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A PVC substance flow analysis for Sweden

Part III: Methodological backgrounds

Report for:
Norsk Hydro

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CONTENTS PART III

1 Impact assessment (theoretical aspects)

1.1 Introduction

One of the objectives of the study is to prioritize the risks to humans and the environment due to environmental impacts from the PVC chain. To make a comparison based on risks to humans and the environment the emissions will be assessed on the basis of their potential contribution to specific environmental problems. Such an assessment can be divided into three stages:

1. *inventory analysis* of the emissions and environmental impacts caused by the system being investigated;
2. *classification* of the environmental impacts by type of effect or environmental problem, followed by *quantification* of the contribution of this type of problem;
3. optionally, *normalisation* of the effect scores by expressing them as a fraction of the overall magnitude of the problem in a defined period in a region, and an *evaluation* in which the effect scores, whether normalised or not, are weighed and then combined to a single environmental index.

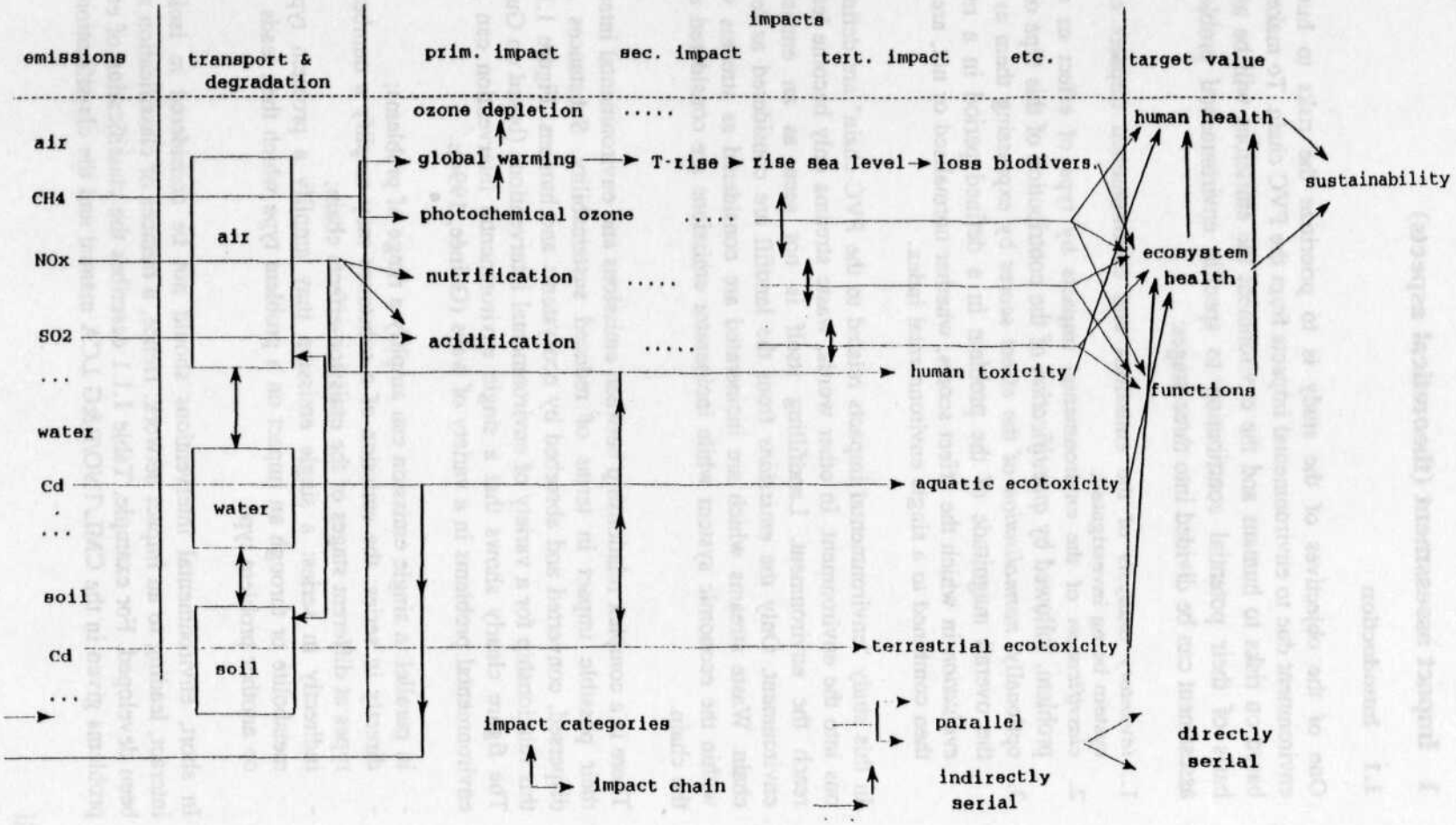
In this study "environmental impacts related to the PVC-chain" are defined as emissions into the environment. In other words, waste streams only become leaks once they reach the environment. Landfilling itself is not seen as an emission into the environment. Only the *emissions* from the landfill are considered as leaks from the chain. Waste streams which are incinerated are considered as streams which remain within the economic system while incinerator emissions are considered as leaks from the chain.

There is a complex relationship between emissions and environmental interventions and their possible impact in terms of reduced sustainability. Substances are released, dispersed, converted and absorbed by ecosystems and humans. Figure 1.1.1 illustrates this relationship for a variety of environmental interventions (based on Guinée (1994)). The figure clearly shows that a single environmental intervention can contribute to environmental problems in a variety of ways (Guinée, 1994):

- in parallel: a single emission can amplify a range of problems;
- directly in series: the emission of a substance may amplify a number of problem types at different stages of the emission-effects chain;
- indirectly in series: a single emission may amplify a problem type through a metabolite or through an impact on a problem type which then leads to an impact on another problem type.

In short, environmental interventions should not be considered in isolation - they interact, leading to an impact network. Hence, a number of classification systems have been developed. For example, Table 1.1.1 describes the classification of environmental problems given in the CML/TNO/B&G LCA manual and the classification later

Figure 1.1.1: Relationships between emissions and target values (based on: Guinée, 1994)



developed by the Netherlands Ministry of Housing, Spatial Planning and Environment (VROM) to monitor the effects of environmental policies (Heijungs, 1992; Adriaanse, 1993). Recently, a third classification method was developed under the Eco-indicator project (Goedkoop, 1995), which is also included in the table.

Table 1.1.1: Comparison of the classification systems based on the VROM themes, Eco Indicator Project and the CML Guide

Eco Indicator Project	CML Guide	VROM themes
Heavy metals in the atmosphere	Human toxicity	Dispersion
Heavy metals in water		
Carcinogens		
Pesticides	Aquatic ecotoxicity	
	Terrestrial ecotoxicity	
Acidification	Acidification	Acidification
Depletion of the ozone layer	Depletion of the ozone layer	Theme: climate Sub-theme: ozone depletion
Enhanced greenhouse effect	Enhanced greenhouse effect	Climate change
Summer smog	Smog	-
Winter smog		
-	Odour nuisance*	Disturbance
-	Noise nuisance*	
-	Space use	Disposal
Eutrophication	Eutrophication*	Eutrophication
-	Resource depletion*	Squandering
-	Misc. other categories*	-

* Not relevant in the context of this study

A classification scheme has be selected with some regard to its application. As the evaluation often requires some form of multicriteria analysis the following requirements are often imposed on the classification scheme (Heijungs, 1994b; Assies, 1994; Tukker, 1994b; Finnveden, 1994; Udo de Haes, 1995):

- the problem types should be homogenous whenever possible (i.e. they should cover environmental interventions causing the same impacts);
- there should be the greatest possible independence between the problem types and they should not overlap.

Figure 1.1.1 clearly shows that in practice it is almost impossible to select a classification scheme which fulfils all these requirements. For practical reasons the scheme described in the CML manual was adopted for this study. Given the above criteria the CML classification is also somewhat more attractive than the classification based on the various VROM themes which combine a wide variety of impacts (e.g. odour, noise and various toxic effects) into one parameter. This results in an implicit weighting within the confines of the various themes.

The sections below cover the classification, normalisation and evaluation steps of this study. The assessment is based on a subset of the categories of environmental problems from the CML guide listed in table 1.1.1. Noise nuisance, odour nuisance, resource depletion and miscellaneous other categories (like radiation, etc.) have not been taken into account. Noise and odour are very locally orientated problems, which seems less relevant at the level of this SFA. Resource depletion is not really relevant for the PVC-chain in a narrow sense, since it uses chlorine as a raw material, which is abundantly available. Other resources (like oil, gas, and other raw materials) can not be properly taken into account due to the system boundaries chosen in this regional material SFA. We have indicated the consumption of auxiliary materials, mainly intermediates, used in the production processes of the PVC chain itself. However, we did not follow the life cycle of these materials backwards. We thus have no list of the use of truly primary raw materials related to the chain of production, use and waste management of PVC, and thus cannot calculate a value for resource depletion. This would only be possible if a *functional* rather than *regional* SFA had been performed (v.d. Voet, 1996).

1.2 Characterization[1]

1.2.1 Introduction

The survey of the environmental interventions is followed by the calculation of their quantitative contribution to a given environmental theme. The weighting is mainly based on the equivalence or classification factors from the Product Lifecycle Assessment method (Heijungs, 1992). The predecessors of these classification factors were previously used in the McKinsey study "Integrated Substance Chain Management" (VNCI/McKinsey, 1991). The LCA classification factors are closely related, and sometimes identical, to the theme indicators used as environmental policy indicators in national or international environmental monitoring systems (Adriaanse, 1993). Both quantify the burden on the environment. However, a major difference between them is that the theme indicators express the *actual burden* on the environment while the classification factors express the *potential contribution* of emissions to a given environmental problem. Thus, the emissions in Sweden are considered irrespective of the location

[1] Paragraph 1.2 rely heavily on textes earlier published in the CML Guide for LCA and the study LCA Impact Assessment of Toxic Releases (Heijungs, 1992; Guinée, 1996)

where they lead to an actual impact. The classification factors used for each environmental impact type are introduced below.

1.2.2 Enhanced greenhouse effect

A number of models have been developed for the enhanced greenhouse effect, to quantify the contribution of emissions of various substances to this effect. *Global warming potentials* (GWP) were developed to compare greenhouse gas emission scenarios. The GWP of a substance is the ratio of the heat absorption due to the instantaneous (i.e. impulse) emission of 1 kg of a greenhouse gas integrated over time compared to the heat absorption of 1 kg carbon dioxide (CO_2) emission. This study used the GWPs specified by the Intergovernmental Panel on Climate Change (IPCC; Houghton et al., 1991 and 1992) which are widely accepted internationally. The method used to calculate the GWPs indicates that they depend on the time horizon used. This study is based on GWPs based on a 100 year horizon: GWP_{100}. The effect score of a given emission in terms of the enhanced greenhouse effect is calculated with the formula:

$$\text{enhanced greenhouse effect} = \sum_i GWP_i \times m_i \qquad (1)$$

where:

enhanced greenhouse effect is the number of CO_2 equivalents in kg/y;
m_i is the atmospheric emission in kg substance per year;
GWP is the Global Warming Potential relative to CO_2 (dimensionless).

1.2.3 Depletion of the ozone layer

Ozone depletion potentials (ODP) have been defined for substances which deplete the ozone layer and are defined similarly to the GWPs. The ODP is defined as the ratio between the equilibrium ozone depletion due to the annual release (flux in kg/y) of a given quantity of a substance into the atmosphere and the equilibrium ozone depletion due to the same quantity of CFC-11. This study used the widely accepted ODPs determined by the Scientific Assessment Panel (WMO, 1989) which includes all leading scientists in this field. The ozone depletion effect score is calculated with the formula:

$$\text{depletion of the ozone layer} = \sum_i ODP_i \times m_i \qquad (2)$$

where:

depletion of the ozone layer = CFC-11 equivalents in kg/y;
m_i = emissions to air, in kg substance per yr;
ODP = ozone depletion potential, dimensionless.

1.2.4 Human toxicity (HT and HTP)

'Old' equivalency factors: HT

A range of models have been developed to determine the contributions of various substances to the theme of human toxicity. The CML Guide is based on a provisional model using HCAs, HCWs and HCSs - human toxicological classification factors for the atmosphere, water and soil. A disadvantage of these factors is that the environmental fate of substances, i.e. their distribution and transformation, is not considered in the calculations (Heijungs, 1992). Though *not applied in this study*, they are described here as a contrast to the 'new' factors.

Provisional classification factor for air:

The provisional classification factor is the product of the provisional exposure factor and the provisional effects factor. Thus, the provisional *human toxicological classification factor for air* (HCA) is:

$$HCA = B_a \times E_a = \frac{\dot{V}_a \times W \times M}{V_a \times (ACA \text{ c.q. } AQG) \times \dot{V}_a} \text{ c.q } \frac{\dot{V}_a \times W}{V_a \times (TDI \text{ c.q. } ADI)} \quad (3)$$

where:

HCA is the provisional classification factor for the atmosphere (kg body weight · kg^{-1} substance);
$\dot{V}_a$ is the human respiratory volume (= 20 m^3 air · day^{-1} · $person^{-1}$);
W is the world population (= $5 \cdot 10^9$);
M is the human body weight (= 70 kg · $person^{-1}$);
ACA is the Acceptable Concentration in the Atmosphere (kg substance · m^3 air);
AQG is the Air Quality Guideline (kg substance · m^3 air);
TDI is the Tolerable Daily Intake (kg substance · day^{-1} · kg^{-1} body weight);
ADI is the Acceptable Daily Intake (kg substance · day^{-1} · kg^{-1} body weight).

Provisional classification factor for water:

The provisional *human toxicological classification factor for water* (HCW) is calculated similarly to that for the atmosphere:

$$HCW = B_w \times E_w = \frac{\dot{V}_w \times W}{V_w \times (TDI \text{ c.q. } ADI)} \quad (4)$$

where:

HCW is the provisional classification factor for water (kg body weight · kg^{-1} substance);
$\dot{V}_w$ is the human water consumption (2 l water · day^{-1} · $person^{-1}$);
W is the world population (= $5 \cdot 10^9$);
V_w is the water volume in the world model ($3.5 \cdot 10^8$ l);
TDI is the Tolerable Daily Intake (kg substance · day^{-1} · kg^{-1} body weight);
ADI is the Acceptable Daily Intake (kg substance · day^{-1} · kg^{-1} body weight).

Provisional classification factor for soil

The provisional *human toxicological classification factor for soil* (HCS) is calculated with the formula:

$$HCS = B_a \times E_a = \frac{M \times W \times N}{V_a \times C\ value-} \qquad (5)$$

where:

HCS is the provisional classification factor for the soil (kg body weight · kg^{-1} substance);
M is the human body weight (= 70 kg body weight);
W is the world population (= $5 \cdot 10^9$);
N is the uncertainty reduction factor for the TDI;
V_S is the soil mass in the world model ($2.7 \cdot 10^{16}$ kg dry matter).

Calculation of the effect score

The provisional human toxicological classification factors for the environmental media air, water and soil are listed in a table in Appendix B to the LCA Guide. The sources of the toxicity data were: Vermeire *et al.* (1991), FAO/WHO (1990), Staarink & Hakkenbrak (1985 and 1987), WHO (1987), Kleijn & Van der Voet (1991), Van den Berg (1991) and Van den Berg & Roels (1991).

When undertaking a practical study the effect score of each substance is calculated by multiplying the emissions into the various environmental media per functional unit by the relevant provisional classification factors. The effect scores of the emissions into the atmosphere, water and soil are added to produce the overall effect score for human toxicity:

$$\text{human toxicity} = \sum_i ((HCA_i \times m_{a,i}) + (HCW_i \times m_{w,i}) + (HCS_i \times m_{s,i})) \qquad (6)$$

where:

human toxicity is the contaminated body weight (kg body weight);

$m_{a,i}$ is the emission into the air (kg substance *i*);

$m_{w,i}$ is the emission into water (kg substance *i*);

$m_{s,i}$ is the emission into the soil (kg substance *i*);

HCA_i is the provisional human toxicological classification factor for the atmosphere (kg body weight · kg^{-1} substance *i*);

HCW_i is the provisional human toxicological classification factor for water (kg body weight · kg^{-1} substance *i*);

HCS_i is the provisional human toxicological classification factor for the soil(kg body weight · kg^{-1} substance *i*).

This can be interpreted as the total human body weight contaminated up to the maximum acceptable limit by the functional unit. As the exposure factors used are only *provisional* (set at 1) it is emphasised that the effect score should only be considered as an indication.

'New' equivalency factors: HTP

To offset these disadvantages Guinée and Heijungs (1993) proposed a toxicity model for use in the LCA method. This includes distribution and decomposition in the calculations of the Human Toxicity Potentials (HTP). The proposed method is very similar to that used in the Uniform System for the Evaluation of Substances (USES). USES was developed by the RIVM (National Institute of Public Health and Environmental Protection) for rapid, general human toxicological and ecotoxicological risk assessments of a wide range of compounds. This approach has been worked out recently by CML and RIVM, giving a list of 100 USES-based equivalency factors for the toxicity themes (Guinée, 1996). *These factors have been applied in this study.*

In brief, for an emission of a substance to water, soil or air using partition co-efficients and degradation rates the steady-state distribution over the various compartments is calculated using Level III MacKay models (MacKay, 1991). This results in a *Potential Environmental Concentration* (PEC) in a standard 'unit world' environment (see figure 1.2.1). For human toxicity, a total exposure for man is calculated, given this distribution over the media making standard assumptions on 'average realistic' exposure (and thus 'average realistic presence of receptors') via different intake routes, like food consumption, inhalation and skin contact (see figure 1.2.2). By comparing this 'Predicted daily intake' (PDI) with the Acceptable daily intake (ADI) USES calculates a Margin of Safety (MOS; in fact the ADI/'PDI'). E.g. substances with low ADIs, low degradation, low immobilisation in sediments and high bioconcentration factors (BCFs) for foodstuffs will have per kg emitted a high contribution to the human toxicity potential. One difference with normal MOS-calculations is that in LCAs the location and duration of the emission are not known. Guinée et al. solved this by introducing a

Figure 1.2.1.: Compartments and processes described in the regional distribution model applied by Guinée et al. (1996) in calculating toxicity equivalency factors

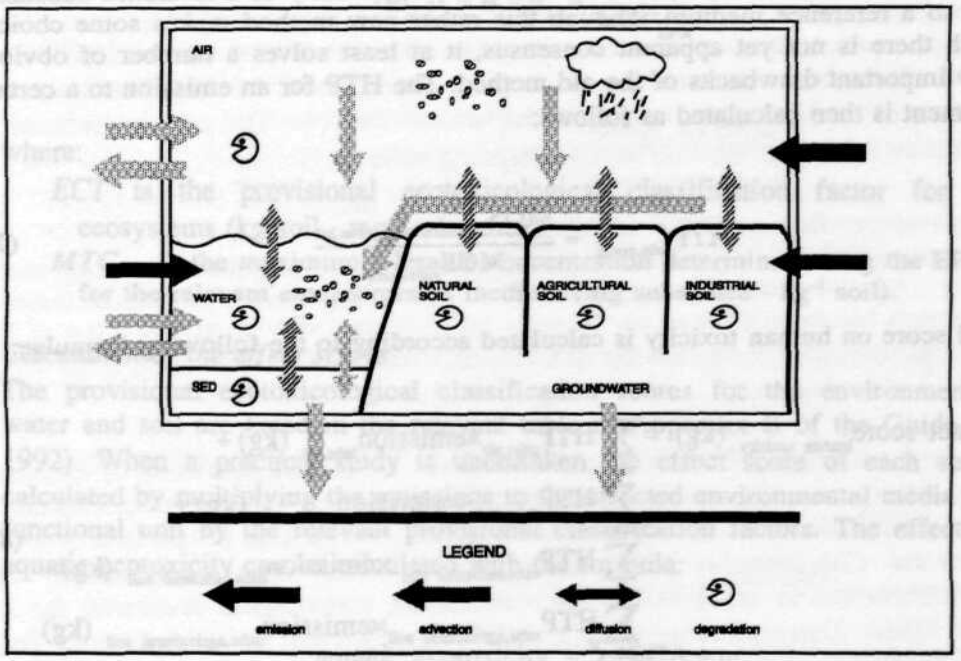
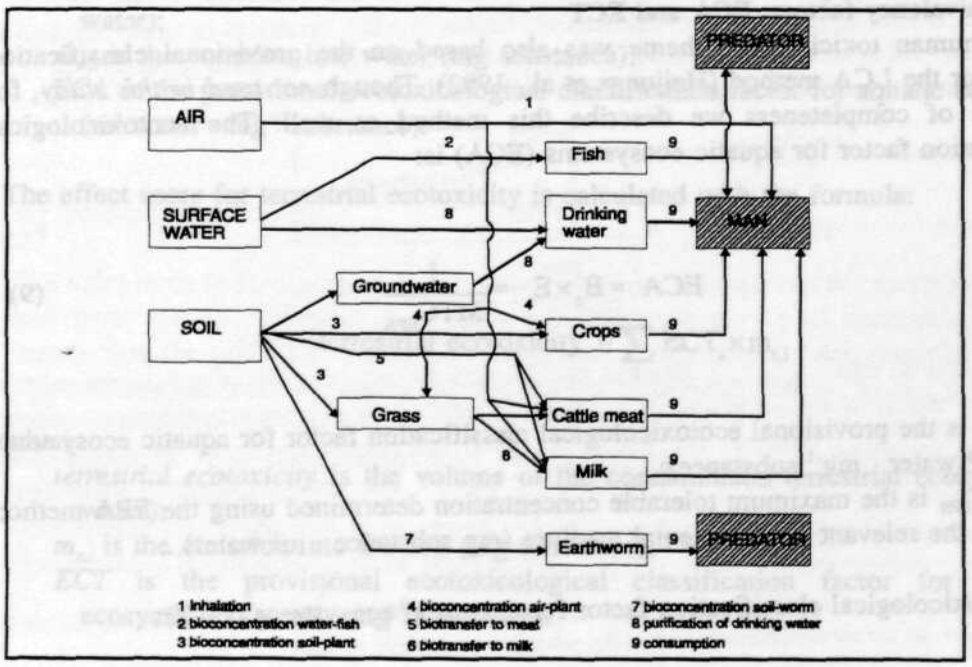


Figure 1.2.2: Schematic representation of the exposure of man and predators via the environment (from: Guinée et al., 1996).



reference substance (1,4 dichlorobenzene). HTP can then be described as an equivalency factor giving the potential contribution of a given quantity of a defined substance to the human toxicity, relative to an equal quantity of a reference substance released to a reference medium. Though this rather new method makes some choices on which there is not yet apparent consensus, it at least solves a number of obvious and very important drawbacks of the old method. The HTP for an emission to a certain compartment is then calculated as follows:

$$HTP_{subs,comp} = \frac{MOS_{1,4\text{-dichlorobenzene},air}}{MOS_{subs,comp}} \quad (7)$$

The total score on human toxicity is calculated according to the following formulae:

$$\begin{aligned} \text{impact score}_{\text{human toxicity}}\ (\text{kg}) = & \sum_{subs} HTP_{subs,air} \times \text{emission}_{subs,air}\ (\text{kg}) + \\ & \sum_{subs} HTP_{subs,water} \times \text{emission}_{subs,water}\ (\text{kg}) + \\ & \sum_{subs} HTP_{subs,\text{industrial soil}} \times \text{emission}_{subs,\text{industrial soil}}\ (\text{kg}) + \\ & \sum_{subs} HTP_{subs,\text{agricultural soil}} \times \text{emission}_{subs,\text{agricultural soil}}\ (\text{kg}) \end{aligned} \quad (8)$$

1.2.5 Ecotoxicity

'Old' equivalency factors: ECA and ECT

As for human toxicity, this theme was also based on the provisional classification factors for the LCA method (Heijungs et al., 1992). Though *not used in this study*, for the sake of completeness we describe this method as well. The ecotoxicological classification factor for aquatic ecosystems (ECA) is:

$$ECA = B_a \times E_a = \frac{1}{MTC_{EPA}} \quad (9)$$

where:

ECA is the provisional ecotoxicological classification factor for aquatic ecosystems (m^3 water · mg^{-1} substance);

MTC_{EPA} is the maximum tolerable concentration determined using the EPA method for the relevant environmental medium (mg substance · m^3 water).

The ecotoxicological classification factor for terrestrial ecosystems (ECT):

$$ECT = B_t \times E_t = \frac{1}{MTC_{EPA}} \quad (10)$$

where:

ECT is the provisional ecotoxicological classification factor for terrestrial ecosystems (kg soil · mg^{-1} substance);

MTC_{EPA} is the maximum tolerable concentration determined using the EPA method for the relevant environmental medium (mg substance · kg^{-1} soil).

Calculation of the effect scores

The provisional ecotoxicological classification scores for the environmental media water and soil are listed in the relevant table in Appendix B of the Guide (Heijungs 1992). When a practical study is undertaken the effect score of each substance is calculated by multiplying the emissions to the affected environmental media due to the functional unit by the relevant provisional classification factors. The effect score for aquatic ecotoxicity can be calculated with the formula:

$$\text{aquatic ecotoxicity} = \sum_i ECA_i \times m_{w,i} \quad (11)$$

where:

aquatic ecotoxicity is the volume of the contaminated aquatic ecosystem (m^3 water);

$m_{w,i}$ is the emission into water (mg substance);

ECA is the provisional ecotoxicological classification factor for aquatic ecosystems (m^3 water · mg^{-1} substance).

The effect score for terrestrial ecotoxicity is calculated with the formula:

$$\text{terrestrial ecotoxicity} = \sum_i ECT_i \times m_{t,i} \quad (12)$$

where:

terrestrial ecotoxicity is the volume of the contaminated terrestrial ecosystem (kg water);

$m_{s,i}$ is the emission into the soil (mg substance);

ECT is the provisional ecotoxicological classification factor for terrestrial ecosystems (kg soil · mg^{-1} substance).

The units of the resulting *terrestrial ecotoxicity* and *aquatic ecotoxicity* values are kg soil and m^3 water, which may be interpreted as the volume of terrestrial or aquatic material contaminated to the MTC_{EPA}. Thus, the *critical volumes* approach is also used for ecotoxicity classification in this provisional method. Again, given the fact that *provisional* exposure factors are used the effect score should only be considered as an indication. With the implementation of the envisaged model for the classification of toxic substances a closer approximation has become feasible. You are also referred to the discussion on human toxicity in the preceding section.

'New' equivalency factors: TETP ad AETP

To offset these disadvantages Guinée and Heijungs (1993) proposed a toxicity model for use in the LCA method, as explained in the preceding section on human toxicity. This approach has been worked out recently by CML and RIVM, giving a list of 100 USES-based equivalency factors for the toxicity themes (Guinée, 1996). *This is the approach used in this study.*

In brief, for an emission of a substance to water, soil or air using partition co-efficients and degradation rates the steady-state distribution over the various compartments is calculated using Level III MacKay models (MacKay, 1991). This results in a *Potential Environmental Concentration* (PEC) in a standard 'unit world' environment. The potential effects due to exposure are calculated using toxicological standards, the *No (adverse) Effect Concentrations* (NEC). NECs are derived by extrapolation from toxicity data for specific species. A variety of extrapolation methods may be used. For this study the method developed by the US *Environmental Protection Agency* (EPA) was selected. Although this method is not the most advanced available, it has the advantage of providing data on the greatest number of substances. As with human toxicity, an emission of 1,4 dichlorobenzene to the reference compartment has been chosen as a reference. *viz.* The equivalency factor for the chemical at stake emitted to air is now calculated by dividing its PEC/PNEC ratio by the same quantity of 1,4-dichlorobenzene emitted to water:

$$AETP_{subs,comp} = \frac{\left(\frac{PEC_{water,subs,comp}}{PNEC_{aquatic\ ecosystems,subs}}\right)}{\left(\frac{PEC_{water,1,4\text{-}dichlorobenzene,water}}{PNEC_{aquatic\ ecosystems,1,4\text{-}dichlorobenzene}}\right)} \qquad (13)$$

In this formula, $PEC_{water,subs,comp}$ denotes the predicted concentration of substance *subs* in water as a result of the emission of 1000 kg/d *subs* to the release compartment *comp*.[2] $PNEC_{aquatic\ ecosystems,subs}$ denotes the predicted no-effect concentration of substance *subs* for aquatic ecosystems. The equivalency factors are called aquatic ecotoxicity

[2] Observe that there are two subscripts relating to compartments: one for the compartment in which the PEC can be found, and one for the compartment to which the substance was initially released.

potentials, abbreviated as AETP.

A similar procedure is used to calculate equivalency factors for terrestrial ecotoxicity, the terrestrial ecotoxicity potentials (TETP). The reference substance is again 1,4-dichlorobenzene, but the reference compartment is industrial soil. The formula is then:

$$TETP_{subs,comp} = \frac{\left(\frac{PEC_{agricultural\ soil,subs,comp}}{PNEC_{terrestrial\ ecosystems,subs}}\right)}{\left(\frac{PEC_{agricultural\ soil,1,4\text{-}dichlorobenzene,industrial\ soil}}{PNEC_{terrestrial\ ecosystems,1,4\text{-}dichlorobenzene}}\right)} \quad (14)$$

Calculation of the impact scores now proceeds according to the following formulae:

$$\begin{aligned} \text{impact score}_{aquatic\ ecotoxicity}\ (kg) = & \sum_{subs} AETP_{subs,air} \times emission_{subs,air}\ (kg) + \\ & \sum_{subs} AETP_{subs,water} \times emission_{subs,water}\ (kg) + \\ & \sum_{subs} AETP_{subs,industrial\ soil} \times emission_{subs,industrial\ soil}\ (kg) + \\ & \sum_{subs} AETP_{subs,agricultural\ soil} \times emission_{subs,agricultural\ soil}\ (kg) \end{aligned} \quad (15)$$

for aquatic ecotoxicity, and

$$\begin{aligned} \text{impact score}_{terrestrial\ ecotoxicity}\ (kg) = & \sum_{subs} TETP_{subs,air} \times emission_{subs,air}\ (kg) + \\ & \sum_{subs} TETP_{subs,water} \times emission_{subs,water}\ (kg) + \\ & \sum_{subs} TETP_{subs,industrial\ soil} \times emission_{subs,industrial\ soil}\ (kg) + \\ & \sum_{subs} TETP_{subs,agricultural\ soil} \times emission_{subs,agricultural\ soil}\ (kg) \end{aligned} \quad (16)$$

for terrestrial ecotoxicity.

The units in these formulae deserve special attention: the equivalency factors (HTP, AETP, and TETP) are all expressed in kg reference substance per kg of toxic substance. This means that the unit "kg" on the left-hand side of the above three formulae is in fact "kg reference substance", while the "kg" on the right-hand side is "kg toxic substance". Including this in the notation makes the formulae more difficult to read, but facilitates interpretation. The final expressions on the left-hand side represent the amount of reference substance that would yield a toxic impact at the end point (humans, aquatic ecosystems and terrestrial ecosystems) which – under the LCA conditions described in this report – is considered to be equivalent.

1.2.6 Formation of photochemical oxidants

POCPs were developed in order to assess different emission scenarios for volatile organic compounds (VOC). These are similar to GWPs and ODPs (Derwent and Jenkins, 1990). The UNECE defines POCP: the POCP of a specified emission is the ratio between the change in ozone concentration due to the emission of one kg of that substance and the change due to the emission of 1 kg of ethene. The UNECE uses a model to calculate POCPs. However, important aspects of this model require improvement. As the UNECE provides a relatively comprehensive list of POCPs, and in view of the international context in which these values have to be placed, this list was used for this study. The effect score for photochemical oxidant formation is calculated with the formula:

$$\text{photochemical oxidant formation} = \sum_i \text{POCP}_i \times m_i \qquad (17)$$

where:

photochemical oxidant formation is the number of ethene equivalents (kg/y);
m_i is the emission to air (kg substance/y);
POCP is the Photochemical Ozone Creation Potential (-).

1.2.7 Acidification

The H^+ release potential relative to sulphur dioxide (SO_2) is the measure of acidification. Thus, the acidification potential (AP) is a measure of the relative contribution of a substance to acidification, relative to the reference substance, SO_2. This is similar to the GWPs and ODPs. The effect score for acidification is calculated with the formula:

$$\text{acidification} = \sum_i \text{AP}_i \times m_i \qquad (18)$$

where:

acidification is the number of SO_2 equivalents (kg/y);
m_i is the emission to air (kg substance/y);
AP is the acidification potential.

1.2.8 Landfilling and space use

Waste streams are not always leaks in the chain. One process's waste stream may be another process's feedstock, as in recycling. Furthermore, a significant proportion of the waste streams is incinerated. The resulting emissions contribute to the environmental problems discussed above and are quantified as such. The resulting "leak" consists of the waste streams which are landfilled. One of the effects of landfilling is space use, which can be expressed in m^3 or in tonnes. Additionally, landfilling will lead to emissions into the environment. The scale of these emissions depends on the landfilling methods used and is not estimated in this study. However, for practical reasons the landfilling volume is introduced, expressed in kg PVC plus additives. For incineration, the amount of chlorine landfilled with slag, fly ash and flue gas treatment residue is taken as a measure. The amount of flue gas residue is corrected for the fact that chlorine, resulting from PVC, is generally neutralised with NaOH or $Ca(OH)_2$. In that case landfilling of the Na or Ca-ion is logically caused by the chlorine content of the waste, and thus has also been allocated to the PVC-chain.

1.3 Normalisation

1.3.1 Introduction

To gain more insight into the significance of effect scores they can be divided by the total magnitude of the relevant problems expressed in identical scores. This step, normalisation, was developed for LCA. Normalisation can provide information about the extent to which the problem under consideration contributes to the overall magnitude of environmental issues. It can also uncover differences in this area between the various effect scores (Guinée, 1995). This can be expressed in a formula:

$$N_i = S_i / A_i \qquad (19)$$

where:

N_i is the normalised score;
S_i is the score of the system under consideration (e.g. the PVC chain) on theme i;
A_i is the overall annual score of all activities in a specified area on theme i.

Normalisation may be undertaken at various levels of scale. Given the non-location-specific nature of studies of this type, normalisation for LCAs is often carried out at the world scale (see e.g. the world total scores of Guinée,1993 and 1995).

For this study it was decided to implement normalisation for Sweden only. The SFA results in the fact that Sweden is the system boundary for the PVC chain. Given this geographical delineation of the system the obvious choice is to use the overall emissions in the same geographical system (i.e. Sweden) for comparison. Thus, all emissions are expressed as a percentage of the overall emissions in Sweden.

1.3.2 Normalisation data and their uncertainties for Sweden

The calculation of Swedish normalisation data is discussed in a separate report (Tukker & Kleijn, 1996b). Here we merely review the result (see table 1.3.1).

An existing normalisation database with Dutch data from CML was used as a basis. Where possible, TNO and CML have entered specific Swedish emission figures in this database. For this purpose, a large amount of literature was kindly provided by the Swedish EPA and the Swedish Chemicals Inspectorate (KemI). If no specific Swedish emssion data were available, they were in general extrapolated from the Dutch data on the basis of relative GNPs. The table gives the derived Swedish score that was used. Realistic alternative values given by other authors are presented as well. Finally, the table summarizes the uncertainty in each theme score. Once again, it has to be underlined that these values are **truly provisional** and should be handled and used with **insight in the strengths and weaknesses of LCA Impact Assessment.** Further improvement of the emission database is highly recommended.

Table 1.3.1: Overall scores per environmental theme, Sweden (1992) and the world (1988).

THEME	WORLD	SWEDEN (used in the PVC-study)	SWEDEN (other sources)	Uncertainty	UNIT
human toxicity (HTP)	-	$4.10\ 10^{12}$		factor 50	kg bw.y^{-1}
aq. ecotoxicity (EATP)	-	$8.10\ 10^{8}$		factor 5	m^3.y^{-1}
terr. ecotoxicity (TETP)	-	$2.49\ 10^{13}$		factor 10	kg.y^{-1}
hum. toxicity (old, HT)	$3.21\ 10^{11}$	$8.11\ 10^{8}$		factor 2-3	kg bw /y
aq. ecotox. (old, ECA)	$9.08\ 10^{14}$	$1.92\ 10^{12}$		factor 5	kg/y
terr. ecotox. (old, ECT)	$1.26\ 10^{15}$	$1.21\ 10^{13}$		factor 10	kg/y
acidification (AP)	$2.86\ 10^{11}$	$4.86\ 10^{8}$	$7.60\ 10^{8}$ [a]	30 %	kg SO_2.y^{-1}
nutrification (NP)	$8.87\ 10^{10}$	$5.23\ 10^{8}$	$6.40\ 10^{8}$ or $8.32\ 10^{7}$ [a]	factor 2-3	kg P-eq.
ozone depl. (ODP)	$1.00\ 10^{9}$	$1.25\ 10^{6}$	$4.10\ 10^{6}$ [a]	factor 3	kg CFC 11.y^{-1}
global warming (GWP)	$3.77\ 10^{13}$	$7.52\ 10^{10}$	$9.00\ 10^{10}$ [a]	30 %	kg CO_2.y^{-1}
smog (POCP)	$3.74\ 10^{9}$	$5.91\ 10^{7}$	$2.31\ 10^{8}$ [b]	factor 5	kg ethene.y^{-1}
odour	$6.28\ 10^{17}$	$3.81\ 10^{15}$		very high	m^3.y^{-1}
landfill volume	-	$35\ 10^{10}$	$46\ 10^{10}$ [c]	30 %	kg.y^{-1}

[a] Baumann et al. (1994)

[b] See Annex 3 in Tukker & Kleijn (1996); comparable with the result of Baumann et al. (1994)

[c] SNV (1995a)

1.4 Normalisation data for individual substances

1.4.1 Introduction

Especially for the toxicity themes, the LCA methodology is still under development. This means that LCA scores for the toxicity themes must be interpreted with care. Comparisons with something like a total national toxicity score have only a very limited meaning.

In order to gain insight into the contribution to toxicity from another perspective, one can compare e.g. data for a number of individual compounds for the PVC-chain with the corresponding data for the total Swedish society. In this way, the normalisation is not performed for themes but for a number of individual substances.

For this comparison, we have selected a number of substances which are seen as most representative of its toxicity in the PVC debate. These substances are:

- dioxins;
- mercury;
- lead;
- (organo)tin;
- phthalates.

It is possible to compare the emissions and flows related to PVC with total Swedish emissions and flows for:

a. emissions to water, air and soil;
b. total new use in 1994;
c. total accumulation in society.
d. total available in waste in 1994;

Points b, c and d are not relevant for dioxins and mercury, since these substances are only emissions from the production and/or waste management stage. The substances do not 'travel' with PVC through the chain. For phthalates, it is not necessary to further elaborate the comparison with Swedish totals. Virtually all phthalates are used to plasticise PVC. So all emissions, new use, waste and accumulated amount of phthalates relate for 100 % to PVC. The next paragraph gives Swedish totals for the other compounds.

1.4.2 Total emissions of mercury, lead, dioxins, and (organo)tin

Data for the emissions to water and air of dioxins, mercury, lead and organotin are given in table 1.4.3. Emission figures for mercury and lead to air are from 1992 and reported in the Environmental Statistics of Swedish EPA (SNV, 1995b). A forecast for

2000 is also given. Emissions of lead to water are given in the same source. They are for 1990; a forecast is given for 1995. The mercury emission to water is given in an SFA for mercury by KemI (1994e). Dioxin emissions to air for 1991 are given in Landner (1995). His average values for 1991 have been used, including the indicated expected improvements for waste incineration. It has been assumed that hospital incinerators will be decommissioned. Organotin emissions to water are also given in a KemI report (1994d). According to KemI, organotin emissions to air can be seen as negligible.

No specific Swedish data were available for other emissions. They have been estimated as follows.

The emissions to air and water for tin have been extrapolated on the basis of Dutch data from the Dutch Emission Record database. The relative GNPs of the Netherlands and Sweden have been used as a basis for the extrapolation (see Tukker & Kleijn, 1996). The dioxin emissions to water have been roughly estimated by comparing the (known) Dutch emissions to air and water. Bremmer (1994) estimated the Dutch emissions to water in 1991 at 3 gram: 0.8 gram from hazardous waste incineration, 1.5 gram from sintering processes and 0.5 gram from chemical production processes. For the latter, reduction measures have already been taken, for the first two some reduction still seems possible. The total dioxin emissions to air in the Netherlands in 2000 will be about 58 grams; a future emission to water of somewhere between 1 and 2.5 grammes would mean that the emission to water will be about 2 to 5 % of the emission to air. *If* such a ratio were also valid for Sweden, this would mean a dioxin emission to water of between 0.5 to 1.3 grammes. This value is of course quite uncertain; we think an uncertainty factor of 2-3 but below a factor of 10 is realistic.

1.4.3 Total new use and accumulated amount for lead and (organo)tin

Lead

For lead, KemI has produced a number of reports that indicate the total Swedish use. KemI gives both the (concentrated) use in the form of easy to recycle metal and the more diffuse applications (KemI, 1994c; Hedelman, 1994). The application as stabiliser in PVC is a diffuse application. Here we make a comparison with both the *diffuse* lead applications and the *total* lead applications in Sweden. Sometimes the diffuse lead applications are seen as a meaningful basis for comparison, since the lead in the other applications like batteries can easily be recycled (KemI, 1994c). But on the other hand, for such applications a small percentage not recycled can mean an significant loss to the environment. Table 1.4.1 reviews the lead use in such applications in 1992 (KemI, 1994c). With a very rough average life-time, estimated by TNO/CML, we calculated the total accumulated amount as well. In our opinion, the data for new use have a maximum uncertainty of 50 %. The uncertainty in the calculated accumulated amount is much higher, and could be a factor of 2 to 3.

Table 1.4.1: (Diffuse) lead use and accumulation in Sweden (1992, in ton).

Lead use	Tonnes, 1992	Life-time	Accumulation
Glass (crystal)	1,320	20	27,400
Paint and rust protection	90	20	1,800
Lead shot and bullets	1,200	0	-
Lead sinkers for angling and net fishing	600	5	3,000
Plastic additives	2,000	15	30,000
Electronics (solder, screens, etc.)	1,300	10	13,000
Alloys	900	20	18,000
Gasoline	340	0	-
Other (diffuse ?) applications	150	10	1,500
Subtotal diffuse applications	7,900		94,700
Chimney fittings	500	20	10,000
Weights	1,000	10	10,000
Accumulators	22,000	10	220,000
Cable sheathing	3,000	10	30,000
Grand total	34,400		364,700

Tin

According to KemI, the tin consumption in Sweden in 1975 (apparently the most recent data) was about 1,300 tons. About 74 % was used in diffuse applications like solder, tinned plate, and other applications (Eriksson, 1991). In brief, assuming that the situation has not changed dramatically since 1975 the total use of new tin in diffuse applications can be estimated at somewhat under 1,000 tons a year. Table 1.4.2 gives a breakdown of uses and estimated accumulated amounts on the basis of life-times.

Table 1.4.2: (Diffuse) tin use and accumulation in Sweden (1975, in ton).

Tin use	Tonnes, 1975	Life-time	Accumulation
Solder	338	10	3,380
Tinned plate	286	5 ?	1,430
Other metal applications	117	5 ?	585
Other applications, partially organotin	221	See organotin	1,600
Subtotal diffuse tin	962		6,995
Copper alloys	338	10 ?	3,380
Total tin	1,300		10,375

Organotin
KemI has produced several estimates of the new use of organotin, all somewhere between 230 and 280 tons a year (KemI, 1994b and KemI, 1994c)[3]. As a general remark KemI indicated that the large majority of this amount was used as stabiliser in PVC. Other uses consisted of application as a pesticide. The low volume in the latter application is confirmed by KemI's pesticide report (KemI, 1995c). The pesticides tributyltinoxide and tributyltin metacrylate accounted for 7 tons in 1994, used as anti-fouling and vacuum-impregnant. In part 1 we made a specific estimate of the total new use of organotin in PVC: 235 tons. Combining these data, we come to a new use of organotin in 1994 of 242 tons. It is unclear whether organotin in PVC has a higher accumulation rate than in antifouling, etc. Here we assume that the percentage accumulated in PVC is equal to the percentage of new use in PVC.

1.4.4 Total amounts in waste for lead and (organo)tin

Specific Swedish data on the total concentrations of lead and (organo)tin in waste are not available. For lead only, Edelman (1991) gives a rough estimate of 1,770 ton in the waste stage. Almost 100 % of the amount of organotin in waste will be from PVC, since the new use for PVC is also about 100 %. A very rough idea of the amount of lead and tin in waste can be obtained by using Dutch waste concentrations. According to Tukker (1996a), mass balance calculations for the amount of lead and tin in MSWI-slag, fly-ash and emissions to air and water show that the initial concentrations in Dutch incinerable household waste are 376 mg/kg (lead) and 64.2 mg/kg (tin). The total amount of Swedish household waste is about 3.2 million tonnes (SNV, 1995a). This results in a rough estimate of 1,200 tons lead (reasonably in line with Edelman, 1991) and 205 tons tin. If the concentration is valid for the *total* volume of Swedish waste of 35 million tons, the amounts can be up to a factor 10 higher. No value for household waste can be extrapolated for other waste without additional indications. In order not to underestimate the contribution of PVC, we used the data calculated for household waste only.

It is unclear whether organotin in PVC has a higher accumulation rate or whether, compared to the new use, it shows up in different amounts in the waste stage than in antifouling etc. Here we assume that the percentage in waste flows from PVC is equal to the percentage new use in PVC.

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[3] A report from 1991 still indicates high uses of organotin for other purposes than PVC (KemI, 1991). We have assumed that these data are outdated and that newer KemI reports give a better view on the situation.

1.4.5 Review

Table 1.4.3 gives a review of the data generated in this paragraph. The uncertainties are indicated as well. Phthalates are not indicated, since they are only used for PVC. All emissions, accumulation and new use is thus for 100 % caused by PVC.

Table 1.4.3: Emissions, new use, accumulation and waste by substance

Substance Aspect	Dioxins	Mercury	Lead	Tin	Organotin
Emission to air (1992)	26.5 g	1,300 kg	365 ton	1.7 ton[2]	~ 0
Emission to air (2000)	id.	1,100 kg	30 ton	?	~ 0
Emission to water (1990)	0.5-1.3 g[2]	220 kg	14 ton	7.2 ton[2]	300 kg
Emission to water (1995)	id.	?	11 ton	?	?
New use (total)	-	-	34,400	1,300 ton	242[4] ton
New use (diffuse)	-	-	7,900	962	[4]
Accumulation (total)	-	-	364,700[1]	10,375[1]	[4]
Accumulation (diffuse)	-	-	94,700[1]	6,995 ton[1]	[4]
To waste	-	-	1200[3]	205 ton[3]	

- Not relevant for this study
- ? Targets unknown

Uncertainties are lower than a factor 1, unless stated otherwise below.

1 Uncertainty factor 2-3

2 Uncertainty factor 2-3 or higher

3 Uncertainty factor 10, probably too low an estimate

4 The relative new use in PVC is 97 % of the Swedish total. It is assumed that for accumulation and waste this figure is similar.

1.5 Distance to target weighting factors

1.5.1 Introduction

Weighting normalised scores on environmental themes is one of the most controversial aspects in LCA. First, it is to a large extent a subjective procedure. Second, the uncertainties in the inventory and characterization are often already so important, that calculating the 'single number for environmental damage' will just result in a figure that gives a false sense of certainty. This was one of the reasons why weighting was *not included in this study*. For the sake of completeness, in this methodological part we merely review some commonly applied weighting methods.

Classification and normalisation result in scores for each theme which are expressed as fractions of the overall score for each environmental theme over a given period in a defined region. These scores have to be weighted if they are to be expressed in a uniform unit. There are a number of options for weighting environmental themes (Lindeijer, 1995a)[4]:

1. panel methods;
2. monetary methods based on the extent of the damage;
3. methods based on target levels for the various themes.

The first method was used in the VNCI McKinsey study. A panel was asked to assign weights to various environmental themes (VNCI/McKinsey, 1991). The second method assumes that the score on a given environmental theme can be expressed in terms of damage (i.e. costs). In essence, this method continues the emission-impact chain to the actual impact and its costs. These are then used for weighting. In the last method it is assumed that a weighting factor is a function of the current load level and a target level on a given theme. The assumption is that a major discrepancy from the target level is relatively serious and therefore means that a relatively high weight is assigned to the theme. At present, the Netherlands Ministry of Housing, Spatial Planning and Environment (VROM) is investigating whether the third method should be adopted as a standard in the Netherlands (RMB, 1994).

1.5.2 Distance to target principle

The distance to target principle (DTT) assumes that the weighting factor for a theme is a function of the current load level and a target level. Such a function can be described by a wide range of formulas (e.g. (Heijungs, 1994a and 1994b), (Tukker, 1994a and 1994b), (Mueller Wenck, 1995) and (SETAC-WIA, 1994)). A widely used formula, which has been proposed by VROM for adoption as a standard is (Adriaanse, 1993; Sas, 1994; RMB, 1994):

$$W_i = \frac{A_i}{T_i} \qquad (20)$$

Where:

W_i is the weight of theme i;

A_i is the current total score in Sweden on theme i (in 1990);

T_i is the target for the total score in Sweden on theme i (e.g. a level of sustainability or policy objective in the year 2000).

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[4] Other weighting methods are based on the costs or exergy needed to prevent an intervention. In this case assumptions have to be made about the prevention techniques used. In these methods, the evaluation is not based on a valuation of the environment at such, instead, they focus on "utility" in terms of costs or exergy requirements.

The extent to which the intrinsic magnitude of a given environmental problem is expressed in a DTT weighting factor is currently under discussion. In Formula 20 above, it is assumed that at the target level the seriousness of the environmental problems with respect to theme A are equal to those with respect to theme B. In that case the weighting factors for themes A and B will both be equal to 1. This assumption can be defended if it is assumed that the intrinsic seriousness of an environmental problem is incorporated in the policy targets. It is assumed that a stricter standard will be imposed on intrinsically serious problems than on problems which are intrinsically less serious. In addition to the DTT weighting factor the CE uses an inter-effect factor of 1 for the intrinsic seriousness of a given environmental problem. This factor was adopted because VROM considers all policy themes to be equally important (Sas, 1994). According to Guinée (1995), the inter-effect factors are not currently available and determining them would require a separate study. For this reason he used an inter-effect factor of 1 for all environmental problems. This approach was also adopted in this study.

The distance to target approach can be used to determine a large number of weighting sets. The basis for the selection of the target level may vary, e.g. a policy objective or level of sustainability. Alternatively, a range of geographical levels at which the current and target levels are compared could be chosen. Recently, the Dutch Council for Environmental Management proposed the use of levels of sustainability as a basis for the weighting process. However, this has not yet been put into practice in terms of proposed weighting factors or levels of sustainability (RMB, 1994).

1.5.3 Motivation for excluding weighting in this study

For information, table 1.5.1 gives a set of DTT weighting factors developed in the Dutch chlorine chain study and which has been extended for the work on the Dutch National Hazardous Waste Management plan (Tukker et al., 1995b; Tukker, 1996a). The weighting factors are based on the environmental pressure in 1990 and the political targets for 2000 in the Netherlands.

However, the text in the preceding paragraphs makes clear that there are still many normative choices in weighting. At this moment, there is no scientific, social or political agreement on how to perform the weighting step, not even in the Netherlands itself, let alone at European level. Maybe such consensus will never be reached. For these reasons, the Nordic Guidelines on LCA (Lindfors, 1995) are quite hesitant about

Table 1.5.1: Distance to target weighting factors, inter-effect factors and the resulting total weights per theme on the basis of Dutch policy goals for 2000

THEME	LEVEL 1990[1]	TARGET 2000[1]	DTT FACTOR	INTEREFF. FACTOR	WEIGHT	WEIGHT REL. TO GREEN-HOUSE EFF.
hum. toxicity	$242\ 10^{17}$	$139\ 10^{17}$	1.7	0.5	0.85	0.7
aq. ecotoxicity	id.	id.	1.7	0.5	0.85	0.7
acidification	$1.02\ 10^{9}$	$396\ 10^{6}$	2.6	1	2.6	2.2
ozone depl.	$11.3\ 10^{6}$	$0.32\ 10^{6}$	35.3	1	35.3	30
greenhouse eff.	$2.44\ 10^{11}$	$2.05\ 10^{11}$	1.2	1	1.2	1
smog formation	$447\ 10^{6}$	$194\ 10^{6}$	2.3	1	2.3	1.9
nutrification	$961\ 10^{6}$	$345\ 10^{6}$	2.8	1	2.8	2.3
odour	1	0.5	2.0	1	2.0	1.8
landfill	15.5/16.7	5.0	3.1	1	3.1	2.6

[1] Units: see text. The classification and characterization for the Dutch *policy* themes are not equal to the CML *LCA*-themes for toxicity, smog formation and odour. DTT factors for these LCA themes derived on the basis of Dutch Policy goals are very provisional (see also Tukker, 1995b). In this PVC-study we split up toxicity in three instead of two themes, like in the Dutch chlorine chain study: human toxicity, aquatic ecotoxicity and terrestrial ecotoxity. We therefore used an intereffect factor of 0.33 instead of 0.3, resulting in a weight of 0.57 instead of 0.85.

the presentation of weighted results in an LCA study. In this project, we felt that presenting unweighted scores provided sufficient insight to formulate policy conclusions with regard to the environmental management of the PVC chain. Adding tables or graphs with weighted scores would not lead to additional insight, and might possibly only lead to additional discussion and/or confusion. We therefore did not include a weighting step in the main report. However, we decided to present some thoughts about the weighting step in this background document, so that anyone interested can perform a weighting themselves.

1.6 Benchmarking the environmental performance of an economic sector

1.6.1 Introduction

To assess whether a target group performs well or poorly in environmental terms its environmental performance has to be compared with that of other target groups. Such a benchmarking system provides a correction for the fact that large-scale activities will naturally have a greater environmental impact than small-scale activities. Thus, a way has to be found to establish a link between the environmental impact of a target group

and the scale of its economic activities. A number of allocation systems could be used (Buitenkamp, 1992; Buise, 1993; Smeets, 1994):

- added value of the production;
- number of employees;
- use of raw materials;
- physical production (e.g. expressed in tonnes of materials).

Naturally, the selected basis for comparison will always leave some room for discussion. Such allocation methods are also inflexible. Conceivably, each target group might be assessed on the basis of an average, acceptable environmental impact. Taken to extremes, the production of drinking water or steel might be compared with the average water or steel consumption in e.g. the European economy. In macroeconomic terms it might be more useful to permit defined sectors a higher or lower than average environmental impact on a given theme depending on their function. This is similar to the differences between capital-intensive and labour-intensive industries. The essence is that the policy targets (or alternatively, the levels of sustainability) are not exceeded at the macro level. A Pareto optimum may be found between these limits, at which the impact on each theme varies greatly between the target groups.

1.6.2 A benchmark for the PVC chain

The overall PVC throughput in Swedish society is about 100 kt/y; including additives and plasticisers this might be about 150 kt/y. The total material throughput of the Swedish society is approximated by the throughput of the most important bulk materials, listed in Table 1.6.1.

For *coal, oil and gas* the values in PJ for domestic use given in the energy-balance for Sweden in table 117 of the Statistisk Årsbok have been recalculated in tons (SCB, 1996). The transfer factors for this calculation have been taken from Sas (1994). *Wood production* has been taken from table 100 in (SCB, 1996). This value is given in m^3; a specific weight of 0.9 tons/m^3 has been used to calculate the mass flow in tons. The production of *iron ore, copper ore* and *concrete* have been taken from table 108 on industrial production in (SCB, 1996). *Livestock production* (animal food) and *yield* (grains and vegetables) have been taken from tables 91 and 86 respectively. The yield is given in metric tons; it has been assumed that this equals about 1 ton weight. All production figures of other materials and goods in the Statistisk Årsbok are at least an order of magnitude lower than each of the bulk materials listed here.

The total mass flow calculated is about 95,000 ktons. The wood production accounts for half of this figure. If we had taken the figure for *processed* wood in Sweden instead of produced wood this figure would have been considerably lower. This fact alone indicates that the figure as such is not very certain. However, we have the

impression that the 95,000 could be accurate to within at least a factor of 2 or 3, which is sufficient for the purpose of this report.

With a total PVC flow of about 150 ktons, PVC amounts to approximately 0.15 % of the material flows in Sweden. This can be considered as an environmental impact benchmark. Given the method by which this percentage was determined it should be regarded as a provisional, indicative figure.

Table 1.6.1: Production-related flow of materials in Sweden

BULK MATERIAL	Producti-on/use	Transfer-factor	Mass flow in Sweden in ktons/yr in 1994
oil	737 10^{15} J	1 ton = 42,5 GJ	17.300
coal	98 10^{15} J	1 ton = 29,3 GJ	3.300
natural gas	32 10^{15} J	1 ton = 39,9 GJ	802
concrete/cement			3.000
iron ore			10.350
copper ore			332
wood (1992/1993)	51.3* 10^6 m^3	1 ton = 0,9 m^3	46.170
food/fodder			
- animal			4.049
- vegetable			10.633
TOTAL			95.000

Another approach is possible on the basis of economic value. The Swedish national product is about 1,400 billion kroner (in 1991 prices) or 220 billion U.S. dollars (SCB, 1996). Table 1.6.2 gives an estimate of the turnover for pipes, flooring and cables of 2,900 MKr, taken from Drakenberg (1994). Those PVC applications cover about half of the market. If it is assumed that the other half of the market has a similar turnover, the total turnover could be about 5,800 MKr. However, this turnover can certainly not be allocated entirely to the system being studied here. For example, in the cable production the use of copper is not included. In the flooring production, the use of all kinds of auxiliary materials is not included. Even for the materials and inputs *inventoried* in the study, like energy and plasticisers, the production stage is not included in the study and thus the *value* of these materials and inputs should be left out. Thus only a part of the turnover given here can be allocated to the material PVC as such. Making a reliable estimate would demand a considerable effort. However, in our opinion taking 25 to 50 % of the 5,800 MKr as the added value of the system studied would mean a maximum mistake of a factor of 2 to 3. This seems adequate for the purpose of the calculation. With this correction factor, an added value of 1,450 to

2,900 MKr can be calculated, which is 0.1-0.2 % of the Swedish GNP. This is more or less in accordance with the value calculated on the basis of mass flows above.

Table 1.6.2: Added value of the system under study

MARKET	Amount	Turnover
Pipes	25,400	475 MKr
Flooring	15,000	725 MKr.
Cables	12,900	1,700 MKr.
Other (estimate)	50,000	2,900 MKr.
SUBTOTAL		5,800 MKr
Correction factor for allocation to the studied PVC system		25-50 %
TOTAL		1,450-2,900 MKr.

2 Reviewed toxicity equivalency factors for a number of substances

In this study new equivalency factors for human- and ecotoxicity, developed by CML and RIVM, are used (Guinée, 1996). The former equivalency factors given in the CML Guide (Heijungs, 1992) were based only on intrinsic toxicity of the compounds (e.g. No Effect Concentrations or NECs). In the CML/RIVM report new equivalency factors were calculated with the aid of the USES (Uniform System for the Evaluation of Substances) computer model which was developed at RIVM. This model makes allowance for diffusion, degradation and persistence which can be considered an important improvement in the calculation of equivalency factors. However, during this study it became clear that for some substances which are important within the study, better input data for the USES model was available. Therefore the input data for USES that were used for these substances in the CML/RIVM report were controlled against a number of available databases. This was done for the substances that made the highest contribution to toxicity scores: the phthalates DEHP, DIDP and BBP, VCM, EDC, hexachlorobenzene and dioxins. The databases we used were:

- UICLID: database of hedset's of the EC
- ISIS: substance database of the engineering firm HASKONING
- Verschueren: handbook of environmental data
- DOSE: database of environmental data
- Howard: handbook of degradation rates

The comparison made clear that for 4 of the 7 substances checked a number of the input parameters of USES had to be changed. On the basis of the comparison we chose a 'most probable' average value for the the input parameters, and calculated new equivalency factors on that basis. These values will be further referred to as the (new) USES-set.

The ECPI (European Council for Plasticisers and Intermediates) delivered their own set of data for phthalates which was based on even more recent, partly unpublished work. For the solubility and the bioconcentration factors large differences were found between the databases and the ECPI data. Since these parameters are very important for the calculation of the equivalency factors we decided to use two sets of equivalency factors for the toxicity themes: the ECPI set and the set from the databases mentioned above.
The ECPI dataset was based on the work of Staples et al. (1996). Staples et al. find that the differences between the solubility data for phthalates found in literature is mainly due to a number of experimental difficulties:

- inability to separate colloidal emulsions of undissolved chemical in the aqueous phase
- contamination from laboratory plastics
- withdrawing samples through the surface film in which phthalates accumulate

All these problems can lead to experimental artifacts that yield measured values that overestimate the true water solubility (Staples et al, 1996).

Furthermore, Staples' group find that predictions of BCFs that rely on simple correlations with chemical hydrophobicity (as used in USES) are inappropriate when applied to phthalates since these models are based on data for persistent chemicals. Lowenbach et al. (1995) suggested an alternative approach by extrapolating BCFs found with rodents to cattle (Staples et al, 1996). The BCFs found via this approach are a factor 1000 to 10,000 lower than the BCFs calculated by USES on the basis of hydrophobicity.

Table 2.1 reviews the USES input parameters and resulting equivalency factors in the original work of Guinée (1996), alternative input parameters from the cited literature and databases, our choice and the resulting equivalency factors, and the ECPI set and the resulting equivalency factors. The table makes clear that for 4 of the 7 substances checked a number of the original USES parameters are inappropriate, resulting in high differences in equivalency factors for similar substances like DEHP and DIDP. The 'new' USES set results for VCM, DIDP, DEHP and BBP in quite different values than with the 'old' USES set. The rather difficult to explain differences between the phthalates from the 'old' USES set are greatly diminished. Only BBP has a relatively high equivalency factor for emissions to water, which is logical due to its relatively high water solubility compared to DEHP and DIDP. The differences in solubilities in the new USES set and the ECPI set have little effect on the scores. The differences in BCFs are only important for human toxicity, and result in equivalency factors up to a factor 1000 or more lower for the ECPI set.

Reviewing this analysis and the comparison in the tables, we can conclude that the following problems exist with the calculation of the toxicity equivalency factors:

1. in some cases, like here EDC, just wrong basic data might have been used;
2. in some cases, like here VCM and the phthalates, inappropriate default values for substance properties have been used;
3. in other cases, literature is not unanimous about the value of key substance properties that form an input in USES;
4. finally, for some substances it is questionable if the current literature values of substance properties have been assessed with an appropriate method (like here solubility and BCFs for phthalates).

All these problems can lead to high uncertainties in equivalency factors. CML has written a letter that warns the users of the equivalency factors specifically for the problems related to point 2 (CML, 1996).

Table 2.1: Toxicity equivalency factors from Guinée et al. 1996, corrected after comparison with databases for physical chemical properties and factors based on phys.chem. properties given by the ECPI

Substance	CAS-no.	Equivalency factor	Guinée et al.	corrected Guinée et al.	ECPI	Guinée et al.	corrected Guinée et al.	ECPI	Guinée et al.	corrected Guinée et al.	ECPI
		for:	Air			Surf. water			soil generic		
di(2-ethylhexyl)phthalate	117-81-7	aquatic ecosystem AETP	2,90E-01	3,91E-03	6,53E-03	4,70E+01	2,49E-01	2,49E-01	8,30E-04	2,99E-09	7,93E-09
		terrestrial ecosystem TETP	1,10E+02	2,07E+03	4,09E+03	2,60E+01	2,71E+00	9,05E+00	3,60E+03	1,30E+04	1,30E+04
		human HTP	1,40E+01	9,19E+04	6,45E+01	5,90E+01	2,84E+02	1,57E-01	2,90E+00	1,69E+00	1,46E-01
diisodecylphthalate	26761-40-0	aquatic ecosystem AETP	4,50E+01	1,69E-02	1,69E-02	1,50E+03	5,88E-01	5,88E-01	9,20E-02	6,42E-10	6,80E-09
		terrestrial ecosystem TETP	6,40E+05	4,59E+02	4,56E+02	5,30E+01	8,24E-04	3,81E-01	1,70E+06	1,30E+03	1,30E+03
		human HTP	4,60E+02	1,97E+05	6,86E+01	1,60E+02	2,22E+02	6,18E-02	1,50E+02	5,59E+00	1,51E-01
benzylbutylphthalate	85-68-7	aquatic ecosystem AETP	3,10E+01	7,71E-01	7,38E-01	4,80E+02	2,73E+01	2,73E+01	6,40E-01	2,50E-05	2,65E-05
		terrestrial ecosystem TETP	1,20E+05	2,85E+02	2,61E+02	7,60E+04	3,75E-02	6,69E-02	3,60E+05	8,54E+02	8,54E+02
		human HTP	4,30E+02	3,76E+02	3,43E+02	1,00E+02	4,26E+00	1,59E-01	8,30E+01	1,91E-01	1,91E-01
1,2-dichloroethane	107-06-2	aquatic ecosystem AETP	1,20E-02	6,73E-02	6,73E-02	5,70E-01	7,29E-01	7,29E-01	1,20E-02	5,52E-02	5,52E-02
		terrestrial ecosystem TETP	4,20E+00	1,14E+01	1,14E+01	4,20E+00	1,14E+01	1,14E+01	2,00E+02	4,15E+02	4,15E+02
		human HTP	6,90E+01	8,46E+01	8,46E+01	6,90E+01	8,68E+01	8,68E+01	6,80E+01	7,97E+01	7,97E+01
hexachlorobenzene	118-74-1	aquatic ecosystem AETP	4,30E+00	3,20E-03	3,20E-03	1,60E+02	1,27E+02	1,27E+02	3,40E+00	3,20E-03	3,20E-03
		terrestrial ecosystem TETP	4,00E+05	6,15E+00	6,15E+00	3,10E+05	5,00E+00	5,00E+00	2,40E+06	2,17E+01	2,17E+01
		human HTP	4,40E+03	2,77E+02	2,77E+02	7,30E+03	3,42E+03	3,42E+03	1,10E+04	2,85E+02	2,85E+02
2,3,7,8-TCDD	1746-01-6	aquatic ecosystem AETP	7,20E+06	7,23E+06	7,23E+06	1,50E+08	1,48E+08	1,48E+08	2,10E+06	2,06E+06	2,06E+06
		terrestrial ecosystem TETP	2,60E+08	2,62E+08	2,62E+08	2,90E+07	2,89E+07	2,89E+07	5,10E+08	5,10E+08	5,10E+08
		human HTP	2,60E+10	2,57E+10	2,57E+10	3,20E+09	3,15E+09	3,15E+09	4,70E+10	4,65E+10	4,65E+10
vinylchloride	75-01-4	aquatic ecosystem AETP	3,10E-03	3,88E-05	3,88E-05	4,60E-01	4,59E-01	4,59E-01	3,10E-03	3,85E-05	3,85E-05
		terrestrial ecosystem TETP	4,50E-02	6,46E-04	6,46E-04	4,40E-02	6,42E-04	6,42E-04	1,10E+00	1,26E+00	1,26E+00
		human HTP	5,50E+02	6,78E+00	6,78E+00	5,50E+02	1,11E+01	1,11E+01	5,40E+02	8,40E+00	8,40E+00

DEHP

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point Celsius	vapour pressure Pa(20-25C)	octanol-water p.c. -	water solubility mg.l-1	photodeg in air DT50 in d	hydrolisis in water DT50 in d	biodeg in water DT50 in d	biodeg in soil DT50 in d	biodeg in STP DT50 in d	BCF fish l.kg-1	BCF worm kg.kg-1	BCF plant stem kg.kg-1	BCF plant root kg.kg-1	BCF plant air m3.kg-1	BCF meat d.kg-1	BCF milk d.kg-1
used as USES	**-50**	**0,0001**	**7,5**	**0,04**	**1**	**1000000**	**5**	**90**	**0,01**	**1,47E+06**	**29,37**	**5E-07**	**0,06683**	**6,80E+05**	**0,7943**	**0,2512**
used as ECPI	**-50**	**1,3E-05**	**7,5**	**0,003**	**1**	**1000000**	**5**	**90**	**0,01**	**120**	**29,37**	**0,02**	**0,02**	**7830**	**0,0002**	**0,00006**
ECPI	-50	1,3E-05	7,5	0,003	0,2-2	?	5	90	0,01	120		0,02	0,02	7830	0,0002	0,00006
Guinée et al.	-50	0,00086	5,24	0,045	1	1000000	5	295,9	0,00963	8083	29,37	5E-07	0,06683	53860	0,7943	0,2512
IUCLID	-50	0,00086	4,8	1,007-0,04	0,75	1934,5	14-21	95		842						
	-46	0,0001	5,03	<0,01	1	194,9	32	(140)		155-886						
	-42	800-1000	7,861	0,041	0,6-2,2		11	(218)		460						
			8	0,041			>100	(47)		114						
			8,39	0,046			30	(43)								
			9,64	<0,34			14									
							275									
ISIS	-50	4,5E-05	3-4	0,35	22		7									
Verschueren	-55		7,86	0,04			5			130						
			9,64	0,285			14			850						
										886						
										300						
DOSE	-50	0,00086	5,11	0,3	143		5									
Howard					0,1-1		5-23	5-23								

= default
= estimated by USES
= estimated by USES given the input 'readily biodegradable'

BBP

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point Celsius	vapour pressure Pa	octanol-water p.c. -	water solubility mg.l-1	photodeg in air DT50 in d	hydrolisis in water DT50 in d	biodeg in water DT50 in d	biodeg in soil DT50 in d	biodeg in STP DT50 in d	BCF fish l.kg-1	BCF worm kg.kg-1	BCF plant stem kg.kg-1	BCF plant root kg.kg-1	BCF plant air m3.kg-1	BCF meat d.kg-1	BCF milk d.kg-1
used as USES	**-35**	**4,1E-05**	**4,91**	**2,7**	**2,75**	**1000000**	**2**	**59,2**	**0,01**	**3781**	**29,36**	**0,00804**	**0,2645**	**1,21E+05**	**0,00204**	**0,00065**
used as ECPI	**-35**	**0,00008**	**4,91**	**2,7**	**2,75**		**2**	**59,2**	**0,01**	**19**	**29,36**	**0,00804**	**0,2645**	**14150**	**0,00204**	**0,00065**
ECPI	-35	0,00008	4,91	2,7	0,5-5	?	<2(2)	59,2	0,01	19						
Guinée et al.	-35	0,0011	4,91	2,7	160	1000000	1000	27690	1000000	3781	29,36	0,00804	0,2645	14150	0,00204	0,00065
IUCLID	-35	4,1E-05	4,91	2,7		>100										
		0,00008	4,78	0,71												
			3,57													
ISIS							<3,5	59		663						
Verschueren	-35	0,00115	4,91	2,9	>100	>100	<3,5			772						
			4,78							663						
			4,05													
DOSE	-35	0,00115	4,91	1,2			<2			663						
Howard					0,25-2,5	not imp.	1-7	1-7								

= default
= estimated by USES
= estimated by USES given the input 'not readily biodegradable'

DIDP

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point Celsius	vapour pressure Pa	octanol-water p.c. -	water solubility mg.l-1	photodeg in air DT50 in d	hydrolisis in water DT50 in d	biodeg in water DT50 in d	biodeg in soil DT50 in d	biodeg in STP DT50 in d	BCF fish l.kg-1	BCF worm kg.kg-1	BCF plant stem kg.kg-1	BCF plant root kg.kg-1	BCF plant air m3.kg-1	BCF meat d.kg-1	BCF milk d.kg-1
used as USES	**-46**	**4E-06**	**8**	**0,5**	**1**	**1000000**	**7**	**90**	**0,01**	**4,65E+06**	**29,37**	**4,1E-08**	**0,05128**	**2,55E+07**	**2,512**	**0,7943**
used as ECPI	**-46**	**4E-06**	**8**	**0,001**	**1**	**1000000**	**7**	**90**	**0,01**	**100**	**29,37**	**0,02**	**0,02**	**7830**	**0,0002**	**0,00006**
ECPI	-46	4E-06	>8	0,001	0,2-2	?	7	90	0,01	100		0,02	0,02	7830	0,0002	0,00006
Guinée et al.	-37	4,3E-07	4,91	1,19	160	1000000	1000	27690	1000000	3781	29,37	4,1E-08	0,05128	54170	2,512	0,7943
IUCLID	-41	10	4,91	0,4	0,4					<3,6						
			<11,77	0,1						<14,4						
			11,8	1,19												
ISIS						no data										
Verschueren						no data										
DOSE						confirmed biodegradable										
Howard						no data										

= default
= estimated by USES
= estimated by USES given the input 'not readiy biodegradable'

EDC

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point	vapour pressure	octanol-water p.c.	water solubility	photodeg in air	hydrolisis in water	biodeg in water	biodeg in soil	biodeg in STP	BCF fish	BCF worm	BCF plant stem	BCF plant root	BCF plant air	BCF meat	BCF milk
	Celsius	Pa	-	mg.l-1	DT50 in d	DT50 in d	DT50 in d	DT50 in d	DT50 in d	l.kg-1	kg.kg-1	kg.kg-1	kg.kg-1	m3.kg-1	d.kg-1	d.kg-1
used	**-35**	**8500**	**1,48**	**8700**	**100**	1000000	**1000**	**21,71**	**1000000**	**1,405**	**13,91**	**1,391**	**2,278**	**0,01774**	**7,59E-07**	**2,40E-07**
Guinée et al.	-98	24000	1,92	5500	80,21	1000000	1000	39,75	1000000	3,869	20,93	1,056	1,74	0,00727	2,09E-06	6,61E-07
IUCLID	-35 -36	8700 8600 8500 8330	1,46 1,45	1000-9000	12-120	10000000										
ISIS	-35	8500	1,48	8600	119		14 21 175									
Verschueren	-35	8131		8690				34								
Howard					12-122	402	100-180	100-180								

= default
= estimated by USES
= estimated by USES given the input 'not readily biodegradable'

VCM

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point Celsius	vapour pressure Pa(20-25C)	octanol-water p.c. -	water solubility mg.l-1	photodeg in air DT50 in d	hydrolisis in water DT50 in d	biodeg in water DT50 in d	biodeg in soil DT50 in d	biodeg in STP DT50 in d	BCF fish l.kg-1	BCF worm kg.kg-1	BCF plant stem kg.kg-1	BCF plant root kg.kg-1	BCF plant air m3.kg-1	BCF meat d.kg-1	BCF milk d.kg-1
used	**-160,2**	**333000**	**1**	**1100**	**2**	**1000000**	**1000**	**14,83**	**1000000**	**0,4651**	**6,743**	**1,414**	**2,691**	**0,00091**	**2,51E-07**	**7,94E-08**
Guinée et al.	-160,2	333000	0,6	1100	160	1000000	1000	12,78	1000000	0,1852	3,115	1,129	2,84	0,00086	1,00E-07	3,16E-08
IUCLID	153,8	333000	1,36	1100	2,2-2,7	jaren				<10						
	153,7	340000	1,58	915	2,4											
ISIS	153,7	337000		2500	2-77											
Verschueren	160	355000		1100	1,2-4											
	153															
Howard							30-180	30-180								

= default
= estimated by USES
= estimated by USES given the input 'not readily biodegradable'

HCB

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point	vapour pressure	octanol-water p.c.	water solubility	photodeg in air	hydrolisis in water	biodeg in water	biodeg in soil	biodeg in STP	BCF fish	BCF worm	BCF plant stem	BCF plant root	BCF plant air	BCF meat	BCF milk
	Celsius	Pa	-	mg.l-1	DT50 in d	DT50 in d	DT50 in d	DT50 in d	DT50 in d	l.kg-1	kg.kg-1	kg.kg-1	kg.kg-1	m3.kg-1	d.kg-1	d.kg-1
used	**230**	**2,40E-04**	**5,4**	**,0,005**	**11,41**	**1000000**	**1000**	**85500**	**1000000**	**1,17E+04**	**29,37**	**0,00196**	**0,2036**	**0,2177**	**0,00631**	**0,002**
Guinée et al.	230	0,0015	5,4	0,005	11,41	1000000	1000	85500	1000000	1,17E+04	29,37	0,00196	0,2036	271,6	0,00631	0,002
IUCLID	225-231	0,00011 0,0000145 0,000017 0,000012 0,0000241 0,000025	5-6,92	0,007 <0,01		geen	geen	geen								
ISIS	231		6,18	0,01	23 not likely	not likely	not likely	not likely								
Verschueren	229		6,18	0,006						7760 1160-3740 14454-954993						
Howard					156-1400		986-2080	986-2080								

= default
= estimated by USES
= estimated by USES given the input 'not readily biodegradable'

TCDD

	physico-chemical properties				degradation		biodegradation			Bioaccumulation						
	melting point Celsius	vapour pressure Pa	octanol-water p.c. -	water solubility mg.l-1	photodeg in air DT50 in d	hydrolisis in water DT50 in d	biodeg in water DT50 in d	biodeg in soil DT50 in d	biodeg in STP DT50 in d	BCF fish l.kg-1	BCF worm kg.kg-1	BCF plant stem kg.kg-1	BCF plant root kg.kg-1	BCF plant air m3.kg-1	BCF meat d.kg-1	BCF milk d.kg-1
used	**305**	**4,5E-06**	**6,8**	**0,0002**	**180**	**1000000**	**1000**	**2150000**	**1000000**	**4,00E+04**	**29,37**	**1,17E-05**	**0,09684**	**4047**	**0,1585**	**0,05012**
Guinée et al.	305	4,5E-06	6,8	0,0002	180	1000000	1000	2150000	1000000	4,00E+04	29,37	1,17E-05	0,09684	4047	0,1585	0,05012
IUCLID																
ISIS	305-306			0,0002		not likely	not likely	not likely								
Verschueren	305-306					600										
Howard					1-3,4		380-450	380-450								

= default
= estimated by USES
= estimated by USES given the input 'not readily biodegradable'

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Annex 2: Graphs with some provisional theme scores

Due to the fact they contain relatively new information, we decided to publish several graphs with theme scores not in the main report. This annex reviews the following figures:

1. A breakdown of (normalised) theme scores by PVC chain section for a number of themes that were not shown in the main report.

2. A breakdown of weighted theme scores by PVC chain section, making use of DTT weights on the basis of Dutch policy goals. The figure is not suitable for publication outside the context of this annex and the rest of the report.

3. Contribution of the scores of the PVC chain to the Swedish total. This figure is in principle suitable for comparing the environmental burden with those of the 'average' Swedish economic activity. The toxicity themes especially have very high uncertainties, either in the Swedish total or the score of the PVC chain.

Figure A2.1: DtT-weighted scores as % of Swedish total, USES-factors, 1994 (white: PVC; black: ethene/electricity)

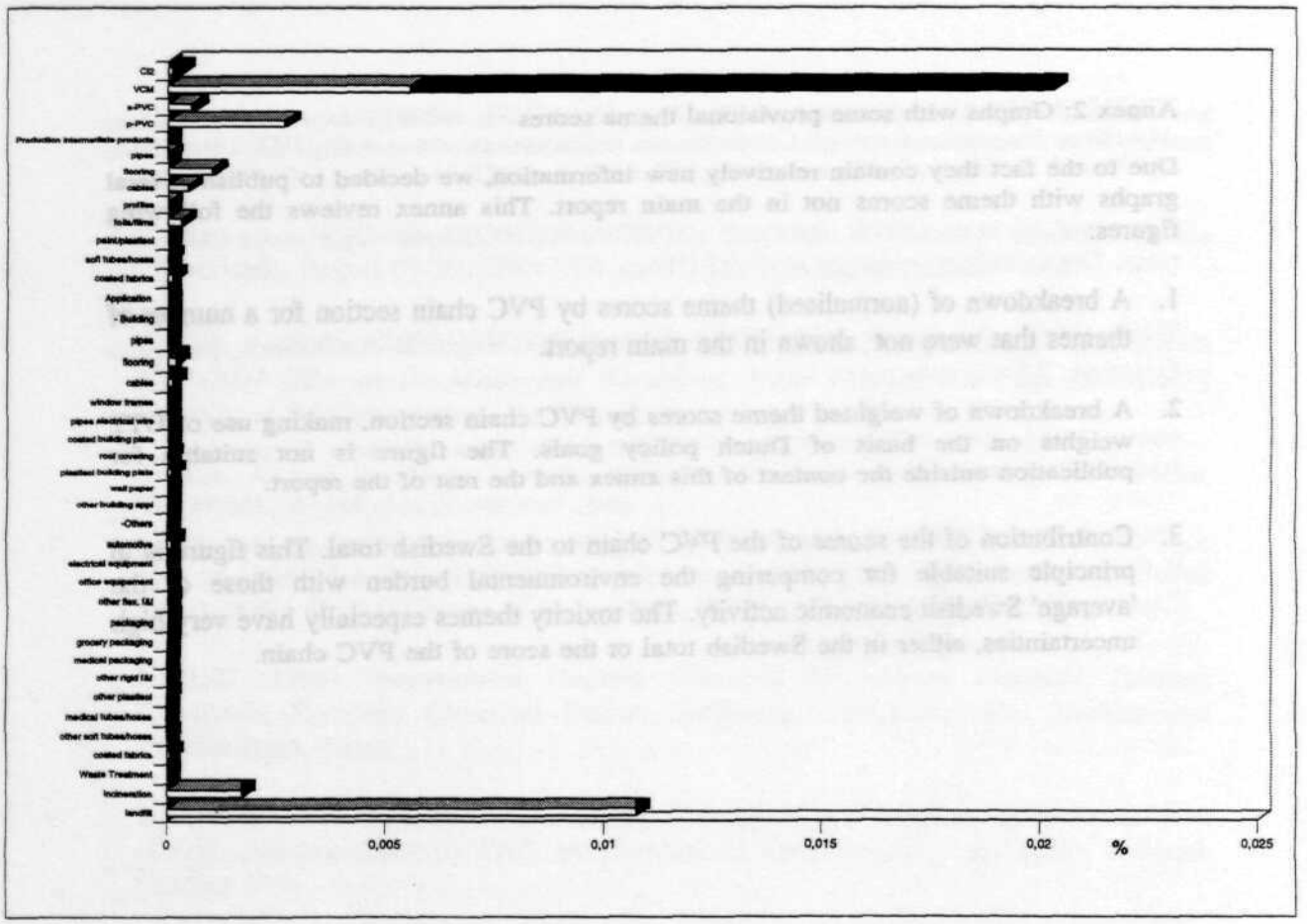


Figure A2.2: DtT-weighted scores as % of Swedish total, ECPI-factors, 1994 (white: PVC; black: ethene/electricity)

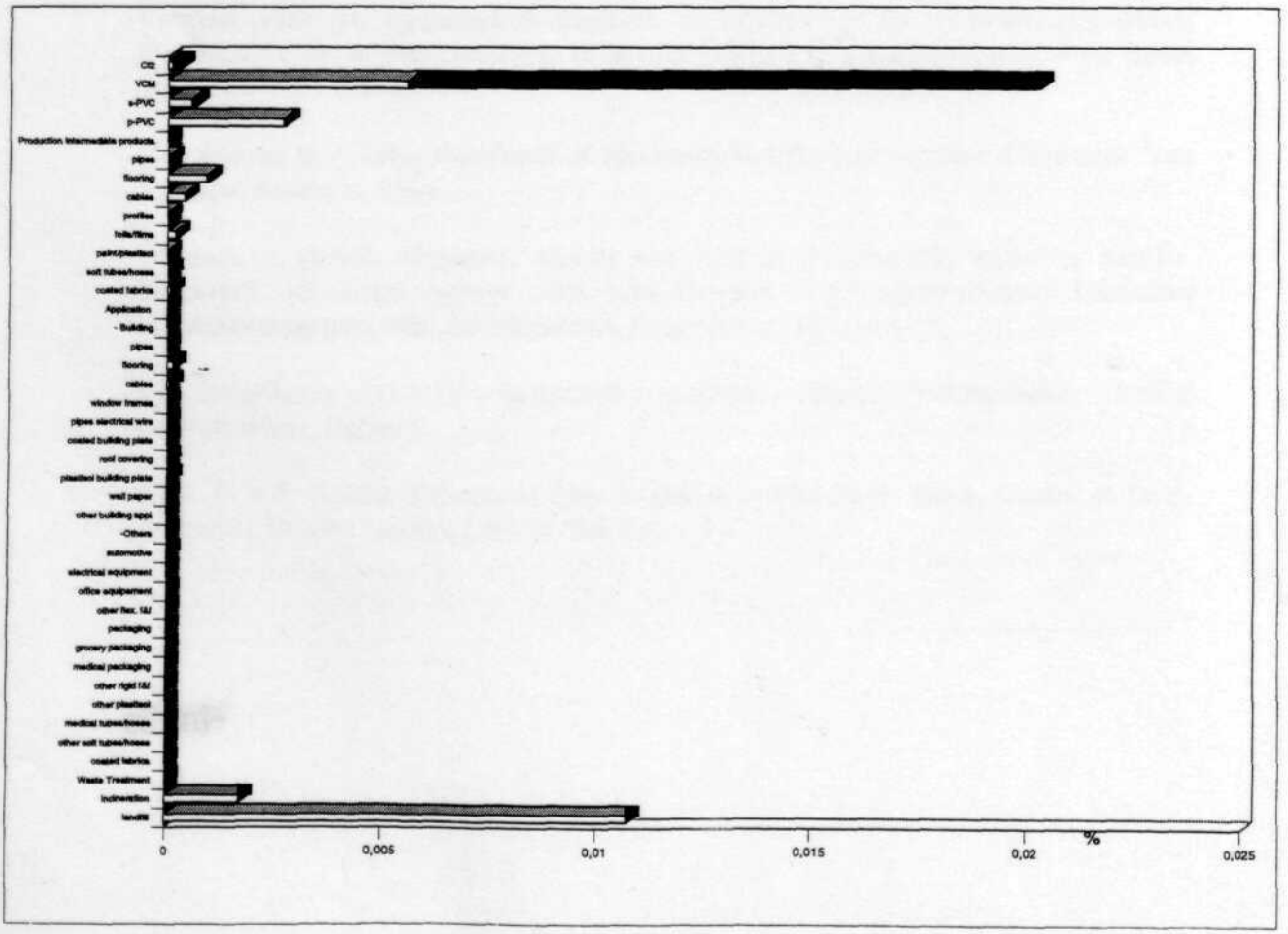


Figure A2.3: Contribution to Swedish total theme scores, USES-factors, 1994 (white: PVC; black: ethene/electricity)

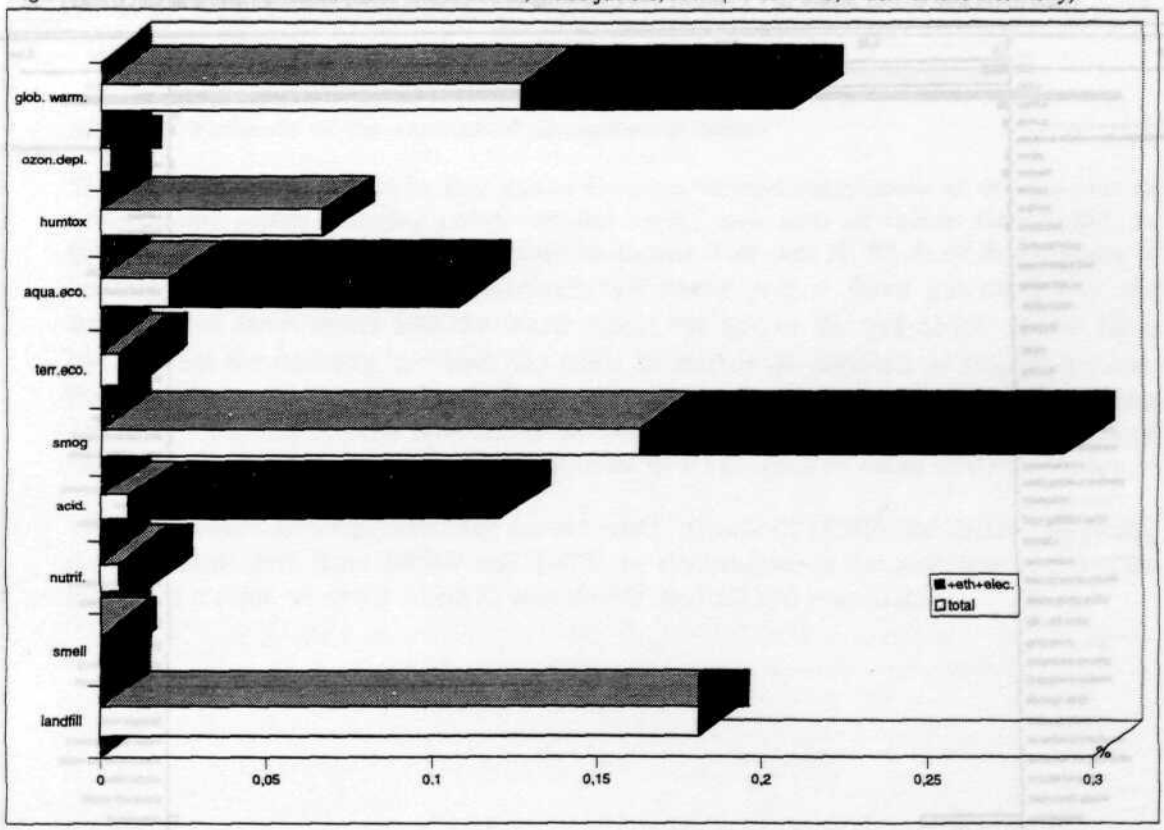


Figure A2.4: Contribution to Swedish total theme scores, ECPI-factors, 1994 (white: PVC; black: ethene/electricity)

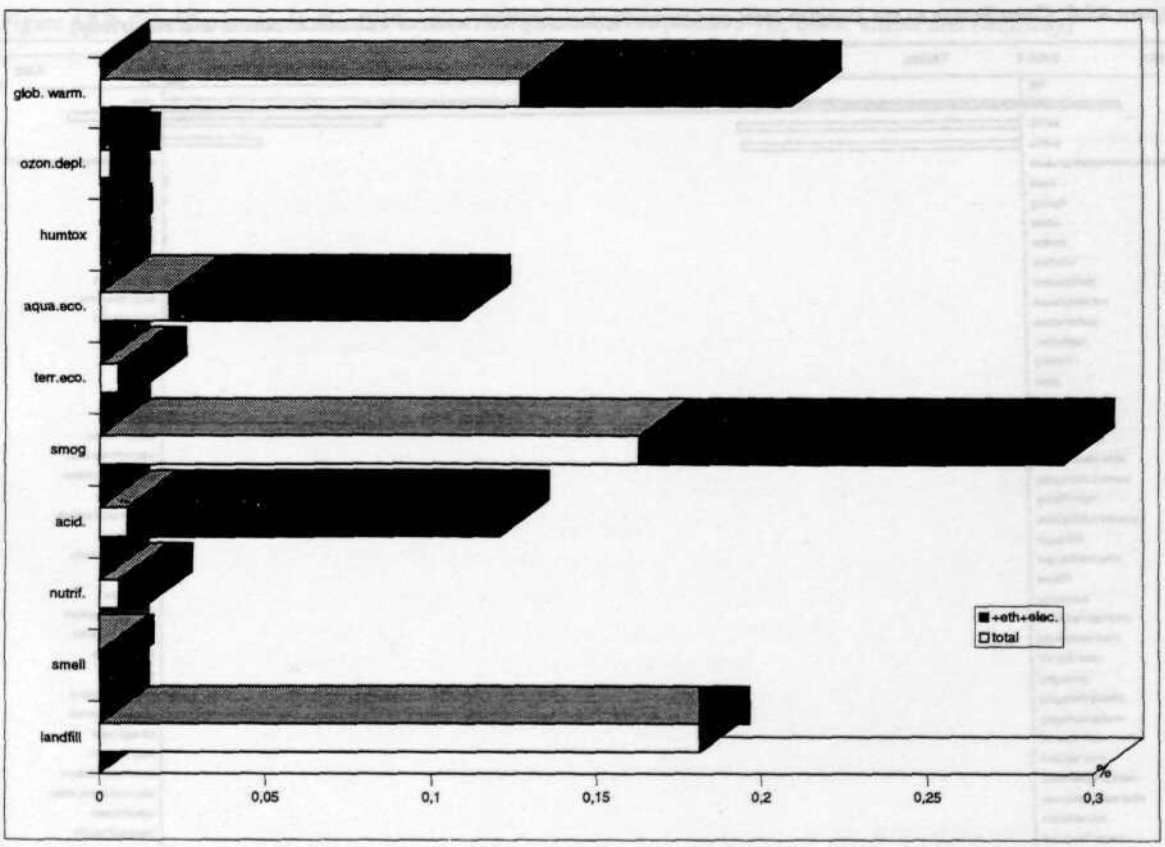


Figure A2.5: Contribution to the Swedish total on Global Warming, 1994 (white: PVC; black: ethene and electricity)

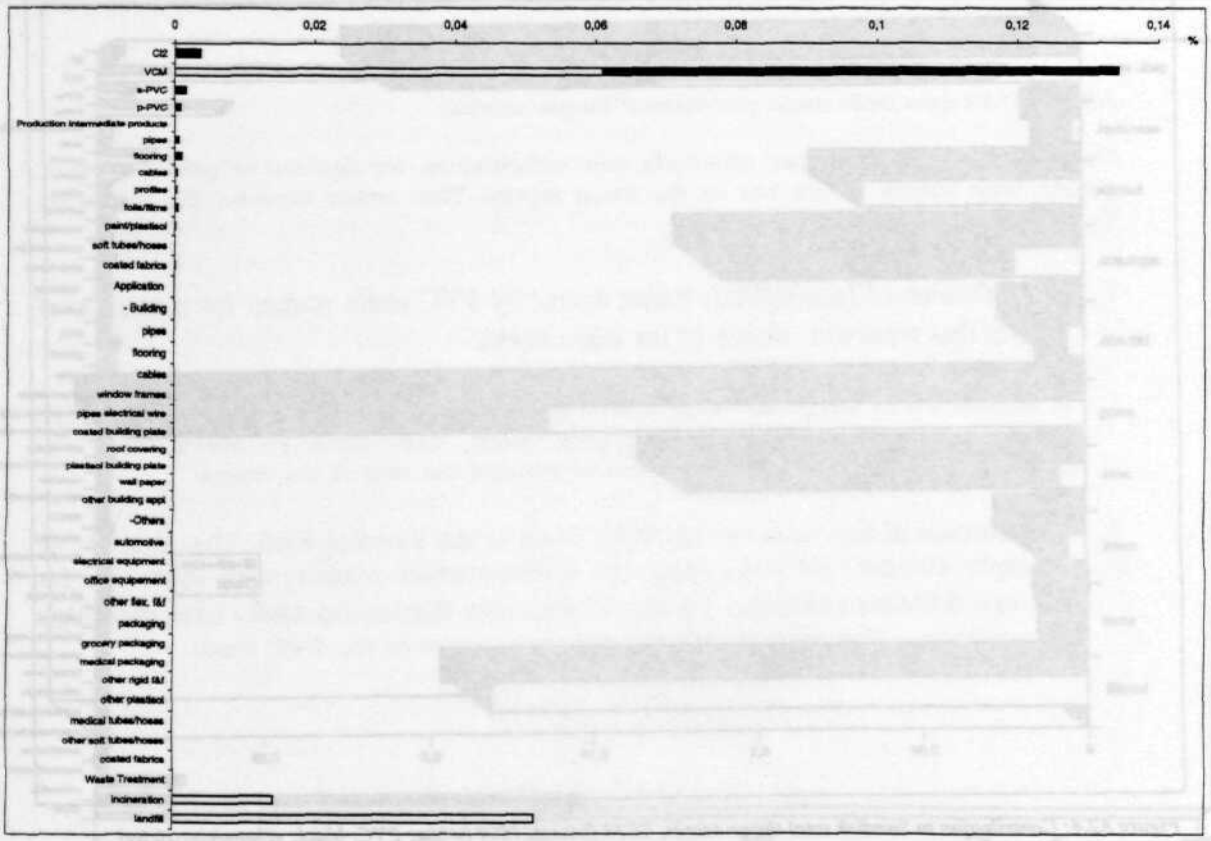


Figure A2.6: Contribution to the Swedish total on Ozone Depletion, 1994 (white: PVC; black: ethene and electricity)

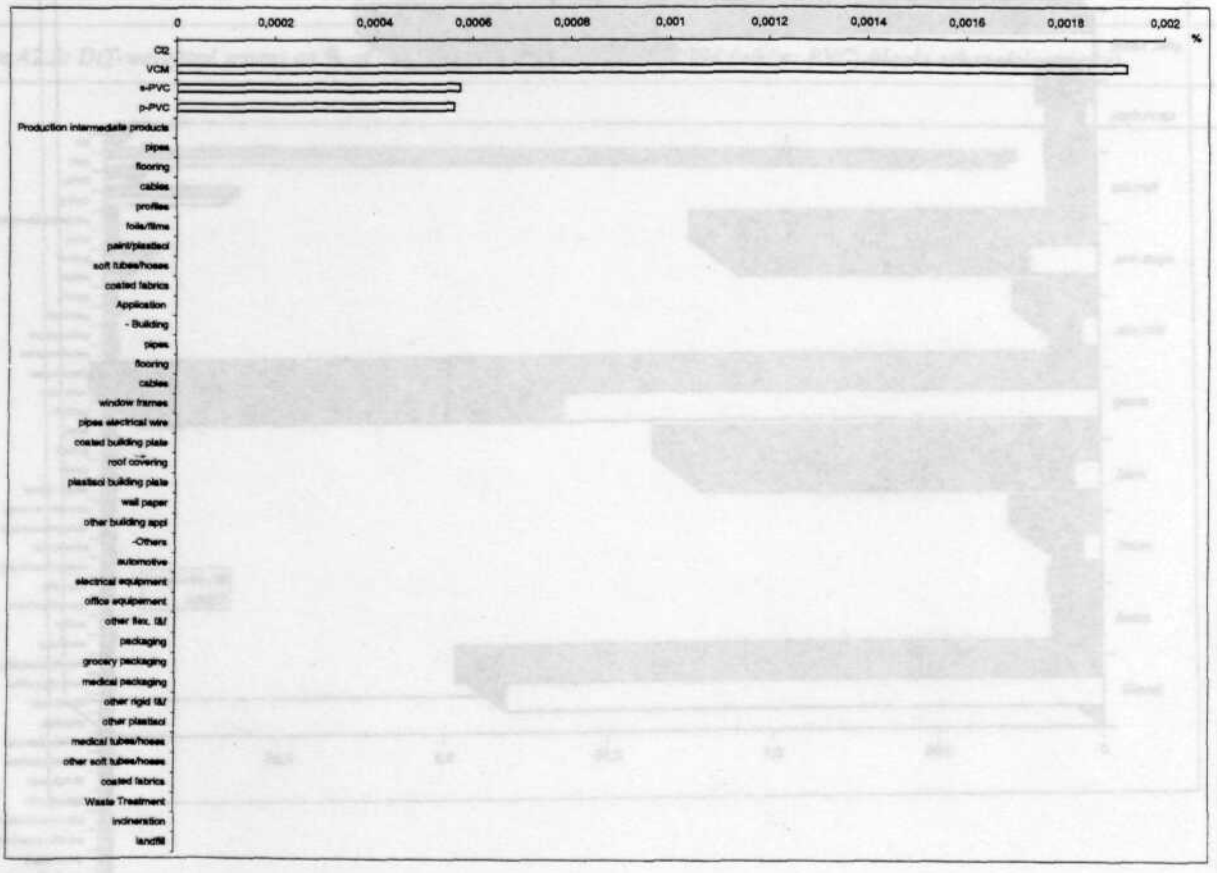


Figure A2.7: Contribution to the Swedish total on Acidification, 1994 (white: PVC; black: ethene and electricity)

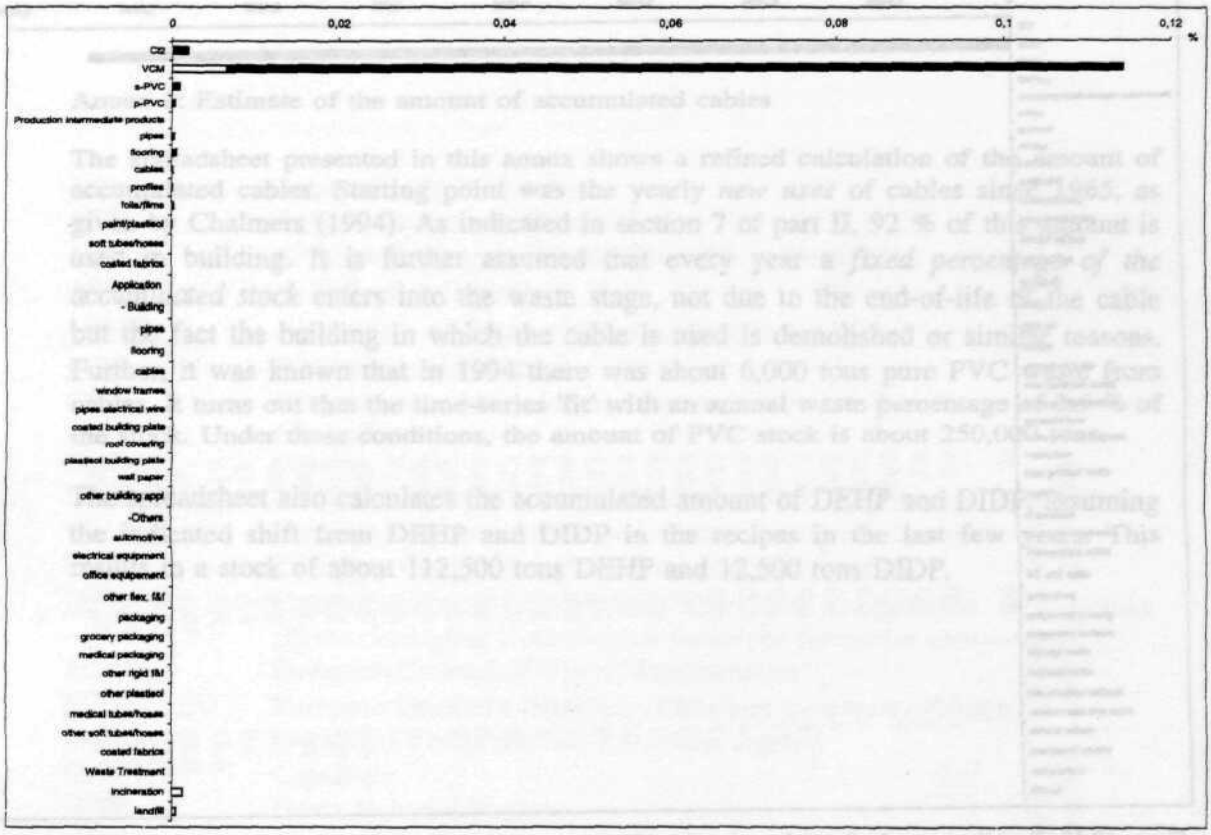


Figure A2.8: Contribution to the Swedish total on Nutrification, 1994 (white: PVC; black: ethene and electricity)

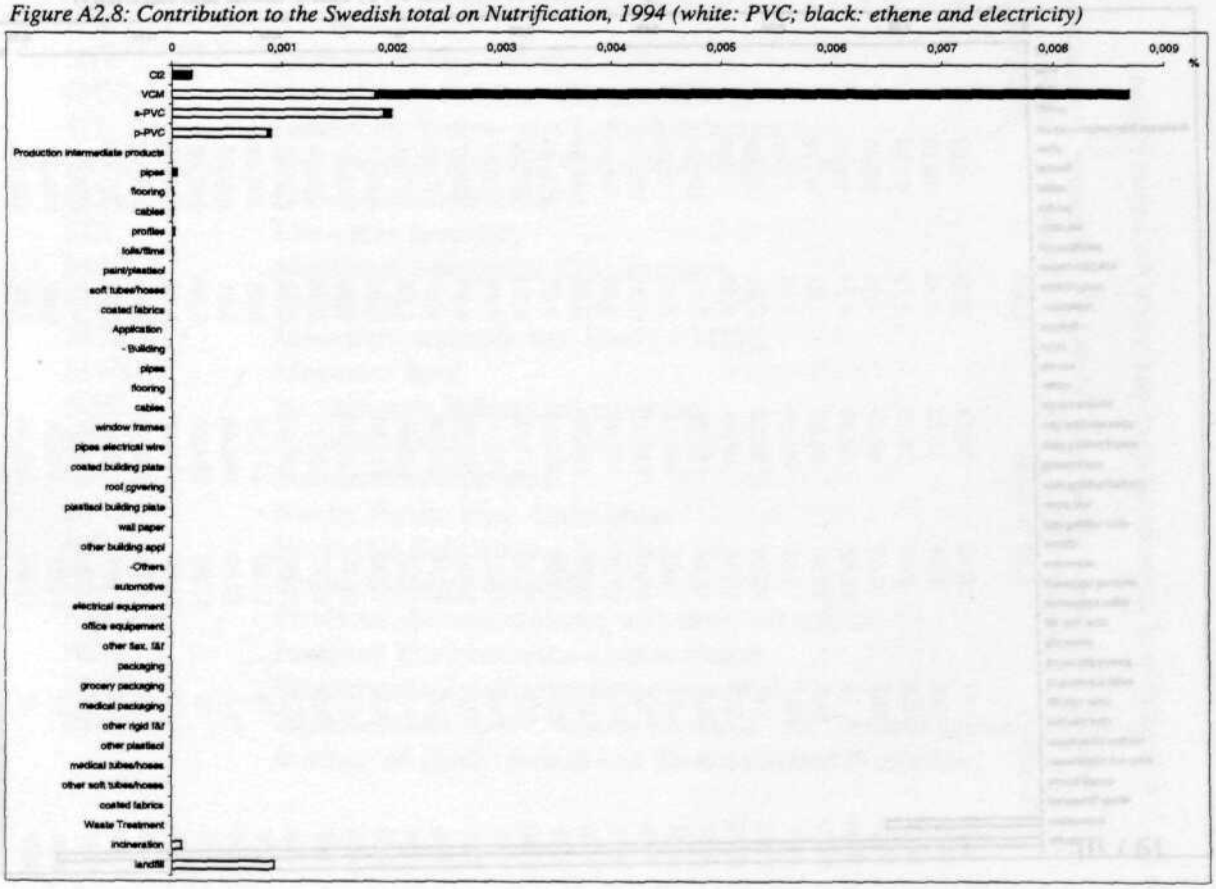


Figure A2.9: Contribution to the Swedish total on Odour, 1994 (white: PVC; black: ethene and electricity)

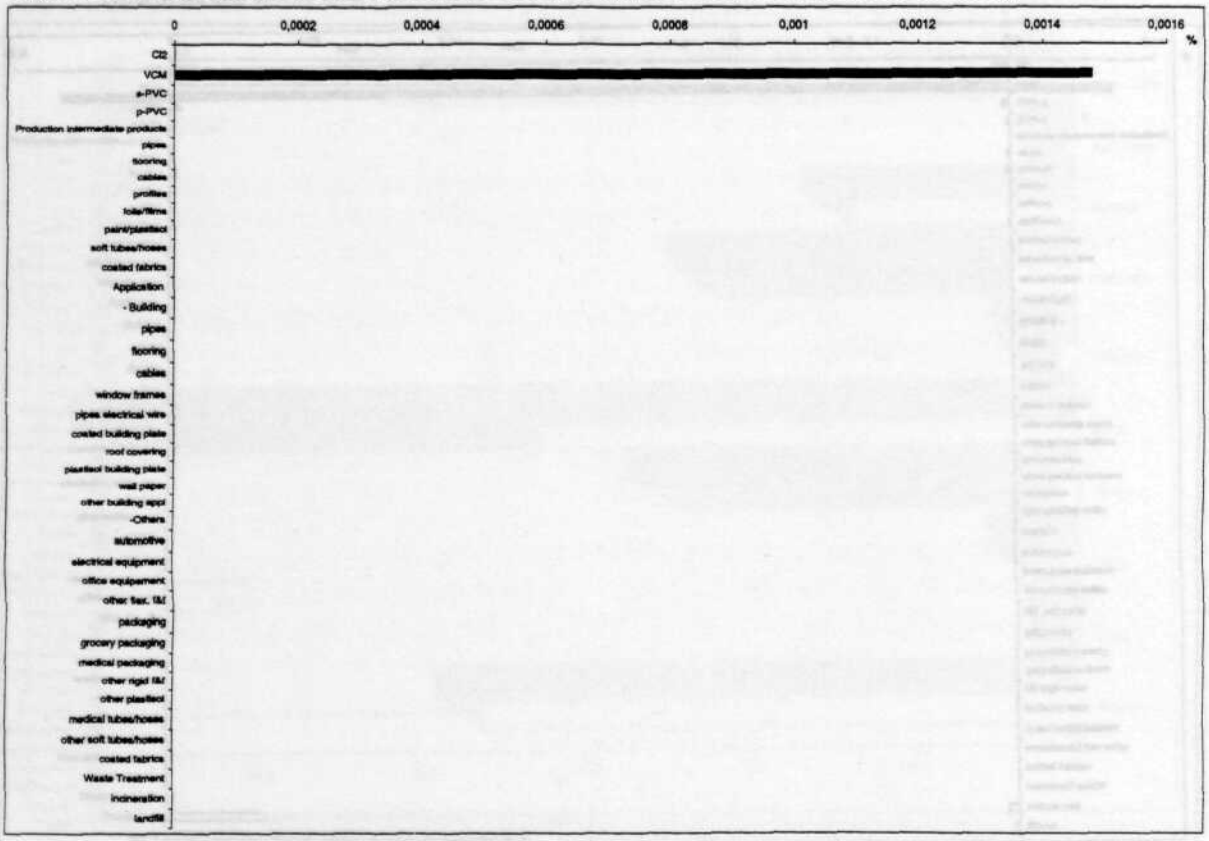
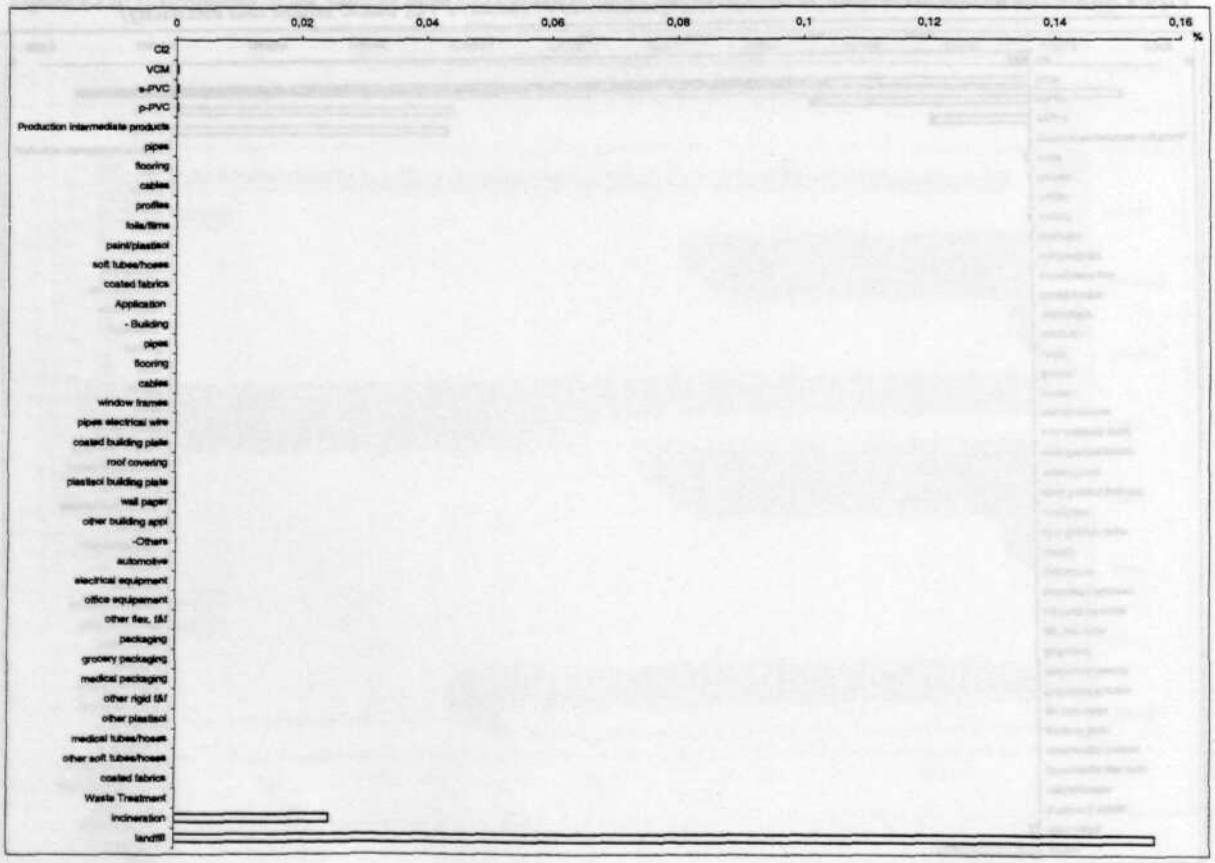


Figure A2.10: Contribution to the Swedish total on Landfill volume, 1994 (white: PVC; black: ethene and electricity)



Annex 3: Estimate of the amount of accumulated cables

The spreadsheet presented in this annex shows a refined calculation of the amount of accumulated cables. Starting point was the yearly *new uses* of cables since 1965, as given by Chalmers (1994). As indicated in section 7 of part II, 92 % of this amount is used in building. It is further assumed that every year a *fixed percentage of the accumulated stock* enters into the waste stage, not due to the end-of-life of the cable but the fact the building in which the cable is used is demolished or similar reasons. Further, it was known that in 1994 there was about 6,000 tons pure PVC waste from cables. It turns out that the time-series 'fit' with an annual waste percentage of 2.5 % of the stock. Under these conditions, the amount of PVC stock is about 250,000 tons.

The spreadsheet also calculates the accumulated amount of DEHP and DIDP, assuming the indicated shift from DEHP and DIDP in the recipes in the last few years. This results in a stock of about 112,500 tons DEHP and 12,500 tons DIDP.

Table B3.1: Calculation of amounts of stock and waste for cables (in ton)

Year	Production	New use in building	Stock year x	Waste	Remaining stock x+1	Fraction plasticised with DIDP	DEHP	Stock DIDP (50 phr)	Stock DEHP (50 phr)
	% building	0,92	% waste	0,025		(TNO-estimate)		50	50
1965	11	10,12	10,12	0,25	9,87	0	1	0,0	5,1
1966	10,6	9,75	19,62	0,49	19,13	0	1	0,0	9,8
1967	10,3	9,48	28,60	0,72	27,89	0	1	0,0	14,3
1968	13,7	12,60	40,49	1,01	39,48	0	1	0,0	20,2
1969	14	12,88	52,36	1,31	51,05	0	1	0,0	26,2
1970	14,5	13,34	64,39	1,61	62,78	0	1	0,0	32,2
1971	14,7	13,52	76,31	1,91	74,40	0	1	0,0	38,2
1972	15	13,80	88,20	2,20	85,99	0	1	0,0	44,1
1973	15,2	13,98	99,98	2,50	97,48	0	1	0,0	50,0
1974	14,5	13,34	110,82	2,77	108,05	0	1	0,0	55,4
1975	14,2	13,06	121,11	3,03	118,08	0	1	0,0	60,6
1976	15,7	14,44	132,53	3,31	129,21	0	1	0,0	66,3
1977	14,4	13,25	142,46	3,56	138,90	0	1	0,0	71,2
1978	15,2	13,98	152,89	3,82	149,06	0	1	0,0	76,4
1979	15,3	14,08	163,14	4,08	159,06	0	1	0,0	81,6
1980	13,5	12,42	171,48	4,29	167,19	0	1	0,0	85,7
1981	13,6	12,51	179,71	4,49	175,21	0	1	0,0	89,9
1982	13,2	12,14	187,36	4,68	182,67	0	1	0,0	93,7
1983	13,2	12,14	194,82	4,87	189,95	0	1	0,0	97,4
1984	14,2	13,06	203,01	5,08	197,94	0	1	0,0	101,5
1985	14,7	13,52	211,46	5,29	206,17	0,1	0,9	0,7	105,1
1986	14,1	12,97	219,14	5,48	213,67	0,2	0,8	2,0	107,6
1987	13,9	12,79	226,45	5,66	220,79	0,3	0,7	3,8	109,4
1988	14,7	13,52	234,32	5,86	228,46	0,4	0,6	6,4	110,7
1989	14	12,88	241,34	6,03	235,31	0,5	0,5	9,5	111,2
1990	14,3	13,16	248,46	6,21	242,25	0,53	0,47	12,7	111,5
1991	13	11,96	254,21	6,36	247,85	0,67	0,33	16,4	110,7
1992	12,3	11,32	259,17	6,48	252,69	0,67	0,33	19,8	109,8
1993	12	11,04	263,73	6,59	257,14	0,67	0,33	23,0	108,9
1994	12	11,04	268,18	6,70	261,47	0,67	0,33	26,1	108,0
1995	12	11,04	272,51	6,81	265,70				
1996	12	11,04	276,74	6,92	269,82				
1997	12	11,04	280,86	7,02	273,84				

Annex 4: Glossary

ADI	Acceptable Daily Intake
AETP	Aquatic Ecotoxicity Potential
AOO	Afvaloverlegorgaan *[Waste Consultancy Body]*
AP	Acidification Potential
APME	Association of Plastics Manufacterers in Europe
BCF	Biocincentration Factor
CML	Centrum voor Milieukunde Leiden *[Centre for Environmental Science Leiden]*
DM	German Mark
ECA	Ecotoxicologische classificatiefactor voor aquatische ecosystemen *[Ecotoxicological classification factor for aquatic ecosystems]*
ECPI	European Council for Plasticisers and Intermediates
ECT	Ecotoxicologische classificatiefactor voor terrestrische ecosystemen *[Ecotoxicological classification factor for terrestrial ecosystems]*
ECVM	European Council of Vinyl Manufacturers
EG, EC, EU	Europese Gemeenschap/Unie *[European Community/Union]*
(S)EPA	(Swedish) Environmental Protection Agency
GJ	GigaJoule
GNP	Gross National Product
GWP	Global Warming Potential
HT	Human Toxicity
HTP	Human Toxicity Potential
IPCC	International Panel on Climate Change
IVL	Institut för Vatten- och Luftsvårdsforskning
KemI	The Swedish National Chemicals Inspectorate
LCA	Life-cycle assessment
LCI	Life-cycle inventory
MAC	Maximum Acceptable Concentration
MJ	MegaJoule
MTR	Maximum tolerable risk level (= MTR)
MWh	Megawatt hour
NEC	No (adverse) Effect Concentration
NEI	No-effect Intake
NP	Nutrification Potential
NPG	Nordic Plastic Pipe Association
NR	Negligible Risk level
ODP	Ozone depletion potential
PBT's	Persistent, bioaccumulating and toxic substances
PEC	Predicted Environmental Concentration
POCP	Photochemical oxidant creation potential
RIVM	Rijksinstituut voor Volksgezondheid en Milieuhygiëne *[National Institute of Public Health and Environmental Protection]*

RIZA	Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwater *[Institute for Inland Water Management and Waste Water Treatment]*
SCB	Statistiska Centralbyrån (Statistics Sweden)
SETAC	Society of Environmental Toxicology and Chemistry
SFA	Substance Flow Analysis
SKr	Swedish Kroner
SNV	Statens Naturvårdsverket (Swedish EPA)
SPIN	SamenwerkingsProject Industriele procesbeschrijvingen in Nederland *[Collaborative Project for the Description of industrial processes in the Netherlands]*
TDI	Tolerable Daily Intake
TEQ	Toxicity equivalent
TETP	Terrestrial EcoToxicity Potential
TJ	TeraJoule
TNO	Nederlandse Organisatie voor Toegepast Natuurwetenschappelijk Onderzoek *[Netherlands Organisation for Applied Scientific Research]*
UK	United Kingdom
USES	Uniform system for the evaluation of substances

Chemical compounds

AOX	Adsorbable organic halogens
Ba	Barium
BOD	Biological oxygen demand
BBP	Butylbenzylphthalate
Ca	Calcium
5CB	Pentachlorobenzene
CH_4	Methane
CFC	Chlorofluorocarbons
Cl_2	Chlorine
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CPs	Chlorophenoles
Cr	Chromium
DEHP	Diethylhexylphthalate
DIDP	Diisodecylphthalate
EDC	Ethylenedichloride
EOCl	Extractable organic chlorine
EOX	Extractable organic halogens
HCB	Hexachlorobenzene
N	Nitrogen
NaClO	Sodium Hypochlorite
NaOH	Sodium Hydroxide
NM-VOC	Non-methane VOC

NO_x	Nitrogenoxide
OCS	Octachlorostyrene
P	Phosphorus
Pb	Lead
PCB	Polychlorobiphenyl
PCP	Pentachlorophenol
PVC	Polyvinylchloride
p-PVC	Paste PVC
s-PVC	Suspension PVC
Sn(S)	Tin(sulfide)
TCB	Trichlorobenzene
TCDD	Dioxins
VCM	Vinylchloride monomer
VOC	Volatile organic compounds

RIZA	Rijksinstituut voor Integraal Zoetwaterbeheer [National Institute for Inland Water Management]
SCB	Statistiska Centralbyrån (Statistics Sweden)
SETAC	Society of Environmental Toxicology and Chemistry
SFA	Substance Flow Analysis
SKr	Swedish Kronor
SNV	Statens Naturvårdsverket (Swedish Environmental Protection Agency)
SPIN	SamenwerkingsProject Inductrie [Collaborative Project for the Dutch Industrial processing in the Netherlands]
TDI	Tolerable Daily Intake
TEQ	Toxicity equivalent
TEDP	Terrestrial EcoToxicity Potential
TJ	Terajoule
TNO	Nederlandse Organisatie voor Toegepast Natuurwetenschappelijk Onderzoek [Netherlands Organisation for Applied Scientific Research]
UK	United Kingdom
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Chemical compounds

AOX	Adsorbable organic halogens
Ba	Barium
BOD	Biological oxygen demand
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Ca	Calcium
C_6H_6	Benzene
CH_4	Methane
CFC	Chlorofluorocarbons
Cl_2	Chlorine
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CPs	Chlorophenoles
Cr	Chromium
DEHP	Diethylhexylphthalate
DIDP	Diisodecylphthalate
EDC	Ethylenedichloride
EOCl	Extractable organic chlorine
EOX	Extractable organic halogens
HCB	Hexachlorobenzene
N	Nitrogen
NaClO	Sodium Hypochlorite
NaOH	Sodium Hydroxide
NM-VOC	Non-methane VOC

Annex 5: Some definitions

Chlorinated micro-pollutants:	Unidentified emissions which have at present neither been shown to cause harm, nor to be harmless.
Daywater	Water from paved surfaces, roofs of buildings, etc., discharged via a sewage system
Emission:	release of a substance from a production process or application, into the environment (soil, water or atmosphere), excluding the flow of chloride (i.e. salt) into the water.
Swedish total score:	score of all emissions in Sweden for a given theme (both chlorine-containing and chlorine-free).
Normalisation:	expressing the score of an emission for a theme as a fraction of the Swedish total score on that theme.
PBTs:	emissions of persistent, bioaccumulating and toxic substances which were not traced.
Process:	production process or application in which a chlorine compound is used for a single, defined application.
Score:	number calculated with the LCA classification method developed by the CML, it is the contribution of an emission to the specified environmental theme.
Segment:	production or application (consumption) process, or cluster of linked production or application processes in the chlorine chain; the description in Volume II was based on this structure.
Utilities:	General processes in production plants, like the central sewage treatment plant.

Annex 6: Participants at meetings of the reference group and peer review committee

Reference group
The following persons participated in one or more meetings of the reference group

B. Bergfald, Bellona, Oslo, Norway
T. Claesson, Hydro Plast AB, Stenungsund, Sweden
M. Friberg, Tarkett AB, Ronneby, Sweden
P. Hagström, Naturvårdsverket (NV, Swedish Environmental Protection Agency), Stockholm, Sweden
H-A. Haugen, Norsk Hydro ASA, Porsgrunn, Norway
A. S. Jasser,Norsk Hydro ASA, Porsgrunn, Norway
A. Konow, Tarkett AB, Ronneby, Sweden
C-M. Larsson, Kemikalieinspektionen (KemI, Swedish Chemicals Inspectorate), Solna, Sweden
E. Ljung,Kemikalieinspektionen (KemI, Swedish Chemicals Inspectorate), Solna, Sweden
L. Lundberg, PVC-Forum, Plast- och Kemibranscherna, Stockholm, Sweden
K. Mingarini, Kemikalieinspektionen (KemI, Swedish Chemicals Inspectorate), Solna, Sweden
R. Niklasson, Hydro Plast AB, Stenungsund, Sweden
P. Richardson, Kemikalieinspektionen (KemI, Swedish Chemicals Inspectorate), Solna, Sweden
A-L. Rykfors, Neste Oxo AB, Stenungsund, Sweden
M-C. Gren-Uhrgård, Hydro Plast AB, Stenungsund, Sweden

Peer review committee
dr. L.-G. Lindfors, IVL, Stockholm, Sweden (chairman)
prof. dr. R.U. Ayres, INSEAD, Fontaineblau, France
prof. dr. W. Klöpffer, C.A.U., Dreieck, Germany

Annex 7: Peer review

Critical Review Report of the Study "A PVC Substance Flow Analysis for Sweden"

Review panel: Prof. Robert U. Ayres, INSEAD, France
Prof. Walter Klöpffer, C.A.U., Germany
Mr. Lars-Gunnar Lindfors, IVL, Sweden (Chairman)

The task of the review panel

The review panel has been given the task to judge on the scientific value of the study, namely to (i) assess if the applied methodology is suitable for the purpose of the study, (ii) if a reasonable data quality is achieved, and (iii) if the methodology itself is applied properly with sufficient insight in its strengths and weaknesses.

The review panel has interpreted that task as to (i) assess the feasibility of the methodology itself in principle to meet the goals of the study, to (ii) assess how the applied methodology and assumed quality of collected data may meet the goals of the study, and (iii) if the conclusions reached are justified.

The review process

The review panel has met four times to discuss the draft reports in various stages:

On 19 April 1996, together with the commissioner of the study, Norsk Hydro, and the practitioners, CML and TNO, when the study, then close to being finalised, was introduced to the review panel.

On 30 May 1996 when the panel discussed and reviewed the first draft reports

On 10 June 1996 when the panels recommendations as how to improve the quality of the study and the report were discussed with the commissioner and the practitioners. It was then decided to extend the study to take those recommendations into consideration.

On 2 October 1996 when the second draft reports were reviewed by the panel, and some remarks of mostly editorial nature were made.

The panel received the final draft reports on 28 October 1996 and agreed on this review report on 4 November 1996.

General comments

The panel fully appreciates the short time frame given to the consultants for this project. The results obtained are impressive when this is taken into consideration. The study is basically a level I study. This means that most up-stream processes outside the main PVC chain are

omitted, and that generic or estimated data are used whenever specific data are not easily available. Various omissions forced by the limited time frame are commented upon in the reports. However, the subsequent comments made by the panel are based on the reports own merits.

In the first draft reports the panel was mainly concerned by the omission of releases of priority pollutants such as chlorinated dioxins and mercury, and emissions associated with the production of ethylene and electricity. These have been taken into consideration in the final draft reports, and the study has thereby been considerably improved. The panel did also recommend not to use a single toxicity characterisation (score) method in this type of assessment, but to base the discussion on inventory data alone, or to use several characterisation methods in parallel in order to reveal and assess the uncertainty associated with the different methods. This has also been included in the final draft reports. However, results from toxicity score calculations are still given in the final draft reports despite the fact that the researchers in their discussion clearly justify why the single score approach should not be used. That may confuse some readers. However, the uncertainties associated with that approach are discussed in detail in the reports

The panel has not checked the numerical values given in the reports. However, the panel is convinced that accurate data have been used in this study, based on the reported data sources. The calculations may be reproduced by a reader, however not easily, since all data used are reported. The methodology itself is described to a reasonable extent. In fact, methods, such as valuation methods, not used in the study and strongly recommended not to be used are also described in the reports.

Feasibility of the methodology

The study has three goals defined in the main report:

1. "To perform a Substance Flow Analysis for the Swedish situation for the material PVC and indicate the relative emissions from processes in this chain"

2. "To give a quantitative indication of the relative importance of related impacts and indicate priority PVC-chain sectors from an environmental point of view"

3. "To place the impacts related to the PVC-chain in perspective to impacts related to those of other activities in society, e.g. by comparing the environmental burdens of PVC with the national total burden and comparing the economic market share of PVC with the GNP"

The panel has no reason to believe that a full "level III-study" should not be able to fulfil these goals. However, this "extended level I-study", may according to the panel only fulfil the first goal, and the second one only for certain impact categories (environmental themes). It cannot totally fulfil the third and final goal. This is due to a combination of the applied methodology and unavoidable data gaps or major data uncertainties.

Comments on the applied methodology

Major comments by the panel are as follows:

- Upstream burdens associated with some raw materials, auxiliary materials and energy supply are deliberately omitted in a level I study. This may cause substantial deviations in estimated emissions as compared to the life-cycle perspective, as shown in the extended part of the study, where some of these have been included. This is acknowledged in the reports, but a level I study may only conclude on the relative importance of parts inside the PVC chain, restricted by the system boundaries. It may not be used to compare the PVC chain with other substance chains. A study like this one may be interpreted qualitatively, however, not quantitatively

- The methods used to aggregate emissions of eco- and human toxic compounds into "toxicity scores" have been only recently introduced to the scientific discussion. They are not agreed upon or even fully evaluated. They should not yet be used or reported in an externally directed study. (As noted, the conclusions of the study are not based on such results).

- The attempts to derive total Swedish scores for different impact categories is the weakest part of the study. This weakness is not the fault of the researchers, but is due to unavoidable uncertainties and/or data gaps. Thus, the use of normalised scores based on aggregated contributions to impact categories (environmental themes) may only be justified for GWP, acidification, eutrophication and possibly ODP

- The study would have gained from a sensitivity analysis of assumptions and omissions

Comments on the conclusions

The practitioners have clearly recognised most of the weak parts of the methodology. The conclusions are not based on the impact assessment part of the study but mainly on inventory data. The review panel finds the conclusions generally well balanced, if it can be assumed that the underlying inventory data are correct. The panel has no reason to believe that this is not the case, based on the reported data collection and quality check procedure

Recommendations

It is recommended to publish the study as completely as possible, i.e. not just the main report.

A glossary would improve the readability of the documents

More strictly defined terms should be used, e.g. polychlorinated dibenzodioxins (PCDD´s), not "dioxins"

A sensitivity analysis of assumptions and omissions would improve the quality of this study.

Concluding remarks

The researchers have taken most of the recommendations made by the review panel into consideration. In particular, the conclusions of the study are justified by the analysis itself, including the applied methodology and the assumed data quality.

The panel is impressed by the open minds of the commissioning organization and the researchers shown to the panel during the review process.

Stockholm 4 November 1996 on behalf of the review panel

Lars-Gunnar Lindfors
Chairman