RAPPORT

Comparison of the environmental impacts from utility poles of different materials

— a life cycle assessment

Martin Erlandsson

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John Munthe
Vice President, Research

IVL Swedish Environmental Research Institute

Organization

IVL Swedish Environmental Research Institute AB

Address

Box 21060
100 31 Stockholm

Telephone

+46 (0)8-598 563 00

Report summary

Project title

LCA on utility poles including composite

LCA för ledningsstolpar inklusive komposit

Project sponsors

Svensk Energi (Swedenergy), Telia Sonera Skanova Access, Jerol Industri, Svenska Träskyddsföreningen,

Author

Martin Erlandsson

Title and subtitle of the report

Comparison of the environmental impacts from utility poles of different materials — a life cycle assessment

Summary

This report describes a method for environmental assessments of telegraph poles made of different materials from a life cycle perspective. Poles of creosote-impregnated wood, steel, composite and concrete have been compared with the help of life cycle analysis methodology. There are several type of life cycle assessment (LCA) and since this was a comparison analysis we chose to use a robust methodology which we call product LCA: This methodology is characterised by allocating all emissions from a process to the products of the process. If one sums the environmental impact of all such products then this would ideally be equivalent to the emissions globally. In other words, a product LCA describes the environmental impacts allocated to products based on the way it is in the physical reality. Socio-economic allocation principal, margin approach or so-called system expansion etc. is not applied in a product LCA approach.

Environmental impacts are described using the common environmental impact categories and USES LCA 1.0 which is the method most widely spread and is better at assessing metals then newer methods. The result of the LCA shows that the most significant environmental aspects of all pole types is emissions of metals from steel poles during the life cycle, which impacts ecotoxicity and human toxicity. The steel pole was also the pole type which had the largest contribution to other environmental impact categories. A sensitivity analysis was undertaken with respect to the attributed service life of the poles, but this uncertainty factor is judged not to impact the "ranking" of the different alternatives studied.

The result of the LCA calculations is only applicable to the chosen products and the assumptions upon which the assessment was made. The conclusions which can be drawn regarding emissions of metals should be treated with some caution with consideration for the model uncertainty in handling human and ecotoxicity in an LCA. Supplementary risk assessments therefore provide valuable additional information. Creosote-impregnated wood has been judged by The Swedish Chemicals Agency in such a risk assessment, according to the methods applied in the Biocide directive, and according to this assessment creosote-impregnated products are approved for professional use in certain applications such as sleepers. A continued use of creosote for impregnation of poles requires national approval in Sweden and several other EU countries if these kind of products shall be installed after 2013.

Keywords

Composite pole, creosote-impregnated pole, creosote-impregnated leaching, utility poles, life cycle assessment (LCA), material choice, product LCA, chemical assessment, risk minimisation, steel pole

Bibliographic details

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1 Introduction

It is always interesting to compare the environmental performance of different products where different options are available. In order to make a fair comparison of the environmental impact of the products must be assessed from a life cycle perspective. Naturally, all products studied must meet a defined fundamental technical performance and thereby have fundamentally comparable functionality, in order that the alternatives studied are comparable.

It is also important that the comparison assesses as many environmental impact categories as possible in order to avoid a situation where a decision to replace one product by another leads to the replacement of one environmental problem by another. Similarly, it is not desirable to move a problem from one stage of a product's life to another (for example, the environmental impacts of one product may be large in the manufacturing phase, whilst another product's impacts may be large in the usage phase). A life cycle assessment (LCA) is a powerful analytical tool in this context for handling environmental impacts. In addition, normalisation using environmental quality objectives provides an assessment of the ecological sustainability from a life cycle perspective.

In general, in the process of selecting which material to use, a comprehensive assessment is made which includes sustainability aspects along with business and technical factors etc. All factors are weighed into the decision. If all sustainability aspects are to be considered (i.e. also social and socioeconomic factors) then additional information must supplement the LCA. Furthermore, an LCA does not handle all types of environmental impacts, e.g. work environment, which may be of interest in this case study. An LCA can be supplemented with other data in order to cover more environmental aspects.

Often the results of an LCA show that products from different materials burden the environment in different ways and in different parts of the life cycle. Assessment of the environmental impacts of wood preservatives is covered by the Biocide Directive. Assessment according to the Biocide Directive is done for the whole European market and mainly consists of a risk assessment of the handling and use of the product with regards to human and ecotoxicity. This results in a list of substances that can be accepted for certain purposes covered by the directive. The EU has assigned the Swedish Chemicals Agency with the task of undertaking such an assessment of creosote in relation to the different areas that creosote impregnated products are used.

In its investigation[1] the Swedish Chemicals Agency chose to broaden the information on which to make the assessment as far as possible within the scope of the Biocide Directive, by commissioning an LCA where the environmental impacts of the different material choices were evaluated by LCA i.e. comparison with other materials in a life cycle perspective. IVL undertook the LCA (Erlandsson et al 2009), which looked at different types of poles, and which formed part of the information provided to the EU[2] by the

[1]http://circa.europa.eu/Public/irc/env/bio_reports/library?l=/review_programme/ca_reports/wood_preservatives/creosote_versionpdf/_EN_1.0_&a=d
[2]http://europa.eu/rapid/pressReleasesAction.do?reference=IP/11/925

Swedish Chemicals Agency. Poles were chosen as the subject of the study because this also provides information which is relevant for concrete and wooden sleepers, and also includes a comparison with steel products. The Biocide Directive approves the use of creosote impregnated wood for certain applications for a further five years until 1st May 2013[3,4] (after that date a new assessment will be made following the method used for other biocides). It is therefore highly interesting to develop new alternatives to creosote impregnated wood such as wood impregnated with other preservatives or completely new material choices such as composite poles or combinations of materials.

3 http://ec.europa.eu/environment/biocides/creosote.htm
4 http://www.kemi.se/templates/News____6640.aspx

2 Objectives, aims and limitations

The aim of the project is to produce a comprehensive report based on a product LCA for the different material alternatives available today for poles. This LCA builds on the calculations already reported for concrete, steel and creosote impregnated wood in the IVL report B 1865 (Erlandsson, Almemark 2009). The calculations in that LCA have been supplemented and extended, and now further supplemented with data for one of the composite pole alternatives now on the market. The aim of the investigation is to gain an understanding of the environmental impacts of a composite in comparison to the other alternatives that have already been analysed.

The composite pole alternative selected for the analysis is a fibreglass pole with a coating (shell) of polyethylene, which makes it possible to climb the pole in the same way as a wooden pole. Other composite pole alternatives are also on the market, but the differences between them from an environmental point of view were not great enough to warrant inclusion in this case study.

The data on manufacture of the composite poles is based on actual data from a production plant (Jerol plant in Tierp, Sweden). The environmental impacts of the raw materials used in the manufacturing are based on sources from a number of commercially available so-called LCA databases. Sensitivity analysis has been undertaken on the different data sources and the data which is considered most representative for the purchased raw materials has been selected.

In the previous LCA on which this study builds, toxicity was handled using the USES LCA 1.0 method. In the last few years new assessment methods have been developed, such as USES LCA 2.0 and USEtox (which both build on the same basic method assumptions). We have calculated toxicity based on USEtox and compared results with USES LCA 1.0. We conclude that the new methods are not applicable for metals, which is also stated in the documentation provided for the new models. We have therefore chosen not to use these new methods in our LCA and have instead used USES 1.0[5] and the normalisation method developed by IVL for ecotoxicity (linked to this assessment method) which we consider to have more robust method assumptions and which thereby better mirrors the potential environmental impacts.

The work environment is not normally assessed in an LCA and has therefore not been included in this study. Some efforts were made to include work environment but there is currently no generally accepted method available.

[5] That is based on a steady-state model instead of a dynamic model integrating the impact over infinite time

3 Life cycle assessment methodology

3.1 Introduction to LCA

3.1.1 Product LCA

There are several different LCA methods which meet the LCA standard ISO 14044. In this context the terms *attributional LCA* and *consequential LCA* are usually used. This method choice can also be described in terms of the *system perspective*, i.e. in which way the system should be analysed, depending on the question which the study is trying to answer.

In this report, *product LCA* is synonymous with an LCA method which aims is to describe the environmental impact of products as clearly as possible. This means that the method choice is driven by the need to describe and model processes and events in a way that can be verified in real life, i.e. the environmental loads match the loads found in inventories, both geographically and temporally. An important characteristic of a product LCA is that all environmental loads that arise are allocated to the processes from which they arise, both concerning manufacturing and in material recycling. In other words, if the environmental impacts that are allocated to each separate manufacturing process in a product LCA are summed together, then the total should equal the global environmental load that arises. Another consequence of this method is that the environmental data is *modular*, i.e. the environmental impact from each sub-process and each raw material, supplied energy wares, auxiliary products etc can be added together.

A well-performed product LCA should therefore not include any double-counting, nor any processes where the related environmental impact is not allocated to its products. In practice, some double counting does occur, but usually not to such an extent that it has significant impact on the final results.

The conditions on which a consequential LCA is based are different and the assessment is based on different premises. A consequential LCA assesses a product system and how a marginal change in production affects the total system. This type of LCA provides completely different results which are dependent on scenario assumptions about how linked systems changed when production increases or decreases. This type of approach is not appropriate in this case and has not been included in the analysis, which is based on the product LCA methodology.

3.1.2 Life cycle inventories

Life cycle assessment is an analytical tool; the environmental impact is calculated based on an inventory of environmental loads which arise in the analysed system. This so-called *life cycle inventory* analysis is undertaken for all processes in the product system studied. Simplified, a product's life cycle can be divided up into a number of life cycle phases as illustrated in the figure below.

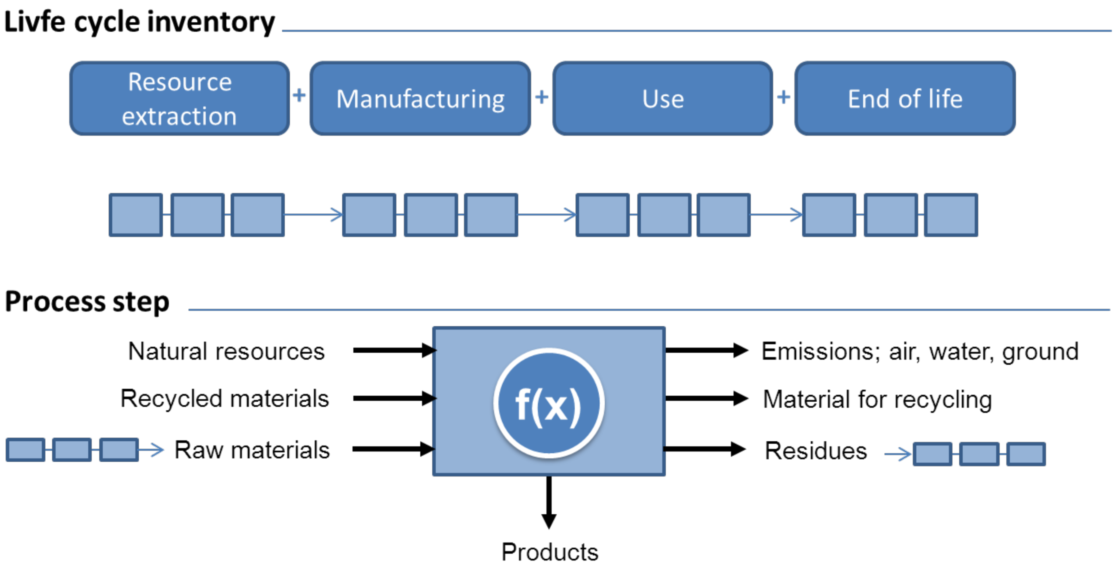

Figure 1 Life cycle inventory analysis, divided into four life cycle phases which together describe the product system and its underlying process steps.

The environmental load, in the form of resource use and emissions to the environment, is described for every process in the product system. The raw materials to the production process give rise to inventory analysis to assess its upstream environmental burden (i.e. the environmental loads from producing this product). In the same way, the inventory analysis must also look at the waste products that arise from the product system (i.e. what is the environmental impact of processing waste products including waste disposal). Of course, this means that in practice even the life cycle inventory of a relatively simple product is quite extensive.

Ideally, all flows in an inventory should be followed back to source, i.e. to an extraction of a natural resource, and followed forwards to emissions. In a product LCA there is also a need to assess which resources are made of recovered products and which can be recycled.

An important general characteristic of an LCA is that the environmental load which arises from a sub-process must be allocated to the products of the process. There is an international standard, ISO 14044, which describes how this allocation should be implemented. This standard defines the stepwise priority for allocation and can be interpreted in different ways, with the result that the choice of allocation method is one aspect which can differ between different LCAs. From experience, the choice of method for process allocation can lead to significant differences in results, and if different data sources are being used then it is important to check that they follow the method chosen for the current study. In this study, we follow ISO's main rule and do the process allocation based on the physical cause-effect relationship, which in practice is often simplified so that the environmental load is allocated by mass (i.e. all products arising from one process are allocated the same environmental impact per unit mass).

Figure 1 shows that material resources from and to society are included in the inventory, or put more correctly, a 'simply cut' between different product system that use the same recycled material resources is introduces. This implies that the recycled material is free of environmental loads raised in the previous product life cycles, when the material s used in a new product. This type of inventory analysis is the most common way of handling material use from worn out products in an LCA-based *Environmental Product Declaration* (an '*EPD*'). In an EPD it is therefore normally assumed that it is 'free' to use recycled products, and consequently that there is no 'bonus' from recycling products, apart from the waste disposal avoided when a product is recycled. This way of handling environmental impact describes the actual environmental load that arises (without, however, consideration for socio-economic aspects).

Other methods for handling material recycling between different product systems are also permitted by the ISO standard for life cycle assessments. The aims of alternative methods are, for example, to describe the socio-economic relationship, where future credits for recycling metals are, in principle, assimilated to the product today – in cases where there is a high likelihood that the input material will be recycled in future.

Another method for handling material recycling between different product systems (which is also in line with the international LCA standard) is to address the question: When the product is recycled, what production is avoided? In other words, what avoided production and which environmental impact can be deducted from the product system in question? When a material is combustible it is assumed that the material replaces another fuel. For renewable material like wood, this means that the resultant environmental impact can be negative, since the so-called marginal fuel is often a fossil fuel. This analysis result does not mean that the emissions are negative in reality (or *avoided emissions* as they are sometimes referred to), but that the analysis contains an 'extra' function and answers the question: What is the environmental impact of my product and its initial use, assuming that the product will be used in the place of another fuel when it is worn out?

There is not currently a generally accepted method for dealing with material recycling in LCAs. In the LCAs in this report we have choosed to follow the methodology which is generally used in EPDs. Put simply, one can say that it is not a completely correct method, but that it has sufficiently relevance without introducing subjective choices which can have large impacts on the analysis results. Furthermore, this is the methodology that will be applied for all building materials in the EU and which will follow the EPD rules defined in the Building Products Directive (see prEN15804:2011).

3.1.3 Life Cycle Impact Assessment

Inventory analysis results in a summary of the different environmental load factors such as emissions (SO_2, CO_2 etc.). In order to interpret the inventory results a *Life Cycle Impact Assessment* (*LCIA*) is undertaken. For each category of environmental impact a characterisation model is developed which describes the environmental impact. This characterisation model can describe a potential effect, converted to a category indicator (e.g. CO_2-equivalents). The ISO standard for LCA permits everything from an inventory result (often called a *midpoint*) to a *category endpoint* as a category indicator. A characterisation model at a midpoint generally has lower model uncertainty, but handles only parts of the environmental mechanism. Similarly, a category endpoint has high environmental relevance, but at the cost of higher model uncertainty. Ideally, a category endpoint describes potential consequences on areas of protection (see figure below).

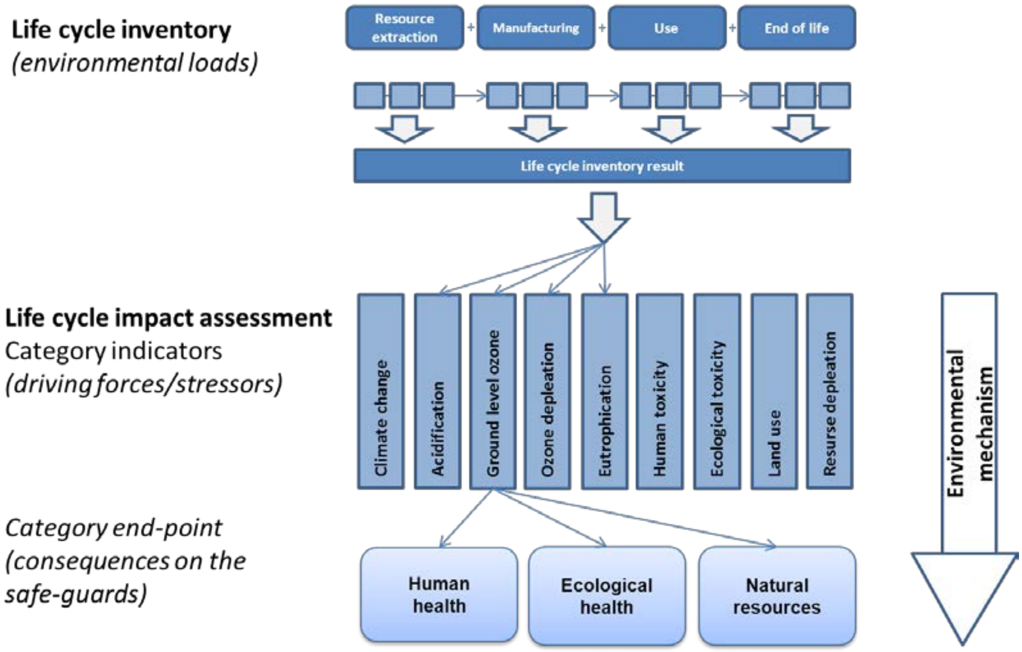

Figure 2 *Environmental assessment based on inventory results which are transformed into contributions to environmental impact categories through a so-called categorisation model and its characterisation factors. A category indicator can be found anywhere in the range from a midpoint to a category endpoint, the latter of which has higher environmental relevance at the expense of higher model uncertainty.*

The ISO standard describes some different possibilities which can increase understanding and interpretation of inventory data through environmental impact assessment over and above conversion to different category indicators: normalisation and weighting. A critical factor in weighting is the evaluation of the importance of the different area of protection. It is common to normalise results in an LCA and the 'classic' method is to normalise against the alternative that has greatest environmental impact in each separate category. A disadvantage of this method is that is no relative importance between the environmental impact categories is obtained. This relative importance can be obtained where, for example, the normalisation is based on all emissions in the region. Another alternative which has greater environmental relevance is to normalise with regards to what the natural environment can tolerate, as described in our environmental quality objectives. IVL has developed one such normalisation method (Erlandsson 2003) and the environmental impact assessment leads to a result where the relative importance between the different environmental impact categories is illustrated. Such a method and illustration builds on the concept that ecological sustainability can only be achieved when all environmental aspects are met (and are thus considered to have equal importance, which is in itself an indirect valuation).

The ISO standard specifies how LCAs involving comparisons should be reported. The results should be reported for each environmental impact category separately and no directly subjective valuation should be included if the normalisation method is chosen. In order to make the results as easy as possible to interpret, and without at the same time introducing subjective weighting methods, the results of this LCA study are reported with the normalisation method that IVL has developed.

3.2 Emissions of toxic substances

There are characterisation models available which enable an assessment of how toxic emissions of different substances to air or water in an LCA. The most common method approach assumes that emissions are released in a unit world which is a simplification of reality. This unit world divides emissions up in different levels, from local level, to a regional and even global scale. This enables local emissions to be analysed on a global level. Historically, USES LCA 1.0 is possibly the most common and most applied characterisation model, since it is recommended by CML and the ISO guidance which has been developed for the ISO 14040 series standards. USES LCA 1.0 is built on the same calculations as EUSES, which is the model for risk assessment developed in the EU to handle, for example, large-scale risk assessments of chemicals according to REACH.

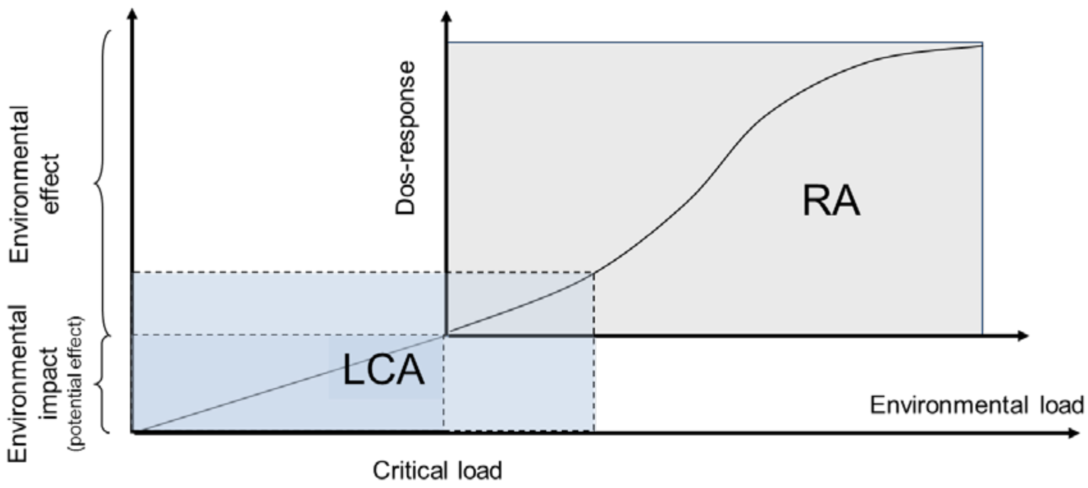

Figure 3 Relevant working areas for an LCA which are applicable for risk minimisation and where a risk assessment can be made in order to ensure that the emissions in the analysis give rise to acceptable levels, i.e. levels under or around the critical load (see the blue field).

An LCA groups emissions from different sources and different life cycle phases and in the model it ”emits them into the same unit world”. This means that an LCA gives a relative value on a potential effect and is therefore useful for risk minimisation and is not an absolute risk assessment. It is therefore preferable that a risk assessment is first undertaken to check whether the expected concentrations in nature and human exposure is acceptable (i.e. fundamentally, that the environmental quality objective is met). In an LCA the contribution to human and ecotoxicity is seen as a linear contribution which is proportional to the emissions to a specific recipient. This simplification can been seen as acceptable as long as the "actual" levels are below acceptable guideline values, or approximately equal to them, see Figure 3.

USES LCA 1.0 was the characterisation model used to handle toxicity in the previous LCA case study performed. In recent years new characterisation models have become available, such as USES LCA 2.0 and USEtox. In general it can be said that these new models are build on the same or similar basic calculation concept (i.e. multimedia models), but new method assumptions have been applied. For example, in USES LCA 1.0 emissions are calculated as a continuous flow with equilibrium concentration (steady-state), whilst in USES LCA 2.0 and UseTox emissions are integrated over infinite time and integrate effects/exposure over time. This refinement makes a large difference for more persistent substances and metals. A normal timespan for human toxicity risk assessment is usually 70 years in comparison. Another important difference is that instead of using PNEC (predicted no effect concentration), the newer models make as assessment based on a risk level which gives fifty percent of the maximal effect i.e. EC50 (half maximal effective concentration). In other words, if one of the new characterisation models is used then the values are based on a non-linear dose-response relationship-. in the context of an LCA, this method development can be explained with the wish to be damage oriented, which simplifies any weighting.

We have undertaken calculations using these new methods and have concluded that they are not appropriate for metals. This is also stated in the documentation from the model developers. We have therefore chosen not to use these new methods in this LCA and have instead used USES LCA 1.0 and the normalisation method which IVL developed for ecotoxicity, which we judge to have more robust method assumptions and thereby better reflect potential environmental impacts appropriate for risk minimisation. The normalisation method which IVL developed is built on critical load limits and where damage oriented methods are thus not so relevant, and where the assessment is based on the ecologically sustainable conditions that we are striving for.

3.2.1 Normalisation

The normalisation used here is based on a judgement of what can, with current knowledge be regarded as an acceptable environmental load. In the normalisation this acceptable annual environment load is divided by the number of individuals in the analysed system (i.e. geographically). In this way an annual quota can be obtained which corresponds to the maximum emissions that one person may give rise to, assuming that everyone is allowed the same emission quota. This per capita emission is called a person equivalent [Pe]. CML (Heijungs 1992) and UMIP (Hauschild 1996) pioneered this concept. It has been found that this type of normalisation give a numerical value which is easy to communicate and intuitively easy to understand the meaning of. If one person's total annual consumption, and it environmental impact, is calculated and normalised in this way and the result is less than 1 Pe for all environmental impact categories then one can say that person's consumption corresponds to a sustainable lifestyle. In all other applications, the numerical number describes how many peoples' annual emissions (emission quota) the product loads the environment by.

The normalisation procedure can be separated into two steps, see Table 1. The normalisation factor specifies how much a person can impact on the environmental *annually* without jeopardizing a sustainable future. The factor is defined as follows:

$$nf_i = \frac{\sum I_{acc}}{\sum \text{individuals}} \tag{1}$$

where:

I_{acc}	load that is judged to be a sustainable total (annual) environmental impact or environmental state [load equivalents]
individuals	number of people existing in the system [Pe]
nf_i	normalisation factor for environmental impact category ***i*** [load equivalent/Pe]

The normalised environmental impact for a product, activity, system etc. is then calculated according to the following equation:

$$Inorm = \frac{Ieq_i}{nf_i} \tag{2}$$

Where,

Inorm	normalised environmental impact [Pe]
Ieq_i	potential contribution to environmental impact category ***i*** [load equivalent]
nf	normalisation factor for the environmental impact category [load equivalent/Pe]

Where,

$$Ieq_i = \sum_n m \cdot Ch \tag{3}$$

Where,

m	emission of a substance [g]
Ch	characterisation factor for a specific substance which describes its contribution to an environmental impact category **i** [load equivalent/g]

Table 1 provides the normalisation factor used in the LCA calculations in order to judge the relative significance of different environmental impact categories. The sustainable environmental impact used in this normalisation method is described in a number of reports from the Swedish EPA et al. (SNV 4995, 4999, 5000, 5002, 5003). See also Erlandsson (2003a, 2003b) for more information on the underlying assumptions.

Table 1 Normalisation factors (nf) for different environmental impact categories.

Environmental impact category	Normalisation factor (nf)	Critical load equivalent
Climate change	4 500	kg CO_2-eq/person
Acidification	29	kg SO_2-eq/person
Tropospheric ozone	1 150	ppb h km^2/person
Eutrophication	39	kg NO_3^-eq/person
Human toxicity	1 634	kg 1,4-dichloro-benzene eq.
Ekotoxicity	1	[-]

3.3 Assumptions in the case study

3.3.1 Functional unit

The functional unit describes the base for product comparison between the different alternatives, and in this study the functional unit is:

- One 9 m utility pole with a lifetime of 50 years, corresponding to its service life

Nine metre poles are usually used for 0.4 kV power transmissions lines and telephone lines (i.e. representative for the distribution grid). This is therefore a very commonly used type of pole, which is the reason it was chosen for the case study[6]. Nine metre poles are also a better alternative than longer poles (i.e. those used for the transformation grid) if one ”only” wants to compare pole material, since the considerably longer poles require another kind of foundation. For a nine metre pole the contribution from the foundations is less significant and is not the main determining factor in the environmental performance, as it is for longer poles. A ”medium” pole for power lines is certainly longer – often between 10 and 11 metres – but uses the same kind of foundation as 9 metre poles. Therefore the 9 metre pole is a preferable choice if the material choice for the pole itself is the focus of the study.

The service life of the pole is set at 50 years and all poles alternatives are assumed to have a lifetime that meets this service life. Service life refers to the period of time that the pole is expected to be in service in a grid, whilst the technical lifetime of the pole can be longer. It should be noted that there are no public statistics on the technical lifetime and service life[7] of telephone[8] and power grids and poles.

[6] Due to the product alternatives that meet the functional unit description, the results of this study can in principle also be used to draw parallel conclusions for concrete and wooden sleepers. However, it should be noted that the service life of sleepers is shorter than poles and that the leaching from sleepers during use is therefore lower, which should be taken into consideration in such a comparison.

[7] Personal communication Ulf Wagenborg, Svensk Energi, October 2011.

[8] Personal communication Conny Wallerius , Skanova, October 2011.

In a changing world we can expect that the lifetime of the lines themselves in smaller power grids and telephone networks will become shorter and that it may perhaps not be the technical lifetime of the pole that is the determining factor for replacement, rather it may be that lines need to be moved or modified than any other reason. As a result of this assumption a sensitivity analysis has been undertaken to analyse the effect on results of an alternative scenario.

3.3.1.1 Average lifetimes

Skanova states that they count on creosote impregnated poles have an estimated service life of about 40 years and that the first inspections for rot are made after 25 or 30 years8. Similarly, Svensk energi states that they consider creosote impregnated poles to have a service life of at least 40 years, but the lifetime of individual poles can be up towards 80 years7. Maintenance staff at Vattenfall judge that creosote impregnated poles have generally had a lifetime of over 50 years, but some poles already need replacing due to rot damage after 30-35 years[9].

Fibreglass based products have has been used industrially since the early 1940s and there is long practical experience of this raw material. Lighting poles made of polyester-reinforced fibreglass were manufactured in 1960 and 1961 and installed in Finland (Ekenäs Energiverk 1998). They are still in use, which means that they have been exposed to a climate similar to the Swedish climate for 50 years[10]. The fibreglass pole that is assessed in this study comes from Jerol Industri and is a fibreglass pole covered with a polyethylene coating, which both makes it possible to climb the pole and stops the fibreglass from being broken down by UV light or weather in the way that a pure fibreglass poles does. The manufacturer's estimate is therefore a lifetime of at least 80 years, based on experience of installed poles and on the fact that the composite pole in this study has a weatherproofing coat of polyethylene10. According to Jerol some other manufacturers on the composite pole market state a lifetime of 120 years for their composite poles, which are furthermore not coated with polyethylene. A lifetime of 80 years should therefore be a cautious estimate.

3.3.1.2 Sensitivity analysis

If one assumes that 50 years is the lifetime which we can expect for creosote impregnated poles, then it can be of interest to calculate results where the lifetimes for the alternatives are assumed to be longer. This sensitivity analysis is undertaken to illustrate the importance of the choice of (average) lifetime/service life for the different pole alternatives. The sensitivity analysis is based on assumptions, and the results should only be used to illustrate the consequences of assuming the lifetime of creosote poles to be 50 years and the lifetime of the alternatives to be longer. In these alternative calculations we have set the lifetime of creosote impregnated poles to be 50 years, concrete poles to be 60 years and composite and steel poles to be 80 years. Note that the leaching from steel poles in this case will take place over a longer time period and that the protective coating should therefore be thicker than the coating on the steel pole used in this analysis. In these calculations we ignore this

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[9] Personal communication Rikard Jernlås, SwedPower, October 2011

[10] Personal communication Rolf Jernström, Jerol Industri, October 2011.

issue but note that the total emissions will increase during the service life. In other words the environmental impact from the full life cycle will be split up over the lifetime given above in order to recalculate to 50 years (an alternative method would have been to state the environmental impact per year, but this does not affect the result).

The functional unit for sensitivity analysis is therefore changed to:

- One 9 m telegraph pole with a comparable lifetime of 50 years

3.3.2 The analysed pole materials

The different pole materials include in this study are briefly described below:

- **Steel:** of 50% recycled steel and foundation of 1.5 m made of concrete (C35). Data for manufacture is based on similar industrial manufacture.

- **Concrete:** High performance concrete (C100) and reinforcements made of 100% recycled steel. Data for manufacturing comes from Abetong and is a centrifugally cast pole with pretensioned reinforcement (this manufacture is no longer in operation).

- **Wood:** A turned wooden pole of pine which is impregnated with creosote WIE Type B, with an retention of 110 kg/m^3 sapwood according to the NTR-A manufacturing standard. With a conservative assumption this has total store of 60 kg/m^3 wood. Data for manufacture comes from Scanpole in Norway.

- **Composite:** The pole is manufactured from a pipe with a core of polyester-reinforced fibreglass and a shell of solid 3-4 mm coloured polyethylene which covers the outside of the pole completely. Manufacturing data has been provided by Jerol Industri in Tierp, Sweden.

All alternatives except the steel pole are assumed to have a foundation depth of 2 metres.

3.3.3 System limits

This study is limited to the poles themselves and does not consider the fastening devices nor any other differences in a power or telephone grid caused by the choice of material. Nor does the work involved in installing and removing lines is included since we assume that this is approximately the same for all of the alternatives studied and of little importance from a life cycle perspective. However, all other transportation during the products' life cycles is included in the inventory.

In the inventory work all construction material is traced back to its origins in the form of the natural resources needed for the different raw materials and energy sources. The handling of waste products includes the transportation of the material to the plant or storage yard for waste product treatment. The environmental impact downstream of waste product treatment is then assigned to future products. It is assumed that all material is recycled in some way, i.e. reuse, material recycling or energy recovery. According to the LCA methodology applied in this study, it is not necessary to consider which products are produced from this recycling; it is only necessary to consider that it is likely that they can be recycled, i.e. that there is a product system that will recover these worn out products. We can expect that creosote poles are used as fuel for energy recovery, steel poles undergo material recycling, composite poles are reused as poles or in other applications such as road culverts, and concrete poles are reused in the form of ballast in plant construction. In a product LCA the environmental impacts that arise after the material arrives at the waste product treatment business and further downstream (i.e. to further recycling or, if such is not possible, to future waste treatment) is to be assigned to the new products of the recycling process.

The assessment of environmental impacts is limited to the impact categories in Table 1. There are no characterisation factors available for resource consumption, which is compensated by also reporting energy use. In this way this case study does illustrate the differences between sending to landfill and recycling a wooden pole, for example as fuel.

4 Results

The result from this LCA calculation is illustrated in 4. 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 mPe) 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.

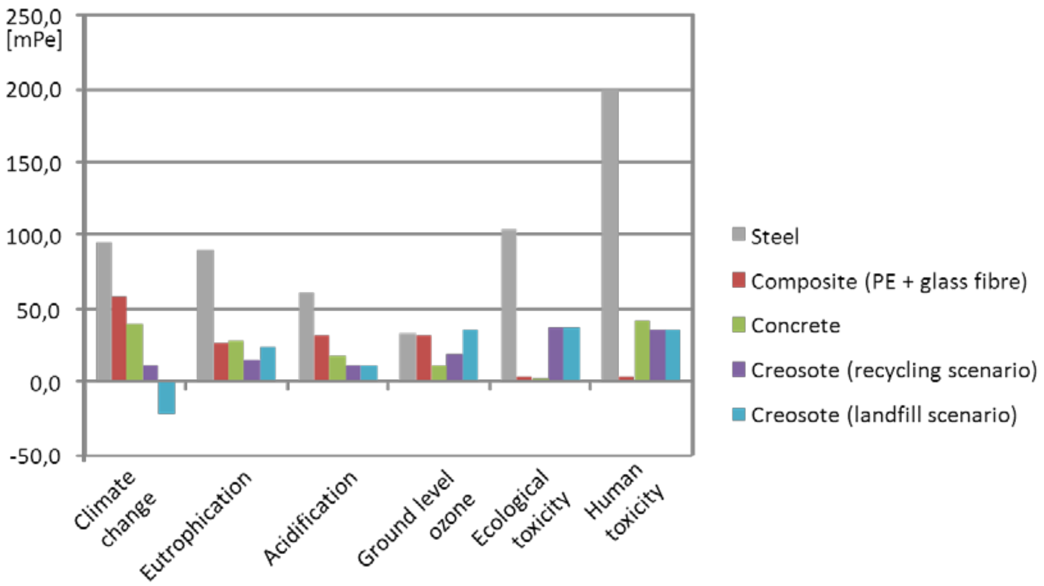

Figure 4 Environmental impact [mPe] for different pole alternatives per environmental impact category and assuming a service life of 50 years for all products.

It should be noted that the original data on steel production from the EcoInvent LCA database were changed in a way that reduces the overall impact to human toxicity from the steel pole from 445 mPe to 174 mPe. 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 only specify a total chromium emission, so the relevance of this assumption has not been validated. Thus there is no evidence to support this modification as reasonable, which should be taken into consideration when interpreting the results in Figure 4.

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 and Environmental Product Declaration (EPD).

Composite poles have generally similar environmental performance to concrete poles but concrete poles have greater impact on eutrophication and composite poles have greater impact on climate change. A very small proportion of the acetone and styrene emissions which arise during the manufacture of composite poles are released into the environment, and these emissions do not give rise to significant impacts on human toxicity in the LCA, see Figure 4. Emissions from the composite pole materials during the use phase have not been analysed and no literature could be found on this subject. If one assumes that the quality of the polyethylene and the pigment used in the composite poles is the same as those used in drinking water pipes, then it is reasonable to assume that any material emissions are negligible, and likewise their potential toxic effects. With regards to leaching from fibreglass, there is literature documenting leaching of different substances (including boron) from fibreglass tanks which are exposed to water at high temperatures. Jerol poles have a coating of polyethylene which protects them from weather exposure. Exposure to water in the part of the pole below ground should be negligible since the pole features a plastic plug at the bottom and an aluminium hat at the top. For these reasons the inventory does not contain any emissions from fibreglass.

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 37 mPe to human toxicity from the concrete pole. If the chromium emission had been 60 % hexavalent and the remaining trivalent (that is equal with the distribution of chrome in the solid cement before reduction), the contribution to human toxicity from the concrete pole would have been 218 mPe instead in the impact assessment. This fact should be borne in mind when analysing the result in Figure 4.

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 mPe 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). This type of oil is therefore assumed to be generally sold and used.

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, and for this reason we will use the typical value given by Rütgers in this LCA. 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 mPe as is illustrated in Figure 4.

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 (i.e. other fractions), which will remain the same irrespective of the content of low-boiling substances.

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.

For renewable natural resources we have assumed that growing trees absorb CO_2, which is normally emitted during energy recovery and therefore leads to zero emissions in total. Figure 4 also includes an alternative of landfilling the poles, something which is practiced in other countries. Since decomposition in a landfill is incomplete over a foreseeable timescale, not all bound carbon is released in the landfill alternative (a carbon sink is created), which leads to a negative climate change impact (formation of methane has been considered). This ”positive” effect takes place to the cost of resource use.

There is currently no generally accepted assessment method for handling resource efficiency or other energy sources. If a general resource efficiency index was established, it would then be enough if such resource efficiency index assesses the valuation of different energy sources, since one can argue that if energy is limitless then there is no resource shortage. IVL is working on developing such an energy resource index which will become available during 2011. Whilst waiting for such a method to become available the most common way of handling valuations of different energy sources in an LCA is to calculate accumulated energy use as a part of the inventory. This so-called primary energy use is often divided into renewable and non-renewable energy use (in an EPD it is also relevant to differentiate between the use of energy-bearing raw materials are used as fuel and when they are used as construction material i.e. *feedstock*). Table 2 shows the inventory results for primary energy use. If one compares the total energy use for creosote-impregnated poles which are recycled with those that are sent to landfill there is a relatively large energy use for the landfill alternative. This can be interpreted that there is bound energy in the pole which is lost when sent to landfill. In this way landfill is clearly a much worse alternative for dealing with the waste product.

Table 2 LCA result given category by category for the investigated pole alternatives complemented with human toxicity potentials calculated with USEtox and with life cycle inventory (LCI) result on energy ware consumption.

		Climate change	Eutro-phication	Acidi-fication	Photoch. ozone formation	Eco-toxicity	Human toxicity - USES LCA 1.0	Human toxicity - USEtox *	Energy ware consump., renewable	Energy ware consump., fossil
Type of result		LCIA [mPe]						LCIA [Pe]	LCI [MJ]	
Steel pole	Production	91	86	58	31	3,6	112	1 500	197	6992
	Service life	2,2	3,1	2,1	1,4	100	86,8	258 000	2	163
	End of life	1,2	1,6	1,1	0,7	<0,0	0,3	1,4	5	94
Composite pole	Production	57	19,5	31	31,2	3,2	3,9	33	133	4314
	Service life	0,4	0,5	0,3	0,2	<0,0	0,1	0,8	1,3	30
	End of life	0,2	0,2	0,1	0,1	<0,0	0,1	0,4	0,2	14
Concrete pole	Production	34	20	13	7,6	1,0	39,0	371	87	1457
	Service life	3,1	4,6	2,9	1,9	0,1	0,8	3,4	2	225
	End of life	0,8	1,3	0,8	0,6	<0,0	0,2	0,7	7	71
Creosote pole, recycling for energy recovery	Production	9,2	11,7	9,9	18	<0,0	2,5	7,1	2522	1743
	Service life	0,9	1,3	0,9	0,6	37	33	3,4	0,7	67
	End of life	0,8	1,4	0,9	0,7	<0,0	0,2	0,6	-2394	51
Creosote pole, landfill	Production	9,2	12	9,9	18	<0,0	2,5	7,1	2522	1743
	Service life	0,9	12	0,9	18	37	33	3,4	0,7	67
	End of life	-32	0,6	0,4	0,6	<0,0	0,1	0,5	0,3	32

**Values recalculated via 1,4-DCB eq.*

4.1 Sensitivity analysis regarding service life

These alternative calculations are based on the *assumption* that poles of steel, concrete and composite have longer service life than 50 years, which is the number set for creosote-impregnated poles based on historical *experience data* and which is also assumed to be a reasonable service time for assessing a transmission line within the study.

The sensitivity analysis is based on ***assigned*** service lives for steel, concrete and composite poles. The analysis also assumes that the technical service life of the product is used in full and the operational need of the transmission lines last as long. This is a condition which is most probably not true today nor in the future. On the other hand, it could be the case that the lines are moved, but the poles are taken with them. The sensitivity analysis therefore even describes this case where the poles are moved and fulfil their service life, which is assumed to be the ***assigned*** technical service life. This scenario assumes that dismantling and erection work is similar for the different alternatives (or small in relation to the total environmental impacts of the life cycle), which is a reasonable assumption.

The results of the sensitivity analysis show that emissions of metals from steel poles still dominates the total environmental impacts in the form of human and ecological toxicity. Since the contribution to ecotoxicity from the steel poles is dominated by emissions during use and decomposition of the galvanisation is assumed to be linear (constant per unit of time) over the service life, the comparable environmental impact is the same as in Figure 4. However, the human toxicity is lower in this sensitivity analysis in relative terms since only part of the contribution comes from leaching during the use phase, and the emissions from different stages of the manufacturing process give rise to lower contributions relative to the main scenario.

A comparison between the main scenario (Figure 4) and the sensitivity analysis does not identify any large relative changes that would result in a chance in 'ranking'. However, one can say that the results for the steel pole alternative gets closers to the concrete pole alternative, and that the steel pole's relative contribution to human toxicity, eutrophication and climate change thereby is lower. Similarly, once can say that the result of a comparison between the composite and creosote-impregnated poles is smaller if it can be shown that the assigned service lives are correct. In the sensitivity analysis climate change is the category that is most significant for the composite poles; human and ecotoxicity are most important for creosote-impregnated poles.

5 Conclusions and further work

This report describes how LCA can be used as a tool for decision support using a product LCA and a normalisation method which is based on what the natural environment can tolerate. A simplified interpretation of the results of this case study is that poles made of creosote impregnated wood are the most competitive alternative based on the environmental aspects covered in the LCA. Poles made of composite and concrete are equal next best performers. Two aspects that are not addressed by LCA are notable in this case study: work environment and resource efficiency. A simplified indicator for resource efficiency used in this study is primary energy, divided into renewable and non-renewable energy resources. In total, composite poles are the alternative in the study that has the lowest contribution to human and ecotoxicity. Steel poles are the alternative which has the highest contribution to human and ecotoxicity.

Furthermore, the following conclusions can be drawn:

- LCA can be used to assess a non-toxic environment in a life cycle perspective, i.e. for risk minimisation in a life cycle perspective, and is therefore a complement to a risk assessment.

- The method used to assess human en ecotoxicity is USES LCA 1.0 and application in the case study shows that the contributions from steel poles are the most significant environmental aspect for all material alternatives. If newer methods, such as USES LCA 2.0 or USEtox were used then this picture would be further strengthened. These methods are, however, not fully applicable for metals and persistent substances and for this reason USES LCA 1.0 was used instead.

- LCA give indications of important areas for environmental improvements for all material choices.

An elaboration to this LCA is to analyse the new or revived alternatives that are now on the market, such as tube poles of plywood (which does not have any impregnation apart from melamine based glue) or glulam (Comwood) (which can be impregnated with an alternative preservative). The possibility of chemically modified wood or development of new preservatives can lead to improved environmental performance for wooden poles. Furthermore, the following method developments would contribute to better data for decision support:

- Further development of normalisation methods to assess resource efficiency and other general updates (including climate change amongst others).

- There is a need to analyse why new assessment methods for toxicity give ”unrealistic results” and how this situation can be improved, or whether alternative ways of handling these aspects must be developed.

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