REPORT

Background data and assumptions made for an LCA on creosote poles

Working report

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Summary

This report outlines one way to make a holistic ecological assessment taken the different product functionalities into account. The report describes the data used and methodological assumptions made for the life cycle assessment (LCA) on creosote poles. The creosote pole is compared to competitive alternatives made of steel and concrete. The use of the report is twofold;

1) it is aimed to be followed up by a manuscript to a scientific paper.

2) it will be used as the background LCA for an environmental product declaration (EPD).

A system approach referred to as 'product LCA' is defined here as a suitable methodology approach for both life cycle assessment (LCA) and when the result is communicated as an environmental product declaration (EPD). This methodology approach has the potential to handle recycling and disposal in an objective way and generates a result that is verifiable concerning both a temporal and spatial scale.

When LCA is used for comparative purposes and assertions, it is important that an impact assessment method is used has an environmental significance without including value-choices. Following the LCA standard ISO 14044 the presented EQO normalisation approach generates an interface where the relative importance between different environmental impact categories can be evaluated. Special attention in the EQO approach is made to generate an integrated value on ecological toxicity.

The result of the LCA illustrate that the significant aspect for poles made of steel an concrete is the contribution to climate change, while the significant aspect for creosote pole is the contribution to human toxicity. However, the over all assessment will favour the creosote poles as the ecological most sustainable alternatives in respect to the environmental quality objectives used for normalisation.

Keyword

Creosote, environmental quality objectives (EQO), EQO normalisation method, life cycle assessment (LCA), normalization, poles, transmission poles, USES-LCA.

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Foreword

The current work is initiated by Swedenergy (Svensk energi), Skanova (branch of TeliaSonera AB), Swedish Wood Preservation Institute as a supplementary work to potentially contribute to the currently ongoing work with the evaluation of creosote according by the European Commissions Biocide directive. Swedish Chemicals Agency (KEMI) is the authorized body of the work on the evaluation for approval on creosote. KEMI:s review report is a decision support on the possible inclusion of creosote in Annex I of Directive 98/8/EC as a part of the review programme referred to in Article 16(2) of the Biocide Directive.

The work performed here shall be regarded as a contribution describing one way on how a societal weighting of benefits could be performed, at least concerning ecological sustainability following the well known and scientifically established standards on the system analytic tool life cycle assessments (LCA). There are no specifications on how this weighting in the Biocide directive should be performed more then the following text as outlined in Technical Notes for Guidance (TNsG) for Annex 1 Inclusion (ECB 2002):

> "The benefits of products containing the active substance should be considered. Especially in cases where there are concerns about the acceptability of the risks, the need for and benefits of biocidal products containing the substance should be considered carefully and weighed against the acceptable level of risk."

This is a working report that will be communicated to interested parties that will make it possible to receive criticism on data used and assumptions made, before the final result is presented. The final report is aimed to support the manuscript for a scientific paper.

Stockholm, October 2009

The others

Content

1 Background

Creosote is used a wood preservative as a building material in different construction works such as poles, posts, railway sleepers and in different foundations. Creosote is based on coal tars that are by-products of the carbonization of coal to produce coke and (coke-owen) gas. This report is focusing on creosote poles which mean that the alternatives materials are mainly concrete and steel. These competitive materials have different environmental performance, where the benefits for creosote wood poles are that it is made from a renewable resource and that the discarded product is an excellent fuel. The significant environmental performance for steel and concrete is characterised by the higher energy use and its contribution to climate change by the use of fossil fuels or indirect via the electricity used in the manufacturing process. Also emissions from the iron ore and lime burning in the manufacturing process are related with carbon dioxide emissions.

Since the product alternatives have different environmental performances that contribute in different degree to the impact categories analysed, there will be a need to evaluate the overall environmental impact, i.e. the relative importance of different impact categories. Furthermore, the different functionality of the different product alternatives has to be accounted for which is handled in LCA by introducing a so called functional unit. Such a result will give a more holistic decision support that includes impact not only from the use phase but from all life cycle phases in the product's life cycle.

In order to evaluate the relative contribution from different impact categories a normalisation method was developed by IVL based on environmental quality objectives (EQO) (Erlandsson 2003a, Erlandsson and Lindfors 2003). This impact assessment method is based on assumptions on ecological sustainable conditions defined for each impact category. The EQO normalisation method follows the requirements by the LCA standard ISO 14044 on how product comparisons for public assertion can be done. The EQO normalisation method makes it possible to evaluate the relative order between different impacts categories included. This kind of system analytic result that in basic is based on a substance or material flow analysis will complement traditional environmental Risk Assessment (RA).

Concerning creosote poles in a societal perspective it is also interesting to evaluate its potential effect on human and ecological toxicity in a life cycle perspective, and moreover, the relative importance compared to other impacts. In LCA it is common to evaluate impact on human health as an integrated indicator result for all substances emitted in the product's lice cycle. In the EQO normalisation method this has also been possible for ecological toxicity by introducing a parallel methodological approach analogous to that for human toxicity. This implies – as for human toxicity – that the effect from a single substance on the ecosystem may be evaluated via pathways and an acceptable daily intake (ADI) etc. This is possible for ecological toxicity by using predicted no effect concentrations (PNEC). The results in the context of an LCA will then be expressed on the same, scale, making it possible to determine the relative importance between emissions via different recipients. The result is in an integrated assessment of ecological toxicity in the LCA case study.

This report outlines one way to make a holistic ecological assessment taking the different product functionalities into account. The report describes the data used and methodological assumptions made for the life cycle assessment (LCA) on creosote poles. The creosote pole is compared to competitive alternatives made of steel and concrete. The use of the report is twofold;
1) it is aimed to be followed up by a manuscript to a scientific paper.
2) it will be used as the background LCA for an environmental product declaration (EPD).

2 Goal and scope

The goal of the LCA in this report is to evaluate the environmental performance of creosote poles and to compare with alternative products. The object is to evaluate the relative order between the different impact categories analysed by using a life cycle impact assessment method that is based on a normalisation method that uses assumptions of environmental quality objectives (EQO). This means that direct subjective elements will be avoided and the result will therefore be suitable according to the LCA standard ISO 14044 for public assertion.

The LCA methodology applied will be based on a so called attribution LCA approach (also called book keeping LCA), that is characterised by the fact that if the emissions from all products (service and goods) were summarised, the total amount from the LCA inventory will then correspond to the amount of substances that actually is emitted in the reality. The alternative approach is the consequential approach (also called margin approach) that includes so called system expansion (an analysed system larger than the individual product system itself) and therefore includes scenario techniques like "what happens if". The LCA methodology applied here shall, when relevant, be compatible with the environmental product declaration standards for building industry ISO 21930, and the forthcoming European standards prEN 15804.

Moreover, since creosote is a biocide the potential contribution to toxicity aspects has to be included in the impact assessment analysis, in order to cover all significant environmental aspects. Complementary characterisation factors suitable for the evaluation of creosote in an LCA will therefore have to be elaborated. The scope is to use the so called EQO normalisation method developed by IVL as impact assessment method in the case study. This impact assessment method makes it possible to evaluate the relative importance of different impact categories included without introducing direct subjective elements. The normalisation method, therefore, follows the requirements of the LCA standard ISO 14044 for impact assessment results suitable for public assertion (Erlandsson and Lindfors 2003).

Besides the creosote wooden poles, alternative product solutions covered in the LCA case study are poles made of concrete or steel. The scope is limited to typical 0,4-kV transmission lines in order to minimise the contribution from different foundation alternatives which would be required for high-voltage transmission lines with higher poles. For 0,4-kV transmission lines 9-meter poles are sufficient. Poles of this height are commonly used for utility telephone lines as well. The study is limited to the poles themselves and does not include the power line or telephone wire and potential different means of attachments. Also the work to rise and demolish the lines is neglected since it will be assumed to be of equal importance for all studied alternatives.

3 General settings for a comparative product LCA

3.1 Dividing the lifecycle for generic reporting

Even if a complete LCA is reported in an EPD it is mandatory according to ISO 21 930 and prEN 15804, it is to mandatory to specify the environmental performance of the production (cradle-to-gate). For consumer products with a short and more or less well defined usage stage, it is feasible to include the usage of the product as well as the actual end-of-life fate. For some long-lived product or product will numerous filed of application, is maybe hard to define a full life cycle. However, for creosote poles the scenario settings on usage stage and end-of-life will be dependant on which country the product will bought and the national conditions for recycling or disposal.

To define a generic reporting format suitable for all kinds of products we have here merged the requirements from building EPDs (ISO 21930 and prEN 15804) and climate declaration/carbon footprint (PAS 2050) according to Table 1.

Table 1 Generic lifecycle for a product applicable for reporting the environmental performance in the context of an EPD and characterization of bases for the data.

Product stage	Usage stage			End-of-life stage		
Manufacturing, transports and raw material supply	Transport to costumer	Construction 1)	Use and maintenance 1)	Disassembly 1)	Transport of waste	Disposal or recycling
Site specific data, if available	Scenario	Scenario	Scenario	Scenario	Scenario	Scenario

1) These phases are only valid if the EPD is for a product with a well-defined application and therefore usage.

In PAS 2050 it is required to divide the manufacturing into raw materials and manufacturing. For the manufacturing it is commonly required in EPD-schemes that the manufacturing shall be based on actual site specific environmental data, while for upstream manufacturing and production of raw materials is commonly accepted to use literature or generic LCA data.

Concerning the consumer use stage for creosote poles the leaching of creosote will have to be accounted for. An emission of zinc is assumed for the steel poles. The rise of the lines and demolition works is, according to the goal and scope definition, not accounted for in this comparative LCA. The most reasonable end-of-life scenario, depending on the country where the material is bought will desire the current disposal and recycling praxis for the country selected in the scenario.

3.2 Allocation of impact between product systems when recycled

3.2.1 'Product LCA' for an unbiased evaluation

An EPD shall include as few subjective elements as possible, be verifiable and is therefore here made with a methodological analytic approach that we will call '*product LCA'* that can be characterised as an attribution approach. To start with, this implies that the environmental impacts are allocated to the products that are generated from the same process. This product LCA approach is valid also for e.g. EPDs, and means that a clear generally applied boundary setting has to be defined between the products system that share the same material resource in a recycling cascade, in order to generate an subjective information module. This boundary setting appears for material recycling and energy recovery.

The most simple product life cycle scenario appears when the product is made of primary resources and after usage left at a landfill or incinerated (i.e. without anyone using the energy generated). In these cases, all environmental burdens will be allocated to the initial and only product system. In the case of recycling, however, a specification has to be made that states how different environmental properties shall be accounted for properly, and the how and when environmental properties (i.e. burden or benefits) shall be allocated between the primary product system in relation to secondary product systems utilization of the extracted and recycled material. This dilemma on accounting for such different environmental properties is still under discussion, why the approach suggested in this report may be regarded as a contribution to this development.

Concerning open loop recycling, a number of allocation procedures may be applied under the LCA standard ISO 14044. This standard is more like a framework than a precise description concerning this methodological issue. Moreover, the standard for EPDs (ISO 14025) does not give any specification on this issue either. This might be explained by the fact that when the standard was developed, the goal of the EPDs was often limited to cover environmental quantitative LCA information only reporting for a cradle-to gate lifecycle perspective. When the aim is enlarged, an EPD as nowadays may also account for and report the performance on a full lifecycle, or with other words cradle-to-grave. Therefore, we have to make specifications on recycling. Recycling accounts in this context for open loop recycling and energy recovery, i.e. when for instance a scraped wooden product is used as fuel (if no use of the energy appears we here use the word incineration).

3.2.2 Accounting of different environmental properties

The first item that has to be accounted for in a product LCA are material flows from and to the technosphere. Secondly, it will be required to handle the inherent environmental properties (or attributes) that are associated with the material recycled or energy recovered. The most straightforward way is to state that,

all inherent properties that can be detected for a material shall follow the environmental burden or benefit of the material.

Consequently,
environmental impacts that cannot be detected as a material property (to the secondary product) will be allocated to the product system where they appear (originally product system)

The meaning of the word detected above is,
an environmental product attribute that is stored and therefore traceable as an inherent property of the material.

This means that such environmental product attributes shall be traceable and measurable and therefore without information of the history of the manufacturing etc, such as impact and benefits from the harvesting (including land use) when recycling wood.

The product LCA system suggested here will follow the attributional system approach, also known as book keeping LCA, since it analyses and reports environmental impacts that are allocated, so the sum of the environmental emissions etc from all products will be equal to the actual environmental global emissions[1].

Wood is a renewable resource that contains biogenic carbon, and its associated environmental benefit as well as possible burden will be allocated to the specific product system that consumes the resource, according to the definition given above. Thus, recycling of a primary wood material will not generate any "energy consumption", only parts that will not be recycled (for instance lost or used in the process for energy at the sawmill/impregnation plant), see Figure 1.

Consequently, the carbon within the wood will not cause a surplus (negative) energy balance since the inherent energy content will follow the recycled material (see the recycling case in Figure 1 where this aspect in the recycling case follows the "materials for recycling" but reduce the "use of primary resources"). If the recycled material is then used as fuel, it will generate a zero-balance with the carbon that was stored in the wood in the growing process, see Figure 1. The benefit with the inherent stored biogenic carbon is first recognized as a benefit when the wood resource is consumed This is illustrated in Figure 1 where the net carbon emission from the manufacturing is balanced by the carbon fixation in the landfill. The major amount of the carbon will be left in the landfill which will dominate and therefore generate a coal sink. Also when the impact is accounted for, i.e. when the emission in the impact assessment analysis is accounted for as CO_2-equivalents, the landfill alternative for solid wood will be a carbon dioxide sink. To exemplify the landfill model from the Wisard™ software produced by Ecobilan gives that 15% of the wood is degraded and creates 85% carbon dioxide and 15% methane, which will result in a net contribution to climate change (i.e. a carbon sink) of about 2 kg CO_2-equivalents per kg wooden coal.

1 The alternative is a consequential approach, also called margin approach, which will not fulfil this requirement and, furthermore, is dependant on assumptions on the margin, why the approach describes "what's happens if" on a marginal basis.

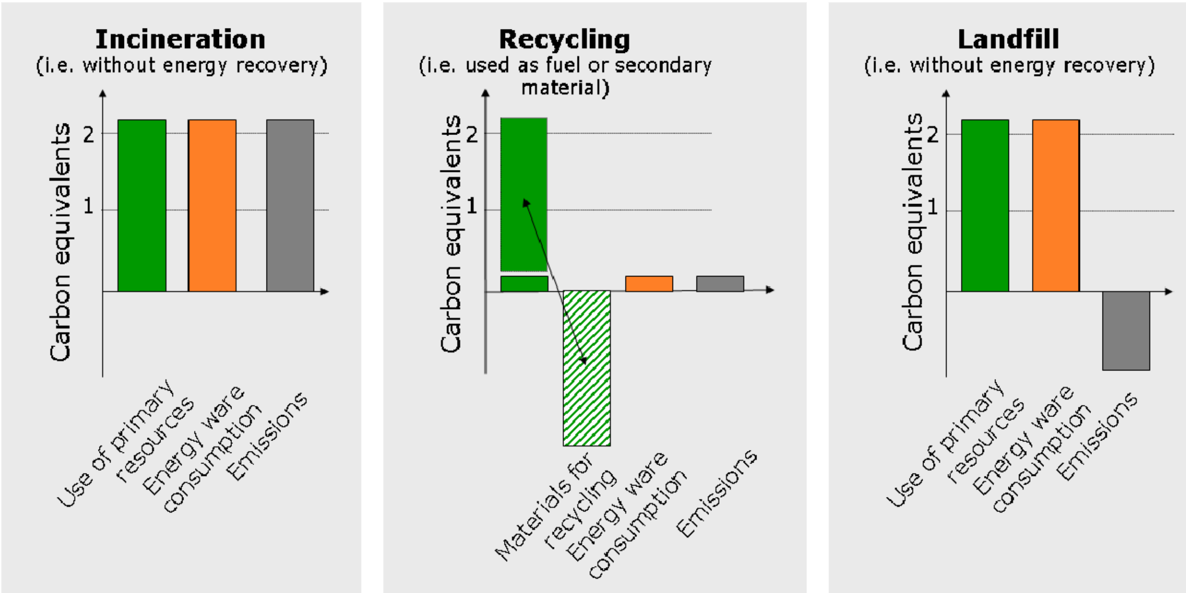

Figure 1 An illustrative example of accounting for some significant environmental aspects related to wood, including the scope of the inventory covering a full product life cycle, i.e. a cradle-to-grave perspective. The impact is given in carbon equivalents where 2 units is equal to the amount harvested in the forestry (inherent coal), where another 0.2 units are used in the processing (sawmill, etc.) and generating the same amount of carbon mostly in form of carbon dioxide. The landfill scenario in this example includes that only 40% of the carbon is emitted from the landfill in a foreseeable time horizon, why a carbon sink appears. Note that the use of primary wood will almost be balanced with the materials for recycling, in the recycling case above.

When the environmental performance is reported for an inventory covering a cradle-to-gate life cycle perspective, the used resources have to be accounted for without any reductions from end-of-life recycling. The resource use can only be reduced by taking an end-of-life scenario into account and when reporting for the environmental impact for the cradle-to-gate, the gross resource use will be reported, see Figure 2**,** combined with the net consumption reported on when a full life cycle is accounted for as illustrated in Figure 1.

When one compare the reported EPD result cradle-to-gate (Figure 2) with the full lifecycle (Figure 1) it is obvious that it is easy to interpret the inherent fixed carbon accounted as use of primary resources as an burden, which it will not become in the case of recycling (that includes energy recovery). Note that this 'possible' burden will be the same if the product was made of recycled wood as of primary wood resources, indicated in the right chart in Figure 2, since the 'possible' burden follows the material. Figure 1 is also an example on that the inherent property (fixed carbon) in wood is allocated to the product that makes use of the resource. In the use of wood the fixed carbon is like a 'hot potatoes' were the biogenic carbon is a positive aspect but will be accounted for as un environmental burden (i.e. energy consumption) if it just incinerated (without energy recovery) or land filled and eliminated if it is recycled.

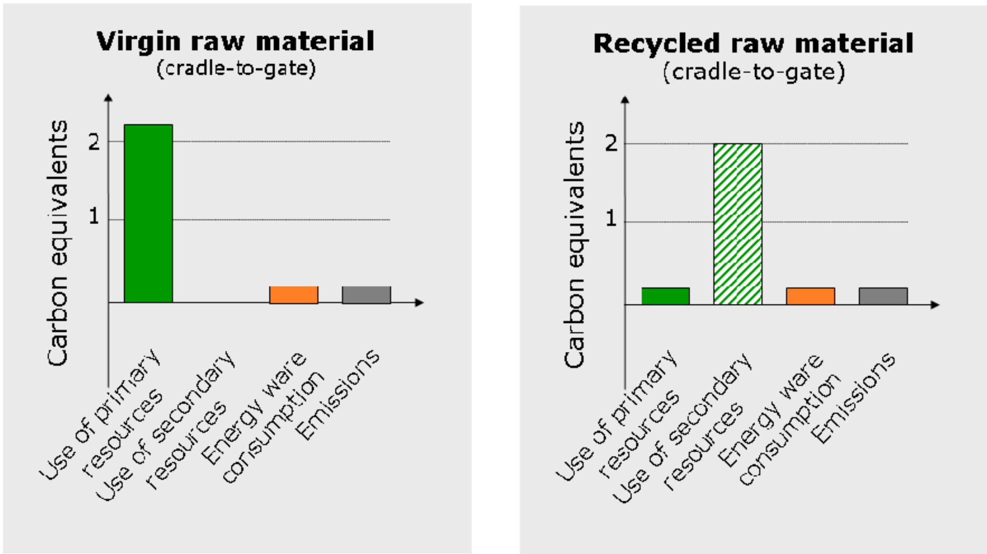

Figure 2 The accounting of some significant environmental aspects related to wooden products as a result of the inventory work that represents a cradle-to-gate perspective (i.e. not a full lifecycle). The value is given in carbon equivalents where 2 units is equal to the amount harvested in the forestry, where another 0.2 units are used in the processing (sawmill etc.) and generating the same amount of carbon as mostly carbon dioxide.

The accounting system applicable for product LCA, typically relevant for EPDs, given here gives a result that indicates that the preferable environmental end-of-life fate will be recycling (Figure 1). Then landfill will be better than simple incineration. This is a result that can be accepted as reasonable. In the illustrative exemplified cases given here this is exemplified by carbon equivalents as an indicator that is valid for emission as well as energy as a common indicator.

When the product LCA approach is applied it does not matter if the product is used for energy recovery or material recycling (substituting a primary wood resource). The fact that there is not any difference between environmental performance measured in carbon equivalents for energy recovery and material recycling is also adequate for a product LCA and therefore for an EPD. For this reason, if a system analytic approach including so-called system expansion would be applied instead, it would be possible to generate a difference between these two alternatives. However, scenarios used in a system expansion approach are depending on assumptions regarding what material the recycled wood replaces or saves by the material recycling respectively energy recovery. This specification is often not known and will in practice more or less always be made on subjective assumptions, wherefore such a system approach should be banned to use in EPDs that will be used in business communication and to end-users in market communications.

The system expansion approach is often referred to as consequential LCA and has the rebound effect that it can not be part of a holistic LCA approach where the environmental impact from different products can be added and will then be equal to all the impact that actually occurs in the real word. Therefore, consequential LCA seems not rational to use in EPDs since it is based on subjective assumptions and does not give an additive result where each product takes an environmental responsibility that is correct in spatial and temporal scale.

3.2.3 Consequences of the 'product LCA' recommendation

The suggested metrological approach, here referred to as 'product LCA', and its implementation will result in a holistic system where each product and service will be responsible for the environmental impact to which it can be physically related to, i.e. processes that it actually is part of. It is only a product LCA approach that imply to a system where all environmental burden prom products and serviced may be added to the actual global emissions and resource use, why the approach also is called bookkeeping LCA.

For waste handling alternatives valid for wooden products, when the product LCA approach is used the analyse result will be dependent of improvements in the actual product or service process rather than assumptions of what material the recycled waste will substitute or what energy source the recovered energy will save.

3.3 Applied impact assessment method

3.3.1 The aim of life cycle impact assessment (LCIA)

The outcome from the inventory step of an LCA is the LCI-profile (including emissions to air, water ground etc) which environmental relevance is evaluated in the cycle impact assessment (LCIA) step, i.e. the emissions contribution to climate change, acidification etc. The aim of LCIA, in brief, can be said to be to give the LCI-profile an environmental dimension and to generate the decision support for further interpretation. In this context a methodological trade-off always exists between condensing the result of an LCIA into as few figures as possible and maintaining environmental relevance. A major aspect with the aimed normalisation method is the need for a market acceptance of the method and the potential possibility to communicate the final output result (following the requirement of the ISO 14044 standard).

3.3.2 Two approaches to achieve environmental significance

A category indicator representing a potential environmental effect can be chosen anywhere along the environmental mechanism between the inventory results and the category endpoints, see Figure 3. Two approaches to achieve an increased environmental relevance of the inventory result namely the damage approach and the environmental quality objective (EQO) normalisation approach are included below.

A category endpoint will, in reality, cause a number of effects. Traditional LCA does not actually include direct quantification of such (absolute) effects, only potential effects (i.e. impacts) reported within the LCIA-profile. The individual category endpoint effects are in reality dependent on each other (see the dotted line in Figure 3 and Figure 4). The damage approach therefore includes different potential effect-related endpoints where spatial and temporal damage functions are essential. A number of operational "damage-oriented" approaches are frequently used, for instance in ExternE (1995, 1997), Eco-Indicator (Goedkoop and Spriensma 2000) and EPS (Steen and Ryding 1992; Steen 1999), and EDIP2000 (Hauschild & Potting, 2001).

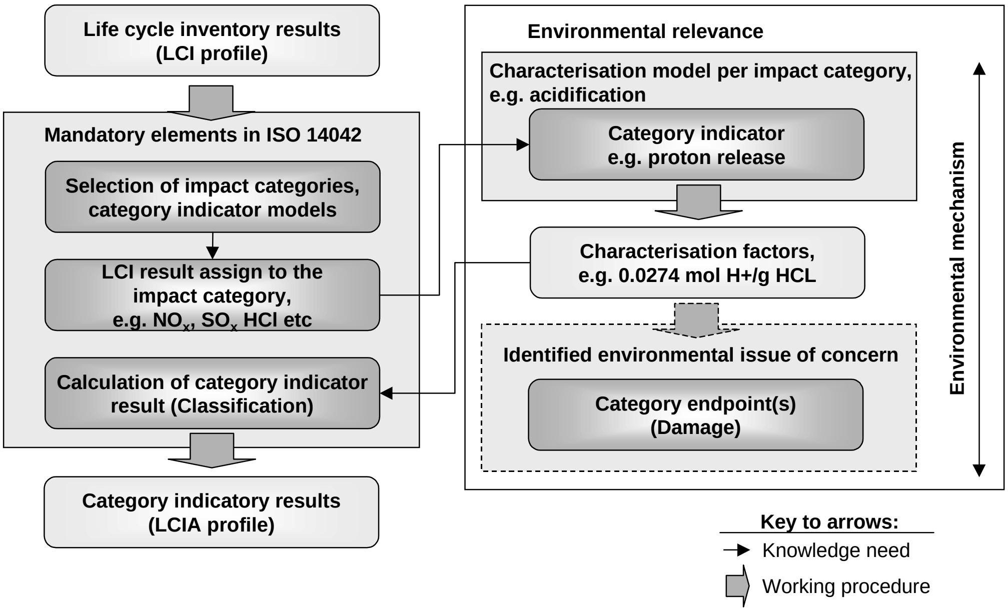

Figure 3 The concept of a category indicator and the characterisation factors applied to the inventory result to estimate the contribution to an impact category.

An alternative method to improve the environmental relevance in the LCIA step, which does not include a value choice-based damage approach, can be achieved by an EQO normalisation (Erlandsson 2003a, 2003b). An EQO indicates environmentally acceptable conditions that can be regarded as ecologically sustainable, defined by a critical load function. EQO includes estimations, which preferably should be based on scientific common knowledge, which is why the EQO will then be of a normative character.

In order to apply the EQO approach in the LCIA, it is necessary that the critical load also can be expressed as a mass flow rather than a recipient concentration, etc. Such EQO are put forward in a holistic way in the so-called Swedish environmental quality goals (Swedish Government 2001). Other approaches such as the EU's ceiling directive (EC 2001b) are a mixture of EQO and political emission targets, and furthermore do not cover all impact categories. A comprehensive record of all kinds of EQO (called Sustainable Reference Value by EEA) and emission targets can be found in the EEA-STAR database (EEA 2004). Also the German UBA valuation method use a mixture of EQO (critical load) and political emission targets per impact category as part of the assessment method (Schmitz and Paulini 1999). An other method based on the same approach operating on critical levels (based on Swiss environmental legal targets) on individual flows is The Ecological Scarcity Method (Frischknecht et al 2009).

The question of who has the permission to contribute to this environmental impact is an additional issue related to the EQO normalisation approach. This is a matter constantly on the political agenda. Therefore, different international agreements result in a quota system where the acceptable amount per contributor is restricted on a national level. However, it will still be a normative decision in the LCIA to decide, for instance, if the acceptable annual emissions contribution to climate change should be normalised per global individual citizen, or if national or regional politically acceptable quota should be applied (as according to the Kyoto protocol, etc.). Because of this specific issue, LCIA always includes some socio-economic elements, direct or indirect (as in a normalisation).

When EQO are put into operation in the LCIA, it is found appropriate to include the precautionary principle within the normalisation step that also in fact streamlines the result. The precautionary principle is based on the fact that the system's vulnerability is to a great extent determined by its weakest part. In this respect, for instance, for the acceptable load for acidification is determined by the safeguard for ecological health rather than the safeguard for human health, since an ecological threshold will be exceeded first. A normalisation based on EQO would then facilitate the determination of the contribution to the safeguards, as illustrated in Figure 4 (i.e., the single line from each impact category to a safeguard). In a damage-oriented approach, in contrast, it is required that each individual contribution from an impact category (potentially) affecting all safeguards is accounted for via a number of category endpoints.

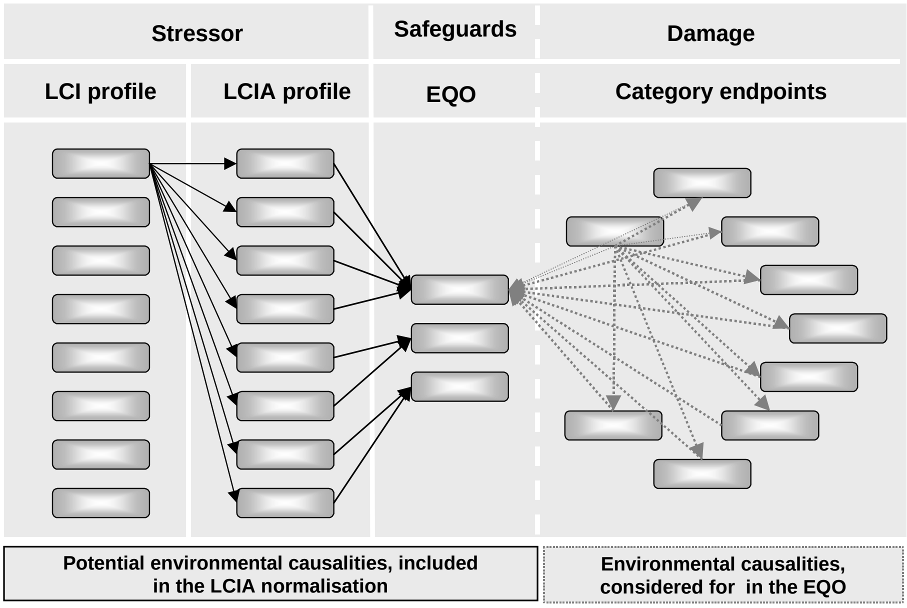

Figure 4 A normalisation based on environmental quality objective, including the precautionary principle, will streamline the LCIA profile without including weighting elements. Damages are indirectly accounted for via EQO i.e. dotted lines (Erlandsson et al 2003).

A normalisation based on EQOs forms a scenario where a number of acceptable environmental category endpoint conditions are safe, acceptable or tolerable on a scientific basis. The collection of all EQO together describes future sustainable environmental conditions where the safeguards are protected. This implies that the category endpoint's contribution to a safeguard must be met if an ecologically sustainable society in its turn shall be realised. In this respect, the normalised impact categories contribute equally to the safeguard. In contrast, in a damage-oriented approach, it would be necessary to phase the ranking of all the different effects included against each other, and this therefore includes elements of value choices.

3.3.3 The EQO normalisation method

The EQO normalisation method is based on the possibility of defining a normalisation factor, which constitutes a yearly impact permit per impact category and person, where the EQO is protected, as specified in equation (1).

$$nf_i = \frac{I_{EQO}}{individuals} \tag{1}$$

where

nf_i	is the normalisation factor for a specific impact category i [impact equivalents/Pe/year]
I_{EQO}	is the yearly critical impact that can be permitted and related to a spatial system and where the environmental quality is protected [impact equivalents/year]
Individuals	is the number of persons that are part of the spatial system and contribute to the critical load [Pe].

It is then possible to present the result in a common unit, namely a so-called *person equivalent* [Pe] according to equation (2).

$$Inorm_i = \frac{\sum_{j=1}^{n} m_j \cdot Ch_j}{nf_i} \tag{2}$$

where

$Inorm_i$	is the normalised potential environmental impact of the impact category i [Pe·year]
m_j	is the emission of a substance j [g]
Ch_j	is the characterisation factor of a specific substance j, which describes its contribution to a specific impact category i [impact equivalent/g]
nf_i	is the normalisation factor for a specific impact category i [impact equivalents/Pe/year]
n	is the number of emitted substances included.

The currently included impact categories and normalisation characteristics in the EQO normalisation method are specified in section 3.4 that is based on the source data given in section 7.1.3. The other applied characterisation factors are found in Erlandsson (2003a).

Table 2 Characteristics used in order to establish the normalisation factor (Erlandsson 2003a, 2003b).

Impact category	Category indicator [equivalents per year]	Type of characterisation model	Basis for normalisation, critical load
Climate change	4500 kg CO_2 eq.	Momentary change of radiation balance of an emitted substance by constant boundary settings and integrated over 100 years.	EU, and the long term emission goal that is equal to a concentration of 450 ppm CO_2 eq. Note that this corresponding normalisation flow will still lead to a climate change.
Eutrophication	39 kg NO_3^- e.q.	Inherent property of oxygen demand by using the Redfield molar ratio of 106:16:1 (C:N:P), with no spatial or temporal resolution.	SE, accepted emissions where both human health and ecosystems are protected.
Acidification	29 kg SO_2 e.q.	Inherent property by stoichiometric formation of H^+, with no spatial or temporal resolution.	SE, accepted emissions where both human health and ecosystems are protected.
Stratospheric ozone depletion	0,27 mg CFC-11 e.q.	The amount of stratospheric ozone destroyed by a specific emitted gas over the entire atmospheric lifetime at a steady state in relation to CFC-11.	SE, accepted emissions due to leaching from current installed equipment.
Photochemical ozone creation	1150 $ppb{\cdot}h{\cdot}km^2$ e.q.	The method to calculate the average ozone load in $ppb{\cdot}h{\cdot}km^2/kg$ (NO_x and VOC) emitted, based on calculations of the amount of ozone produced along trajectories that follows four days after a local emission source.	SE, accepted emissions where both human health and ecosystems are protected.
Human toxicity	1643 kg 1,4-dichloro-benzene e.q.	Estimation of a predicted environmental concentration (PEC) based on a nestled multimedia model with settings valid for EU and evaluated with a predicted no effect concentration (PNEC) in the impact media.	EU, based on an acceptable emission of only benzene, which is based on WHO's air quality guideline with an additional lifetime cancer risk of 10^{-5} *). Note that the normalisation procedure implies a conservative evaluation, since this is based on an acceptable emission of only one substance. All emissions sources, therefore, contribute to this substance-specific critical load.
Ecological toxicity	1 Pe e.q.	Estimation of a predicted environmental concentration (PEC), based on a nestled multimedia model with settings valid for EU and evaluated with a predicted no effect concentration (PNEC) in the impact media.	EU, based on the maximum acceptable emission of each emitted substance where the risk characterisation ratio is not exceeded in any receiving environment PEC/PNEC≤1.

*) Note that the life time risk in the calculation of an individual characterisation factor for an individual substance in USES-LCA is (still) 10^{-6}, which means that carcinogenic substances relative impact compared to other effects follows traditional acceptable risk assessment. An alternative assumption in the normalisation approach is to say that the life time risk of 10^{-6} relevant for a population is selected combined with an assessment factor of 10, since all substances contributes to the quota of the emission permit of one single substance.

For the life cycle impact assessment (LCIA) characterisation factors from 'CML 2002' are selected as they are implemented in the LCA software GaBi (as version 'CML 2002' December 2007), except for ecological toxicity and photochemical ozone creation. The characterisation factors from CML '2002' can be found in the "Handbook on Life Cycle Assessment. Guide to the ISO Standards" (Guinée 2002). The characterisation factors from CML are commonly used and more or less a de facto set of indicators in the LCA community. It should be noticed that USES-LCA developed by Huijbregts et al (1999) is part of the characterisation factors given in the "Handbook on Life Cycle Assessment. Guide to the ISO Standards".

The characterisation factors applied in the case study for photochemical ozone creation and ecotoxicity from emissions to air, fresh water, sea water and agricultural soil are normalised reference values in g person equivalents/year as defined in Erlandsson (2003a). Ecotoxicity is reported as the sum of the ecotoxicities in all receiving media.

3.3.4 Risk minimisation and risk assessment

LCA was originally developed as a risk minimisation tool because it is a relative approach-based method and no actual concentrations are estimated. In reality, only a number of limited environmental loads beneath the critical load cause a potential effect, e.g. particles, cancerous substances and ionising radiation (radon). Even if an emitted substance can be proved to have no effect beneath a threshold, this emission can be regarded as a contribution to a concentration that may exceed the threshold in the future. A normative decision can therefore be applied that supports a method where all emissions are regarded as potentially or actually contributing to the exceeding of a threshold and should therefore be considered as part of the environmental burden. This can only be justified for the purpose of *risk minimisation* and justifies model linearity between the threshold and the origin, see Figure 5. Here the linearity is being referred to as a *critical load function*. This model assumption can be likened to the common sink problem, where the 11th litre of water causes the sink having a capacity of 10 l to overflow. The question is whether it is only the last litre that is responsible for the overflow. If LCA operates in the potential effect area that is indicated in Figure 3 or just above the critical load, it is suggested here that all 11 l would carry the same burden for causing the sink to overflow. This would at least be correct when the normalisation based on EQO is applied, when no adverse effects are likely, i.e. only potential or minor effects appears (see the area marked LCA in Figure 5).

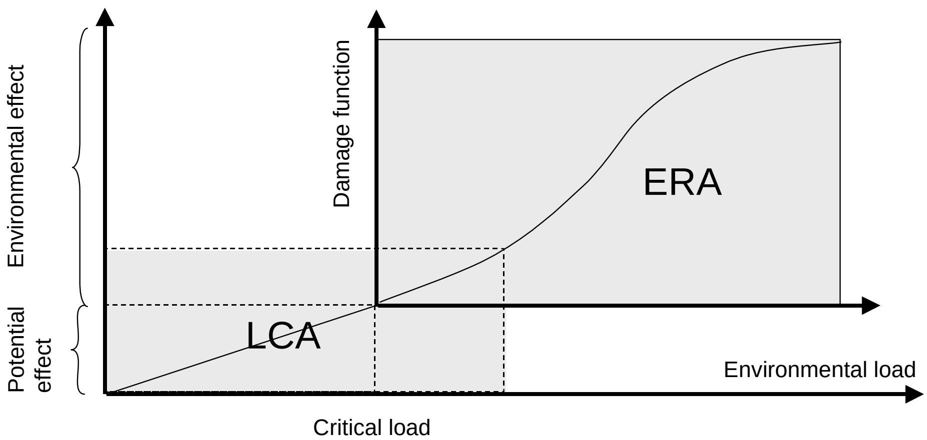

Figure 5 The relation between environmental load and potential or defined effect described by damage functions. In the figure, the critical load is indicated, which states the reference value for the normalisation and critical load function.

If a true damage-oriented approach is used, the non-linear "damage function" that is indicated in Figure 3 would be a relevant working area concerning the environmental effect. Furthermore, if the analysis of the emitted substance concludes that no environmentally critical load will be exceeded in reality, the substance will not contribute to any environmental impact in the LCA result, and therefore this approach will not be relevant for risk minimisation. This is shown in Figure 5 as the 'Environmental effect', i.e. PEC/PNEC > 1. If this is the case other quantitative tools like environmental risk assessment (ERA) should be applied instead. Even if this situation is likely to occur, the risk minimisation approach will still be relevant. In this case, the EQO normalisation procedure suggested here could be utilised in parallel to cover the risk minimisation perspective, since the risk assessment methods don't have any life cycle perspective.

3.3.5 The EQO normalisation approach elaborated for toxicity

To handle risk minimisation in an LCA, and in the EQO normalisation approach in particular, an analytic impact assessment model that handles site dependence has to be introduced. Different such models are developed within the LCA society based on EUSES that is a common risk assessment application developed by commission by the European Commission (http://ecb.jrc.it/). The applied multi-media based risk assessment method in the EQO normalisation concept uses the USES-LCA concept developed by Huijbregts (1999).

The USES-LCA model makes it possible to assess different emitted substances integrated into one common value for human toxicity. Concerning human toxicity in the EQO normalisation approach, no further calculation is therefore required besides setting a yearly acceptable impact that protects the human health. This is made by a backward calculation of the yearly maximal benzene emissions. Benzene is chosen since the impact of this substance is well known and documented by e.g. WHO guidelines (see Table 2). It should be noticed that this is a very conservative basis since all emissions are assumed to contribute to a single substance's specific-critical load. This normalisation approach should therefore be treated with respect concerning the LCA result when relatively high figures on human toxicity appear. If this is the case in a case study the analysis should then be supported by more sophisticated methods such as ERA for the critical emission sources in the product's life cycle.

Ecological toxicity is evaluated by considering all initial emission sources in the category indicator model (e.g. air, seawater, agricultural soil, industrial soil etc) one at the time (Erlandsson 2003b). The maximal yearly mass flow is then calculated under the condition that the concentration in any receiving compartment shall be (less or) equal to the maximal acceptable concentration (i.e. predicted no effect concentration, PEC=PNEC). This implies that an *Acceptable Initial Compartment Flow* (AICF) can be established, see Figure 6. The AICF are determined by an iterative calculation in the multi-media model for each substance included initial emission source, which then constitutes the characterisation factor.

For each impact category the impact assessment calculation uses a characterisation factor for each specific substance and initial emission source specified in a common unit. An integrated assessment of ecological toxicity as a whole is therefore now possible. This is the innovative part of the procedure compared to the originally developed EUSES-based concept suggested by Huijbregts et al (2000). The EQO normalisation procedure is based on the acceptance of the precautionary principle, where the most vulnerable receiving compartment affected by the emitted substance limits the overall acceptable emission. This procedure means that the EQO utilised for the normalisation will be modelled on the bases of the different initially included emission sources.

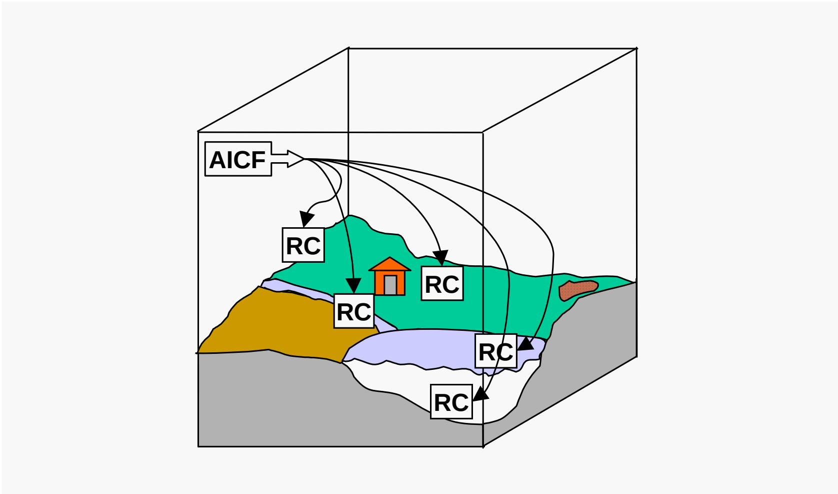

Figure 6 The dimensioned mass flow (AICF) corresponding to a risk characterisation ratio in the receiving compartment (RC) less or equal to 1 (PEC/PNEC ≤ 1).

This is in fact very similar to how human toxicity is modelled in all ERA concepts where different exposure pathways are accounted for. It is then possible to estimate e.g. a daily tolerable intake that is compared to the actual intake caused by the emission scenario, where different human health effects are included within the same concept.

3.4 Toxic characterisation factors for creosote

In this study we assume that the creosote oil is divided and characterised into two blocks that are evaluated separately within the LCA. These two blocks are a volatile part that typically has a boiling temperature below 270 ºC and the residual oil fraction that consists of heavier high-boiling substances.

The dominant substances in the volatile block of substances in creosote oils are naphthalene and 1- and 2 methylnaphthalene. The risk assessment report from KEMI (2009) analysed and evaluated the so called "Grade B, BPD composite, ATE 8300" creosote mixture. In this product the naphthalene content is 6 % while it is 8% and 4% for 1- methylnaphthalene and 2- methylnaphthalene respectively. Concerning the representation of the volatile block in the LCA it is assumed that these substances and the proportion given below will be adequate and will therefore be used to characterise this block of the creosote mixture.

The high-boiling substances in the residual block is characterised by its content of benzo(a)pyrene. The UNECE POPs Protocol specifies four substances that belong to the heavier molecules in the PAH group. They are regarded as the most carcinogenic ones and are referred to as PAH-4. They are used as indicators for air emissions. The four substances are benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene and indeno(1,2,3-cd)pyrene. Compared to the composition of the "Grade B, BPD composite, ATE 8300" (Kemi 2009) none of the PAH-4 substances of the POPs Protocol occurs in a detectable amount. If one consults the technical specifications from two suppliers of creosote a typical value is around 20 to 35 ppm benzo(a)pyrene (Rütgers 2008, Koppers 2009).

However, in order to use the benzo(a)pyrene content as an indicator for the whole creosote composition a factor of 5 has to be applied. This factor is based on cancer studies on mice where the creosote compositions were found to be 3-5 times more potent than what could have been expected based on their BaP content (Fraunhofer Institute, 1999, see KEMI (2009) DOC III A6/B6, point A6.7). The result on creosote was based on a composition with a BaP content of 10 ppm. This follows the presumptions made in the risk assessment by KEMI (2009) which are the same as those made by CSTEE (CSTEE. Scientific Committee for Toxicity, Ecotoxicity and the Environment. March 4 (1999). Therefore, in the characterisation of the residual block of the creosote oil in the LCA a benzo(a)pyrene content of 10 ppm is used, based on the same assumption as made by KEMI (2009).

On order to calculate characterisation factors to be applied in the impact assessment of toxicity in the LCA the USES-LCA model requires chemical-physical data as well as toxicological information (Huijbregts 1999).

Concerning ecological toxicity and benzo(a)pyrene the main references for these data are KEMI (2007) that corresponds to information in KEMI 2009[2]. and information from JRC in Ispra (2003). For human toxicity caused by chronic inhalation of B(a)P, information from the WHO (2000) Air quality Guideline for Europe (second edition) gives an acceptable inhalation of 0,012 ng/m^3 associated with a lifetime cancer risk of 10^{-6}. A tolerable daily intake of B(a)P at a lifetime risk of 10^{-6} is reported by the Swedish EPA. Data based on WHO (1996) give $2{,}3 \cdot 10^{-12}$ kg/kg bw/d and another reference US EPA (IRIS 2002) gives $0{,}14 \cdot 10^{-12}$ kg/kg bw/d, where the latter is used in the USES-LCA calculations.

Data on naphthalene is available in USES-LCA by Huijbregts (1999).and the chemical-physical data is used without any changes. In USES-LCA a tolerable daily intake of 50 µg/kg bw/day (Vermeire 1993) is used. Newer data from RIVM (1998) gives 40 µg/kg bw/day (lifetime cancer risk 10^{-6}). The US Department of Health and Human services (2005) gives an oral semi-chronic value of 6 µg/kg bw/day (risk 10^{-6}). The latter alternative is used (although it is not a chronic value) in the USES-LCA calculations.

The improved data on the chronic inhalation value will generate an increased characterisation factor (a more dangerous substance) in the LCA impact assessment, based on new knowledge. The original data in EUSES-LCA referrer to a non-cancer value from US EPA (1998) of 3,0 ng/kg/day (non-cancer). An updated reference from the US Department of Health and Human services (2005) gives an inhalation chronic value of 7 ng/kg bw/day (risk 10^{-6}). Based on information from the California EPA (2005) a value 8,3 ng/kg/day (risk 10^{-6}) can be calculated. The values for the inhalation pathway from the US Department of Health and Human services and the California EPA may be recalculated to 24,5 and 29 ng/m^3 respectively (risk 10^{-6}). The value on acceptable inhalation of naphthalene based on information from US Department of Health and Human services is selected in the USES-LCA calculations.

The resulting characterisation factors on human end ecological toxicity are summarised below.

[2] Since the IVL inventory work started in the beginning of 2009 KEMI has published an updated version of the risk assessment in February. However, the data on ecological toxicity and chemical-physical data are in basic the same.

Table 3 Calculated characterisation factors for the impact assessment of human and ecological toxicity, where the figures for ecological toxicity is based on a BaP content of 10 ppm and an assessment factor of 5 to give an representative figure for the entirely oil composition.

Ecological toxicity [Pe/kg]	Air	Fresh water	Sea water	Agricultural soil
Creosote, WEI B	1,8E-06	5,9E-03	4,4E-06	3,7E-03
Naphthalene	1,8E-06	3,2E-02	3,7E-05	3,8E-03
Human toxicity [kg DCB eq./kg]	**Air**	**Fresh water**	**Sea water**	**Agricultural soil**
Creosote, WEI B	1,0E+00	1,4E+01	2,3E-01	4,1E+00
Naphthalene	6,8E+01	4,7E+01	1,6E+00	4,0E+01

The source data for the USES-LCA calculations of characterisation factors are available in section 7.1.3.

3.5 Emission of creosote during service

There exist very few literature references on leaching of creosote from poles and studies reported on sleepers are not considered to be relevant. Firstly, the exposure from weathering is quite different since the poles are standing up and the biological activity is focused above the ground line. This up-standing position implies a vertical redistribution of the creosote to the lower part forced by gravity. Secondly the wooden material for sleepers is commonly made of oak and beech. The uptake of creosote is lower in these wood species compared to pine. Pine is used for transmission poles world wide but for sleepers only in northern Europe.

The following literature data on leaching on creosote poles was found:

- Finland, 10 years of service (Nurmi 1990)
- Germany (Sürzelberg), 20 years of service (WEI 1985)
- Sweden (Simlångsdalen), 40 years of service (Holmroos and Bergqvist 1994, Holmroos 1994)
- USA, 100 years of service (Betts 1982).

The field test in Sürzelberg and Simlångsdalen are part of the same project coordinated by WEI.

A general problem with the literature references are the limited number of samples involved, furthermore the original amount and the specification of the creosote oil is limited or just missing. Moreover, the creosote mixture and its content vary as well as the preservation and the technical guidance of the retained amount required for different applications. To make a qualified estimation of the creosote leachable the following assumption and interpretations are made:

The sample from Finland consists of ten poles installed at a power line and therefore in service during the measuring. The original retention was not measured, why an amount of 190 kg/m3 sapwood is assumed (based on the measured remaining content after one year). The technical specification of the creosote oil is not reported and the emission is given as one over all figures with no details on individual substances or boiling intervals. To calculate a figure for the leached oil from the poles it is assumed that as an average ¼ of the pole is below or jus above the ground line why it is represented by the test wooden samples taken from the ground level and a depth of 0,5 m below ground. ¾ of the leaching from the pole is calculated as a average if the test sample taken from the pole 1 m above ground and at the top.

Twelve poles were selected from the test field in Sürzelberg for analysis of the remaining part of creosote after about 20 years of service. The poles were not full-size (3 m) and samples of the remaining creosote were taken from points 0,6 meter below ground, 0,3 meter above ground and 0,4 meter from the top. In order to calculate the leachable part of the creosote the average of the test pieces from the three sample heights was used to calculate the overall emission from a pole. The creosote retention was determined after the poles were impregnated. Since the retention was determined by fractionated distillation WEI has corrected the original measurements according to new technology based on gas chromatography. Six types of creosote oils were used (i.e. two poles per oil type was analysed). Oil type II in the test is the one which best corresponds to WEI type B, why these figures are used here.

The same creosote oil type II was used in all WEI test fields at Sürzelberg and Simlångsdalen. In the WEI measurement program a gas chromatography analysis of the original creosote preservative oil was included, as mentioned above, This analysis was performed in 1974 with the aged oil from 1951 - 53. These figures on the creosote content is used for the treatment of the data on leaching of creosote reported on the analysed pole that was treated with oil type 2 (i.e. the one which best corresponds to WEI type B) from Simlångsdalen. The poles in Simlångsdalen test field are not full size (3 meter long). The average leachable amount is calculated by using the average of the four sample slices that are selected from 0,8 m below ground (the bottom end of the pole +/- 0,1 meter, an average of these two sample slices is used) and 0,6 m from the top. Note that the same total leachable amount will be the result if all samples from an individual pole are weighted equal (i.e. if the two samples around the ground line were not aggregated). But the distribution of the remaining substances will then be changed.

All measured figures on the retained creosote components (i.e. 10 substances of the creosote formulation) are used in the calculation to characterise the leaching, with the exception of the chrysene figures since they also include other substances. For this reason, the chrysene figures overestimate the leachable amount, but the figures may be used as a bottom line (that indicates that more than 50 weight-% of this and other heavy substances are fixed in the wood after 40 years of service.

The pole analysed by Betts (1984) is only one pole with no information on original or planned retention, why the result is only given here as an indicative value. It is worth mentioning that the pole was not suffering from decay when it was taken out from the transmission line. The resulting figures of leaching based on the information from the references given above are illustrated in Figure 1 .

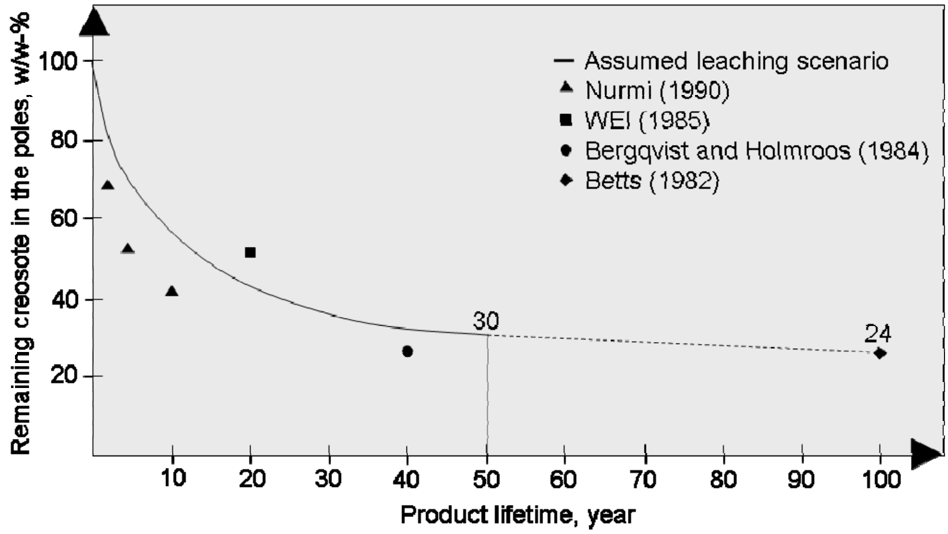

Figure 7 Estimated remaining amount of creosote in service based on literature data. Note that the references are based on historical creosote formulation and new formulations with less volatile components would significantly reduce the leachable fraction.

Based on the literature data on leaching the retained amount of creosote will here be assumed to be at least 30 weight-% after 50 years of service life. If the volatile part of creosote in today's WEI type B creosote (defined in EN 13991, App B) is not more than 20 volume-%, or if specific type B formulations are used, the volatile contribution from naphthalene and 1-, or 2- methylnaphthalene will be less than 7 weight-% (see Grade B/ATE 8515, Table 1.1.1 in KEMI 2009, or RÜTGERS Chemicals (2008)). If everything else remains the same but the content of naphthalene and 1- or 2-methylnaphthalene is reduced to 7 weight-%, and it is assumed that these components represent at least 25 weight-% of the creosote formulations referred to here, the reduced leaching of such modern type B creosote can be estimated to be no more than 40 % during an average service life of 50 years.

An additional study by Evens et al (1984) on the leaching during the first year measured an emission of creosote components of about 10 to 25% that was assumed to be volatile substances emitted to air.

In the LCA it is assumed that 70 weight-% of creosote is leached. Of this amount 15 weight-% is assumed to be emitted to air during the service life. These volatile emissions (<270 °C) can be assumed to take part mainly during the early years of the pole's lifetime. These emissions to air will in the LCA be characterised as consisting of 5% naphthalene, 7% 1- methylnaphthalene and 3% 2-methylnaphthalene. The rest of the leached creosote will in the LCA be assumed to be emitted to ground, equal with 55 weight-% of the original creosote retention. This residual emission is characterised in the LCA with a content of 10 ppm benzo[a]pyrene.

4 The LCA

4.1 The case study specifications

4.1.1 Functional unit

- **One 9-m pole (0,4-kV transmission) with a service life of 50 years**

Nine-meter poles are typically used for 0,4-kV transmission lines or for utility telephone lines. This kind of transmission line with 9-meter poles is commonly used and therefore selected for the case study. For 9-meter poles the contribution from the foundation will not have a decisive influence on the environmental performance compared to taller ones, why this size is preferable to illustrate the difference between the alternative pole materials.

4.1.2 System description

Poles of the following materials are compared in the study:

- **Steel**: made of 50% recycled steel and a concrete foundation of 1,5 m made of ordinary building concrete (C35)
- **Concrete**: high-performance concrete (C100) and reinforcement made of 100% recycled steel
- **Wood**: made of pine and treated by creosote type WEI B, with an uptake of 110 kg/m^3 sapwood according to the manufacturing standard NTR-A, which by conservative assumption gives an overall retention of 60 kg/m^3 wood.

The lifetime of the transmission line is set to 50 years and all material alternatives will last for this specification. In reality the average lifetime for this kind of transmission lines will probably be shorter in many cases compared to the lifetime of the poles.

4.1.3 System boundaries

The study is limited to the poles themselves and does not account for the power line or telephone wire and potential different means of attachments. Also the work to rise and demolish the lines is neglected since it will be assumed to be of equal importance for all studied alternatives (and of minor importance in a life cycle perspective).

The construction materials are followed backwards to their origin in nature, as are necessary auxiliary materials and energy wares. End-of-life of the poles after the service life is not included in the figures reported in this working report. Nevertheless end-of-life will be included by incineration with energy recovery for the creosote poles, material recycling for the steel and downgrading for concrete that means that is will be used for land fill. The impacts from the incineration are allocated to the generated energy and not to the poles following the so called product LCA approach applied here. End-of-life scenarios will be more elaborately accounted for in a follow-up article based on the basic LCA presented here.

Transports from the factory gate to the site of use and from there to the site of final disposal are not included at this point. This will be complemented in the forthcoming article. The complementary work is not expected to drastically change the LCA result given below, which is supposed to reflect the environmental impact of a full life cycle

The environmental impact assessment is limited to the emission categories described in Table 2. Use of resources is not reported as no characterisation model is established on resource use or material consumption.

4.1.4 Inventory

The inventories of the different pole alternatives included in the study are summarised in the Appendix section 7.1. Such inventories of energy and auxiliary materials that have been collected from public sources are available on request.

4.2 Result and interpretation

The result from this initial LCA calculation is illustrated in Figure 8. The result of the LCA is given as a normalised result where the relative importance of the impact categories is given. The dominating impact category is human toxicity, where the contribution from the steel pole dominates. There are two major sources behind this environmental performance, namely an emission of leached zinc during the service life (88 Pe) and emission of metals from the steel production. These metal emissions are also the dominant source behind the contribution to ecological toxicity from the steel pole.

It should be noticed that the original data on steel production from the EcoInvent LCA database were changed in a way that reduces the overall impact from the steel pole from 445 Pe to 174 Pe. This was done in the final LCA by changing the emission to air of all reported hexavalent chromium to only 5% of the total emission. The remaining 95% are instead assumed to be emitted as trivalent chromium. This distribution between trivalent and hexavalent chromium is equal with the monitored background emission in Swedish air (Woldegiorgis et al 2007). Other data sources do not specify the chromium emission, why is as reported as "unspecified". This modification of reported hexavalent chromium will thus at least reduce the consequences of choosing different data sources. There is no evidence that this is correct why this fact must be borne in mind when analysing the result given in Figure 8.

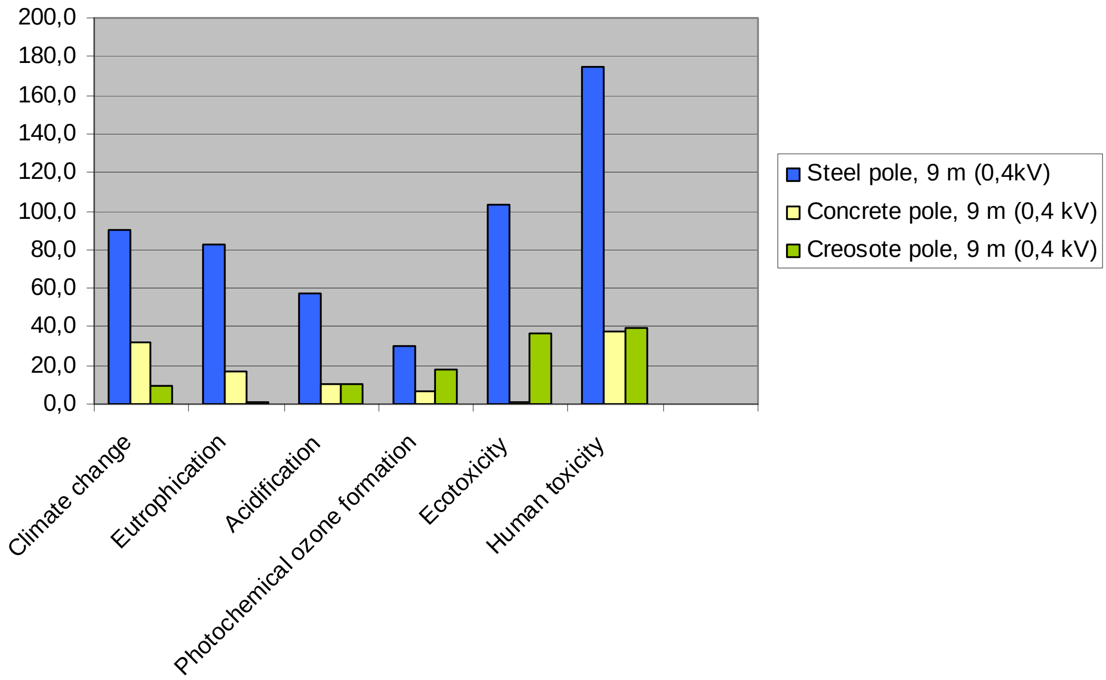

Figure 8 The relative importance of different impact categories included in the LCA study and normalised according to the EQO normalisation approach. The LCA covers the life cycle from raw material production to the use phase, where the emission significantly contributes to the performance of the steel and wood poles.

Steel is the material alternative which has the highest impact on climate change, acidification, eutrophication and photochemical ozone formation. These impact categories are the basic ones that are included in most impact assessment analyses of a LCA.

Creosote poles and concrete poles do not differ that much. Concrete poles contribute more to climate change and eutrophication, while treated wood has a higher impact on photochemical ozone formation and human and ecological toxicity. It must be noticed that the inventory data for the concrete pole (based on updated manufacturing sources) include an important assumption, namely the contribution of hexavalent and trivalent chromium emitted from the cement kiln. The assumption is that only 5 % of the emitted chromium is hexavalent, and the rest trivalent. The same assumption is made for steel. This yields a contribution of 61 Pe to human toxicity from the concrete pole. If the chromium emission had been 100 % hexavalent, the contribution to human toxicity from the concrete pole would have been 254 Pe instead in the impact assessment. This fact should be borne in mind when analysing the result in Figure 8.

In the initial LCA performed it was assumed that the naphthalene content was 6 weight-% as an average for European creosote WEI type B, corresponding to the generic composite “Grade B, BPD composite ATE 8300” according to KEMI (2009). Almost all low boiling components of the creosote including naphthalene will be emitted during the service life of the poles. This emission will generate a contribution of 65 Pe to human toxicity when the effect of the generic composite grade B is accounted for. In this case 60% of the creosote human toxicity potential comes from the naphthalene emission. The impact from the creosote poles on human toxicity therefore to great extend depends on the specific formulation of the creosote oil. Creosote oil WEI B from Rütgers typically contains less than 1 weight-% of naphthalene (Rütgers 2008). Information from Koppers on individual batches delivered to ScanPole in Norway also claims that Koppers may produce this kind of low-content naphthalene creosote type B (Koppers 2009).

Since naphthalene has a significant contribution to human toxicity in this LCA of creosote poles it seems important to use a creosote composition that will reflect the current situation rather than the generic figures given by KEMI (2009). The figures on naphthalene content by KEMI (2009) are not crucial in their study in respect to the context defined by their risk assessment. Koppers and Rütgers are the two dominant suppliers of creosote oil in Europe, why we will use the typical value given by Rütgers in this LCA. This assumption is also supported by the delivery information to ScanPole from Koppers. (The formulation may, however, vary between different batches). The consequence of using this low-naphthalene creosote oil with a maximum content of 1 weight-% naphthalene will change the contribution of the creosote to human toxicity. The human toxicity potential from the creosote pole will thus be reduced to 37 Pe as is illustrated in Figure 8.

The low-boiling substances in creosote are important in the preservation process but not for the durability. The reduced naphthalene content will not have the same effect on ecotoxicity, since the dominating source of ecotoxicity is the leaching of creosote, which will remain the same.

Somewhat unexpectedly the steel pole seems to have a higher ecotoxicity impact than the creosote pole. The result of the LCA therefore clearly illustrates the importance of a complementary risk minimisation approach, where LCA may be used to cover the full life cycle of an individual product, which can then be compared to alternative products that fulfil the same function.

5 Conclusions

This report gives a description of how LCA can be utilised as a tool to provide decision support based on 'product LCA' and the 'EQO normalisation' approach. LCA can be used to provide decision support for comparative product assertions following the requirements given in the LCA standards ISO 14044. A fair product comparison requires that the functionality of the products is accounted for, that a full life cycle is included, inclusion of the significant impact categories in the impact assessment, and as few subjective methodology assumptions as possible.

The result of the LCA illustrates that poles made of steel or concrete have a higher impact on climate change than creosote poles. The significant aspect of creosote poles is human toxicity. Even so, steel poles have a higher impact than creosote poles on ecotoxicity as well as on human toxicity. An overall assessment will favour the creosote poles as the ecologically most sustainable alternative in respect to the environmental quality objectives used for normalisation. The results presented in Figure 8 probably underestimate the impacts of steel poles and concrete poles on ecotoxicity and human toxicity, since we may have underestimated the contribution of hexavalent chromium. A more detailed analysis of actual emissions and an elaborate impact assessment method are crucial in order to evaluate the environmental performance of metals better and to take the bioavailability and environmental fate into account. Such analyses and assessments are not included in this project.

When LCA is used for comparative purposes and assertions, it is important that an impact assessment method is used that has an environmental significance without including value choices. Following the LCA standard ISO 14044 the presented EQO normalisation approach generates an interface where the relative importance of different environmental impact categories can be evaluated. No restrictions to use such a normalisation method in environmental declarations type III according to ISO 14025 are found. Therefore EQO normalisation would be the ultimate choice to present the result from a LCA also in environmental declarations.

Special attention in the EQO approach is paid to the possibility of generating an integrated value for ecological toxicity by using an approach based on predicted no-effect concentrations PNEC in different receiving compartments (air, water, and ground), taking into account pathways and initial emissions. This paper illustrates a practical case study where the EQO normalisation approach is used and where the focus is on toxic substances. It illustrates that the generic approach, which assumes that the chemical is released in its most toxic form, may lead to unrealistic results. The result may also be significantly affected by new information, which leads to improved assessments of the impacts of e.g. the naphthalene content in creosote and its toxic assessment. Therefore, when toxic effects are taken into account a more careful interpretation of the inventory data from generic databases as well as from primary industry sources is required before the data are used in an LCA.

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7 Appendix

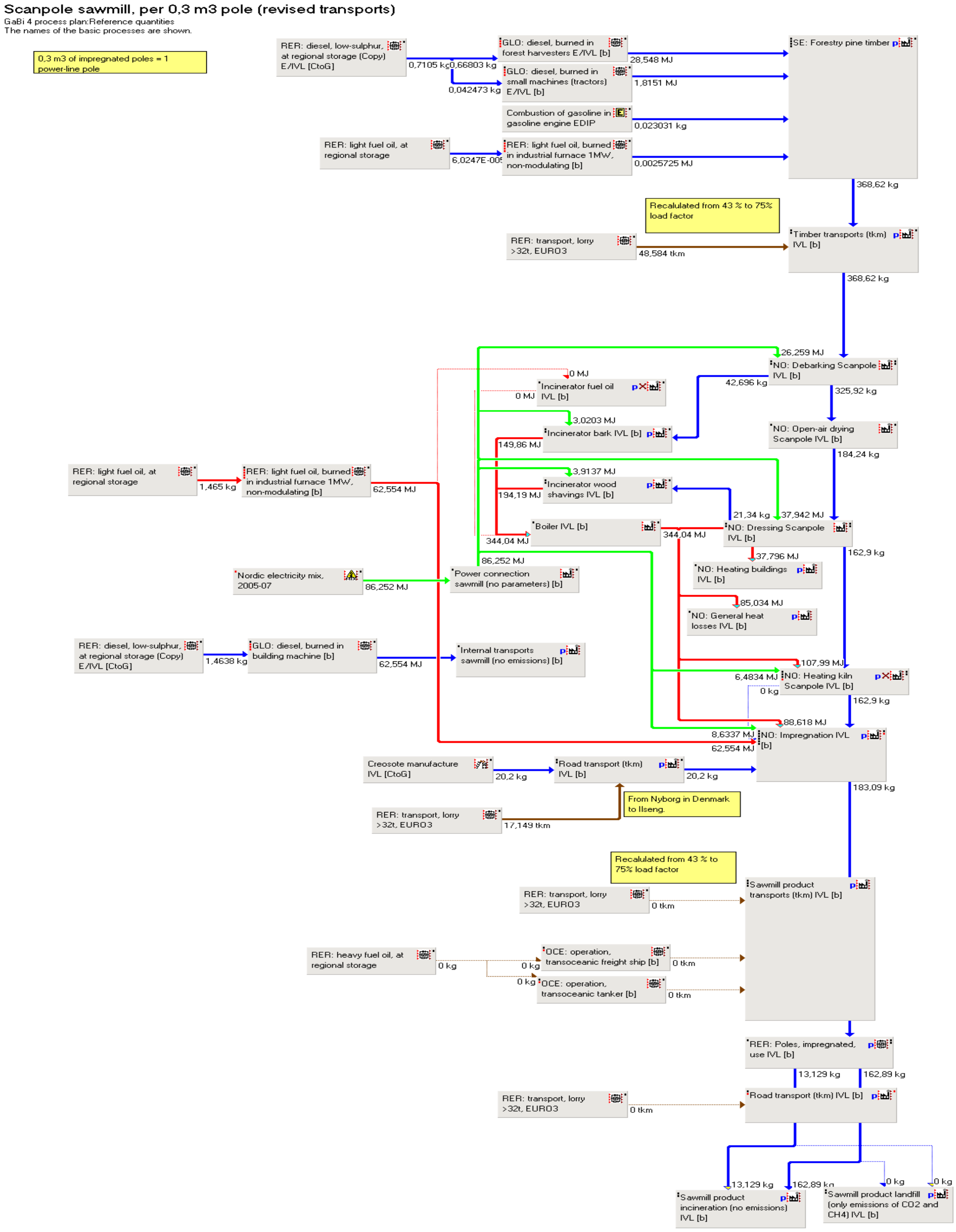
7.1 Documentation of life cycle inventory work

7.1.1 Life cycle inventory of creosote-impregnated poles

The major assumptions behind the LCA of wooden poles are listed below. (Data from ScanPole, Norway).

Process stage	Data sources, other assumptions
Timber-cutting	Consumption of fuels and emissions of CO_2, CO, NMVOC, NO_x, SO_x and dust from diesel engines: Swedish forestry data 2006. Other emissions from diesel engines in machines: Swiss off-road data (Ecoinvent database). Emissions from petrol engines: EDIP database. Emissions from heating with fuel oil (forest nursery): Light fuel oil in an industrial 1-MW furnace (Ecoinvent database).
Log transport to pole factory	Distance: 200 km one way with fully loaded trucks. The trucks have 50 % return load as an average, which gives a load factor of 75 % as an average for the entire transport Emissions from the truck: Data for an average European heavy-duty truck (> 32 tonnes, Euro III) with an average payload of 11,7 tonnes (43 % load factor) (Ecoinvent database). An increase of the load factor from 43 % to 75 % reduces the fuel consumption per tkm to 0,659 of the consumption at 43 % (calculated from data in the GaBi professional data base). This is taken into account by reducing the actual transport distance accordingly.
Pole factory	Creosote type B is used for impregnation with a by NTR required uptake class A of 110 kg/m^3 sapwood that here are assumed to correspond to 60 kg/m^3 (i.e. a conservative estimate, since 50% heartwood generally is accepted) Heating with biofuels (bark and shavings): Emission data for small stationary plants: Boström, C_Å, Flodström, E, Cooper, D. 2004. Emissionsfaktorer för stationär förbränning. Rapportserie SMED Nr 3 2004. Heating with fuel oil: Light fuel oil in an industrial 1-MW furnace (Ecoinvent database). Emissions from internal transports: Swiss off-road data for diesel engines in building machines (Ecoinvent database). Electricity: Average Nordic production mix 2005 – 2007. Emission data for the individual types of power plants from GaBi and Ecoinvent databases. Emissions of VOC from drying wood: Neglected.

Process stage	Data sources, other assumptions
Creosote manufacture	Data for creosote type C partially from F. Werner (2008), "Ökologische Bilanzierung von Eisenbahnschwellen" ,report to Studiengesellschaft Holzschwellenoberbau e.V., Umwelt und Entwicklung, Zürich. Our assumptions: To 1 kg of creosote is allocated the consumption of 1 kg of tar, since mass allocation between the products of tar distillation yields essentially the same result as allocation based on energy content or price (F. Werner). The creosote output is about 5 - 8 % of the total output from tar distillation. Transport of creosote to the mill: 849 km by truck from Nyborg (Denmark).
Transport to customers	
Use	Service life: 50 years Release of creosote to the environment during the use: 70 % of the amount originally in the poles, divided in 15 % assumed to evaporate to the air and 55 % to be discharged to the ground, see section 3.5 for more details.
Transport to waste disposal	
End-of life scenarios	Two alternatives are accounted for: Recycling or energy recovery where incineration with energy recovery is the most likely one. The emissions from the incineration are allocated to the generated energy (i.e. the heat or power) and not to the pole. .
Methodology assumption	When calculating the eutrophication potential it is assumed that the BOD, DOC and TOC parameters (analytical measures to fresh and to sea water) measure the same pollutants as the COD parameter. To avoid a double-count, only the COD value is included in the reported eutrophication potential.

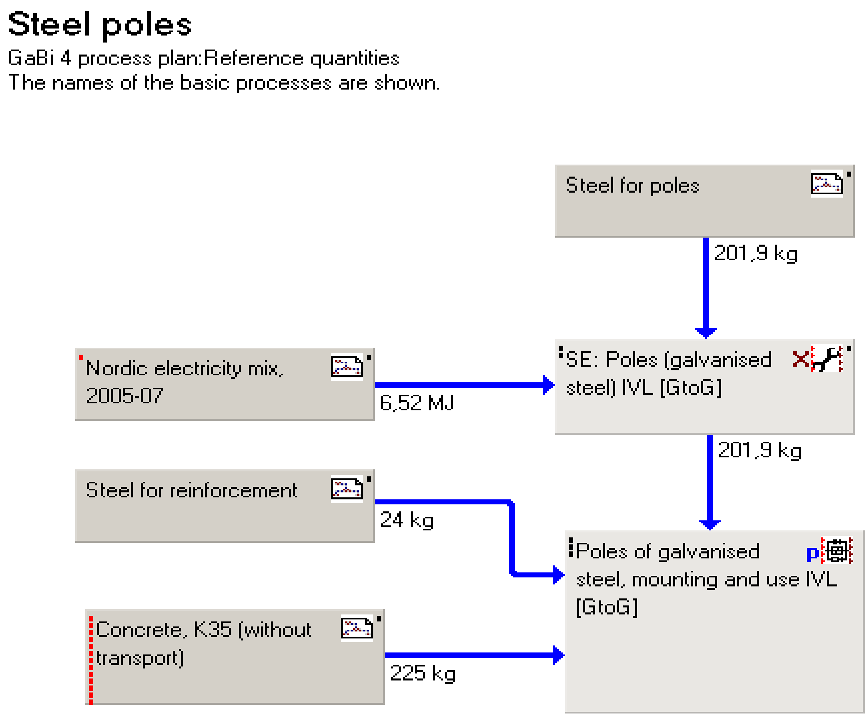


Flowchart of the inventory model of wooden poles in GaBi. Flow data for 1 pole with a size of 0,3 m³.

7.1.2 Life cycle inventory of steel poles

The major assumptions behind the LCA of steel poles are listed below.

Process stage	Data sources, other assumptions
Manufacture of galvanised steel	Each steel pole consists of 200 kg of low-alloyed steel coated with 1,9 kg of zinc. The surface area of the steel framework is assumed to be 60 m^2/tonne. The steel profiles are assumed to be manufactured from 50 % ore-based and 50 % scrap-based steel. Data for steel production (converter and electric arc furnace respectively) pertain to average European conditions (data from the database Ecoinvent). The steel is assumed to be hot-rolled and section-bar rolled (average European data for these processes from Ecoinvent). The galvanising process is assumed to be hot dip-coating performed batchwise (average European data from Ecoinvent). The steel pole is installed in a concrete foundation of 1,5 meter and a weight of 225 kg. The concrete quality C35 is reinforces with 25 kg recycled steel.
Manufacture of concrete	The applied concrete quality C35 consists of 230 kg cement per m^3. The cement is produced at the Cementa manufacturing unit at Slite/Gotland. For more details on the cement see description in section 7.1.3.
Reinforcing steel	Low-alloyed steel manufactured from scrap, hot rolled and section-bar rolled. (Average European data from Ecoinvent).
Transports	
Pole factory	Available data for production of lamp-posts made from low-alloyed steel have been used. Data on pre-processing (cut and bend) is extrapolated from "Birstaverken AB, LCA av väg- och broräcken", VBB VIAK AB, 2000-06-06, i.e. 0.4 MJ electric power per produced lamp-post. Electricity consumption from welding is assumed to be the same as for stainless lamp-posts, 6,12 MJ. (This may be an overestimate, since it is known that welding of stainless steel is more time consuming than welding of low-alloyed steel). Loss of material is neglected.
End-of-life	The technical service life of poles is assumed to be 50 years 1,2 % of the steel is lost during the service life (estimate based on data from the Swedish Corrosion Institute, K. Tjus et al (2004) "Metodstudie för ekologisk produktutveckling", IVL report B 1578). It is assumed that, averaged over the service life of the poles, 90 % of the surface area subjected to corrosion still has a zinc layer. Hence the corrosion causes zinc in water as an emission, to this extent. Low-alloyed steel has approximately the composition 99 % Fe, 0.09 % Cr, 0.09 % Ni, 0.73 % Mn and 0.01 % Mo. It is assumed that corrosion releases metals to water in that proportion from an estimated average zinc free (corroded) area of 10 % over the service life of the poles. The reinforcing steel is assumed to remain in the scrapped concrete.



Flowchart of the inventory model of steel in GaBi. Flow data for 1 pole.

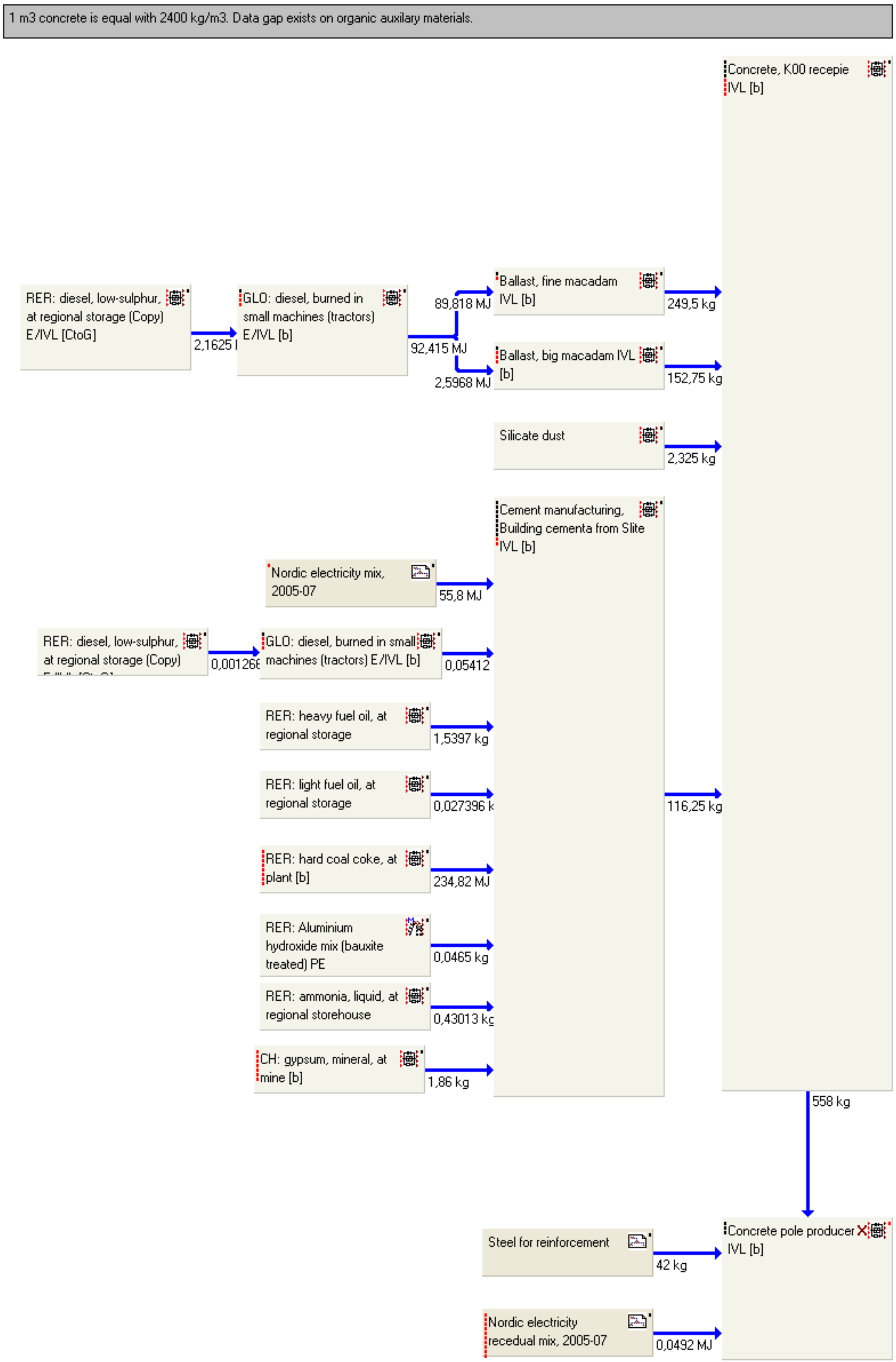
7.1.3 Life cycle inventory of concrete poles

The major assumptions behind the LCA of concrete poles are listed below.

Process stage	Data sources, other assumptions
Manufacture of cement	Specific data from the Cementa manufacturing plant in Slite/gotland is used as data source as it is the largest cement producing plant in Sweden. The impact from the manufacturing site is found in the yearly Environmental report from 2006. These data are representative for the current production in 2009 (p.c. Kerstin Nyberg, Cementa, August 2009). The distribution between chromium III+ or VI in the cement clinkers is 60 respectively 40%. However, it is not sure that this distribution also is representative for the air emission from the cement kiln. Instead it is assumed that only 5% of the chromium is hexavalent that corresponds to the average distribution in air according to IVL report No B1762 (Woldegiorgis A et al 2007). Missing emission data in the eEvironmental report 2006 from Cementa on gases contributing to climate change is complemented, based on generic emission figures used by the Swedish Environmental Protection agency for international reporting (the so called annex 25), which is an update of the emission figures reported by SMED report No 3 2004 (Boström et al 2004). Data for use of energy wares and auxiliary materials is based on generic data found in Ecoinvent.
Manufacture of concrete	The applied concrete quality C35 consists of 230 kg cement per m^3. The cement is produced at the Cementa manufacturing unit at Slite/Gotland. For more details see the cement description in section 7.1.3.
Reinforcing steel	Database data is used for the reinforcing steel in the inventory from Ecoinvent. Generally data on low-alloyed steel manufactured from scrap representing an European average is selected.
Transports	
Pole factory	Each concrete pole consists of 0,25 m^3 concrete C100. The concrete quality C100 have cement content of 500 kg/m^3. Data for production of concrete poles are collected from Abetong (Erlandsson 1991). The concrete poles are centrifugal cased with an electricity consumption of 0,082 MJ/kg. In the calculation it is assumed that the steel reinforcement made of 100 % recycled steel. The final poles have a steel content of 7 wight-% according to Abetong (Erlandsson 1991). Loss of concrete etc for manufacturing is neglected.
Usage phase	No emission on leaching from the concrete service life is accounted for. The leaching of components from concrete such as hexavalent chromium is not accounted for since it is assumed to be limited, at least compared to the contribution from the cement kiln and the emission to air.
End-of-life	The technical service life of poles is assumed to be 50 years. It is assumed that the concrete pole will be used for land filling when it is discharged, i.e. downgraded and not recycled. This implies that the reinforcing steel is assumed to remain in the scrapped concrete. No significant emission from the land fill is accounted for. The transportation to a landfill is assumed to be within 50 km from the transmission line.

Concrete poles, K100 (without transport)

GaBi 4 process plan:Reference quantities
The names of the basic processes are shown.



7.2 Selected data for characterisation factors for creosote in USES-LCA

Table 4 Source data used for calculation of characterisation factors on human and ecological toxicity in USES-LCA. See 3.4 for references.

Compound name	**Name**	**Unit**	**Creosote (WEI B)**	**Naphthalene, improved**
CAS nr.	CAS	-	8001-58-59	91-20-3
Effects assessment				
Acceptable/Tolerable Daily Intake or Reference Dose for man	ADI/TDI/RfD	kg/kg*d	7,45E-08	6,00E-08
Acceptable/Tolerable Concentration in air for man	HACair	kg/m3	6,00E-09	3,00E-09
Maximum Tolerable Concentration for aquatic compartment	MPCwater	kg/m3	1,00E-04	2,90E-07
Maximum Tolerable Concentration for fresh water aquatic compartment	MPCwater fresh	kg/m3	no value	no value
Maximum Tolerable Concentration for salt water aquatic compartment	MPCwater salt	kg/m3	no value	no value
Maximum Tolerable Concentration for sediment compartment	MPCsediment	kg/kg(wwt)	2,00E-06	EP
Maximum Tolerable Concentration for fresh water sediment compartment	MPCsediment water fresh	kg/kg(wwt)	EP	6,72E-08
Maximum Tolerable Concentration for salt water sediment compartment	MPCsediment water salt	kg/kg(wwt)	EP	EP
Maximum Tolerable Concentration for soil compartment	MPCsoil	kg/kg(dwt)	3,00E-07	5,33E-08
Inorganic substance, but no metal?	Inorganic?	yes/no	no	no
Physico-chemical properties				
molecular weight	MW	g/mol	200	128,19
octanol-water partition coefficient	Kow	-	2,69E+04	2,34E+03
melting point	TEMPmelt	C	30	80,5
vapor pressure (25)	VP	Pa	6,60E-01	10,4
solubility (25)	SOL	mg/l	8	31
dissociation constant for acids	pKa	-	1,00E+01	no value
Is the compound a metal?	Metal_x?	yes/no	no	no
Partition coefficients				
Henry's law constant (25C)	HENRY25	(Pa-m3/mol)	3,60E+00	49
organic carbon partition coefficient	Koc	l/kg	9,33E+03	933,254301
solid-water partition coefficient soil	Kp(soil)	l/kg	no value	no value
solid-water partition coefficient sediment	K(sed)	l/kg	no value	no value
solid-water partition coefficient suspended matter	Kp (susp)	l/kg	no value	no value
Aerosol collection efficiency	COLLECTeff		no value	no value
Fraction of aerosol bounded substance	FRass(aerosol)	-	no value	no value
aerosol deposition velocity	AEROSOLdeprate	m/s	no value	no value

Compound name	**Name**	**Unit**	**Creosote (WEI B)**	**Naphthalene, improved**
CAS nr.	CAS	-	8001-58-59	91-20-3
Degradation rates				
reaction half-life in air	DT50air	d	7,45E-08	6,00E-09
hydroxyl radical reaction in air (-10C)	OH rad	cm3/molec-sec	6,00E-09	2,45E-11
hydroxyl radical reaction in air (12C)	OH rad	cm3/molec-sec	1,00E-04	2,90E-07
hydroxyl radical reaction in air (25C)	OH rad	cm3/molec-sec	no value	no value
hydrolysis in surface water (PH=6, 20C)	DT50 hydro water (PH=6)	d	no value	no value
hydrolysis in surface water (PH=7, 20C)	DT50 hydro water (PH=7)	d	2,00E-06	EP
hydrolysis in surface water (PH=8, 20C)	DT50 hydro water (PH=8)	d	EP	6,72E-08
biodegradation in surface water (20C)	DT50bio water (20C)	d	EP	EP
abiotic degradation in soil (PH=6, 20C)	DT50soil abio (PH=6)	d	3,00E-07	5,33E-08
abiotic degradation in soil (PH=7, 20C)	DT50soil abio (PH=7)	d	no	no
biodegradation in soil (20C)	DT50soil bio (20C)	d		
abiotic degradation in the sediment zone (PH=7, 20C)	DT50sed abio (PH=7)	d		
abiotic degradation in the sediment zone (PH=8, 20C)	DT50sed abio (PH=8)	d	200	128,19
aerobic biodegradation in the sediment zone (20C)	DT50sed areobic (20C)	d	2,69E+04	2,34E+03
anaerobic biodegradation in the sediment zone (20C)	DT50sed anearobic (20C)	d	30	80,5
metabolism in plant tissue	DT50plantmetabolism	d	6,60E-01	10,4
photodegradation upon plant tissue	DT50plantphoto	d	8	31
Exposure assessment				
bioconcentration factor in fish relative to contaminant water concentration	BCFfish	l/kg(wwt)	2000	398,107171
Partitioning coefficient between leaves and air	Kplant -air	m3/m3	no value	no value
Conductance	g(plant)	m/s	no value	no value
Transpiration stream concentration factor	TSCF	-	no value	no value
Root concentration factor relative to contaminant porewater concentration in soil	RCF	l/kg wwt	no value	30
bioconcentration factor in plant roots relative to contaminant soil concentration	BCFroot-soil	kg wwt/kg wwt	no value	no value
bioconcentration factor in plant leafs relative to contaminant soil concentration	BCFleaf-soil	kg wwt/kg wwt	no value	no value
biotransfer factor for meat	BAFmeat	d/kg(food)	no value	no value
biotransfer factor for milk	BAFmilk	d/kg(food)	no value	no value
Respirable fraction of inhaled substance	Fresp	-	no value	no value
Bioavailability for inhalation	BIOinh	-	0,29	no value
Bioavailability for oral uptake	BIOoral	-	0,5	1