Published in International Journal of Life Cycle Assessment
http://dx.doi.org/10.1007/s11367-012-0397-8
Article in press

A review of life cycle assessments on wind energy systems

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Abstract

Purpose

Several life cycle assessments (LCA) of wind energy published in recent years are reviewed to identify methodological differences and underlying assumptions.

Methods

A full comparative analysis of 12 studies were undertaken (10 peer-reviewed papers, 1 conference paper, 1 industry report) regarding six fundamental factors (methods used, energy use accounting, quantification of energy production, energy performance and primary energy, natural resources, and recycling). Each factor is discussed in detail to highlight strengths and shortcomings of various approaches.

Results

Several potential issues are found concerning the way LCA methods are used for assessing energy performance and environmental impact of wind energy, as well as dealing with natural resource use and depletion. The potential to evaluate natural resource use and depletion impacts from wind energy appears to be poorly exploited or elaborated on in the reviewed studies. Estimations of energy performance and environmental impacts are critically analyzed and found to differ significantly.

Conclusions and recommendations

A continued discussion and development of LCA methodology for wind energy and other energy resources are encouraged. Efforts should be made to standardize methods and calculations. Inconsistent use of terminology and concepts among the analyzed studies are found and should be remedied. Different methods are generally used and the results are presented in diverse ways, making it difficult to compare studies with each other, but also with other renewable energy sources.

Keywords: life cycle assessment, wind energy, wind power, natural resource use, primary energy conversion, energy accounting

1. Introduction

At the end of 2009, wind energy generated about 1.8% of the global demand for electricity with almost 160 GW of installed capacity (IPCC, 2011). This equals about 0.2% of the total global primary energy demand, or 273 TWh (IPCC, 2011). Wind energy capacity is currently growing at a high rate, with global installed capacity roughly doubling every three years, and growth rates seems to be accelerating (WWEA, 2010). This makes wind energy an important renewable energy option for the future. The IPCC even estimates that wind energy could potentially provide over 20% of the global electricity demand by 2050 (IPCC, 2011), while other more moderate estimations foresee a smaller contribution (Höök et al., 2012).

Energy systems based on wind, as well as other renewable energy sources, are often automatically assumed to be sustainable and environmental-friendly sources of energy in much of the mainstream debate. However, all systems for converting energy into usable forms have various environmental impacts, not to mention a requirement of natural resources. It is essential to have consistent evaluation methods for analyzing all aspects of a given energy source. Without such methods, it is difficult to compare them and make the right decisions when planning and investing in energy systems for the future.

Good decisions on how to construct future energy systems relies on having access to suitable feasibility indicators for different options so one can pinpoint the most optimal pathway. Academics, practitioners, and policy makers continue to debate the benefits and costs of alternative energy sources, primarily the renewable ones. The economics of wind energy have been thoroughly covered by previous studies such as Blanco (2009) and Valenzuela and Wang (2011). However, Welch and Venkateswaran (2009) found that decisions that incorporate both environment concern and investors' desire for shareholder value maximization are more likely to be truly sustainable.

Future growth of any new energy systems, in this case wind power, will require energy, as well as other resources during the expansion phase, and these implications need to be considered when planning future developments. A need for meticulous environmental impact assessments and energy performance evaluations can be seen here. Environmental concerns are perhaps more difficult to quantify than economic issues and many attempts have been made based on different approaches.

1.1 Scope of this study

A popular way of measuring the environmental impact and energy performance of wind energy is *life cycle assessments* (LCA). Many analysts have tried to handle these problems for wind energy based on various approaches. This study focuses on how LCA methodologies are used for evaluating environmental impact energy performance of wind energy systems by reviewing existing publications, discussing differences in methodology and results.

Several recently published LCA studies for wind energy are reviewed (Table 1). Some has been published as scientific articles (Schleisner, 2000; White, 2006; Ardente et al., 2008; Lee and Tzeng, 2008; Crawford, 2009; Martinez et al., 2009a; 2009b; Tremeac and Menuier, 2009; Weinzettel et al., 2009; Guezuraga 2012), while others are conference papers (Lee et al., 2006) or reports released by wind turbine producers (Vestas, 2011). Not all assessments claim to follow the ISO-standards, and some are more of energy and CO_2 assessments than full LCAs. Vestas (2011) has been reviewed by an external reviewer that states that the study is carried in compliance with the currently best LCA practices and in accordance with ISO-standards. The authors would like to highlight several potential issues and call for a methodological discussion concerning how LCA is used for these purposes.

Table 1. *Rated power and general turbine types in the reviewed studies.*

Study	**Type of turbine assessed**
Ardente et al. (2008)	Wind farm, 0.66 MW turbines
Crawford (2009)	0.85 MW turbine and 2 MW turbine
Guezuraga et al. (2012)	1.8 MW gearless turbine, 2.0 MW geared turbine
Lee et al. (2006), Lee and Tzeng (2008)	Three wind farms in Taiwan with 0.66 MW, 0.6 MW and 1.75 MW turbines
Martinez et al. (2009a, 2009b)	2 MW turbine
Schleisner (2000)	Wind farms onshore and offshore with 0.5 MW turbines
Tremeac and Menuier (2009)	4.5 MW turbine and 0.25 MW turbine
Vestas (2011)	3 MW turbines
Weinzettel et al. (2009)	Floating turbine, 5 MW
White (2006)	Three wind farms in the US with 0.345 MW, 0.75 MW and 0.60 MW turbines

2. Methodological differences

The idea of measuring the energy performance of products evolved in the 1970s. The *energy analysis* method was stated at a conference held by the International Federation of Institutes for Advanced Studies in 1974 (Mortimer, 1991). Life cycle assessment (LCA) has many similarities to energy analysis, but is not restricted to just energy. LCA started to evolve in the mid-1980s and became increasingly common during the 1990s when scientific publications started to reach wider audiences. As LCA methodology evolved, many different methods and guidelines emerged. Recent development is well described by Finnveden et al. (2009) or Guinée et al. (2011).

An LCA can be defined as "*the compilation and evaluation of the inputs, outputs and potential environmental impact of a product system throughout the life cycle*" (Guinée, 2001). Life cycle assessments generally follow the same four basic steps: *goals and scope, life cycle inventory, impact assessment* and *interpretation*.

The *goals and scope* phase determines the method, the functional unit, and system boundaries as well as studied environmental impacts and level of detail. Choices made here about methodology, system boundaries, cut-off limits or functional units can have large impact on the final results (Baumann and Tillmann, 2004). This is seen clearly in the reviewed studies.

In the *life cycle inventory* (LCI), inputs and outputs throughout the entire life cycle are estimated, according to the chosen system boundaries and methods. There are several different ways to do this and choices in methodology can have a large impact on final results. Ekvall and Weidema (2004) describe a way to categorize two main methods, namely *attributional* and *consequential* LCI. An attributional LCI describes the physical flows relevant to environmental impact in and out of the life cycle system boundaries, while consequential LCI attempts to generate information about consequences of actions made by describing how the physical flows relevant to environmental impact will change with certain changes in the life cycle. It should be mentioned that there is not always a clear practical distinction between attributional and consequential LCI.

The next step is the *life cycle impact assessment* (LCIA), where results from the inventory are converted into environmentally relevant information (Baumann and Tillmann, 2004). Martinez et al. (2010) compares no less than seven different methods to perform a LCIA. Sometimes an attempt is made to express the impact on a common scale through weighting or further evaluate the results LCIA. This can never be based solely on purely objective factors, as subjective values always must be introduced (Baumann and Tillmann, 2004). As a consequence, LCA is not necessarily a method that meets the standards of strict natural science. This should hopefully be handled in a good way in the *interpretation* of the results.

2.1. Discrepancies in the LCI

While none of the studies can regarded as using consequential LCI methodology, there are two other main approaches that can be used for the LCI: *process chain analysis* (PCA) and *input-output* (I/O) analysis. I/O analysis uses economic data to estimate the resource use in different parts of the economy, while PCA estimates the actual physical flows of mass and energy. In I/O-based LCA, inter-industrial relationships are quantified in an I/O matrix, representing interactions between different sectors of the economy. All of the reviewed studies primarily use attributional approaches based on PCA, but two studies also partly use I/O analysis.

Hendrickson et al. (1997) compared PCA and I/O methods and finds no major differences in results between them. However, Crawford (2009) states that a PCA is generally seen as more accurate and relevant than an I/O analysis, but that it is also accused of missing a great deal of important information that can be accounted for with I/O analysis. Crawford (2009) quantifies this by claiming that the system boundaries for a PCA can be up to 87% incomplete. Both Hendrickson (1997) and Crawford (2009) propose the use of hybrid methods, capable of fusing the advantages of both techniques. However, only two of the reviewed studies (White, 2006; Crawford, 2009) relied on hybridized methods. It appears to be more far common to use PCA.

White (2006) uses I/O analysis for some parts of the lifecycle otherwise lacking data, while Crawford (2009) uses a hybrid method of PCA and I/O. For one of the three wind farms assessed in White (2006), I/O is used for assessing energy use of the operations and maintenance phase and concludes that this phase of the life cycle contributes with about 34% of the total energy use of that wind farm. For the other two farms assessed in White (2006), the operations and maintenance phase contributes with 16% and 11% of the energy use. So, it appears that when using I/O instead of PCA for the operations and maintenance phase, the energy use of that part of the LCA becomes many times greater, even within the same study. Other studies, which do not use I/O methods, appear to present much lower relative numbers for operations and maintenance. For example, Ardente et al. (2008) states that energy usage for operations and maintenance is only 6.5% of the total energy use.

Crawford (2009) uses a hybrid method of PCA and I/O where only 22% of the embodied energy comes from process data, while 78% comes from I/O data. He draws the conclusion that other "*embodied energy analysis methods*" of wind energy can be up to 78% incomplete. This tendency to primarily rely on PCA might very well lead to underestimated figures for energy use, but it is also possible that I/O analysis exaggerates energy use. Further studies on this issue are both required and should be encouraged.

2.2 Impact assessment method dependency

All reviewed studies estimate energy performance, although in slightly different ways. For other environmental impact, some studies present only greenhouse gas emissions (Lee et. al., 2006; Lee and Tzeng, 2008; White, 2006; Crawford, 2009; Guezuraga, 2012). Others use different impact assessments to express environmental impacts, in vastly different ways. Ardente et al. (2008) describe emissions of seven different air emissions, seven water emissions and three kinds of wastes. Schleisner (2000) presents the environmental impact as seven different emissions. Weinzettel et al. (2009) uses the *CML 2 baseline 2000 V2.03*-method and presents the environmental impact in 8 different impact categories. Vestas (2011) claims to use *CML 2009 update* and presents environmental impact in 15 different impact categories. Tremeac and Menuier (2009) uses a method called *Impact 2002+* that links 14 midpoint categories from the LCA to four damage categories: climate change, resources, ecosystem quality and human health.

As an example of methodology dependence, Martinez et al. (2009a) and Martinez et al. (2009b) are done by the same authors, on the same turbine model, at the same wind farm with the same assumed production, but using different impact methodologies giving significantly different results. Martinez et al. (2009a) uses the method *Eco-Indicator 99* using 11 different impact factors. Martinez et al. (2009b) uses *CML-methodology,* presenting environmental impact in 10 different impact categories presented as equivalents of different emissions, as well as cumulative energy demand.

The environmental impact results of these two studies of are difficult to compare, especially since they are presented in different units. However, the most interesting thing is perhaps that the resulting energy performances differ significantly. Martinez et al. (2009a) presents an *energy payback ratio* (EPBT) of 0.4 years, while Martinez et al. (2009b) presents an EPBT of 0.58 years. Basically, it appears that the same people assessing the same thing but using a different methodology gives a 45 % longer EPBT.

It could be added that LCA methodology during the course of these studies has been, and are still evolving. At the same time vastly different methodology are being used today. Clearly, using different methodologies can cause widely different results as illustrated by the Martinez studies. The different methodologies used make it difficult to compare assessments to each other and creates questions if the results using different methodology should be compared at all.

3. Energy-relates issues

In the reviewed studies, there are several issues related to energy. Some include energy use accounting, while others relate to quantification of energy production and the actual presentation of the energy performance.

3.2. Energy use accounting

Some energy carriers, especially electricity, can be valued differently. Electricity may be seen as being produced by other energy carriers, according to an electricity generation mix, and converted into primary energy using thermal equivalents. How and if the electricity should be converted into primary energy is often not obvious, and depending on the electricity generation mix and the heating values used for the calculations, the results can be significantly different.

In most of the reviewed studies, it is difficult to see which different energy resources are used, especially for electricity. In fact, none of the reviewed studies, except Weinzettel et al. (2009), says anything about how much electrical energy compared to direct thermal energy have been used during the life cycle. The electricity is converted into primary energy and the total primary energy use is presented either as one fixed number or divided into various primary energy carriers.

Energy use is frequently only described as one single number in primary energy terms, which summarize all different energy carriers involved, while other studies express the energy use somewhat divided into different fuels (Table 2). Some specify the electricity generation mix that is used for the conversions (Table 2), but what the electricity generation mixes look like and how the energy use is divided into energy carriers is often explained poorly, or not at all. The five studies that specify energy carriers present totally different ones, making comparisons quite difficult.

Table 2. *Electricity generation mixes used for energy inputs in the reviewed studies as well as presented energy carriers.*

Study	**Electricity generation mix**	**Presented energy carriers**
Ardente et al. (2008)	Average European ? *	Not specified
Crawford (2009)	Not specified	Not specified
Guezuraga et al. (2012)	National (Germany, Denmark, China)	Nuclear, brown coal, natural gas, crude oil, black coal, residual biomass, hydro, wind, waste
Lee et al. (2006), Lee and Tzeng (2008)	Not specified	Not specified
Martinez et al. (2009a, 2009b)	Not specified (a) National (Spain) (b)	Not specified (a) Fossil, nuclear, hydro, biomass and others (b)
Schleisner (2000)	National (Denmark for all origins)	Coke, coal, oil, natural gas
Tremeac and Menuier (2009)	Not specified (4.5 MW) National (Finland) (250 W)	Not specified
Vestas (2011)	National (country specific for manufacturing countries)	Crude Oil, hard coal, lignite, natural gas, renewable fuels, wood
Weinzettel et al. (2009)	Not specified	Electricity, oil, diesel and electricity from oil
White (2006)	Not specified	Not specified

** The data for energy and environmental impacts from manufacturing of turbines and towers is claimed to refer to "average European data" without further specifications.*

How the electricity should be converted into primary energy is not obvious and results can be significantly different depending on the electricity generation mix used. Guezuraga et al. (2012) presents different cases with German, Danish and Chinese electrical generation mixes, with the resulting EPBT ranging from 1.15 with German electricity to 2.36 with Chinese electricity, indicating that the choice of electrical generation mix could have a large impact on the resulting energy performance.

The studies that mention electricity mix all use national generation mixes, except for Ardente et al. (2008) that claims to use average European data for energy use for manufacturing of turbines and towers since none of the components are manufactured in Italy. Schleisner (2000) assumes the energy supply system to follow Danish conditions despite the origin of inputs, however, it is pointed out that the aluminum is in fact produced in Norway where the electricity is mainly produced by hydropower. How the energy is divided into fuels, how much of the energy use have been electricity or what "*Danish conditions*" look like are not explained. Vestas (2011) on the other hand claims that "country-specific energy mixes" are used for processes in different countries. So, appears like Ardente et al. (2008) use average European generation mix for processes in other countries, Schleisner (2000) use Danish conditions for all countries and Vestas (2011) uses country specific generation mixes processes in different countries.

The European electrical grid is largely interconnected and the use of national electricity generation mixes is not necessarily the most accurate way to convert the electricity into primary energy, as electrons pay little care to national borders. Two potentially plausible alternatives are to use an average European mix or rely on marginal electricity, often originating from coal. Perhaps national electricity generation mixes are in fact preferable, but in any case, a more transparent and consistent treatment of electrical energy and other used energy carriers would greatly reduce the opacity in LCA methodology. Currently, the energy use and the corresponding LCA results can be significantly affected by the analyst's personal choices for electricity mix and energy conversions, unfortunately often under a lack of transparency. Standardization and improved transparency would likely reduce the spread in results and make LCA studies appear less inconsistent to planners and policy makers.

3.3. Quantification of energy production

LCA results are generally presented as an environmental impact expressed per kWh of produced electricity. Energy performance is also highly dependent on the energy production. This makes the estimated energy production have a huge impact on final results, since everything is expressed in relation to the produced electricity. Basically, a doubling of the production of electrical energy cuts an apparent emission in half and makes the energy performance twice as good.

The amount of electricity that will be produced depends on many factors such as wind class, wind speed, rotor size, turbine power rating and distance to grid. Naturally the actual wind resource at a specific site is a very important factor. Different types of sites can be divided into different wind classes based on average annual wind speed, extreme wind gusts and turbulence (Vestas, 2011). Modern wind turbines are in fact designed for certain wind classes, and Vestas (2011) suggests that comparisons should only be made between wind turbines within a specific wind class.

In this paper, turbines of different sizes and types that are evaluated in different ways are compared to each other. What we wish to do is not to compare different turbines to each other, but rather compare different methods to assess environmental impact and energy performance of wind turbines. The methods used to estimate the annual production of electricity of a wind farm varies significantly between the different studies. To be able to compare estimated production, the conceptual capacity factor is used, which is the ratio between the energy actually produced and the energy possible to produce if running at rated power all the time. Consequently, comparisons are far from trivial and the simple analysis done here does not go into any detail but rather attempts to pinpoint some underlying differences among the reviewed studies.

Lenzen and Munksgaard (2002) reviewed many energy and CO_2 assessments performed since the 1970s, and found capacity factors ranging from 0.08 to 0.50. Modern wind turbines typically lie between 0.20–0.35. In the reviewed assessments, the capacity factors ranges between 0.19 and 0.53 (Table 3). Not all the reviewed studies express a capacity factor, so for some of them a capacity factor have been calculated from given production or assumed full load hours.

Some of the reviewed studies use measured data from actual wind farms while others use more theoretical quantifications of the electricity production (Table 3). For example, Garrett (2011) states that Vestas (2011) use sophisticated real-time diagnostic tools and sensors, operating on around 20 % of global installed wind capacity, which measure individual turbine performance, power output and health status (such as fatigue loading and turbine condition), as a base for quantification of production of electricity for the turbine operating in a certain wind class. In contrast, other assessments have more or less well-motivated assumptions for electricity production.

Table 3. *Capacity factors used in the reviewed studies and method for quantification of production.*

Study	**Capacity factor**	**Method for quantification of production**
Ardente et al. (2008)	0.19 – 0.30	Low estimate: measured operation data of actual wind farm High estimate: design estimation
Crawford (2009)	0.34 (850 kW) 0.33 (3.0 MW)	Estimated from wind data of typical site and characteristic power curve of turbines
Guezuraga et al. (2012)	0.34 (geared) 0.21 (gearless)	Assumed figure from manufacturer (geared) Measured operation data (gearless)
Lee et al. (2006), Lee and Tzeng (2008)	18.9, 30.9, 42.6 for three different wind farms	Measured operation data of actual wind farms
Martinez et al. (2009a, 2009b)	0.23 *	Assumed as equivalent full-load hours
Schleisner (2000)	0.29 (offshore) ** 0.25 (onshore) **	Unspecified
Tremeac and Menuier (2009)	0.30, 0,20***	Assumed
Vestas (2011)	0.43 **	Derived from Vestas statistical database
Weinzettel et al. (2009)	0.53	Assumed
White (2006)	0.26, 0.29, 0.20	Measured operation data of wind farms

** Calculated from given full load hours per year.*
*** Calculated from estimated production.*
**** Based on a 50 W average power of 250 W installed.*

White (2006) and Ardente et al (2008) assess actual existing wind farms and use actual measured production data of the specific location and turbine type. In both cases, actual production data is significantly lower than expected. White (2006) compares projected capacity factors for three wind farms with measurements, which were significantly lower. The projected capacity factors were 0.33, 0.35 and 0.31, while the actual capacity factors were 0.26, 0.29 and 0.20 respectively. Ardente et al. (2008) finds the actual capacity factor measured over a year to be 0.19 compared to the design capacity factor of 0.30. Guezuraga et al. (2012) compares a 2 MW geared turbine with a 1.8 MW gearless turbine. For the geared turbine data provided from the manufacturer of a typical good wind site location is used, giving a capacity factor of 0.34. For the gearless turbine, actual operating data from a turbine measured at a site over 9 years is used, resulting in a capacity factor of 0.21. It is possible that the two different turbines would get somewhat different production if located at the same wind conditions, but comparing two turbines of different types operating under totally different conditions becomes somewhat problematic. This could indicate a tendency to exaggerate expected production rates of wind farms and highlight issues with using expected production numbers for quantification of production.

Communicating emissions and energy use as a fixed value per kWh of produced electricity can itself be problematic. This primarily stems from the vast differences seen in the result depending on the production and the fact that it is not a fixed feature of a certain type of turbine. If measured numbers of a specific wind farm are used, the result can potentially be expressed as a fixed feature for that specific wind farm. However, this does not mean that the specific turbine type will produce the same amount of electricity in other locations. White (2006) states that no two wind farms with the same technology can be expected to have the same energy payback ratio at different locations. Vestas (2011) propose that comparisons should be made between turbines operating in the same wind class. Production should perhaps be quantified not as a fixed production rate, but rather as high and low estimates, giving different scenarios.

3.4. Energy performance and primary energy

The energy performance is usually expressed as *energy return on investment* (EROI), defined as cumulative electricity generated divided by cumulative primary energy required, or *energy payback time* (EPBT), defined as the amount of time it takes to “pay back” the energy used over the life cycle. EROI and EPBT can be good indicators on whether a wind turbine actually produces more energy than is consumed during its life cycle. However, some studies also include a conversion of the produced electrical energy to primary energy. Table 4 contains the energy performance results from the reviewed studies.

A few examples are worth noticing. Weinzettel (2009) states that the EPBT is 13 months, but also converts the electrical energy with an assumed thermal efficiency of 0.40 and gets a *primary energy payback time* (PEPBT) of 5.2 months. Schleisner (2000) also converts the produced electricical energy to primary energy with an efficiency of 0.4 but calls the resulting 4.7 months for an offshore wind farm and 3.1 months for an onshore wind farm, *energy payback time*, instead of *primary energy payback time.* It appears that the EPBT presented is in fact PEPBT, but this is not explained. In contrast, a more holistic presentation of both an “*ordinary*” payback time and several PEPBTs for different electricity generation mixes were presented by Vestas (2011).

When Schleisner (2000) converts electricity production to primary energy but still confusingly calls this the time before the energy is “*paid back”* (i.e. EPBT), instead of PEPBT, this causes great problems with comparing the energy performances from different studies. Weinzettel et al. (2009) denotes payback time after conversion *final payback time*, and compares it to a Vestas study where the EPBT is 6 to 7 months. However, the Vestas reference used in Weinzettel et al. (2009), presents the electricity produced as a direct equivalents, thus two completely different things are compared to each other. Another example is Lee et al. (2006) and Lee and Tzeng (2008) who found and EPBT of 1.3 months, which compared to a few other assessments with EPBTs ranging from 3 to 8 months. It seems like Lee et al. (2006) use direct energy payback time without any conversion, but still receives far better results than for instance Schleisner (2000) that used a primary energy conversion through thermal equivalents, comparing different concepts. This also makes the extremely short EPBT result in Lee et al. (2006) even more odd, since the return ratio gets many times greater after conversion.

There is no consensus on how conversion to primary energy should be done, or even if and why it should be done. Schleisner (2000) converts the produced electricity to the primary energy that would be produced in a conventional power plant with an efficiency of 40%. Tremeac and Meunier (2009) use the efficiency of the French national electric network of 0.35.

Table 4. *Energy return on investment (EROI) or primary energy return on investment (PEROI) and energy payback time (EPBT) or primary energy payback time (PEPBT) of the reviewed studies. When only payback time is given, return on investment is calculated and vice versa. When payback time is given in years, it is converted to months for easy comparisons.*

Study	EROI	PEROI	EPBT	PEPBT
Ardente et al. (2008)	-	40-80	-	6–12 months
Crawford (2009)	-	21 (850 kW) ** 23 (3.0 MW) **	-	11.4 months 10.4 months
Guezuraga et al. (2012)	8.7–33.3	-	7.2–27.6 months	-
Lee et al. (2006), Lee and Tzeng (2008)	185	-	1.3 months	-
Martinez et al. (2009a, 2009b)	50 (a) 34.4 (b)	-	4.8 months (a) 7.0 months (b)	-
Schleisner (2000)	-	51.3 * (onshore) 69.0 * (offshore)	-	4,7 months (offshore) * 3.5 months (onshore)*
Tremeac and Menuier (2009)	11.8 (4.5 MW) 3.1 (250 W)	34.5 (4.5 MW) 8.7 (250 W)	20.4 months (4.5 MW) 78 months (250 W)	7.0 months (4.5 MW) 27.5 months (250 W)
Vestas (2011)	30	68.6–109.1	8 months	2.2–3.5 months
Weinzettel et al. (2009)	18.5	46.2	13 months	5.2 months
White (2006)	11, 24, 28	-	21.8, 10.2, 8.6 months	-

** Expressed as energy payback time, but based on the assumption that the electrical energy produced replaces a conventional power plant with an efficiency of 0.40.*
*** Expressed as energy yield ratio (EYR), but the energy output is earlier described as being converted to primary energy using the factor 0.34.*

Tremeac and Meunier (2009) argue that the PEPBT-method should be used since it is more consistent and claims that different energy forms are compared otherwise. However, this view is not obvious. Wind turbines can only produce electricity and cannot easily replace other energy forms. The primary energy used for producing the replaced electricity can therefore be seen as irrelevant, making the entire focus on primary energy unjustifiable. Also, the direct impact of the produced electricity is difficult to assess (IPCC, 2011). In the short run, produced electricity will probably replace some kind of fossil fuel power plant on the margin. In the long run, introduction of wind energy has wider implications on the electricity grid and may even necessitate expansion of other energy sources (IPCC, 2011). The impact of wind energy on the electrical energy system is complicated and system specific, thus difficult to forecast with any precision (IPCC, 2011). It is also questionable if we now can say what the energy system will look like in the future. Who can accurately say what the electrical energy generation mix will look like in 20 years from now, if we compare electrical energy to the efficiency of today's electrical system?

LCA methods cannot evaluate which other production facility, if any, will reduce production and environmental benefits such production reductions are also difficult to approximate. LCA is simply not a methodology that contains the necessary tools for estimating changes to the electric system layout as a result from increased electricity production from wind power. This can be seen in the variation in factors used for converting the electrical energy to primary energy. A better consensus on how to present produced electricity and how to estimate energy performance of wind power should definitively be aimed for.

If the energy payback time or energy return on investment is still presented in primary energy terms, this must be clearly explained, which is only done in some of the reviewed studies. Also, similar terminology should be used to better portray the differences between primary energy and direct energy performances. The existing confusion regarding terminology, seen in several of the reviewed papers, is actually misleading or even harmful since it adds more confusion than applicability for outsiders trying to use the LCA results for planning or investments. By introducing standardized concepts, studies would be significantly easier to compare. We propose that both "*ordinary*" EPBT or EROI and their primary-energy-based versions should be presented and clearly defined in any LCA that aims to be holistic.

4. Natural resources

Building a wind turbine creates a demand for natural resources, both energy resources and material resources. Stewart and Weidema (2005) state that natural resource use has been included as an important impact category in most LCA impact methods. However, methods to quantify resource depletion within LCA methodology have been under debate and no consensus seems to exist on which one to use. Finnveden (2005) pinpoints the lack of a generally accepted framework and highlights that resource use is important to the overall results for many LCA weighting techniques.

4.1 Natural resource inputs

Most of the reviewed studies express the inputs of material resources as amount of refined resources that are used for building the wind turbine. It could be interesting to discuss what should actually be considered an input. For iron, is the input iron ore or refined forms of iron (cast iron, steel bars, specialized iron-alloys, etc)? The only assessment that mentions amounts of raw material used, instead of refined ones, is Vestas (2011) where masses of unrefined material resources from the LCI are presented in a table. This is an interesting attempt, but could be somewhat problematic since it the quantities can be very different depending on, for instance, assumed ore grades. What is even more problematic with the Vestas (2011) LCI table is that recycling credits is included in the table, and some resources even have negative values of usage for different phases of the life cycle. Although this is somewhat intuitively strange, this can partially be explained with that a substitution approach is used instead of allocation of different co-products (Gbegbaje-Das, 2011). When a metal is mined, the co-products are also assumed to be used, thus replacing other production. For some resources, the end-of-life phase is larger than the other phases, making the total resource use negative. As an example, if the two different iron ore grades presented in Table 5 are combined to a total amount of iron ore it adds up to a total of $-2.61*10^{-4}$ kg iron ore per kWh of produced electricity. This could be interpreted as if you would actually gain 0.26 grams of iron ore for every kWh of electricity produced by the wind turbine, which from a physical perspective on reality is clearly nonsensical. In personal correspondence with Vestas about this issue, representatives claimed that there has been an accounting error related to mass-balancing in Vestas (2011) and that it will be corrected in an updated version of the 2011-report (Garrett, 2011).

Table 5. *Iron ore use in different stages of life cycle of Vestas V112 wind turbine Adapted from Vestas (2011).*

Material resources kg / kWh produced	Total	Wind-turbine	Foundations	Transformer	Wind plant set up	Site operations	End of life
Iron ore (56.86%)	-2.68E-04	1.65E-03	1.53E-04	8.06E-06	3.54E-06	1.25E-04	-2.21E-03
Iron ore (65%)	7.32E-06	-9.39E-07	5.73E-06	-8.97E-10	1.10E-09	-6.28E-08	2.59E-06

Vestas (2011) should get some credit for the effort of trying to present the amount of natural resources used for building a turbine, but an issue with the numbers is that they obviously do not represent the actual amounts of resources used without subtracting substitution or recycling credits in the numbers. This makes it impossible to use the numbers for assessing the amounts of resources used for building a turbine without using these factors. Presenting negative resource use numbers as "*LCI data*" is highly questionable and raises questions about what the numbers are actually used for in the LCA. If negative numbers of resource use have been used for assessing environmental impacts, the validity of the study could be questioned.

4.2 Resource depletion and input bottlenecks

Many of the studies have resource use or depletion included in the LCIA as different impact factors. Tremeac and Menuier (2009) use an impact assessment method called *Impact 2002+* where one of the four damage categories is called natural resource depletion. The resource depletion is expressed in GJ primary non-renewable energy and does not seem to take account for non-energy resources. Martinez (2009b) and Vestas (2011) has an impact category called "*abiotic depletion*" that expresses an abiotic depletion factor for extraction of minerals and fossil fuels in antimony equivalents per kg extraction, based on concentration of reserves and rate of de-accumulation. Vestas (2011) also includes several different types of resource consumption in the impact categories: *Abiotic resource depletion (ADP elements)*, *Abiotic depletion (ADP fossils)* and *water consumption*. There are also impact factors called *recyclability*, *primary energy from renewable raw materials (net calorific value)* and *primary energy from resources (net calorific value)*. ADP fossils is claimed to describe the amount of non-energetic resources that are directly withdrawn from the geosphere to reflect the scarcity of the materials, expressed in antimony equivalents. In Vestas (2011), the use of one single ore, copper-molybdenum-gold-silver ore, accounts for about 75% of the total of this impact category, which is a result worth noticing. An important thing to understand is that many different resources is included and presented in many different units between different studies.

If the use of natural resources is an important aspect of building renewable energy converters, like wind turbines, this should be reflected upon. Klejn and van der Voet (2010) concludes that a growth of wind energy to contribute with 15% of the global primary energy use by 2050 would necessitate substantial amounts of resources, requiring global production of copper, iron ore and cement to increase dramatically. Jacobson and Delucchi (2011) claims that such bulk materials are not likely to be of any immediate concern for a fast expansion of wind energy, but rare earth elements (REEs) such as neodymium (Nd) used for permanent magnets in certain generator designs are likely more problematic materials for future development.

Although REEs are fairly commonly occurring globally, they could still become a limiting factor for a fast growth of wind energy globally (Figure 1). The global REE market is currently totally dominated by China with its 95% share (Chen, 2011), and this dependence might be seen as questionable from an energy security perspective. That standpoint got reinforced when the Chinese government chose to restrict exports, aimed at protecting domestic supply (Tse, 2011). Recent studies have highlighted rare earth elements as critical raw materials for clean energy and found its availability to be associated with high risks in the foreseeable future (Haxel et al., 2002; Long et al., 2010, British Geological Survey, 2010; US Department of Energy, 2010; Moss et al., 2011). Kanawaza and Kamitani (2006) also highlighted how REEs nearly always are associated with radioactive heavy metals, resulting in environmental problems. In fact, one of the key reasons for dismantling of REE mining in the USA was associated radioactive releases (Haxel et al., 2002; Castor, 2008). Recently, the portfolio managers of Morgan & Spitz even decided to exclude neodymium-based direct drive turbines from their environmental fund motivated by severe environmental impacts generated by mining (Renewables International, 2011).

None of the reviewed assessments specifically mentions any use of neodymium, but according to Biggs (2011), the Vestas V112 turbine that is assessed in Vestas (2011) does contain neodymium, which is confirmed by Garrett (2011). No quantities of neodymium are mentioned, but the LCI table of Vestas (2011) contains a post called Rare-earth ore and there is materials listed in the refined materials inventory as *special metals and magnet*, which could include REEs like neodymium. It should also be mentioned that rare earth metals, like neodymium, in the V112–3.0 MW tower magnets results in a saving of around 10 tons of steel per turbine (Garrett, 2011).

None of the other reviewed studies mentions neodymium or any other special metals. For analysts with interest in such questions, few or none of the reviewed LCAs are especially helpful. One reason for this could be that the use of non-energy resources does not seem to be of any significant concern in the reviewed studies. Another reason could be that many studies uses different kinds of cut off criterion and cut off limits, possibly excluding important materials, making up small parts of the total mass. Martinez et al. (2009a, b) claims to take account for elements that together make up more than 95% of the foundations, 95% of the tower and 85% of the nacelle and rotors. Ardente et al. (2008) states that any process of activity that contributes to less than 1% of the total environmental impact of one impact category can be neglected. In general, steel and concrete often make up a total of 95% of the weight of the wind farm. With cut-off limits at this level or lower, certain critical material accounting for only a small part of the mass could potentially be missed in an LCA. It is possible that the other turbines assessed do not contain REEs, like neodymium, but for instance Guezuraga et al. (2012) assesses a direct driven turbine, which almost certainly contains large amounts of REEs.

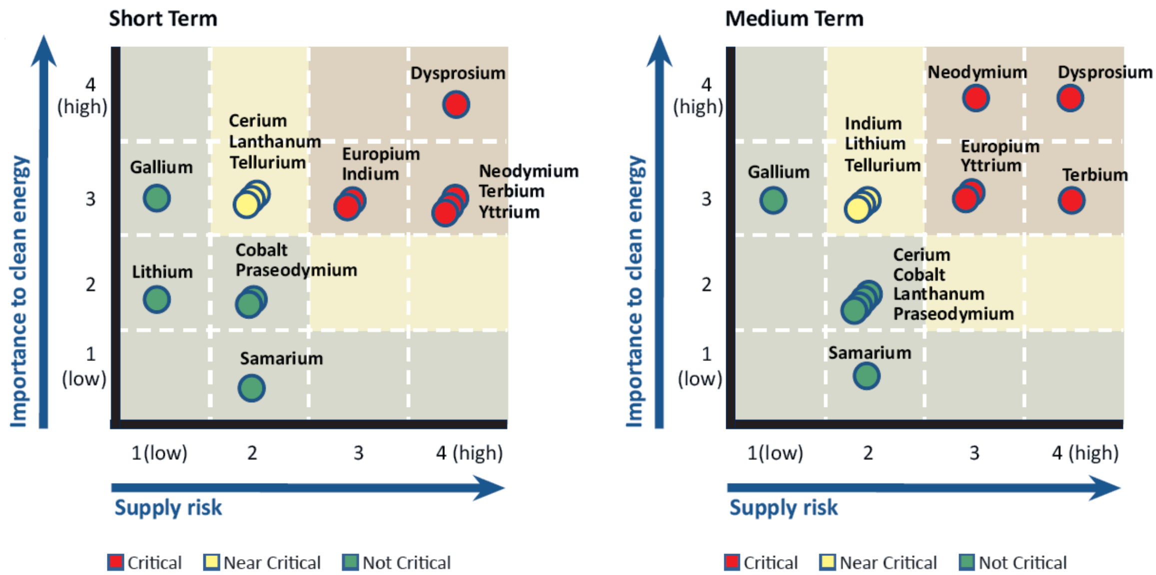

Figure 1. *Short term (0-5 years) and medium term (5-15 years) outlook and risk for neodymium and other elements for clean energy as identified by US Department of Energy (2010).*

Finnveden (2005) argues that LCA is not the proper methodology to handle resource use, but simultaneously states that it could a useful tool for investigating resource use aspects of various products. The demand for material resources for building wind farms if often not of any great concern in an LCA, but if the installed capacity of wind power is projected to grow extremely fast in coming years and the amounts of certain materials that are going to be used for this development can be important for the maximum possible growth rate, the cost of development as well as the apparent feasibility of wind energy in a wider context. Material shortages, production bottlenecks, price spikes of key components due to raw material depletion, and similar events can very well pose looming challenges for future expansion so it may be wise to include material use to a greater extent in LCA studies on wind energy.

4.3 Recycling

International standards for LCA allow for different ways of dealing with recycling, but states that the approach of inflows and outflows of recycled materials should be consistent (Ekvall and Weidema, 2004). Many assessments credit the beneficial environmental impact of recycling the materials in the future to the overall environmental impact of the life cycle. Martinez et al. (2009b) concludes that "*despite the significant amount of material used, its final impact is reduced because of the 90% material recycling in the phase of dismantling and disposal of the turbine*". Tremeac and Meunier (2009) state: "*The reason why dismantling and removal yields impact reductions is that recycling is used to a high extent*". This end-of-life approach for recycling is supported by the metals industry (Atherton, 2006).

It could be questioned how certain it is that the materials will in fact be recycled in 20 years, or more. For some materials making up large parts of a wind turbine, i.e. steel, copper, aluminum and other metals, it is highly likely that the materials will be recycled in the future, but it is not certain. The economics of recycling scrapped wind plants are also uncertain and it is entirely possible that the cost of dismantling and extracting the recyclable parts will be prohibitively high in the future, especially for wind farms located in remote or off-shore areas. For example, the Tehachapi Pass in California contains "*bone yards*" of abandoned wind turbine hardware that has been lying around without being recycled (Pasqualetti et al., 2002). Even if decommission is usually mandatory in operating permits, the total costs of decommissioning may not be covered due to price inflation, low capacity, unexpected circumstances (e.g., hurricane destruction), or a combination of such events (Kaiser and Snyder, 2012). It is possible that recycling can become uneconomic compared to abandonment under certain conditions, which is important to remember as decommissioning is dependent on a number of highly uncertain parameters that can have significant direct or indirect impacts on cost. Thus, material may be "*frozen in*" and unavailable despite their theoretical recyclability. Material recovery at the end of the life cycle cannot be guaranteed as expressed by Crawford (2009), who also stresses that the environmental credit should rather be given to products using the recycled material.

Since actual resource use does not appear to be of any great concern in the reviewed studies, the crediting of recycling mostly have impact on the other things assessed, such as energy performance and different emissions. However, the exact implications of the recycling on final results are generally difficult to follow in the assessments, although some tries to provide more details. In Martinez et al. (2009a) recycling significantly improves the environmental impact of the wind turbine. For the tower, environmental impact is said to improve with 52% due to recycling crediting. In essence this means that the apparent environmental impact of building the tower – mainly consisting of steel, is cut in half by end-of-life recycling crediting that might happen after more than 20 years. Guezuraga (2012) states that when no recycling of materials is considered, energy requirements are increased by 43.4 %, indicating that energy performance would be 43.4% worse if this view of recycling is not used. In Martinez et al. (2009b), "*Characterization results"* are presented as 10 different impact factors in a table, for maintenance, tower, foundation, rotor, nacelle and a total. Another table describes "*Environmental impact prevented by recycling"*, for the same parts of the turbine. For many of the impact factors, the impact prevented by recycling seems to be a large part of the total and for the two impact factors, *Abiotic depletion* and *Photochemical oxidation,* the environmental impact prevented by recycling is larger than the total value in the *Characteration results*-table. This could be interpreted, as building the turbine would actually decrease depletion of abiotic resources, which is intuitively impossible.

This raises questions of how this looks like in other assessments which does not present details how the end-of-life recycling approach affect the energy use and other impacts over the life cycle. Better and more transparent descriptions on how the assumed future recycling affect the final results are called for if LCA is supposed to give reasonable perspectives on sustainability issues. Clearly, there is a significant issue here and readers could easily be misled unless they are cautious.

If natural resource demand of wind power are to be investigated, in a greater context over time, other issues with the end-of-life recycling methods rises. Wind power is often projected to grow at an extremely high pace, causing an increased demand for resources during the construction of the new turbines, no matter how much will be recycled at the end of their life cycles. The materials will still be locked up in the wind turbine during its economic life, typically around 20 years or more, before it potentially can be recycled.

Jacobson and Delucci (2011) states that Earth has somewhat limited reserves of economically recoverable iron ore, over a 100–200 year perspective at current recovery rates, but also mention that most of the steel will be recycled. What is not mentioned is that the steel consumption is already rising fast. ESTP (2009) projects the global steel consumption to be over 2000 Mt by 2050, compared to just below 1400 Mt in 2010. This growth, coupled with the fact that recyclable steel has often been held up for many decades before finally being recycled, makes the total part of steel production coming from recycled steel is fairly low, only around 45% in Europe (ESTP, 2009). Such real world recycling shares appears to be in significant disagreement with some of the very high recycling percentages used in the reviewed studies.

5. Concluding discussion

There is still a matter of controversy concerning the environmental impact of wind energy, as Tremeac and Menuier (2009) pointed out. For several reasons, some of which are addressed in this paper, there is a wide spread in results from different wind power assessments, and even what kind of results that are presented. Compiling 72 studies, Lenzen and Munksgaard (2002) found energy intensities of 0.014 to 1.016 kWh_{in}/kWh_{el} while CO_2 intensities varied from 7.9 to 123.7 g/ kWh_{el}. Kubiszewski et al. (2010) compiled 50 EROI studies and found values ranging from 1.0 to 125.8 with an average of approximately 18. It is difficult to see that the higher figures describe the same concepts as the lower ones. It should be added that many of the results in these studies are old, and that LCA methodology has evolved since. However, a large spread in results is still seen in the fairly new studies reviewed in this paper (Table 3).

The critique expressed here is not directed towards existing ISO-standards concerning LCA and does not question whether or not the reviewed assessments follow these standards. The critique is not directed towards the specific assessments or authors, but rather tries to address a need for discussion on how environmental impact, energy performance and natural resource use of renewable energy resources should be assessed. The impacts are usually derived from different methods presenting different kinds of environmental impacts. What should be asked is, which environmental impacts are actually relevant for energy producing facilities like wind turbines, and what is the best methods to assess this. For future long term planning, the most relevant factors of interest for wind turbines are probably use of resources, both energy and materials.

5.1 Improving the treatment of energy

There is significant problem that EROI or EPBT is sometimes presented as primary energy using thermal equivalents, and sometimes using direct equivalents, making comparisons very difficult, especially since is sometimes difficult to even interpret if the conversion were done. As an example, Lee et al. (2006) and Lee and Tzeng (2008) presents an EPBT of 1.3 months – equivalent an EROI of 185 – widely superior to all other reviewed studies. It seems like they use direct energy payback time without any conversion to thermal equivalents, but still compares their result to Schleisner (2000), who converts produced electricity to primary energy. It can be seen as quite odd that an energy performance many times better than Schleisner (2000) – and literally all other previous LCAs on wind energy – but this is not reflected upon. Instead, it is claimed that performance of wind power systems implemented in Taiwan is among the best in the world (Lee et al. 2006). Drawing these conclusions without analyzing other reasons for the variations, such as methodological differences, should be considered highly questionable. This is just one of example how a LCA study can make flawed and even misleading comparisons and conclusions, and one should be cautious in drawing conclusions from the results. However, the purpose of this paper is not to compare the different results, but merely to look at how wind energy LCAs are commonly done.

Regarding energy use during the life cycle, we find no consensus on how different energy carriers should be treated. How this is done is generally not clearly described in published studies either. The total amount of primary energy used is often presented, and in some cases this is also divided into different energy carriers. However, energy carriers used varies between studies making comparisons difficult. For electricity, national generation mixes are typically used, if anything is mentioned at all. How much of the total energy used was originally electrical energy is not plainly presented in any of the reviewed studies, making it difficult to investigate the impact of using of different electricity mixes. Guezuraga et al. (2012) showed that switching generation mix could alter the results by around 50%, indicating the importance of this factor.

Another example is Schleisner (2000), converting electricity into primary energy that is supposed to be used by a "*conventional power plant*" at an efficiency of 40%. This way of thinking basically takes for granted that the electricity produced will substitute electricity from a "*conventional power plant*" despite the fact that LCA methodology simply does not contain the tools to assess the impact of the produced electricity within a real world energy-economic system. Similarly, other assessments draws conclusions about how much emission will be saved due to the new production. Instead of being used as built-in assumptions in an LCA, this kind of estimations should probably be left for more advanced modeling approaches. One such example is the *Life Cycle Sustainability Analysis* (LCSA) framework and related approaches that broadens the scope of LCA to also include sector and economy levels, thus giving more integrated and holistic assessments (Guinée et al., 2011).

There is simply no consensus on how produced electrical energy should be treated. Some converts the electricity produced in to primary energy using thermal equivalents, while some do not. If the primary energy conversions are to be used it is extremely important that it is explained that the conversion were made and how it was done so that the result is communicated correctly. Among the reviewed studies, we note significant confusion on these concepts and even faulty comparisons. It is not acceptable that published LCA studies mixes up fundamental concepts from energy analysis. As long as different assessments do not use the same method and do not describe which methods that are used properly, this will add to the difficulties of comparing different assessments to each other and make some results more or less useless.

5.2 Improved handling of non-energy resources

The need for non-energy resources does not seem to be seen as an important factor in most studies, and is usually not considered or discussed in any detail. When they are, intricate impact methods expressing resource depletion in antimony equivalents per kg is sometimes used even though this likely will be challenging to grasp for laymen and planners. Material resource use is a trivial issue for LCA according to Weidema (2000). In contrast, Finnveden (2005) suggests that resource use, although it should not be included as an impact factor in the LCIA, could be included in the LCA and states that LCA potentially can be a useful tool for discussing both environmental and resource aspects of products.

Another significant problem is the use of end-of-life recycling crediting. It can be argued, for many reasons, that environmental effects of recycling that may occur in 20 years should not be credited the environmental impacts apparent today. However, most of the reviewed studies credit future recycling in some way. The implications of the recycling crediting on the results are often difficult to interpret, but for some that presents it, the effect appears to be significant. For instance, energy use in Guezuraga et al. (2012) is increased by 43.3% when no recycling of materials is considered.

The amount of refined materials that is needed to build a wind turbine is often presented, but usually often including recycling or substitution crediting. LCA do not have the necessary tools to model real world mineral exploitation or the mechanism of economic substitution, making this situation identical to the shortcomings seen for electricity. If the LCI reflected actual resource use for the product, without end-of-life credits or assumed substitution, the LCA would be improved. With such presentations, it would also be possible to better explore the consequences of other assumptions about recycling and substitution.

From a purely environmental position the actual resource depletion might not be the most important factor, but from a sustainability viewpoint it can be extremely important information required for making the correct choices about future development energy and natural resources. When it comes to evaluating the impact of decisions concerning renewable energy sources, use of material resources can be of major future importance, and LCA methods are likely to be among the most appropriate way to address this. One possible way to deal with some of the issues with natural resource use is to use exergy in a life cycle perspective, which has been proposed by scientists in the last couple of decades. The *life cycle exergy analysis* (LCEA) method is an example of this (Wall, 2011). How, or even if, natural resource use and depletion should be included in LCA of wind energy appears to be largely unsettled, and a continued discussion on this is strongly encouraged.

5.3 Final recommendations

The most troublesome part we found is the lack of transparency regarding fundamental and underlying assumptions, calculations and conversions done in the reviewed LCAs. Mitigating this issue will not only improve clarity, but is also likely to strengthen the credibility of LCA methodology. The LCA society should clearly strive for better agreement on which methods that are to be used for evaluating renewable energy resources. This is not just desirable, but crucial, to be able to accurately evaluate and present the environmental performance of wind energy. Also, the use of natural resources, like REEs, should be clearly mentioned in the assessments to enable evaluating of possible bottlenecks in future production. There are some initiatives to address these issues via LCSA (Guinée et al., 2011), but strong international collaboration is a must to avoid ending up with a multitude of different approaches and methods.

Acknowledgements

We would like to thank two anonymous reviewers for helpful comments. Ehri Gbegbaje-Das from PE International and Peter Garrett and Klaus Rønde from VESTAS have our gratitude for providing assistance and clarifications regarding the Vestas LCA-study. This study has been supported by the STandUP for energy collaboration initiative.

Nomenclature

EPBT = energy payback time
EROI = energy return on (energy) investment
I/O = input-output
LCA = life cycle assessment
LCI = life cycle inventory
LCIA = life cycle impact assessment
LCSA = life cycle sustainability analysis
PCA = process chain analysis
PEROI = primary energy return on (energy) investment
PEPBT = primary energy payback time
REE = rare earth elements

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