-abnehmer B	Trare ind built III-II B	m^2/TJ	1,3
Infra Erdgas HD-abnehmer F	Trare ind built III-II F	m^2/TJ	2,1
Rohgas (Erdgas) - Onshore	Trare extr gas III-II EU	m^2/m^3	0,0000020
Infra Transport Erdgas-Pipeline NL, D, N, Euro and Alg.	Trare ind built III-II EU	m^2/tkm	0,0000071
Infra Transport Erdgas-Pipeline GUS	Trare ind built III-II GUS	m^2/tkm	0,0000057
Infra Strom Hochspannung - Bezug in UCPTE, CH	Trare ind built III-II EU	m^2/TJ	0,016
Infra Strom Mittelspannung - Bezug in UCPTE, CH	Trare ind built III-II EU	m^2/TJ	0,020
Infra Strom Niederspannung - Bezug in UCPTE, CH	Trare ind built III-II EU	m^2/TJ	0,024
Sand fuer Bau	Trare extr sand III-II EU	m^2/kg	0,000040
Processes derived from hard coal chain			
Infra Schiene	Trare rail III-II EU	m^2/tkm	0,000016
Infra Strasse LKW, lieferwagen	Trare road III-II EU	m^2/tkm	0,000014
Infra Strasse PKW Westeuropa	Trare road III-II EU	m^2/tkm	0,000006
Infra Steinkohle Untertagbau ab Bergwerk	Trare coal h mine III-II	m^2/t	0,016
Steinkohle aus Tagbau ab Bergwerk	Trare coal h pit III-II	m^2/t	0,24
Stk aus Untertagbau ab Australien-Bergwerk	Trare coal mine III-II AUS	m^2/t	0,1
Stk aus Untertagbau ab Nord-Amerika-Bergwerk	Trare coal mine III-II USA	m^2/t	0,1
Stk aus Untertagbau ab Ost-Europa-Bergwerk	Trare coal mine III-II oEU	m^2/t	0,1
Stk aus Untertagbau ab Sued-Afrika-Bergwerk	Trare coal mine III-II sAF	m^2/t	0,1
Stk aus Untertagbau ab UCPTE-Bergwerk	Trare coal mine III-II UCP	m^2/t	0,1
Rohbraunkohle ab Bergwerk UCPTE	Trare coal b pit III-II UCP	m^2/t	0,07
Processes derived from Transport LKW 28t chain			
Betonkies	Trare bentonite III-II EU	m^2/kg	0,000040
Other processes related to Oil chain			
Infra Raffinerie [piece]	Trare ind built III-II EU	m^2/piece	0,0016
Infra Foerderung Erdoel Onshore		m^2/piece	0,0015
Nuclear energy			
Uran in Uranerz aus Tagebau-Mine	Trare uran pit III-II	m^2/kg	0,20
Uran in Uranerz aus Tagebau-Mine	Trare uran mine III-II	m^2/kg	0,01

Uran natuerlich in Urankonzentrat	Trare Uconc III-II	m^2/kg	0,05
Infra Anreicherung Diffusion	Trare Udiff III-II	m^2/piece	0,0086
Infra Anreicherung Zentrifuge	Trare Ucentr III-II	m^2/piece	0,0050
Infra Brennelementfertigung	Trare Uelem III-II	m^2/piece	0,01
Infra Endlagerung B	Trare Udump B III-II	m^2/piece	0,5
Infra Endlagerung C	Trare Udump C III-II	m^2/piece	7,6
Infra Kernkraftwerk DWR	Trare ind DWR III-II	m^2/piece	0,15
Infra Kernkraftwerk DWR UCPTE	Trare ind DWR III-II UCP	m^2/piece	0,18
Infra Kernkraftwerk SWR	Trare ind SWR III-II	m^2/piece	0,18
Infra Kernkraftwerk SWR UCPTE	Trare ind SWR III-II UCP	m^2/piece	0,21
Infra Konversion	Trare Uconv III-II	m^2/piece	0,0009
Infra Wiederaufarbeitung	Trare Urecov III-II	m^2/piece	0,03
Infra Zwischenlagerung zu Lager B	Trare ind built EU III-II	m^2/piece	0,56
Infra Zwischenlagerung zu Lager C	Trare ind built EU III-II	m^2/piece	2,80
Schwach radioaktive Abfaelle	Trare dump w radact III-II	m^2/m^3	2,0

Correction in production processes: conversion to Dutch situation for coal and gas power plants

Process - Production	Name	Unit	Landuse
Infra Brenngas Kraftwerk NL	Trans ind built II-IV NL	m^2/TJ	0,8
Infra Stk Kraftwerk NL	Trans ind built II-IV NL	m^2/TJ	0,1

Annex 7 Nomenclature, CORINE formulas and impact assessment scores

		Ecosystem occupation (average respectively maximum reference)					
CORINE formula	Name in TNO database	EO.av,min	EO.av,max	EO.av,av	EOmax,min	EOmax,max	EOmax,av
2111	Occup arable cane int INDI	4	89	**33**	5	96	**36**
2111	Occup arable cane int KEN	10	119	**50**	11	128	**54**
2111	Occup arable int NL	39	206	**113**	42	223	**122**
2111	Occup arable int EU	17	100	**53**	18	108	**57**
2112	Occup arable integr EU	14	119	**58**	14	125	**61**
2111	Occup arable linseed int EU	17	100	**53**	18	108	**57**
2113	Occup arable org EU	12	41	**25**	17	56	**34**
2111	Occup arable soybean int USA	24	131	**71**	26	141	**77**
1321	Occup dump benthos gas	0	0	**0**	0	0	**0**
1321	Occup dump benthos oil	0	0	**0**	0	0	**0**
1321	Occup dump benthos EU	31	79	**52**	48	83	**64**
1323	Occup dump build NL	127	169	**148**	128	170	**149**
132	Occup dump NL	127	169	**148**	128	170	**149**
13222	Occup dump radioac h NL	127	169	**148**	128	170	**149**
13221	Occup dump radioac lm NL	127	169	**148**	128	170	**149**
1322	Occup dump radioac NL	127	169	**148**	128	170	**149**
1322	Occup dump w radact EU	56	82	**69**	57	83	**70**
131	Occup extr Ag	61	133	**97**	62	135	**98**
131	Occup extr Al	40	100	**70**	40	101	**71**
131	Occup extr bentonite EU	56	82	**69**	57	83	**70**
131	Occup extr coal h	63	82	**73**	64	82	**73**
131	Occup extr coal h mine	63	82	**73**	64	82	**73**
131	Occup extr coal h pit	63	82	**73**	64	82	**73**
131	Occup extr coal h mine AUS	40	90	**65**	40	91	**65**
131	Occup extr coal h mine USA	79	107	**93**	79	108	**94**
131	Occup extr coal h mine oEU	55	80	**67**	55	81	**68**
131	Occup extr coal h mine sAF	39	110	**75**	40	112	**76**
131	Occup extr coal h mine UCP	56	82	**69**	57	83	**70**
131	Occup extr coal b pit UCP	56	82	**69**	57	83	**70**
131	Occup extr Cr	28	72	**50**	29	73	**51**
131	Occup extr Cu	101	128	**114**	102	129	**115**
131	Occup extr diabase	56	82	**69**	57	83	**70**
131	Occup extr dolomite	56	82	**69**	57	83	**70**
131	Occup extr EU	56	82	**69**	57	83	**70**
131	Occup extr Fe	69	105	**87**	69	106	**88**
131	Occup extr feldspar	56	82	**69**	57	83	**70**
131	Occup extr fluorite	56	82	**69**	57	83	**70**
131	Occup extr gas	42	58	**50**	43	59	**51**
131	Occup extr gypsum	56	82	**69**	57	83	**70**
131	Occup extr kaoline TEMP	47	64	**55**	47	65	**56**
131	Occup extr kaoline TROP	40	110	**75**	40	111	**75**
131	Occup extr limesand NL	127	169	**148**	128	170	**149**
131	Occup extr Mn	54	103	**78**	54	104	**79**
131	Occup extr Ni	16	23	**19**	16	23	**19**
131	Occup extr oil	23	36	**29**	23	36	**30**
131	Occup extr Pb	61	93	**77**	61	94	**78**
131	Occup extr sand EU	56	82	**69**	57	83	**70**

131	Occup extr Sn	80	168	**124**	81	169	**125**
131	Occup extr stonesalt	56	82	**69**	57	83	**70**
131	Occup extr Ti	29	73	**51**	29	74	**52**
131	Occup extr uran	28	45	**36**	28	45	**37**
131	Occup extr uran pit	28	45	**36**	28	45	**37**
131	Occup extr uran mine	28	45	**36**	28	45	**37**
131	Occup extr Zn	41	64	**52**	41	64	**53**
3111	Occup forest broadl int AFR	-	-	**-**	-	-	**-**
3111	Occup forest broadl int ASIA	-	-	**-**	-	-	**-**
3111	Occup forest broadl int EU	8	28	**17**	12	44	**26**
3111	Occup forest broadl int USA	19	26	**23**	29	40	**35**
3121	Occup forest conif int GUS	-9	-14	**-11**	3	5	**4**
3121	Occup forest conif int nEU	-15	-19	**-17**	5	7	**6**
3121	Occup forest conif int nAm	-44	-60	**-52**	16	22	**19**
3121	Occup forest conif int wEU	-82	-101	**-91**	30	37	**34**
31	Occup forest EU	-3	-5	**-4**	26	39	**33**
3101	Occup forest int NL	40	53	**46**	56	75	**66**
121	Occup ind area AFR	26	70	**47**	27	74	**49**
121	Occup ind area ASIA	93	184	**136**	95	193	**142**
121	Occup ind area EU	47	76	**61**	48	80	**63**
1212	Occup ind built cont NL	127	169	**148**	128	170	**149**
1212	Occup ind built brick NL	127	169	**148**	128	170	**149**
1212	Occup ind built concerete NL	127	169	**148**	128	170	**149**
1212	Occup ind built clinker NL	127	169	**148**	128	170	**149**
1211	Occup ind built EU	56	82	**69**	57	83	**70**
1211	Occup ind built	56	82	**69**	57	83	**70**
1211	Occup ind built D	127	169	**148**	128	170	**149**
1211	Occup ind built I	152	194	**173**	153	196	**175**
1211	Occup ind built A	49	68	**58**	50	68	**59**
1211	Occup ind built CH	62	82	**72**	62	83	**73**
1211	Occup ind built NL	127	169	**148**	128	170	**149**
1211	Occup ind built B	127	169	**148**	128	170	**149**
1211	Occup ind built F	164	193	**178**	165	195	**180**
1211	Occup ind built EU	56	82	**69**	57	83	**70**
1211	Occup ind built GUS	17	24	**20**	17	25	**21**
1211	Occup ind built Uconc	56	82	**69**	57	83	**70**
1211	Occup ind built Udiff	56	82	**69**	57	83	**70**
1211	Occup ind built Ucentr	56	82	**69**	57	83	**70**
1211	Occup ind built Uelem	56	82	**69**	57	83	**70**
1211	Occup ind built U dump B	56	82	**69**	57	83	**70**
1211	Occup ind built U dump C	56	82	**69**	57	83	**70**
1211	Occup ind built DWR	56	82	**69**	57	83	**70**
1211	Occup ind built DWR UCP	56	82	**69**	57	83	**70**
1211	Occup ind built SWR	56	82	**69**	57	83	**70**
1211	Occup ind built SWR UCP	56	82	**69**	57	83	**70**
1211	Occup ind built Uconv	56	82	**69**	57	83	**70**
1211	Occup ind built Urecov	56	82	**69**	57	83	**70**
1212	Occup ind veg EU	10	51	**28**	14	68	**37**
32	Occup other nature NL	0	0	**0**	0	0	**0**
2311	Occup pasture, meadow int EU	16	75	**41**	18	86	**47**
1224	Occup rail embankment NL	15	49	**30**	28	88	**55**
1223	Occup rail network EU	52	93	**71**	52	93	**71**
1223	Occup rail network NL	117	191	**152**	117	191	**152**
1222	Occup road embankment NL	5	197	**89**	19	271	**130**
1221	Occup road network EU	52	93	**71**	52	93	**71**

1221	Occup road network NL	117	191	**152**	117	191	**152**
122	Occup road, rail network EU	52	93	**71**	52	93	**71**
15	Occup urb built cont EU	56	82	**69**	57	83	**70**
112	Occup urb built discont EU	15	54	**32**	19	67	**40**
112	Occup urb built discont NL	35	110	**69**	43	137	**85**
141	Occup urb green EU	8	35	**19**	12	54	**30**
141	Occup urb green NL	17	71	**41**	26	111	**64**
142	Occup urb sport NL	62	206	**126**	64	214	**131**
5121	Occup water artificial EU	31	79	**52**	48	83	**64**
5121	Occup water artificial NL	70	163	**112**	109	170	**138**
	(additional names)						
2.1.1.3	Occup arable org NL	28	84	**53**	39	116	**73**
2.1.1.2	Occup arable integr NL	31	245	**125**	32	256	**131**
1.1.3	Occup constr site urban NL	0	0	**0**	18	46	**30**
1.2.5	Occup constr site indus NL	-1	-2	**-2**	19	44	**30**
2.1.1.42	Occup fibre crop hemp NL	61	173	**111**	65	185	**119**
3.1	Occup forest broadl int NL	40	53	**46**	56	75	**66**
1.5	Occup ind built int NL	127	169	**148**	128	170	**149**
2.3.1	Occup meadow int NL	36	154	**88**	41	176	**100**
2.3.1.3	Occup meadow org NL	-10	-35	**-21**	7	25	**15**
2.2.1.3	Occup orchard org NL	37	93	**62**	49	123	**82**
1.2.1.4	Occup road embankment EU	2	96	**42**	8	132	**61**
1.2.2.4	Occup rail embankment EU	7	24	**14**	12	43	**26**
1.5	Occup urb built cont NL	52	209	**122**	55	219	**128**
131	Occup extr clay foreland trad NL	123	171	**147**	128	173	**150**
131	Occup extr clay indyke NL	123	171	**147**	128	173	**150**
131	Occup extr gravel trad NL	107	159	**133**	122	169	**145**
131	Occup extr limestone NL	-42	21	**-10**	69	119	**94**
131	Occup extr sand indyke NL	107	159	**133**	122	169	**145**
		Ecosystem renaturation (average respectively maximum reference)					
		ERav,min	**ERav,max**	**ERav,av**	**ERmax,min**	**ERmax,max**	**ERmax,av**
1321	Renat benthos III-II 50y EU	0	0	**0**	0	0	**0**
131	Renat bentonite III-II 50 EU	25	61	**41**	29	70	**47**
131	Renat coal h mine III-II 50y	28	61	**43**	32	69	**49**
131	Renat coal h pit III-II 50y	28	61	**43**	32	69	**49**
131	Renat coal mine III-II 50 AUS	18	67	**38**	20	76	**44**
131	Renat coal mine III-II 50 USA	35	79	**55**	40	91	**63**
131	Renat coal mine III-II 50 oEU	24	59	**40**	28	68	**46**
131	Renat coal mine III-II 50 sAF	17	82	**44**	20	94	**51**
131	Renat coal mine III-II 50 UCP	25	61	**41**	29	70	**47**
131	Renat coal b pit III-II 50 UCP	25	61	**41**	29	70	**47**
1322	Renat dump w radact III-II 50	25	61	**41**	29	70	**47**
131	Renat extr gas III-II 50y EU	25	61	**41**	29	70	**47**
131	Renat extr sand III-II 50y EU	25	61	**41**	29	70	**47**
1212	Renat ind veg III-II 50y EU	17	42	**28**	22	55	**37**
1211	Renat ind built III-II 50y D	56	125	**88**	65	144	**100**
1211	Renat ind built III-II 50y I	67	144	**102**	77	165	**118**
1211	Renat ind built III-II 50y A	22	50	**35**	25	58	**40**
1211	Renat ind built III-II 50y NL	56	125	**88**	65	144	**100**
1211	Renat ind built III-II 50y B	56	125	**88**	65	144	**100**
1211	Renat ind built III-II 50y F	73	143	**106**	84	164	**121**
1211	Renat ind built III-II 50y EU	25	61	**41**	29	70	**47**

1211	Renat ind built III-II 50 GUS	7	18	**12**	8	21	**14**
1211	Renat ind DWR III-II 50y	25	61	**41**	29	70	**47**
1211	Renat ind DWR III-II 50y UCP	25	61	**41**	29	70	**47**
1211	Renat ind SWR III-II 50y	25	61	**41**	29	70	**47**
1211	Renat ind SWR III-II 50y UCP	25	61	**41**	29	70	**47**
1223	Renat rail III-II 50y EU	25	61	**41**	29	70	**47**
1221	Renat road III-II 50y EU	25	61	**41**	29	70	**47**
1211	Renat uran mine III-II 50y	12	33	**22**	14	38	**25**
1211	Renat uran pit III-II 50y	12	33	**22**	14	38	**25**
1211	Renat Uconc III-II 80000y	25	61	**41**	29	70	**47**
1211	Renat Uconv III-II 50y	25	61	**41**	29	70	**47**
1211	Renat Ucentr III-II 50y	25	61	**41**	29	70	**47**
1211	Renat Udiff III-II 50y	25	61	**41**	29	70	**47**
1211	Renat Uelem III-II 50y	25	61	**41**	29	70	**47**
1322	Renat Udump B III-II 50y	25	61	**41**	29	70	**47**
1322	Renat Udump C III-II 50y	25	61	**41**	29	70	**47**
1211	Renat Urecov III-II 50y	25	61	**41**	29	70	**47**
131	Renat extr sand sed III-II NL	296	762	**529**	731	1159	**944**
131	Renat extr sand bo III-II NL	13	101	**57**	74	129	**102**
131	Renat extr limestone II-I NL	-6219	-3721	**-4970**	-328	1138	**389**
131	Renat extr limestone II-water NL	-36	23	**-6**	64	111	**87**
131	Renat extr gravel river wl III-II NL	-1605	-2019	**-1812**	-107	249	**70**
131	Renat extr gravel bo III-II NL	13	101	**57**	74	129	**102**
131	Renat extr clay wl III-II NL	-339	568	**270**	1333	2260	**1.797**
131	Renat extr clay bo III-II NL	-60	34	**-13**	50	108	**79**

		Ecosystem Transformation		
		ETmin	ETmax	**ETav.**
1321	Trans benthos II-IV EU	0	0	**0**
131	Trans bentonite III-IV EU	39	95	**64**
131	Trans coal mine II-IV	47	101	**72**
131	Trans coal mine III-IV	44	95	**67**
131	Trans coal h pit II-IV	47	101	**72**
131	Trans coal h pit III-IV	44	95	**67**
131	Trans coal mine II-IV AUS	29	111	**64**
131	Trans coal mine II-IV USA	58	132	**92**
131	Trans coal mine II-IV oEU	40	99	**66**
131	Trans coal mine II-IV sAF	29	136	**74**
131	Trans coal mine II-IV UCP	41	102	**68**
131	Trans coal b pit II-IV UCP	41	102	**68**
1322	Trans dump radact II-IV EU	41	102	**68**
131	Trans extr sand III-IV EU	39	95	**64**
311	Trans forest broadl nat-III AFR	26	63	**44**
311	Trans forest broadl nat-III ASIA	91	164	**128**
311	Trans forest broadl nat-III sAM	55	110	**83**
311	Trans forest broadl nat-III TROP	33	90	**62**
3121	Trans forest conif I-II nAm	1,5	3,5	**2,4**
3121	Trans forest conif III-II nEU	-1	-2	**-2**
1212	Trans ind veg II-IV EU	41	102	**68**
1211	Trans ind built II-IV	41	102	**68**
121	Trans ind built III-IV EU	39	95	**64**
1211	Trans ind built II-IV EU	41	102	**68**
1211	Trans ind built II-IV D	94	208	**146**
1211	Trans ind built III-IV D	88	196	**137**

1211	Trans ind built II-IV I	112	240	**171**
1211	Trans ind built III-IV I	106	225	**160**
1211	Trans ind built II-IV A	37	84	**58**
1211	Trans ind built III-IV A	34	78	**54**
1211	Trans ind built II-IV CH	46	101	**71**
1211	Trans ind built II-IV NL	94	208	**146**
1211	Trans ind built III-IV NL	88	196	**137**
1211	Trans ind built II-IV B	94	208	**146**
1211	Trans ind built III-IV B	88	196	**137**
1211	Trans ind built II-IV F	121	239	**176**
1211	Trans ind built III-IV F	114	224	**166**
1211	Trans ind built III-IV EU	39	95	**64**
1211	Trans ind built II-IV GUS	12	30	**20**
1211	Trans ind built III-IV GUS	12	28	**19**
1211	Trans ind built Ucon II-IV	41	102	**68**
1211	Trans ind built Udif II-IV	41	102	**68**
1211	Trans ind built Ucen II-IV	41	102	**68**
1211	Trans ind built Uele II-IV	41	102	**68**
1211	Trans ind built II-IV UCP	41	102	**68**
1223	Trans rail III-IV EU	39	95	**64**
1223	Trans rail II-IV EU	41	102	**68**
1221	Trans road III-IV EU	39	95	**64**
1221	Trans road III-IV NL	88	196	**137**
1221	Trans road II-IV EU	41	102	**68**
131	Trans uran pit II-IV	21	55	**36**
131	Trans uran mine II-IV	21	55	**36**
	(additional names)			
1211	Trans cont urb III-IV NL	20	43	**30**
1211	Trans ind built int III-IV NL	88	196	**137**
1211	Trans ind built int II-IV NL	94	208	**146**
1211	Trans ind built int II-IV EU	41	102	**68**
31	Trans forest III-II NL	-137	-304	**-213**
32	Trans marsh III-II NL	-88	-196	**-137**
1221	Trans road III-IV NL	88	196	**137**
1221	Trans road II-IV NL	94	208	**146**
1221	Trans road III-IV EU	39	95	**64**
1221	Trans road II-IV EU	41	102	**68**
1214	Trans road embankment III-III NL	-18	-40	**-28**
1214	Trans road embankment II-III NL	50	110	**77**
1214	Trans road embankment III-III EU	-8	-20	**-13**
1214	Trans road embankment II-III EU	22	54	**36**
142	Trans sport III-IV NL	29	65	**46**
112	Trans urb built discont III-IV EU	-52	-127	**-85**
141	Trans urb green III-IV NL	-186	-413	**-289**
131	Trans extr clay foreland II-II NL	0	0	**0**
131	Trans extr clay indyke III-III NL	0	0	**0**
131	Trans extr gravel III-water NL	138	87	**113**
131	Trans extr gravel bo III-II NL	103	-29	**37**
131	Trans extr limestone II-water NL	61	3	**32**
131	Trans extr limestone II-I NL	-109	-208	**-159**
131	Trans extr sand III-water NL	138	87	**113**
131	Trans extr sand bo III-II NL	103	-29	**37**

	Ecosystem transformation due to renaturation	ETRmin	ETRmax	**ETRav.**
1321	Trare benthos III-II EU	-61	-148	**-100**
131	Trare bentonite III-II EU	-61	-148	**-100**
131	Trare coal h mine III-II	-69	-147	**-105**
131	Trare coal h pit III-II	-69	-147	**-105**
131	Trare coal mine III-II AUS	-43	-162	**-93**
131	Trare coal mine III-II USA	-85	-193	**-134**
131	Trare coal mine III-II oEU	-59	-144	**-97**
131	Trare coal mine III-II sAF	-42	-199	**-108**
131	Trare coal mine III-II UCP	-61	-148	**-100**
131	Trare coal b pit III-II UCP	-61	-148	**-100**
1322	Trare dump w radact III-II	-61	-148	**-100**
131	Trare extr gas III-II EU	-61	-148	**-100**
131	Trare extr sand III-II EU	-61	-148	**-100**
1211	Trare ind built III-II EU	-61	-148	**-100**
1211	Trare ind built III-II GUS	-18	-44	**-30**
1211	Trare ind built III-II D	-137	-304	**-213**
1211	Trare ind built III-II I	-164	-350	**-250**
1211	Trare ind built III-II A	-53	-122	**-84**
1211	Trare ind built III-II NL	-137	-304	**-213**
1211	Trare ind built III-II B	-137	-304	**-213**
1211	Trare ind built III-II F	-177	-349	**-258**
1211	Trare ind DWR III-II	-61	-148	**-100**
1211	Trare ind DWR III-II UCP	-61	-148	**-100**
1211	Trare ind SWR III-II	-61	-148	**-100**
1211	Trare ind SWR III-II UCP	-61	-148	**-100**
1212	Trare ind veg III-II EU	0	0	**0**
1223	Trare rail III-II EU	-61	-148	**-100**
1221	Trare road III-II EU	-61	-148	**-100**
131	Trare uran pit III-II	-30	-81	**-53**
131	Trare uran mine III-II	-30	-81	**-53**
1211	Trare Uconc III-II	-61	-148	**-100**
1211	Trare Udiff III-II	-61	-148	**-100**
1211	Trare Ucentr III-II	-61	-148	**-100**
1211	Trare Uelem III-II	-61	-148	**-100**
1322	Trare Udump B III-II	-61	-148	**-100**
1322	Trare Udump C III-II	-61	-148	**-100**
1211	Trare Uconv III-II	-61	-148	**-100**
1211	Trare Urecov III-II	-61	-148	**-100**
131	Trare extr sand sed III-II NL	103	24	**63**
131	Trare extr sand bo III-II NL	103	-29	**37**
131	Trare extr clay wl III-II NL	53	-6	**36**
131	Trare extr clay bo III-II NL	53	-51	**16**
131	Trare extr gravel river-wl III-II NL	-253	-295	**-274**
131	Trare extr gravel river-bo III-II NL	103	-29	**37**
131	Trare extr limestone II-water NL	61	3	**32**
131	Trare extr limestone II-I NL	-109	-208	**-159**

Life Support indicator NPP [g/m2.y] (all average values)

	-extraction NL not separately listed here-	**LO (Life support Occupation)**
2111	Occup arable cane int INDI	**250**
2111	Occup arable cane int KEN	**115**
2111	Occup arable int NL	**550**
2111	Occup arable int EU	**380**
2112	Occup arable integr EU	**380**
2111	Occup arable linseed int EU	**380**
2113	Occup arable org EU	**380**
2111	Occup arable soybean int USA	**205**
1321	Occup benthos EU	**250**
1321	Occup dump benthos gas	**250**
1321	Occup dump benthos oil	**250**
1323	Occup dump build NL	**1200**
132	Occup dump NL	**1200**
13222	Occup dump radioac h NL	**1200**
13221	Occup dump radioac lm NL	**1200**
1322	Occup dump radioac NL	**1200**
131	Occup extr Ag	**1102**
131	Occup extr Al	**912**
131	Occup extr coal h	**805**
131	Occup extr Cr	**692**
131	Occup extr Cu	**875**
131	Occup extr diabase	**1030**
131	Occup extr dolomite	**1030**
131	Occup extr EU	**1030**
131	Occup extr Fe	**913**
131	Occup extr feldspar	**1030**
131	Occup extr fluorite	**1030**
131	Occup extr clay foreland NL	**1200**
131	Occup extr clay indyke NL	**1200**
131	Occup extr gas	**803**
131	Occup extr gravel indyke NL	**1175**
131	Occup extr gypsum	**1030**
131	Occup extr kaoline TEMP	**1020**
131	Occup extr kaoline TROP	**1030**
131	Occup extr limesand NL	**1200**
131	Occup extr limestone NL	**600**
131	Occup extr Mn	**948**
131	Occup extr Ni	**681**
131	Occup extr oil	**502**
131	Occup extr Pb	**744**
131	Occup extr sand indyke NL	**1175**
131	Occup extr Sn	**1434**
131	Occup extr stonesalt	**1030**
131	Occup extr Ti	**1002**
131	Occup extr uran	**688**
131	Occup extr Zn	**649**
3111	Occup forest broadl int ASIA	**572**
3111	Occup forest broadl int EU	**0**
3111	Occup forest broadl int USA	**-176**
3121	Occup forest conif int GUS	**54**
3121	Occup forest conif int nEU	**54**

3121	Occup forest conif int nAm	**54**
3121	Occup forest conif int wEU	**471**
31	Occup forest EU	**198**
3101	Occup forest int NL	**338**
3111	Occup forest broadl int NL	**30**
3121	Occup forest conif int NL	**-145**
121	Occup ind area AFR	**611**
121	Occup ind area ASIA	**1540**
121	Occup ind area EU	**990**
1212	Occup ind built cont NL	**1200**
1211	Occup ind built EU	**1030**
1212	Occup ind veg EU	**830**
32	Occup other nature NL	**0**
2311	Occup pasture, meadow int EU	**130**
1224	Occup rail embankment NL	**1000**
1223	Occup rail network EU	**1030**
1223	Occup rail network NL	**1200**
1222	Occup road embankment NL	**-310**
1221	Occup road network EU	**1030**
1221	Occup road network NL	**1200**
122	Occup road, rail network EU	**1030**
15	Occup urb built cont EU	**1030**
112	Occup urb built discont EU	**830**
112	Occup urb built discont NL	**1000**
141	Occup urb green EU	**830**
141	Occup urb green NL	**1000**
142	Occup urb sport NL	**1000**
5121	Occup water artificial EU	**630**
5121	Occup water artificial NL	**800**
	(additional names ETH)	
1321	Occup dump benthos EU	**250**
1322	Occup dump w radact EU	**1030**
131	Occup extr bentonite EU	**1030**
131	Occup extr coal h mine	**805**
131	Occup extr coal h pit	**805**
131	Occup extr coal h mine AUS	**647**
131	Occup extr coal h mine USA	**855**
131	Occup extr coal h mine oEU	**1170**
131	Occup extr coal h mine sAF	**542**
131	Occup extr coal h mine UCP	**1030**
131	Occup extr coal b pit UCP	**1030**
131	Occup extr sand EU	**1030**
131	Occup extr uran pit	**688**
131	Occup extr uran mine	**688**
1211	Occup ind built	**1030**
1211	Occup ind built D	**1200**
1211	Occup ind built I	**1190**
1211	Occup ind built A	**904**
1211	Occup ind built CH	**983**
1211	Occup ind built NL	**1200**
1211	Occup ind built B	**1200**
1211	Occup ind built F	**1236**
1211	Occup ind built EU	**1030**
1211	Occup ind built GUS	**767**

1211	Occup ind built Uconc	**1030**
1211	Occup ind built Udiff	**1030**
1211	Occup ind built Ucentr	**1030**
1211	Occup ind built Uelem	**1030**
1211	Occup ind built U dump B	**1030**
1211	Occup ind built U dump C	**1030**
1211	Occup ind built DWR	**1030**
1211	Occup ind built DWR UCP	**1030**
1211	Occup ind built SWR	**1030**
1211	Occup ind built SWR UCP	**1030**
1211	Occup ind built Uconv	**1030**
1211	Occup ind built Urecov	**1030**
1211	Occup ind built brick NL	**1200**
1211	Occup ind built concrete NL	**1200**
1211	Occup ind built clinker NL	**1200**
1224	Occup road embankment EU	**-480**
1224	Occup rail embankment EU	**830**
		LR (Life support Renaturation)
131	Renat extr sand sed III-II NL	**-730**
131	Renat extr sand bo III-II NL	**635**
131	Renat extr limestone II-I NL	**-620**
131	Renat extr limestone II-water NL	**985**
131	Renat extr gravel river wl III-II NL	**-5066**
131	Renat extr gravel bo III-II NL	**635**
131	Renat extr clay wl III-II NL	**-13182**
	(additional names theoretical renaturation ETH)	
131	Renat coal mine III-II 50 sAF	**-383**
131	Renat coal mine III-II 50 UCP	**105**
131	Renat coal b pit III-II 50 UCP	**105**
1322	Renat dump w radact III-II 50	**-125**
131	Renat extr gas III-II 50y EU	**105**
131	Renat extr sand III-II 50y EU	**105**
1212	Renat ind veg III-II 50y EU	**330**
1211	Renat ind built III-II 50y D	**275**
1211	Renat ind built III-II 50y I	**265**
1211	Renat ind built III-II 50y A	**-21**
1211	Renat ind built III-II 50y NL	**275**
1211	Renat ind built III-II 50y B	**275**
1211	Renat ind built III-II 50y F	**311**
1211	Renat ind built III-II 50y EU	**105**
1211	Renat ind built III-II 50 GUS	**-158**
1211	Renat ind DWR III-II 50y	**-125**
1211	Renat ind DWR III-II 50y UCP	**105**
1211	Renat ind SWR III-II 50y	**-125**
1211	Renat ind SWR III-II 50y UCP	**105**
1223	Renat rail III-II 50y EU	**105**
1221	Renat road III-II 50y EU	**105**
1211	Renat uran mine III-II 50y	**-125**
1211	Renat uran pit III-II 50y	**-125**
1211	Renat Uconc III-II 80000y	**-125**
1211	Renat Uconv III-II 50y	**-125**
1211	Renat Ucentr III-II 50y	**-125**
1211	Renat Udiff III-II 50y	**-125**

1211	Renat Uelem III-II 50y	**-125**
1322	Renat Udump B III-II 50y	**-125**
1322	Renat Udump C III-II 50y	**-125**
1211	Renat Urecov III-II 50y	**-125**

		LT (Life support transformation)
1321	Trans benthos II-IV EU	**1200**
131	Trans bentonite III-IV EU	**650**
1322	Trans dump radact II-IV EU	**1200**
131	Trans extr coal mine II-IV	**1200**
131	Trans extr coal mine III-IV	**650**
131	Trans extr coal h pit II-IV	**1200**
131	Trans extr coal h pit III-IV	**650**
131	Trans extr coal mine II-IV AUS	**1200**
131	Trans extr coal mine II-IV USA	**1200**
131	Trans extr coal mine II-IV oEU	**1200**
131	Trans extr coal mine II-IV sAF	**1200**
131	Trans extr coal mine II-IV UCP	**1200**
131	Trans extr coal b pit II-IV UCP	**1200**
131	Trans extr gypsum II-III EU	**181**
131	Trans extr sand III-IV EU	**650**
311	Trans forest broadl nat-III AFR	**1**
311	Trans forest broadl nat-III ASIA	**930**
311	Trans forest broadl nat-III sAM	**559**
311	Trans forest broadl nat-III TROP	**380**
3121	Trans forest conif I-II nAm	**495**
3121	Trans forest conif II-II nEU	**390**
1211	Trans ind built II-IV	**1200**
121	Trans ind built III-IV EU	**650**
121	Trans ind built II-IV EU	**1200**
1211	Trans ind built II-IV D	**1200**
1211	Trans ind built III-IV D	**650**
1211	Trans ind built II-IV I	**1200**
1211	Trans ind built III-IV I	**650**
1211	Trans ind built II-IV A	**1200**
1211	Trans ind built III-IV A	**650**
1211	Trans ind built II-IV CH	**1200**
1211	Trans ind built II-IV NL	**1200**
1211	Trans ind built III-IV NL	**650**
1211	Trans ind built II-IV B	**1200**
1211	Trans ind built III-IV B	**650**
1211	Trans ind built II-IV F	**1200**
1211	Trans ind built III-IV F	**650**
1211	Trans ind built III-IV EU	**650**
1211	Trans ind built II-IV GUS	**1200**
1211	Trans ind built III-IV GUS	**650**
1211	Trans ind built Ucon II-IV	**1200**
1211	Trans ind built Udif II-IV	**1200**

1211	Trans ind built Ucen II-IV	**1200**
1211	Trans ind built Uele II-IV	**1200**
1211	Trans ind built II-IV UCP	**1200**
1212	Trans ind veg II-IV EU	**1000**
1223	Trans rail III-IV EU	**650**
1223	Trans rail II-IV EU	**1200**
1221	Trans road III-IV EU	**650**
1221	Trans road III-IV NL	**650**
1221	Trans road II-IV EU	**1200**
131	Trans uran pit II-IV	**1200**
131	Trans uran mine II-IV	**1200**
	(additional names)	
1211	Trans cont urb III-IV NL	**650**
112	Trans discont urb III-IV EU	**450**
31	Trans forest III-II NL	**-662**
1211	Trans ind built int II-IV EU	**1312**
1211	Trans ind built int III-IV NL	**650**
1211	Trans ind built int II-IV NL	**1312**
32	Trans marsh III-II NL	**-550**
1221	Trans road III-IV EU	**650**
1221	Trans road II-IV EU	**1312**
1221	Trans road III-IV NL	**650**
1221	Trans road II-IV NL	**1312**
1224	Trans road embankment III-III EU	**-860**
1224	Trans road embankment II-III EU	**-648**
1224	Trans road embankment III-III NL	**-860**
1224	Trans road embankment II-III NL	**-648**
142	Trans sport III-IV NL	**450**
141	Trans urb green II-IV NL	**450**
131	Trans extr clay foreland II-II NL	**0**
131	Trans extr clay indyke III-III NL	**0**
131	Trans extr gravel III-water NL	**600**
131	Trans extr gravel bo III-II NL	**-100**
131	Trans extr limesand III-water NL	**500**
131	Trans extr limestone II-water NL	**-100**
131	Trans extr limestone II-I NL	**-250**
131	Trans extr sand III-water NL	**600**
131	Trans extr sand bo III-II NL	**-100**

LTR (Life support transformation due to renaturation)

1223	Trare rail III-II EU	**-550**
1221	Trare road III-II EU	**-550**
1321	Trare benthos III-II EU	**-550**
131	Trare bentonite III-II EU	**-550**
131	Trare coal h mine III-II	**-550**
131	Trare coal h pit III-II	**-550**
131	Trare coal mine III-II AUS	**-550**
131	Trare coal mine III-II USA	**-550**
131	Trare coal mine III-II oEU	**-550**
131	Trare coal mine III-II sAF	**-550**
131	Trare coal mine III-II UCP	**-550**
131	Trare coal b pit III-II UCP	**-550**
1322	Trare dump w radact III-II	**-550**

131	Trare extr gas III-II EU	**-550**
131	Trare extr sand III-II EU	**-550**
1211	Trare ind built III-II EU	**-550**
1211	Trare ind built III-II GUS	**-550**
1211	Trare ind built III-II D	**-550**
1211	Trare ind built III-II I	**-550**
1211	Trare ind built III-II A	**-550**
1211	Trare ind built III-II NL	**-550**
1211	Trare ind built III-II B	**-550**
1211	Trare ind built III-II F	**-550**
1211	Trare ind DWR III-II	**-550**
1211	Trare ind DWR III-II UCP	**-550**
1211	Trare ind SWR III-II	**-550**
1211	Trare ind SWR III-II UCP	**-550**
1212	Trare ind veg III-II EU	**-550**
131	Trare uran pit III-II	**-550**
131	Trare uran mine III-II	**-550**
1211	Trare Uconc III-II	**-550**
1211	Trare Udiff III-II	**-550**
1211	Trare Ucentr III-II	**-550**
1211	Trare Uelem III-II	**-550**
1322	Trare Udump B III-II	**-550**
1322	Trare Udump C III-II	**-550**
1211	Trare Uconv III-II	**-550**
1211	Trare Urecov III-II	**-550**
131	Trare extr sand sed III-II NL	**-400**
131	Trare extr sand bo III-II NL	**200**
131	Trare extr clay wl III-II NL	**-400**
131	Trare extr gravel river-wl III-II NL	**-400**
131	Trare extr gravel river-bo III-II NL	**200**
131	Trare extr limestone II-water NL	**-100**
131	Trare extr limestone II-I NL	**-250**

Annex 8 Results case outer wall

Below the detailed results of the cases described in the main report are given, as figures with impact scores.

First, the TNO land use biodiversity impact scores for a concrete outer wall (excluding the use phase!) are presented. Two alternatives are distinguished: **no renat**, using data on aggregate extraction without renaturation and **renat**, assuming extraction techniques including renaturation. The results for occupation using an average reference precede those using a maximum reference. Next, the transformation scores are shown. Finally the normalized results are given for both occupation and transformation. The theoretical renaturation for roads, railways, buildings etc. (as recorded in the ETH database [Frischknecht et al., 1996]) is not taken into account in general. We did perform a sensitivity analysis on this, which resulted in slightly (<10%) higher values for renaturation and lower values for transformation due to renaturation when these theoretical renaturations were taken into account.

Also the land use scores according to the Eco-Indicator 99 [Goedkoop & Spriensma, 1999] are shown, using default values based on [Köllner, 1999] for the various groups of land use. For extractions the local EI 99 impact score 0,96 (discontinuous built) is applied; for renaturation to a high quality level the local score 0,10 (discontinuous to green) is chosen as the most appropriate one. What is called transformation in [Goedkoop & Spriensma, 1999] is actually renaturation [Goedkoop, 2002] and expressed as such. Occupation and renaturation could be added up; this is no done for reasons of transparancy.

The case ends with the presentation of the life support impact scores using NPP as an indicator.

Secondly, the same results are shown for a brick outer wall are presented. Thirdly, the results for a wooden outer wall are presented. The legenda for the impact scores and alternatives is given below.

Legenda for all figures and tables:

EO = Ecosystem Occupation	no renat = extraction without renaturation
ER = Ecosystem occupation during Renaturation	renat = extraction with renaturation included
TO = Total Occupation	a = average reference, no renaturation
ET = Ecosystem Transformation	ar = average reference, incl. renaturation
ETR = Ecosystem Transformation due to Renaturation	m = maximum reference, no renaturation
TT = Total Transformation	mr = maximum reference, incl. renaturation
Can = Canada	Sca = Scandinavia

LOnpp = Life support Occupation using NPP as indicator
LRnpp = Life support occupation during Renaturation using NPP as indicator
LTnpp = Life support Transformation using NPP as indicator
LTRnpp = Life support Transformation due to Renaturation using NPP as indicator

The **uncertainty range** in the biodiversity scores presents only the *impact assessment uncertainty*. It is determined by applying the lower versus the upper boundary values of each factor in the impact assessment scores. It is a gross overestimation of the IA data uncertainly, as no stochastic quenching of the scores is allowed.

The uncertainty range in the life support scores on the contrary, depicts only the *inventory data uncertainty*. This is determined by applying the lower and upper boundary values for each land use intervention in the foreground processes (extraction and production of the building materials). The uncertailty in the background processes could not be assessed as easily, and would result in a similar overestimation due to lack of stochastic quenching. In the foregrond prcesses there is little overlap between the alternative processes in each case, allowing for a fairly acceptable comparison of the alternatives including their uncertainty ranges. Nevertheless, also the uncertainty ranges determined in this manner will result in an overestimation due to lack of stochastic quenching in the foreground processes.

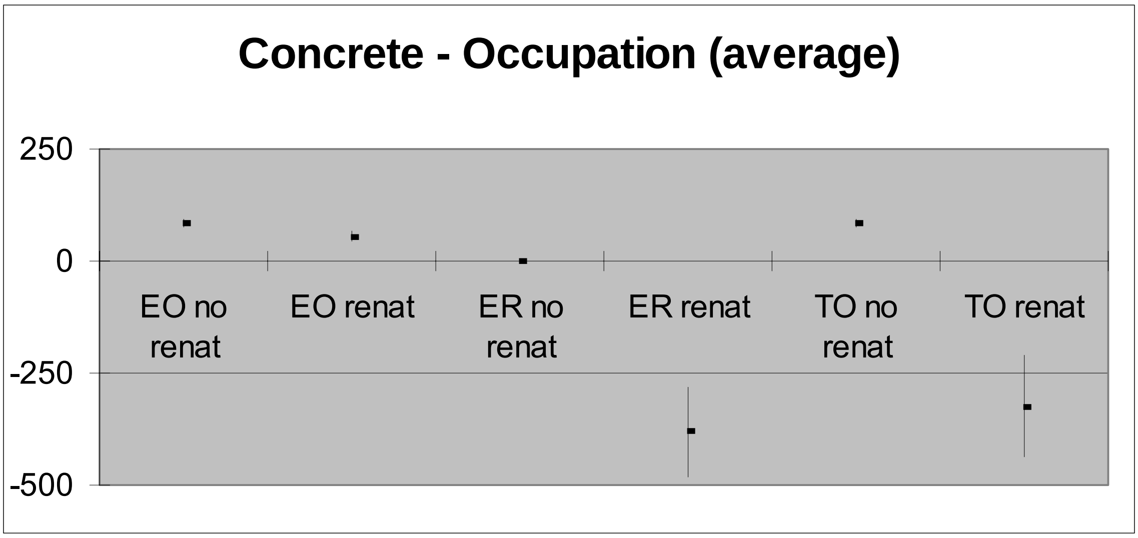
The total data uncertainty can not be determined by adding up (quadratically) the inventory and impact assessment data uncertainty, due to the overestimation in the impact assessment data. This uncertainty assessment is very performed very crudely because of the project focus. A full data uncertainty assessment would require additional software, procedures and effort (see [Ciroth, 2002] for an overview of uncertainty assessment methods for LCA).

8.1 Case concrete outer wall

The traditional concrete wall case is where sand and gravel are extracted from indyke pits, resulting in an artificial lake with recreation or nature development on the borders. For the modern case extraction in riverbeds is assumed, including full renaturation by a coordinated long-term process. This is in accordance to some recent real-life cases in the Netherlands (see annex 4 from Stichting Ark). In the modern case no occupation is included, other than the occupation during renaturation. Renaturation is then stated to start already during the extraction process, according to statements from the Dutch aggregate extraction industry (see annex 3). The occupation results for the biodiversity indicator are shown in figure A8.1.

The biodiversity impacts from occupation during the extraction process (EO) is larger for the traditional case, as can be expected (no occupation without renaturation in the modern case). Renaturation only occurs via backgroundprocesses (mainly energy production and transport). In the modern case only the occupation due to background processes is present. The renaturation score is negative and dominant (EO and ER can be summed up to TO). It is clear from this case that the influence of the way an extraction takes place is large.

Figure A8.1 TNO biodiversity occupation impact scores [m^2.y/FU] using an average reference.



The contribution of major process types to the various scores is given in table A8.1. Energy production (including all underlaying industrial processes) according to the adapted ETH database [Frischknecht et al., 1996] gives large contributions to the biodiversity occupation scores, except when renaturation is performed. Renaturation during aggregate extraction clearly influences the total occupation score, in spite of the large contribution of energy to traditional cases. Industrial production is also an important contributor; part of it is due to the foreground processes (clinker and cement production), and part of it is due to background processes of the ETH database (energy production). These processes have no significant contributions to renaturation.

Table A8.1 Process contribution to occupation in an average reference system (concrete)

Category	EO no renat	EO renat	ER no renat	ER renat
Energy (extraction&production)	44%	47%	100%	0%
Aggregrate extraction	22%	0%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	28%	44%	0%	0%
Transport	5%	9%	0%	0%

Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

In figure A8.2 the results are given when a maximum reference state is chosen. The pattern of the above impact scores for occupation using the 2 reference choices show similarities, as well in the comparison between the alternatives, as in terms of the process contribution per alternative (table A8.2).

However, an important difference is seen when figures A8.1 and A8.2 are compared: the renaturation score (ER) for mordern extractions is higher when the maximum reference is used, and lower when the average reference is used. This is because any renaturation remains an occupation with a significant land use impact when the maximum regional species density is used to compare with. When an average species density is used to compare with, renaturation easily improves against this average, leading to a negative impact. By applying both reference states, it becomes clear how important this methodological choice is, and how its underlaying value statements can influence the results of a case study.

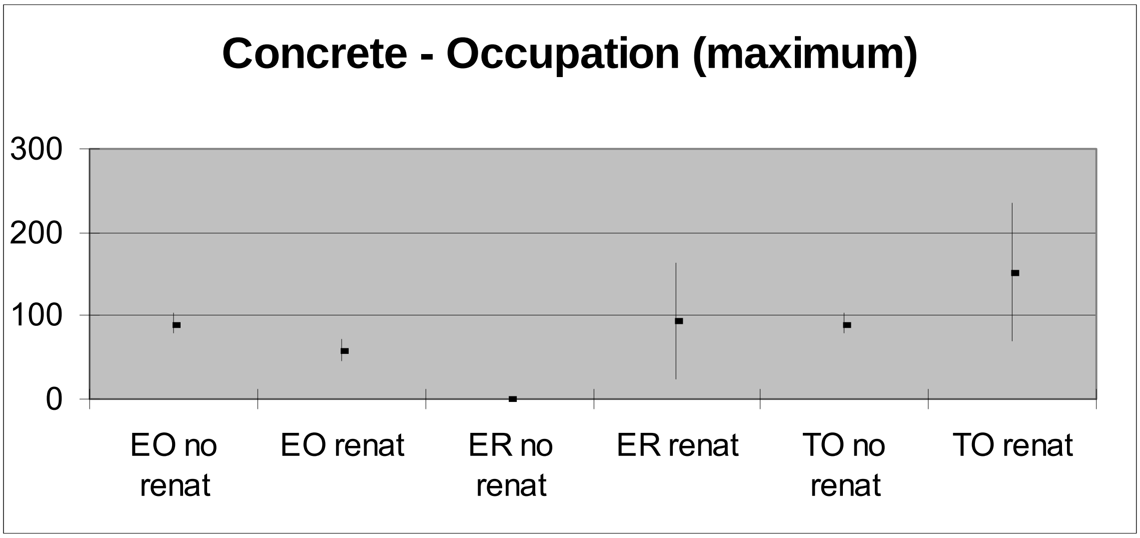


Figure A8.2 TNO biodiversity occupation impact scores [m^2.y/FU] using a maximum reference.

Table A8.2 Process contribution to occupation in a maximum reference system.

Category	EO no renat	EO renat	ER no renat	ER renat
Energy (extraction&production)	42%	46%	100%	0%
Aggregrate extraction	24%	0%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	27%	43%	0%	0%
Transport	6%	11%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

In figure A8. 3 below the results for the biodiversity transformation impacts are depicted. In transformation, the initial state is a kind of reference. Statistical data from Floron has been used to determine the average initial state for each type of aggregate extraction in the Netherlands (see annex 5). The picture is similar to the scores for occupation using an average reference, and shows again the sensitivity of the assessment method. Renaturation results in an environmental improvement (a negative score), and the overall results imply a net increase in biodiversity value when renaturation is applied for concrete resources.

In table A8.3 the contribution to the transformation scores of this case are shown. The contributions are similar as those for occupation. Only for transformation due to renaturation (ETR) there is no significant contribution of the energy processes.

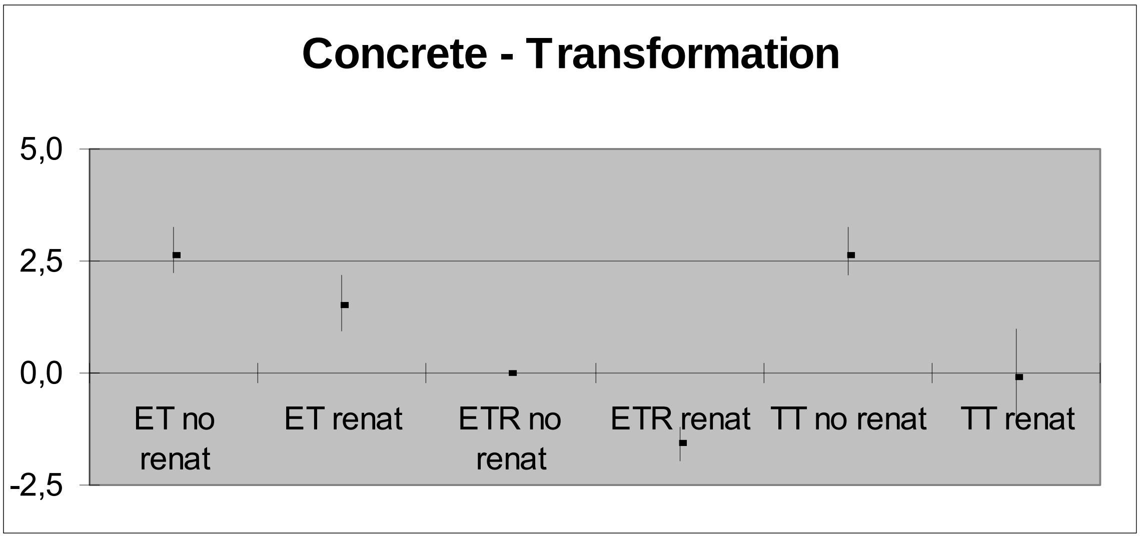


Figure A8.3 TNO biodiversity transformation impact scores [m^2/FU] using the average initial state values according to Floron (see annex 5).

Table A8.3 Process contribution to transformation (concrete)

Category	ET no renat	ET renat	ETR no renat	ETR renat
Energy (extraction&production)	43%	50%	100%	0%
Aggregrate extraction	31%	1%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	27%	49%	0%	0%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

In figure A8.4 a summary of the various total impact scores are shown, as normalisation results[25]. The lower land use impacts when applying modern extraction processes with renaturation are clear. Only when using a maximum reference the renaturation is not regarded an improvement but just a lower impact, resulting in less relevant differences between extraction with and without renaturation.

When this whole picture is viewed, it is clear how important the transformation impacts are for the interpretation of the case results. Using different reference states for the occupation may lead to conflcting interpretations of the occupation results, depending on value choices. When the transformation impacts are included, the expected improvement due to renaturation is clear at once.

[25] Normalisation scores are determined by dividing the case impact score by its score on a national level.

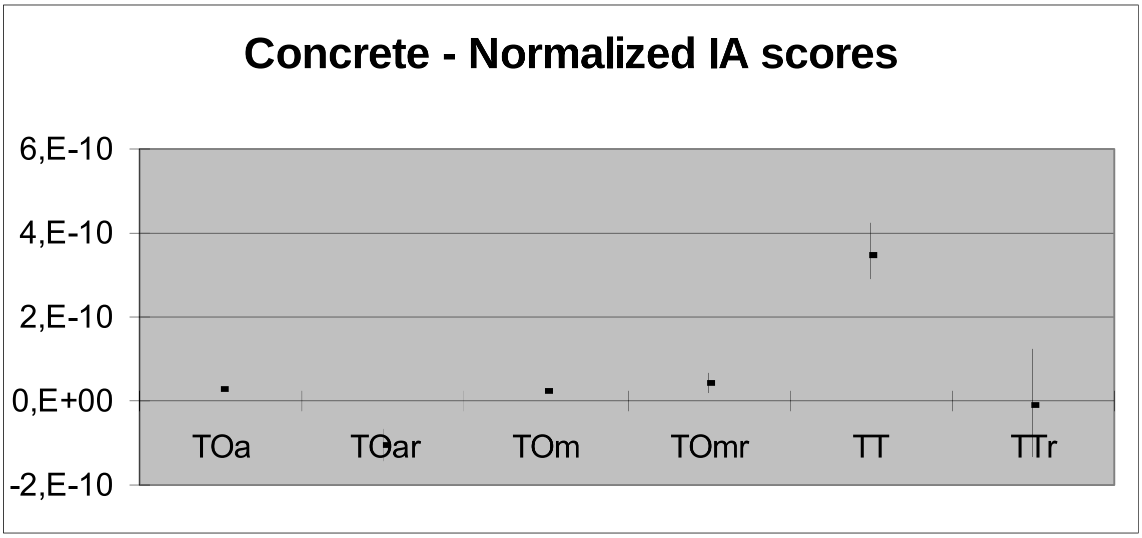


Figure A8.4 Normalisation results for biodiversity occupation (average and maximum reference) and transformation.

When applying the Eco-Indicator 99 land use scores (multiplying the appropriate impact scores from table 5.8 in [Goedkoop& Spriensma, 1999] with the extraction or renaturation time for occupation, and applying their appropriate conversion effect scores, see chapter 4 in the main report), the following results appear:

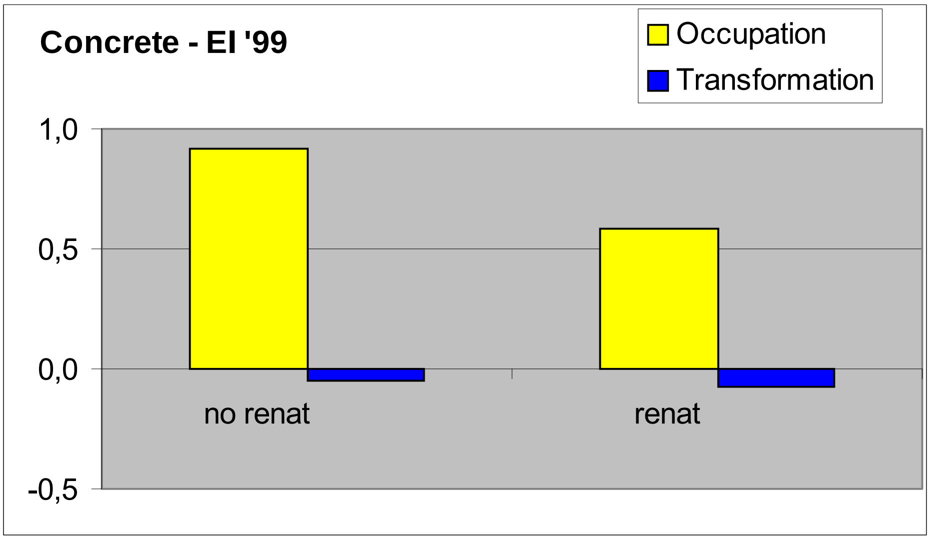


Figure A8.5 Eco-Indicator 99 land use impact assessment scores (occupation and transformation).

Table A8.4 Process contribution to Eco-Indicator 99 land use scores (concrete)

Category	Occ no renat	Occ renat	Tra no renat	Tra renat
Energy (extraction&production)	60%	61%	62%	46%
Aggregrate extraction	17%	0%	32%	49%
Wood extraction	0%	0%	0%	0%
Industrial production	10%	16%	6%	5%
Transport	14%	23%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

The improvement in occupation according to the EI 99 is not as pronounced as when using the TNO average reference (see the negative TO values in figure A8.1)), but it is a clear improvement, in contrast to using the maximum reference (figure A8.2). In fact, the reference used in the EI 99 (broadleaved forest) is in between the ones used by TNO, which explains these results. For transformation, even the traditional case gives a negative impact score, because the contribution of the energy production is higher here (see table A8.4), and negative (including the renaturation process for the background processes). This negative value arises from the interpretation of the ETH land use data as conversted to the TNO format in terms of rougher categories according to [Goedkoop&Spriensma, 1999].

The procedure for dealing with transformation according to the EI 99 report is applied: multiplying the delta impact score (initial state – final state) with the renaturation time to arrive at scores with unit m^2.y, which can be added up with the occupation scores to total land use scores. In fact, this procedure results in renaturation scores [Goedkoop, 2002] and is thus expressed as such.

The fact that the EI 99 has no regional differentiation does not show up in this case, as the main differences are within the Netherlands.

We performed a sensitivity analysis on the extent to which renaturation is attributed to the extraction process. In the above cases, the whole renaturation process (sometimes lasting over 200 years) is attributed to the extraction process. One could also argue (as expressed in sections 3.2.3 and 4.2.1) of the main report) that only part of the renaturation process should be attributed to the extraction process. In this case, the extent to which the renaturation process should be included is unclear. An economic depreciation time could then be applied, as proposed and applied in the EVR method [Vogtländer, 2001]. Here we apply a 30 years depreciation. As the modern extraction process is accompanied with renaturation from the start, 30 years amounts to just after the end of the extraction period. See table 4.3 in section 4.2.1 for the applied case data. The results of this alternative attribution is shown in figure A8.6 below.

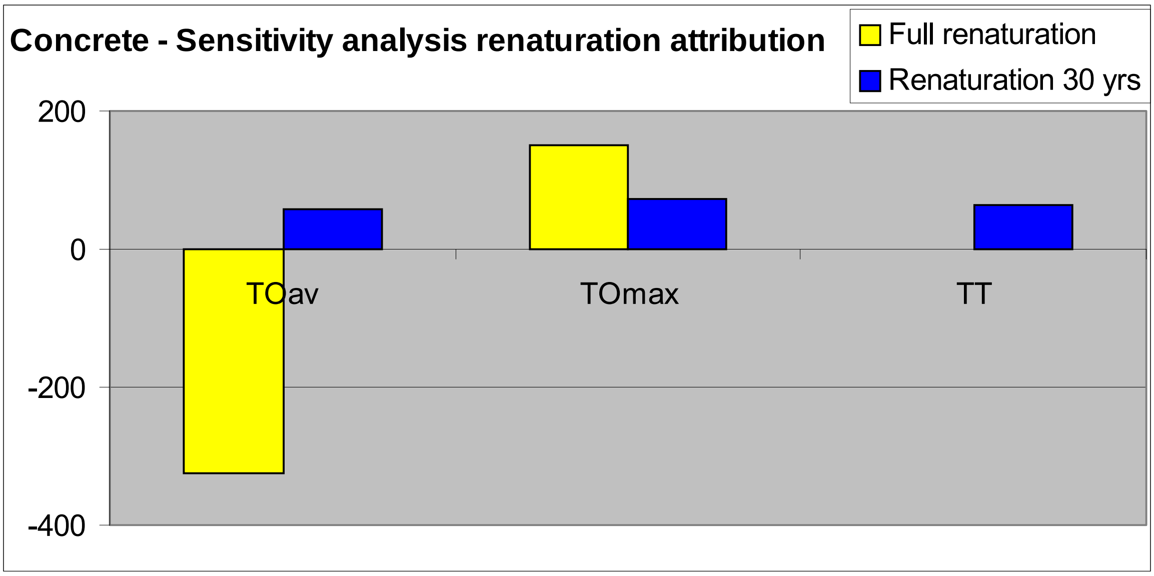


Figure A8.6 Total biodiversity occupation and transformation results using different attribution methods for renaturation.

The difference between full renaturation attribution to the extraction process and attributing only the extraction period is largest for occupation with the average reference and for transformation. Impact scores become much higher when the renaturation process is not or hardly included. When using a maximum reference the whole renaturation period is expressed as an impact to nature, so when a shorter period is attributed, the impact becomes less. When accepting this standpoint, these results are logical. The implication of this sensitivity analysis is that the attribution of the renaturation process is very important for case scores.

Finally, the results for the life support indicator NPP are shown, in figures A8.7 and A8.8 (normalized). The contributions to occupation and transformation are given in tables A8.5 and A8.6, respectively.

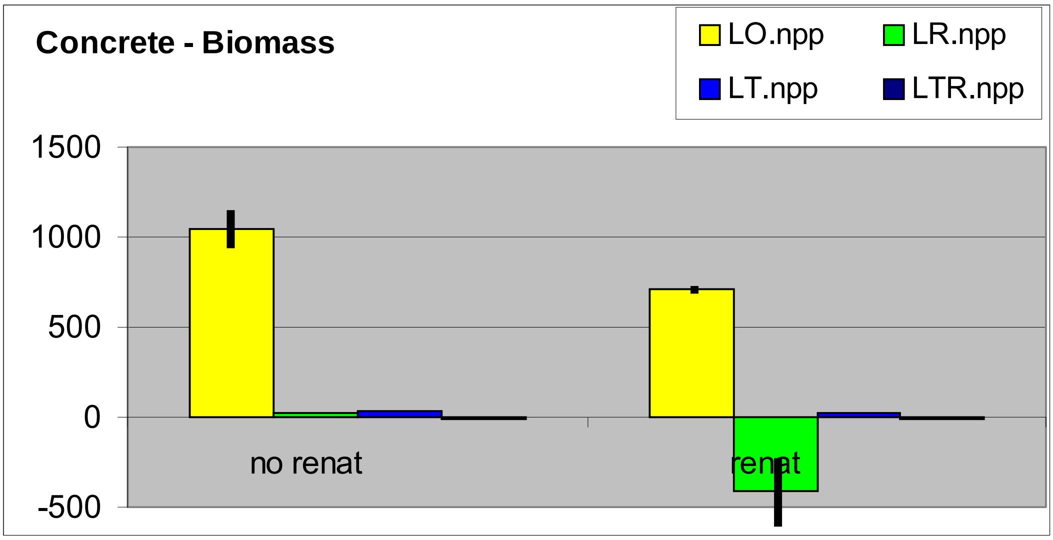


Figure A8.7 TNO life support impact scores using the NPP indicator. Units are [m^2.y times gC/m^2.y] for occupation, and [m^2 times gC/m^2.y] for transformation

For occupation, only an average reference is applied due to data restrictions. The pattern is comparable to that of biodiversity (figure A8.1), although the LR scores are less dominant than the ER scores. This is because the life support biomass values during renaturation are not so far above the average reference in the Netherlands, resulting in a less negative value than with the biodiversity reference.

The approximate uncertainty range due to the inventory of the foreground processes (see the vertical bars in figure A8.7 and the explanatory text in the introduction to annex 8) is clearly lower than the uncertainty in the (biodiversity)[26] impact scores, except for the renaturation process. The area extracted per tonne aggregate is the main cause for this uncertainty.

Table A8.5 Process contribution to life support occupation impacts using an average reference

Category	LO no renat	LO renat	LR no renat	LR renat
Energy (extraction&production)	48%	46%	100%	3%
Aggregrate extraction	17%	0%	0%	97%
Wood extraction	0%	0%	0%	0%
Industrial production	23%	35%	0%	0%
Transport	12%	18%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

Table A8.6 Process contribution to life support transformation impacts (concrete)

Category	LT no renat	LT renat	LTR no renat	LTR renat
Energy (extraction&production)	77%	82%	100%	68%
Aggregrate extraction	11%	0%	0%	32%
Wood extraction	0%	0%	0%	0%
Industrial production	13%	18%	0%	0%

[26] It can be expected that the uncertainty for biomass IA is not lower than that of biodiversity.

Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

The normalised scores for NPP (figure A8.8) give a different picture, because the contribution of the case to the total Dutch scores for life support impacts is much higher for transformation than for occupation, and because in A8.7 both the transformation and occupation scores were depicted on the same scale, although the units differed. In A8.8 the units are equal (contributions to the national total). This gives a higher emphasis to the transformation scores. The trend is of course the same. The relative inventory uncertainty ranges are also the same.

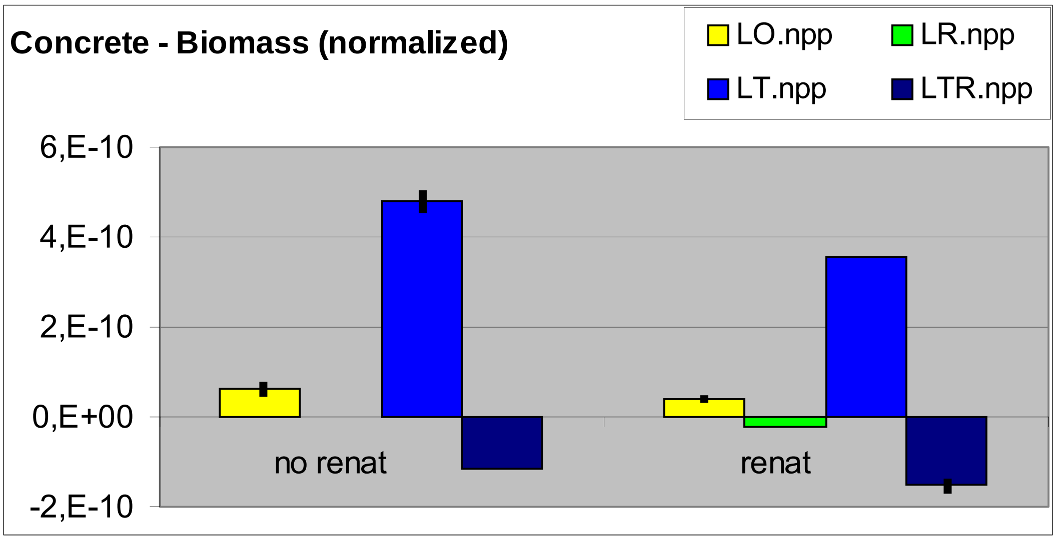


Figure A8.8 **Normalisation results for life support occupation (average reference) and transformation.**

8.2 Case brick outer wall

For the brick outer wall, similar figures are shown for the 6 different biodiversity and 4 life support impact scores. The results are similar too (except for the transformation results), as are the explanations.

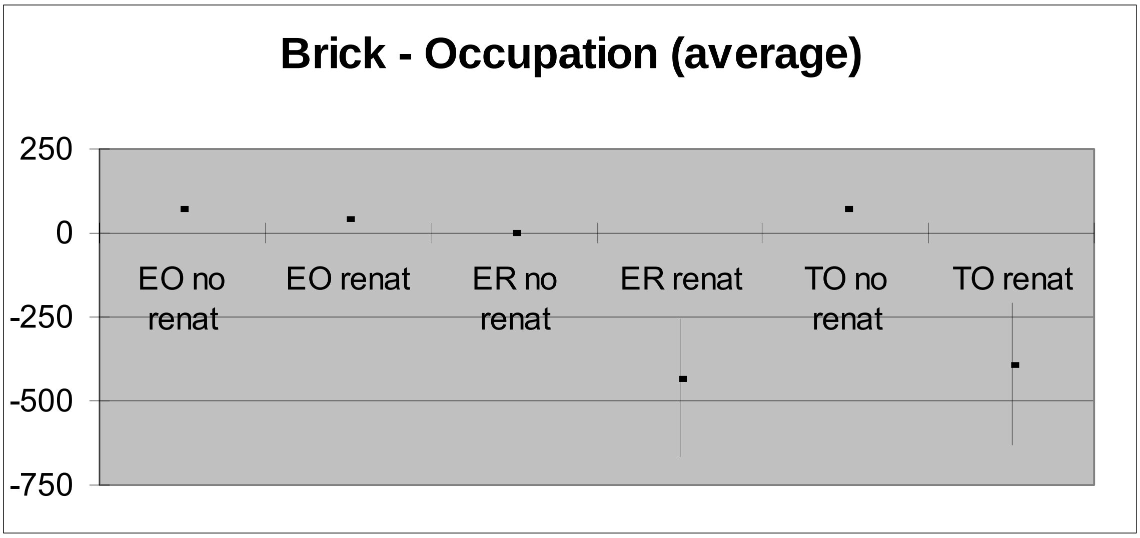


Figure A8.9 TNO occupation impact scores [m^2.y/FU] using an average reference.

Table A8.7 Process contribution to occupation in an average reference system.

Category	EO no renat	EO renat	ER no renat	ER renat
Energy (extraction&production)	26%	35%	100%	0%
Aggregrate extraction	38%	0%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	37%	65%	0%	0%
Transport	0%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

The fact that using a maximum reference implies an impact (due to remaining below the national maximum) even when renaturation takes place, is even more clear in this case. The time and area used for the renaturation process is decisive when a maximum reference is applied. And any deviation from this maximum quality level is considered an impact. leading to a large impact due to renaturation.

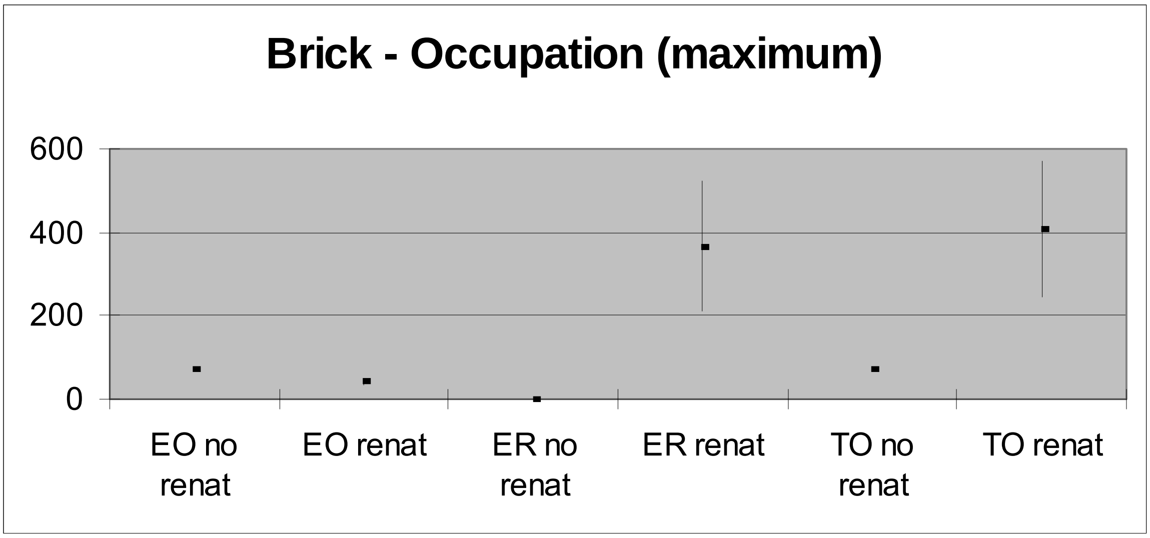


Figure A8.10 TNO occupation impact scores [m^2.y/FU] using a maximum reference.

Table A8.8 Process contribution to occupation in a maximum reference system.

Category	EO no renat	EO renat	ER no renat	ER renat
Energy (extraction&production)	25%	35%	100%	0%
Aggregrate extraction	39%	0%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	36%	65%	0%	0%
Transport	0%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

The transformation impacts of the brick case (figure A8.11) are not clearly negative when renaturation occurs, in contrast to the concrete case. This results from a relatively high initial state (36 species/0.01 ha) and not too high final state (25 to 38 species/0.01 ha), in contrast to the 75 to 100 species/0.01 ha for modern gravel extraction in riverbeds (case concrete). There thus does not seem to be a significant improvement in biodiversity against the traditional case, where no net transformation results, as the situation before the extraction is returned (and most transformation contributions come from background processes).

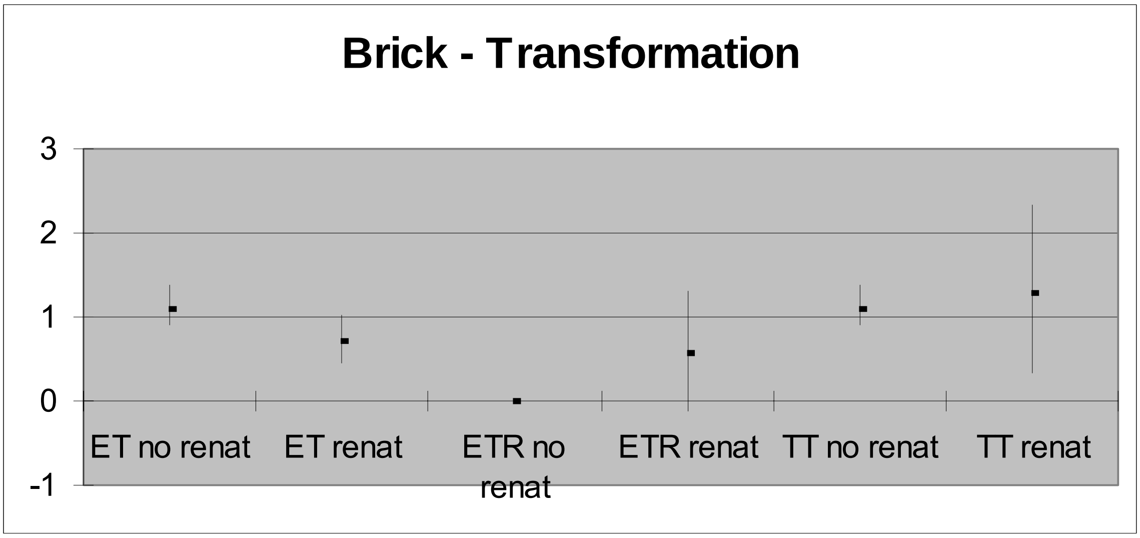


Figure A8.11 TNO transformation impact scores [m²/FU] using the average initial state values according to Floron (see annex 5).

Table A8.9 Process contribution to transformation.

Category	ET no renat	ET renat	ETR no renat	ETR renat
Energy (extraction&production)	41%	46%	100%	0%
Aggregrate extraction	27%	3%	0%	100%
Wood extraction	0%	0%	0%	0%
Industrial production	32%	52%	0%	0%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

In the normalisation figure (A8.12) a summary is given of the above results.

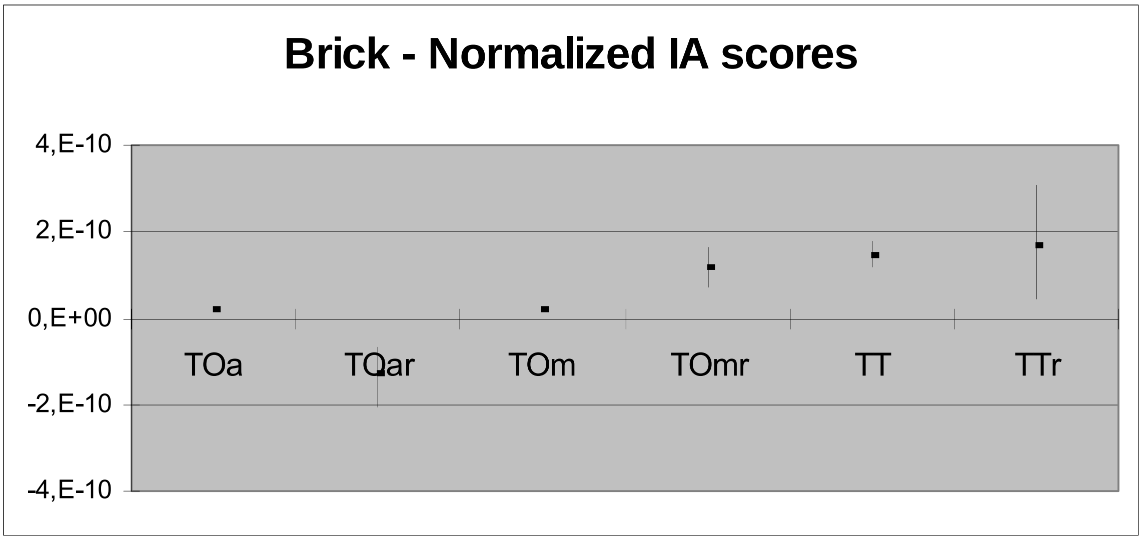


Figure A8.12 Normalisation results for occupation (average and maximum reference) and transformation.

Also for this case, the Eco-Indicator land use scores have been calculated (see figure A8.13). Here, improvement is evident when renaturation is included, for occupation as well as for transformation. The improvement in occupation is similar to when using the TNO method with average reference (see TOav in figure A8.12), although the occupation score does not become negative here. The latter is due to the fact that the renaturation impacts are here included via the transformation impact scores and modern occupation scores result completely from energy production (see the contribution table table A8.8). This also explains why there is a clear improvement in transformation when including renaturation. This is because the transition from extensive meadow to nature is simply classified as positive to the environment (negtive score), and multiplied with a very long renaturation time. The results however become incomparable with those of the TNO method due to this procedure[27].

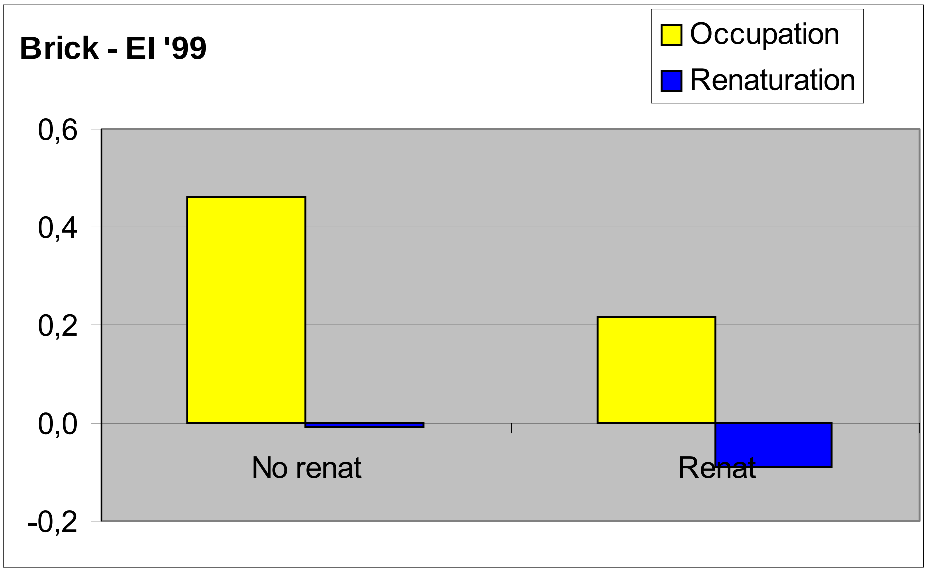


Figure A8.13 Eco-Indicator 99 land use impact assessment scores (occupation and transformation).

[27] The procedure to multiply transformation scores with the renaturation time is methodologically considered inconsistent here, as transformation is thus interpreted as occupation during renaturation.

Table A8.8 Process contribution to Eco-Indicator 99 scores.

Category	Occ no renat	Occ renat	Tra no renat	Tra renat
Energy (extraction&production)	58%	94%	50%	9%
Aggregrate extraction	39%	0%	47%	89%
Wood extraction	0%	0%	0%	0%
Industrial production	2%	5%	3%	1%
Transport	0%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

As for the concrete case, a sensitivity analysis has been performed to assess the difference when not all the renaturation period has been attributed to the extraction process. See figure A8.12. Here, the pattern is the same as for the concrete case. Attributing much less of the renaturation period implies a higher score for transformation (less or no improvement) and for occupation with an average reference. It shows an improvement when renaturation is still considered an impact to nature (when using a maximum reference).

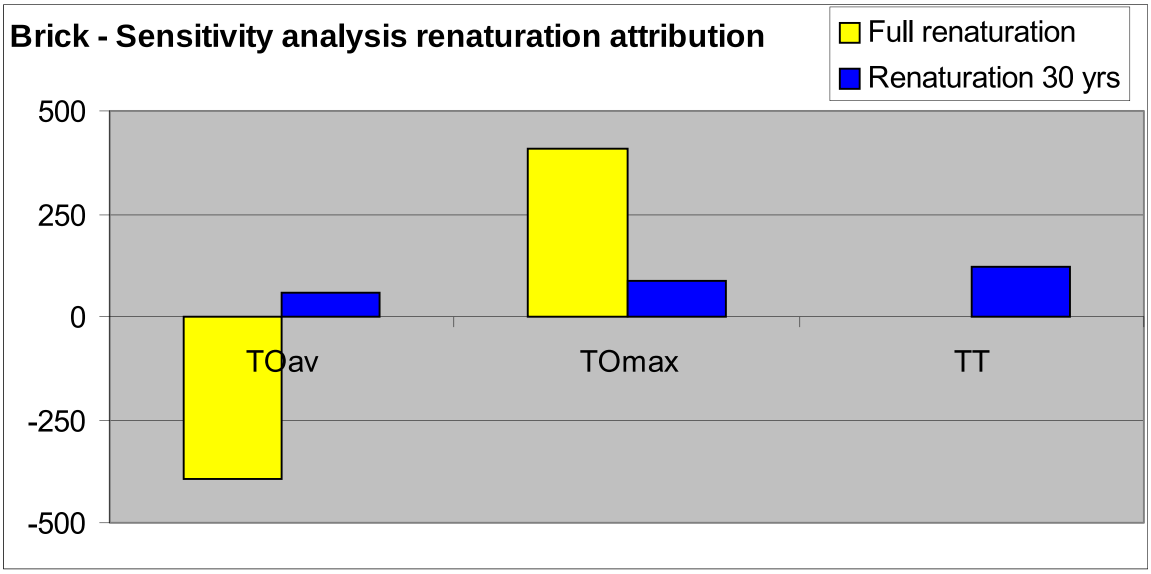


Figure A8.14 Total occupation and transformation results using different attribution methods for renaturation.

The results when using the life support indicator NPP (biomass productivity) are given in figures A8.15 and A8.16 (normalised scores). The gross contributions are given in tables A8.9a and A8.9b.

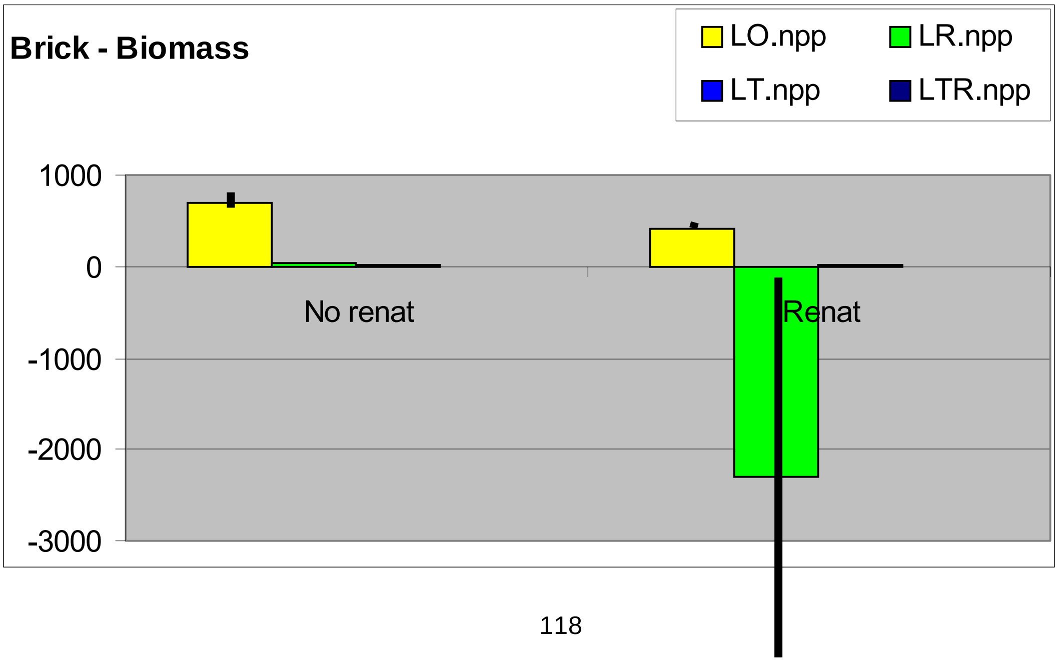


Figure A8.15 TNO life support impact scores using the NPP indicator. Units are [m^2.y times gC/m^2.y] for occupation, and [m^2 times gC/m^2.y] for transformation

As with the concrete case, the uncertainty is largest for the renaturation, where there is a large uncertainty in the area necessary per tonne of clay extracted. In fact, this uncertainty is estimated using an uncertainty rule based on the number of cases with data compared to the total amount of clay extracted in the Netherlands. If much more cases would be examined, or a few representative cases would be taken, this uncertainty would vanish. Presently, only one case is included (see annex 3).

Table A8.9a Process contribution to life support occupation impacts (brick)

Category	LO no renat	LO renat	LR no renat	LR renat
Energy (extraction&production)	36%	47%	100%	1%
Aggregrate extraction	32%	0%	0%	99%
Wood extraction	0%	0%	0%	0%
Industrial production	31%	53%	0%	0%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

Table A8.9b Process contribution to life support transformation impacts (brick)

Category	LT no renat	LT renat	LTR no renat	LTR renat
Energy (extraction&production)	79%	84%	100%	25%
Aggregrate extraction	9%	1%	0%	75%
Wood extraction	0%	0%	0%	0%
Industrial production	12%	14%	0%	0%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

Here, also the same pattern occurs as with the concrete case. The life support improvement in the modern case is clear, for occupation from figure A8.15 and for transformation from A8.16, where all transformation scores are more pronounced due to the larger contribution of transformation to the Dutch normalisation scores.

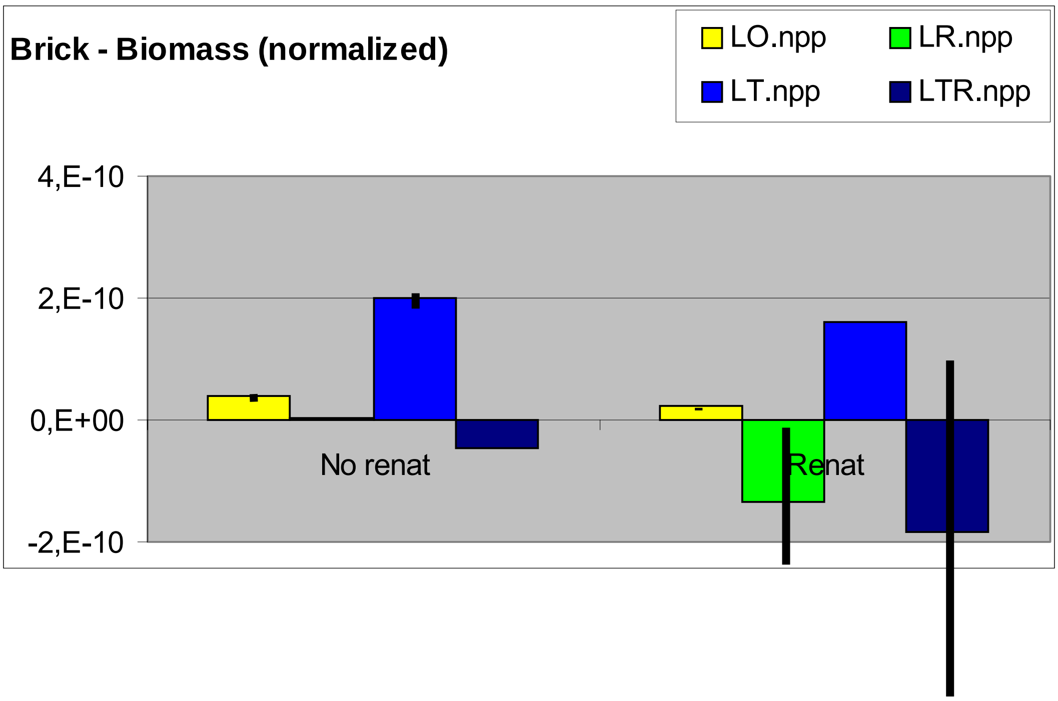


Figure A8.16 Normalisation results for life support occupation (average reference) and transformation

Tentative comparison between concrete and brick outer walls

Although a comparison of these cases was not our aim, it may be illustrative to discuss the differences shortly. Attending readers may have noticed that the TNO land use occupation impact scores for the brick case are about the same as for the concrete case. The total TNO transformation impact scores are slightly lower (on average even negative) for the concrete case. Due to the large uncertainty range in the possible situation after renaturation, differences between the two cases seem insignificant, in spite of the difference in total material use (for this m^2 concrete outer wall 245 kg material was used, whereas for the m^2 of brick outer wall 203 kg material was used). The transformation impacts scores due to renaturation are very different between the different aggregates. Gravel and limestone have rather large negative impact scores due to the renaturation, whereas restauration on clay soil does not lead to very spectacular plant species diversity until after many years, leading to impacts comparable to those before the extraction. Limestone is however used for mortar in bricks as well as in concrete. Different aspects therefore lead to a rather complex picture on the interpretation of the difference between the brick and the concrete case, which is beyond the scope of this project to pursue. Nevertheless, it is illustrated by these cases that when detailed input information is given for case studies, a nuanced picture of the impacts can be produced using the TNO land use impact assessment method, with as main limitation the uncertainties in the impact assessment factors. Next to these limitations, there are the general methodological limitations mentioned in the main report, such as not including differences in landscape impacts, mountain resource loss and desiccation impacts.

8.3 Case wood

For the wooded outer wall alternatives using wood from Canada versus wood from Scandinavia for just over half of the total wood used are chosen (the rest comes from Scandinavia for both alternatives, according to [Kortman et al., 2000]). The cases are therefore labelled Can and Sca, respectively. See the results below (figures A8.17 to A8.22). What becomes clear from figures A8.17 and A8.18 is that the occupation scores for the cases with all Scandinavian wood are much lower (for EOmax, or less negative for EOav) than those for Canadian wood. This is due to the fact that about twice as much land is occupied for the more extensive Canadian forestry, and because ecosystem level factors are higher for Canada than for Scandinavia (the local impact scores are the same for northern Europe and northern America).

What may be less obvious is the very large occupation scores compared to those of the concrete and brick cases. This is due to the large area used for forestry compared to most abiotic extractions. The fact that forests also bear many natural functions (which is a positive aspect) is not apparent from the results, because we compare the functions in a commercial forest to those of the natural forest. For biodiversity there is no improvement, because commercial forests do not exceed the average biodiversity values per unit area, and for NPP the reference is dominated by (natural and commercial) forests anyway. Because the yearly increase in biomass production is lower for production forests than for natural forests in boreal regions (see [Oak Ridge, 2001] and [Lindeijer et al., 1998]), the impact score remains positive (bad for the environment), and very large so due to the high intervention values for occupation. For coniferous wood from central Europe the NPP impact score would have been negative, by the way.

Renaturation scores are zero for wood, as the forest statistics used [FAO, 2001] include regrowth already in the country average values applied. The ER values are therefore determined by the ETH data (see table A8.10).

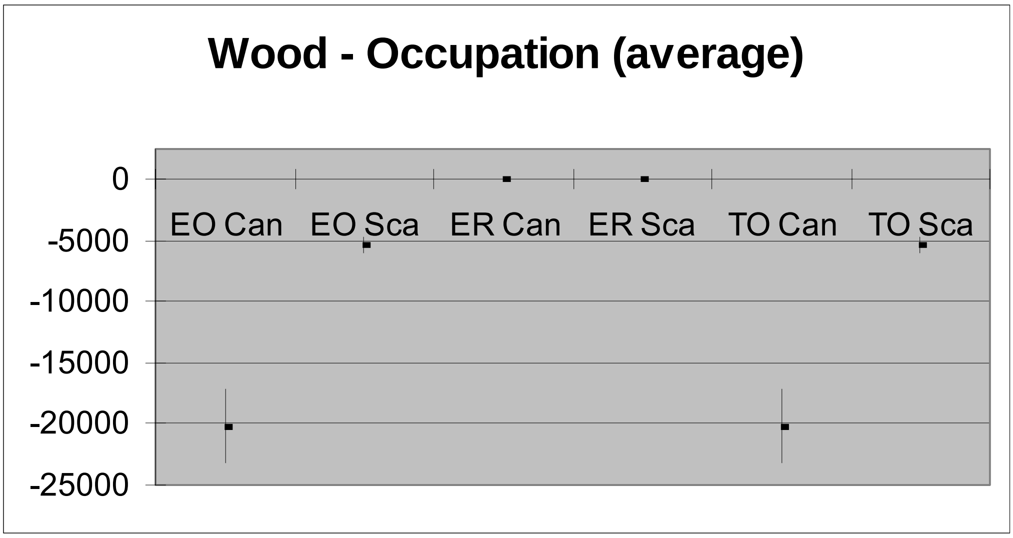


Figure A8.17 TNO occupation impact scores [m^2.y/FU] using an average reference.

Table A8.10 Process contribution to occupation in an average reference system.

Category	EO Can	EO Sca	ER Can	ER Sca
Energy (extraction&production)	0%	2%	100%	100%
Aggregrate extraction	0%	0%	0%	0%
Wood extraction	100%	97%	0%	0%
Industrial production	0%	0%	0%	0%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

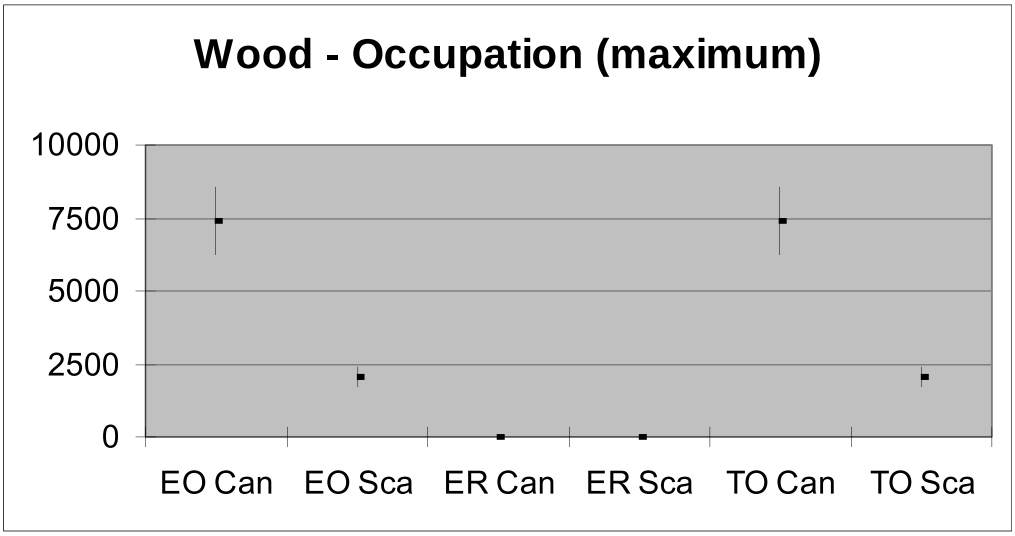


Figure A8.18 TNO occupation impact scores [m^2.y/FU] using a maximum reference.

Table A8.10 Process contribution to occupation in a maximum reference system.

Category	EO Can	EO Sca	ER Can	ER Sca
Energy (extraction&production)	0%	5%	100%	100%
Aggregrate extraction	0%	0%	0%	0%

Wood extraction	99%	93%	0%	0%
Industrial production	0%	1%	0%	0%
Transport	0%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

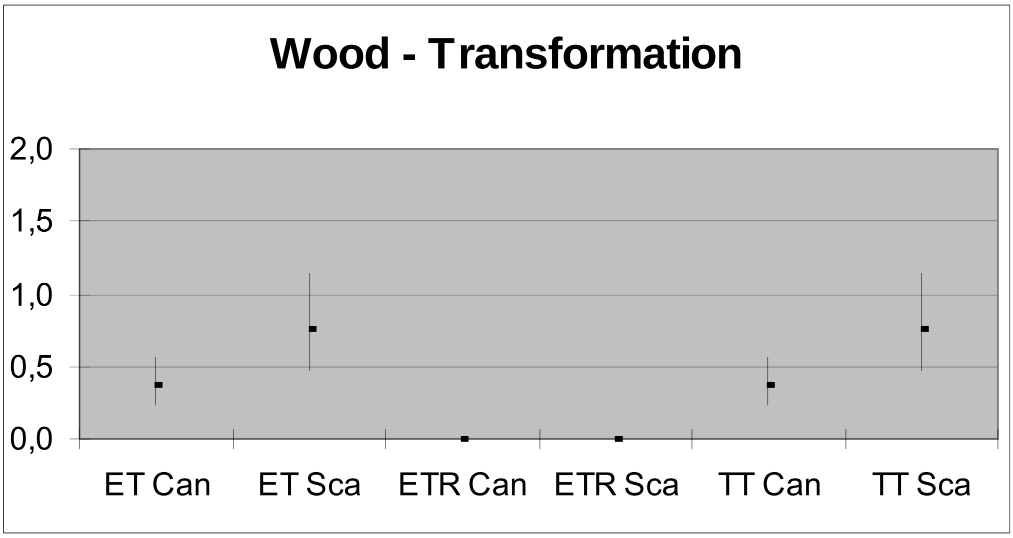


Figure A8.19 TNO transformation impact scores [m^2/FU] using the average initial state values according to [Barthlott, 1997].

For transformation, the intervention values are not much higher than those of aggregate cases, leading to comparable impact scores too. The rather large negative renaturation scores for transformation are due to the energy system (ETH), as can be seen from table A8.11. These background process scores level out the transformation differences between Sca and Can (figure A8.19).

Table A8.11 Process contribution to transformation.

Category	ET Can	ET Sca	ETR Can	ETR Sca
Energy (extraction&production)	73%	77%	100%	100%
Aggregrate extraction	0%	0%	0%	0%
Wood extraction	2%	2%	0%	0%
Industrial production	20%	17%	0%	0%
Transport	5%	4%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

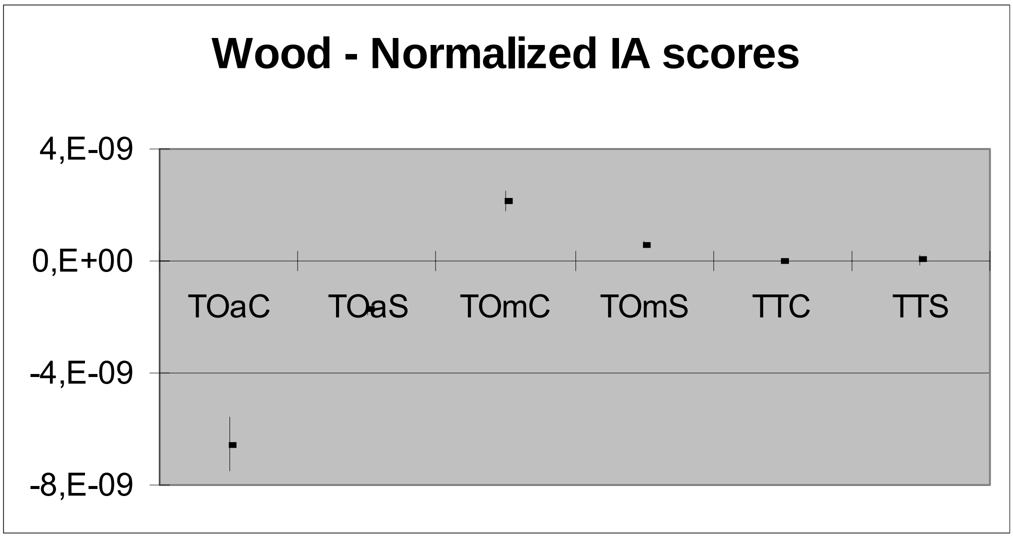


Figure A8.20 Normalisation results for occupation (average and maximum reference) and transformation.

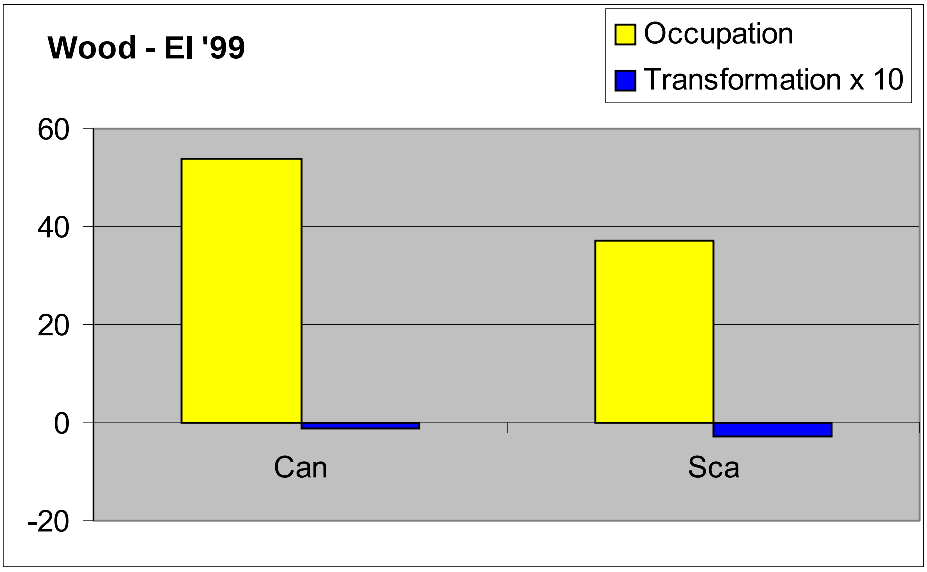


Figure A8.21 Eco-Indicator 99 land use impact assessment scores (occupation and transformation).

Using the Eco Indicator 99 (figure A8.21), the occupation impact scores are not negative, because only for broadleaved forest there are impact scores from [Köllner, 2001], and these are positive (negative for the environment). Transformation scores are negative, as the extensive meadows assumed to be transformed to forestry have a lower species number than (broadleaved) forests.

Table A8.12 Process contribution to Eco-Indicator 99 scores.

Category	EO Can	EO Sca	Renat Can	Renat Sca
Energy (extraction&production)	1%	5%	13%	12%
Aggregrate extraction	0%	0%	0%	0%
Wood extraction	99%	94%	84%	86%
Industrial production	0%	0%	3%	2%
Transport	0%	0%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

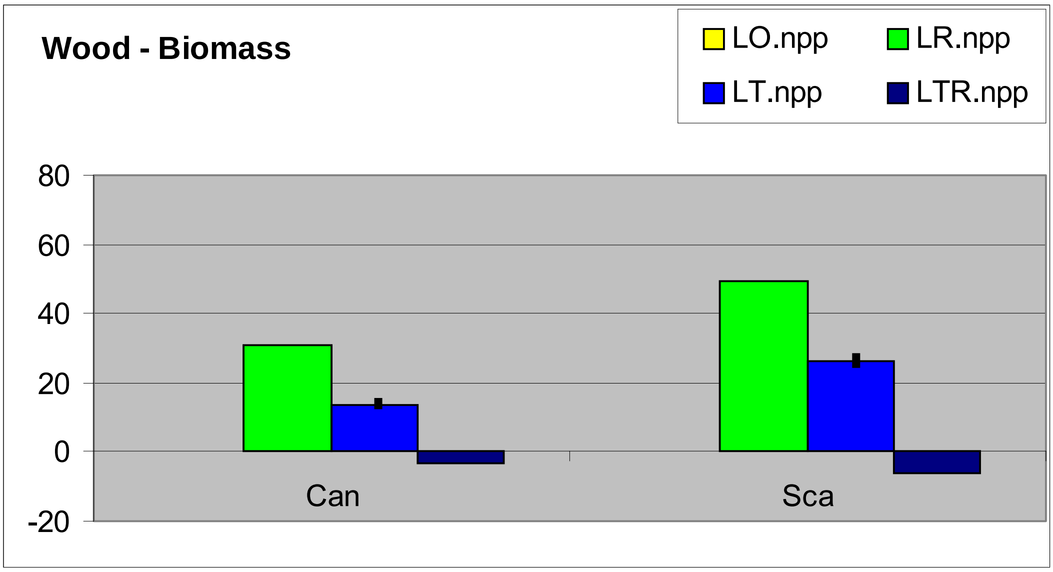


Figure A8.22 TNO life support indicator scores using the NPP indicator. Units are [m^2.y times gC/m^2.y] for occupation, and [m^2 times gC/m^2.y] for transformation.

For LOnpp the scores are very large (26500 resp. 19100 m^2y.gC/m^2y), and could not be shown adequately in the figure. The score for Canada is nearly 1,5 times that for Scandinavia (in spite of similar IA scores) because forestry is more extensive in Canada. The more intensive forestry in Scandiavia leads to lower occupation scores. The occupation renaturation scores (only due to energy production, see table A8.13) are insignificant compared to the occupation due to wood extraction (including renaturation within the forestry practice). For transformation, the scores of wood are of a comparable level to other process types. The uncertainty in the LR, LT and LTR scores is insignificant. The uncertainty in the life support occupation (LO) is shown in figure A8.23 (normalized scores) as these are better shown there.

Table A8.13a Process contribution to life support occupation impacts (wood)

Category	LT Can	LT Sca	LTR Can	LTR Sca
Energy (extraction&production)	1%	9%	100%	100%
Aggregrate extraction	0%	0%	0%	0%
Wood extraction	98%	89%	0%	0%
Industrial production	0%	1%	0%	0%
Transport	0%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

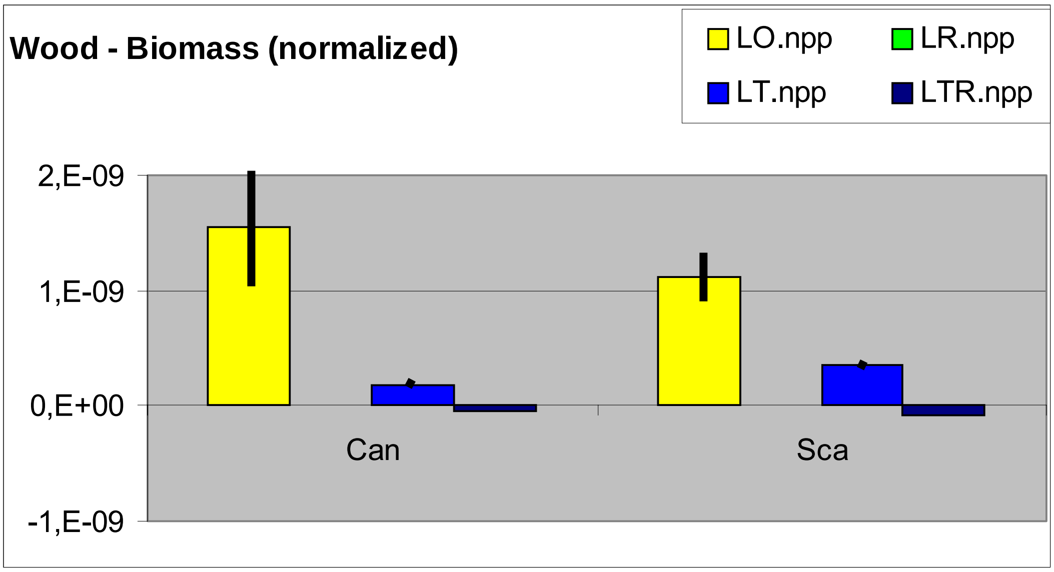
Table A8.13b Process contribution to life support transformation impacts (wood)

Category	LT Can	LT Sca	LTR Can	LTR Sca
Energy (extraction&production)	75%	75%	100%	100%
Aggregrate extraction	0%	0%	0%	0%
Wood extraction	13%	16%	0%	0%
Industrial production	11%	9%	0%	0%
Transport	1%	1%	0%	0%
Waste disposal	0%	0%	0%	0%
Other	0%	0%	0%	0%

Above life support (biomass) results are repeated in figure A8.23, the normalisation results. Here, transformation scores are more prominent than the occupation scores when compared to figure

A8.22. Still, the occupation normalisation scores are highest, and the occupation renaturation scores (LR) are more clearly insignificant. The relative uncertainty is highest for occupation, mainly due to the natural variability in fresh wood density, leading to different areas required for extracting wood.

Figure A8.23 Normalisation results for life support occupation (average reference) and transformation



Annex 9 Glossary and abbreviations

Glossary

Aesthetical (landscape) impacts

Impacts on human health/welfare due to more or less deteriorating landscape sights.

Aggregate extraction

Extraction of (surface) minerals such as sand, gravel and clay, mainly used for their physical rather than chemical properties

Background system/process

A system or process for which secondary data, databases, public referenceds, estimated data based on input-output analysis, are used in an LCA.

Biodiversity (quality indicator)

Biodiversity includes the diversity on the level of ecosystems, species and genes. It has an intrinsic value due to the multitude of life forms implied, and it has a direct relationship with life support functions of the earth; its carrying capacity. The biodiversity quality and impacts on these can be expressed in terms of indicator scores.

Biome

Largest regional grouping of plant and animal habitats discernable at a global scale (largest ecosystem classification).

Characterisation

A step of Impact assessment, in which the environmental interventions assigned qualitatively to a particular impact category (in classification) are qunatified in terms of a common unit for that category, allowing aggregation into a single score: the indicator result; these scores together constitute the environmental profile.

Data quality

A data characteristic relevant for the capacity of the data to satisfy stated requirements.

Ecosystem

System of interactions and dependencies between organisms, mainly classified according to plant and/or animal species (classification possible on different regional levels).

Ecosystem quality

Different aspects of ecosystem quality: the intrinsic value of biodiversity versus the functional value of the global life support system

Environmental impact

A consequence of an environmental intervention in the environment system.

Environmental intervention

A human intervention in the environment, either physical, chemical or biological; in particular resource extraction, emissions (incl. noise and heat) and land use.

Environment life cycle assessment

Part of an overall life cycle assessment in which only the environmental consequences are considered.

Extraction

Withdrawal of a biotic or abiotic resource from the environment in a unit process, considered as an environmental intervention.

Functional unit

The quantified function provided by the product system(s) under study, for use as a reference basis in an LCA, e.g. 1000 hours of light (adapted from ISO).

Impact assessment

The third phase of an LCA, concerned with understanding and evaluating the magnitude and significance of the potential environmental impacts of the product system(s) under study.

Impact score

⇨ Indicator result.

Indicator result

The numerical result of the characterisation step for a particular impact category, e.g. 12 kg CO_2-equivalents.

Input

A product (goods, materials, energy and services), waste for treatment or environmental intervention (including resource extraction, land use, etc.) modeled as ‘entering’ a unit process (adapted from ISO).

Interpretation

The fourth phase of an LCA, in which the results of the Inventory analysis and/or Impact assessment are interpreted in the light of the Goal and scope definition (e.g. by means of contribution, perturbation and uncertainty analysis, comparison with other studies) in order to draw up conclusions and recommendations.

Intersection impacts

Impacts occurring due to the fragmentation of a piece of land due to a more or less line-shaped land use activity.

Intervention

⇨ Environmental intervention

Land occupation

The (partial) unavailability of a given plot of land for alternative uses for a certain period time.

Land quality

Quality of the land expressed in (biodiversity or life support) indicator terms.

Land transformation

The change in the quality of a given plot of land due to a particular mode of human use, measured in terms of changes in biodiversity and life support functions.

Land use

The physical process of using land

Land use activities

Processes involving land use

Life cycle assessment (LCA)

Compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its life cycle; the term may refer to either a procedural method or a specific study.

Life cycle impact assessment

Impact assessment

Life support functions

The ecological structures and processes that sustain the productivity, adaptability and capacity for renewal of lands, water and/or the biosphere as a whole.

Net Primary Production (NPP)

An indicator for various life support functions (potential for food production, biochemical substance and energy cycles, topsoil formation and preservation). The NPP is calculated as the net yearly plant (or carbon) mass production per unit area. NPP addresses mainly the life support for human life.

Normalisation

A step of Impact assessment in which the indicator results are expressed relative to well-defined reference information, e.g. relative to the indicator results for global interventions in 1995.

Occupation

The occupation of land has a duration and an area aspect, and occupation is therefore expressed in area times time [m^2.y] per functional unit of activity output. This functional unit is based on the total performance during the year(s) are accounted for.

Occupation impact

During the occupation of the land, the land has a certain quality. This quality determines the occupation impacts caused by the occupation.

Output

An economic flow (e.g. energy, waste for treatment) or environmental intervention (e.g. pollutant or noise emission) modeled as 'leaving' a unit process (adapted from ISO).

Process

⇨ Unit process (see also: environmental process)

Product

A positively valued economic flow of goods, materials, energy or services produced in a unit process and possibly serving as an input to another unit process.

Product system

A set of unit processes interlinked by material, energy, product, waste, or service flows and performing one or more defined functions.

Reference state

The level to which the land use quality is related to. This may be any predefined level on a certain scale level, such as a maximum reference level, or an average reference level

Relaxation

⇒ Renaturation

Relaxation time

The time necessary for renaturation to be completed.

Renaturation

The process of recovery of land to a more natural (or simply improved) state.

Renaturation occupation
When the human activity is followed by an active human contribution to changing the final state of that activity, there will be occupation during this process of renaturation too, called renaturation occupation. When the renaturation is allowed to occur spontaneously, one needs to decide whether this should be considered an aspect of occupation.

Scarcity
Used here in the context of ecosystem scarcity: the relative present extent of an ecosystem.

Species
Level of (plant or animal) classification; a species expresses a set of unique characteristics on that level.

Species density
Density of species within a certain area.

Species richness
Diversity of species.

Transformation
Changing land quality due to an activity.

Transformation impact
The net change of land quality, comparing only the situation before a change to the situation afterwards. It is measured in m^2 times quality changed [m^2.Δquality] per functional unit.
Negative transformation impact is good for the environment, a positive impact is bad for the environment.

Unit process
The smallest portion of a product system for which data are collected in an LCA.

Vulnerability
Susceptibility of ecosystems to endangering changes from the outside

Abbrevations

α	alpha parameter in $S = \alpha\ LOG^{10}(A_{map})$ from [Lindeijer et al., 1998]
A	area (in [m^2])
a	average reference (sometimes parameter a in $S = aA^b$)
$_{act}$	index for actual land quality
$_{av}$	index for average reference
AFE	Atles Flora Europaeae
A_{map}	unit area used for mapping species richness
ANPP	aboveground biomass productivity
ar	average reference, incl. renaturation
b	species accumulation factor in $S = aA^b$
BUWAL	Bundesamt für Umwelt, Wald und Landschaft
C	carbon
CO_2	carbon dioxide
Can	Canada
^{E}+06, E+6	To be multiplied with a factor 1,000,000 (10^e6)
EI 99	Eco-indicator 99
EO	Ecosystem Occupation
ER	Ecosystem occupation during Renaturation
ES_i	Ecosystem Scarcity factor for biome i
ET	Ecosystem Transformation
ETH	Eidgenössische Technische Hochschule (Zürich)
ETR	Ecosystem Transformation due to Renaturation
$_{fin}$	index for final land quality
fNPP	free NPP = NPP minus amount of biomass extracted for human use
gC	grammes carbon
$g/m^2.y$	grammes biomass per square meter and year
ha	hectare (10.000 m^2)
IA	Impact assessment
$_{ini}$	index for initial land quality
kt	kilotonnes (1,000,000 kg)
LCA	Life cycle assessment
Log, LOG^{10}	Logarithmic -mathematical function- (with base 10)
LOnpp	Life support Occupation using NPP as indicator
LRnpp	Life support occupation during Renaturation using NPP as indicator
LTnpp	Life support Transformation using NPP as indicator
LTRnpp	Life support Transformation due to Renaturation using NPP as indicator
m	maximum reference, no renaturation
$_{max}$	index for maximum reference
mln	milllion (10^e6)
mr	maximum reference, including renaturation
MRPI	Milieurelevante product information (environmentally relevant product information)
$m^2.y$, m^2y	square meter times years (unit of occupying an area of land during some years)
no renat	extraction without renaturation
NO_x	nitrous dioxyde
NPP	Net primary production
$_{ref}$	index for reference land quality
renat	extraction with renaturation included
S	Species richness (number of species)
S_{stand}	Species richness standardised to a unit area
SD	Species density (relative local species richness)
SR_i	Relative species richness factor for biome i
Sca	Scandinavia
SETAC	Society for Environmental Toxicology and Chemistry

t	time (in years)
tkm	tonnes x kilometers (unit of transport performance)
TO	Total Occupation
TT	Total Transformation
WIA	SETAC-Europe Working Group on Impact Assessment
WRI	World Resources Institute
y	years