Assessing the life cycle environmental impacts of wind power: A review of present knowledge and research needs

Anders Arvesen* and Edgar G. Hertwich

Industrial Ecology Programme and Department of Energy and Process Engineering, Norwegian University of Science and Technology

* Corresponding author. Email address: anders.arvesen@ntnu.no

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ABSTRACT

We critically review present knowledge of the life cycle environmental impacts of wind power. We find that the current body of life cycle assessments (LCA) of wind power provides a fairly good overall understanding of fossil energy use and associated pollution; our survey of results that appear in existing literature give mean values (± standard deviation) of, e.g., 0.060 (±0.058) kWh energy used and 19 (±13) g CO_2e emitted per kWh electricity, suggesting good environmental performance vis-à-vis fossil-based power. Total emissions of onshore and offshore wind farms are comparable. The bulk of emissions generally occur in the production of components; onshore, the wind turbine dominates, while offshore, the substructure becomes relatively more important. Strong positive effects of scale are present in the lower end of the turbine size spectrum, but there is no clear evidence for such effects for MW-sized units. We identify weaknesses and gaps in knowledge that future research may address. This includes poorly understood impacts in categories of toxicity and resource depletion, lack of empirical basis for assumptions about replacement of parts, and apparent lack of detailed considerations of offshore operations for wind farms in ocean waters. We argue that applications of the avoided burden method to model recycling benefits generally lack transparency and may be inconsistent. Assumed capacity factor values are generally higher than current mean realized values. Finally, we discuss the need for LCA research to move beyond unit-based assessments in order to address temporal aspects and the scale of impacts.

Keywords: LCA, carbon footprint, sustainability assessment, wind energy, electricity

1 Introduction

Electric power generation by wind turbines is commonly regarded as a key technology in addressing some of the greatest environmental and resource concerns of today, namely man-made climate change and other negative effects of air pollution, and security of energy supply. Among other factors, strong growth in today's markets and prospects of exploiting vast resource potentials at offshore sites contribute to the anticipation that wind power will play a significant role in achieving a shift away from fossil-based power generation towards renewables in coming decades [1, 2]. Wind power likewise features prominently in the current body of climate change mitigation scenarios produced by large-scale integrated assessment models [3, 4]. Even though wind power is driven by a renewably energy flux (that is, the kinetic energy in air streams), in a life cycle perspective there are non-renewable resource demands and harmful emissions associated with it. These environmental and resource pressures can be quantified and assessed by the method of life cycle assessment (LCA).

Surveying LCA studies published from the year 2000 on, this paper synthesizes and critically reviews current state of knowledge about the life cycle environmental impacts of wind power. The work was carried out with the goal of contributing to a wider, comparative study of the environmental and resource impacts of low-carbon energy technologies by the International Resource Panel for the United Nations Environment Programme.

Several literature reviews of wind power LCAs are already available. Lenzen and Munksgaard [5] survey 72 energy and CO_2 analyses of wind power systems published between 1977 and 2001. Kubiszewski and colleagues [6] and Raadal and colleagues [7] extend the work of Lenzen and Munksgaard [5], adding additional analyses, focusing on energy demand and greenhouse gas (GHG) emissions, respectively. In another review in the IPCC Special Report on Renewable Energy Sources and Climate Change [1, 8], 126 estimates from 49 studies are surveyed. The present LCA review aims to supplement the previous assessments, providing new surveys and analyses of results as well as qualitative discussions. In particular, we attempt to make the following original contributions: i) taking a broader view of environmental impacts, focusing not only on cumulative energy demand and GHG emissions [5-7], but on a wider set of impact categories assessed in the LCA literature; ii) discussing important aspects that are not sufficiently treated in previous LCA reviews, including capacity factor assumptions, modeling of recycling benefits, techniques for calculating life cycle inventories (process-LCA or hybrid LCA), and static versus future-oriented LCA; iii) critically assessing the scope and quality of existing studies, identifying areas that are well understood as well as important knowledge gaps; and iv) proposing directions that future research may take in order to gain a more complete and solid understanding of the environmental implications of wind power.

The following section briefly introduces the conceptual basis of LCA and the two prevailing methodological approaches to life cycle inventory analysis. Next, Section 3 describes the construction of the literature database which forms the basis of the survey and review. Results of the literature survey are presented in two sections: Scope, assumptions and methodologies of existing LCA research on wind power are dealt with in Section 4, while Section 5 presents stressor and impact indicator results. A critical evaluation of present knowledge and research needs is given in Section 6. Finally, Section 7 provides concrete recommendations for future research.

2 LCA: conceptual basis and calculation techniques

LCA is a method to explore how the delivery of or demand for a specific product or service (e.g., the delivery of one unit of electricity from wind) initiates processes that may cause environmental impacts. Through a systematic mapping of operations and associated environmental pressures along a product's life cycle, LCA strives to give a complete picture of the environmental burdens caused by one product [9].

Two approaches to quantifying life cycle inventories are in use. In conventional LCA methodology, henceforth referred to as process-LCA, a bottom-up approach is taken to define and describe operations in physical terms. This approach makes possible the use of data that are specific for the operations under consideration, meaning that results can potentially be generated at high levels of detail and accuracy. On the downside, there is a need to apply cut-off criteria to exclude operations that are not expected to make significant contributions. It is known, however, that added together the excluded contributions are significant [10, 11]. The second approach, environmentally extended input-output analysis (EEIOA), is a top-down technique in which inventories are quantified using monetary data at the level of economic sectors. As EEIOA does not require cut-offs to be made, it does not have the same problem with truncation as process-LCA. However, EEIOA operates at a high aggregation level; the sector resolution in EEIOA is generally too coarse for making LCAs of specific products. Hybrid methods – where process-LCA is used to model important operations, and EEIOA is used to model operations that would otherwise be omitted – can potentially exploit advantages of both approaches, but is more challenging to employ [10-12]. Also, depending on the method of hybridization and quality of data [12], most hybrid models may offer limited support for following material flows through product systems.

LCA results may be presented as inventories of individual stressors, or as environmental impact category indicators at 'midpoint' or 'endpoint' levels of aggregation. Midpoint indicators allow for environmental effects of several individual stressors to be assimilated into a single impact category. Endpoint indicators measure impact potentials by endpoints in the effect chain; human health, ecosystem health and natural resources are typically regarded as three such endpoints, but sometimes even one single indicator of environmental damage is used [13, 14].

3 Literature survey

In surveying published LCA research, priority was given to cover publications in peer-reviewed journals, and for the most part, studies were identified through searches in common scientific databases. However, when found appropriate other types of publications (e.g., environmental reports by manufacturers, documentation of LCA databases) that have been known to the authors were included as well. The LCA survey presented here differ from that of past reviews in that studies published prior to 2000 are excluded. The primary reason for this is the strong developments in wind power technologies, LCA methodologies and databases, and background economy characteristics in previous decades. Furthermore, the set of studies reviewed was judged to be large enough to provide interesting insights.

An overview of the reviewed LCA studies on wind power systems is given in Table 1. Of the 44 reviewed studies (Table 1), 34 were selected for quantitative analysis. In general, the following guidelines were followed in constructing the set of observations used for quantitative analysis: i) Only original LCA research was included. ii) Studies of integrated wind power generation and energy storage systems were excluded in the cases where the contribution from the actual wind power system could not be extracted from the inventories presented. iii) For studies presenting a number of results that apply to different systems (e.g., onshore and offshore wind farms, differently sized turbines), all reported results were included. iv) For studies presenting a number of results for one specific system, but with differing methods or assumptions (e.g., different capacity factors, different approaches to modeling benefits of recycling), the default (reference) scenario was surveyed if such a scenario was defined. Conversely, if a default scenario was not defined, an average of reported values was surveyed. Table S1 in the supplementary information provides the raw data for the quantitative analysis in terms of system characteristics, and emission and impact indicator results.

Table 1. Overview of assumptions, methods and scope of reviewed LCA studies. Site: Ons = Onshore; Off = Offshore. Size: S = Small (< 100 kW); M = Medium (100 kW-1 MW); L = Large (> 1 MW). Lifetime: '*' means longer lifetimes for some components. Credits: 'x' means system is credited with indicator values that are perceived to be avoided through recycling at end-of-life; '(x)' means system is credited, but results without credits are also presented. Temporal scope: A blank means static assessment under assumptions of present technologies; non-blank entries indicate future-oriented assessments. Method: Pro = Process-LCA; Hyb = Hybrid LCA; IOA = Analysis that relies fully or in large part on input-output multipliers. Impact categories: C = CO_2 emissions; CC = Climate change; E = Cumulative energy demand; R = Resource requirements, abiotic depletion; A = Acidification; O = Stratospheric ozone depletion; HT = Human toxicity; P = Particulate matter formation, dust; ET = Ecotoxicity; PO = Photochemical oxidation (smog); N = Nutrient enrichment, eutrophication; W = Solid waste generation; L = Land use, land transformation; h = human health endpoint; e = natural environment endpoint; r = natural resources endpoint; s = single score endpoint; α = non-toxic emissions that provide additional information; τ = toxic emissions that provide additional information ('additional' with regards to the impact categories that are accounted in this table column). Characters are underlined if results are presented in generic units only (e.g., 'points').

Citation	Site	Size	Lifetime (years)	Credits	Geographical scope	Temporal scope	Method	Impact categories
[15]	-	-	-		Global	2009-2100	-	C
[16]	Ons	S M	25	x	Canada		Pro	CC E A PO
[17]	Ons	L	20		Germany Denmark China		Pro	CC E
[18, 19]	Ons Off	L	20 (Ons) 25 (Off)		Europe/Global	2007-2050	Hyb	C CC A PO N
[20]	Ons	L	20	x	China		IOA	CC E
[21]	Ons	L	20	x	Denmark		Pro	CC E R A P ET PO N W
[22]	Off	L	20		Germany		Pro	CC E A HT PO N
[23]	Off	L	20		UK		Hyb	C CC
[24]	Ons	M	20	(x)	Europe		Pro	CC E R A O HT ET L
[25]	-	-	20		Brazil	20 years	-	E
[26]	Mix	L	20		Northern-Europe		Pro	C CC A O P PO N W τ
[27]	Ons	M L	20		Australia		Hyb	CC E
[28]	Ons	S	20		Canada		Pro	CC
[29]	Ons	-	30		Denmark		Pro	L
[30-32]	Ons	L	20	(x)	Spain		Pro	CC E R A O HT ET PO N L s
[33]	Ons	L	-	x	New Zealand	100 years	Pro	C E
[34]	Ons	S L	20	(x)	France		Pro	CC E h e r
[35]	Off	L	20	(x)	Norway		Pro	CC E R A HT ET PO N
[36]	Ons	M	20		Italy		Pro	C CC E A O P PO N W τ
[37]	Ons	-	20		Taiwan		Pro	C E
[38]	Off	L	20*		Denmark	2005-2050	Pro	C P L α τ
[39]	Off	L	-		Germany	2005-2020	Pro	C
[40-42]	Ons Off	S M L	20		Switzerland/Europe		Pro	C CC E R A O HT P ET PO N W L h e r s
[43]	Ons	S	25		Turkey		Pro	C CC E
[44]	Ons Off	L	-		Germany	2010	Pro	C CC E A P N α τ
[45]	Ons	S	20	(x)	UK		Pro	C E
[46]	Ons Off	L	20 (Ons) 20* (Off)	x	Denmark		Pro	C CC E A O HT ET PO N W
[47]	Ons	L	20	x	Denmark		Pro	C CC E A O HT ET PO N W
[48]	Ons	M	20-30		US		Pro	C E
[49]	Ons	M	30		Japan		Pro	CC
[50]	Ons	M	20		Canada		Pro	CC E
[51]	Ons Off	L	20 (Ons) 20* (Off)	x	Denmark		Pro	C CC E A O HT N W α
[52]	Ons	M	-		Germany Brazil		Hyb	C E
[53]	Ons	M L	20		Germany		Pro	E
[54]	Ons Off	M L	20		Europe		Pro	C α
[55]	Off	L	20*		Denmark		Pro	CC A HT ET PO W
[56]	Ons	M	15		Japan		Pro	C E
[57]	Ons	M	20		US	40 years	IOA	CC
[58]	Ons Off	M	20		Denmark		Pro	C E α s
[59]	Ons	M	20		Belgium		Pro IOA	CC E

Finally, we note that the set of observations included in the quantitative analysis is not a random sample. The identification of studies did not follow a formal, randomized procedure, and the studies that were identified are sometimes not independent, as they utilize common sets of assumptions or data. Also, the survey involved some subjective choices (e.g., regarding how multiple observations from single studies should be inventoried) that may to some extent have influenced quantitative analysis results.

4 Scope, assumptions and methodologies

The LCA literature covers the whole spectrum of available wind turbine sizes, from hundreds of watts sized units [28, 34] to multi-MW turbines in onshore and offshore locations. As is evident from Table 1, analyses of wind farms operating on land form a vast majority, and there is a predominance of analyses with Europe countries as their reference locations. A fair number of analyses (13) of ocean-based systems were also identified. With exceptions, LCAs of offshore wind power study bottom-fixed wind turbines in relatively shallow waters. Two studies analyze, respectively, a hypothetical wind farm comprised by floating units [35] and an operational wind farm at a water depth of 30 m [22].

Manufacturing of the actual wind turbines is the only life cycle stage that is common to all analyses. In addition, all assessments based on wind turbines with capacities of hundreds of kilowatts and more include the manufacturing of foundations, and the majority model electrical connections (internal cables within wind farm, external cabling and sometimes transformer stations) needed to connect a wind farm to an existing grid. Most studies also take into consideration – though variably –the operation and maintenance of the system, as well as transportation activities. A number of assessments [28, 39, 43, 50, 56] address integrated systems where wind energy converters are supplemented with other power generation technologies and/or technologies for energy storage.

The manner in which the end-of-life phase is modeled varies. Some studies make assumptions to model transport and disposal of waste, others omit this part. End-of-life is unique among the life cycle phases in that it may reduce emissions and resource use: Negative contributions occur when analysts deduct indicator values that are perceived to be avoided when, after the operating lifetime, system components are recycled or incinerated to produce valuable outputs. In this way, the system is credited for returning usable resources (e.g., recyclable steel) to the technosphere – in the LCA literature this is referred to as substitution by system expansion or avoided burden method [13]. LCA studies that employ the avoided burden method are in minority, but nevertheless represent a significant share (Table 1) Decommissioning of a wind farm after the service lifetime is typically modeled as identical to installation.

LCAs of wind power generally assume lifetimes of 20 years, for onshore and offshore wind farms alike (Table 1). Fig. 1 displays capacity factor assumptions by region as a function of power rating. Three overall trends may be observed from Fig. 1, in overall terms consistent with general knowledge and expectations [60, 61]: i) performance in terms of capacity factor increases with wind turbine nominal capacity; ii) offshore wind farms exhibit greater energy capture than onshore farms; and iii) for a given power rating, sites in North America tend to show higher capacity factors than European sites. Across all regions, the assumed capacity factor mean value (± standard deviation) is 18% (±5.4%) for onshore wind turbines with nameplate capacity below 100 kW, 22% (±5.1%) for onshore with capacity 100 kW - 1 MW, 31% (±7.5%) for onshore with capacity > 1 MW, and 43% (±8.4%) for offshore (Table S2 in the supplementary information).

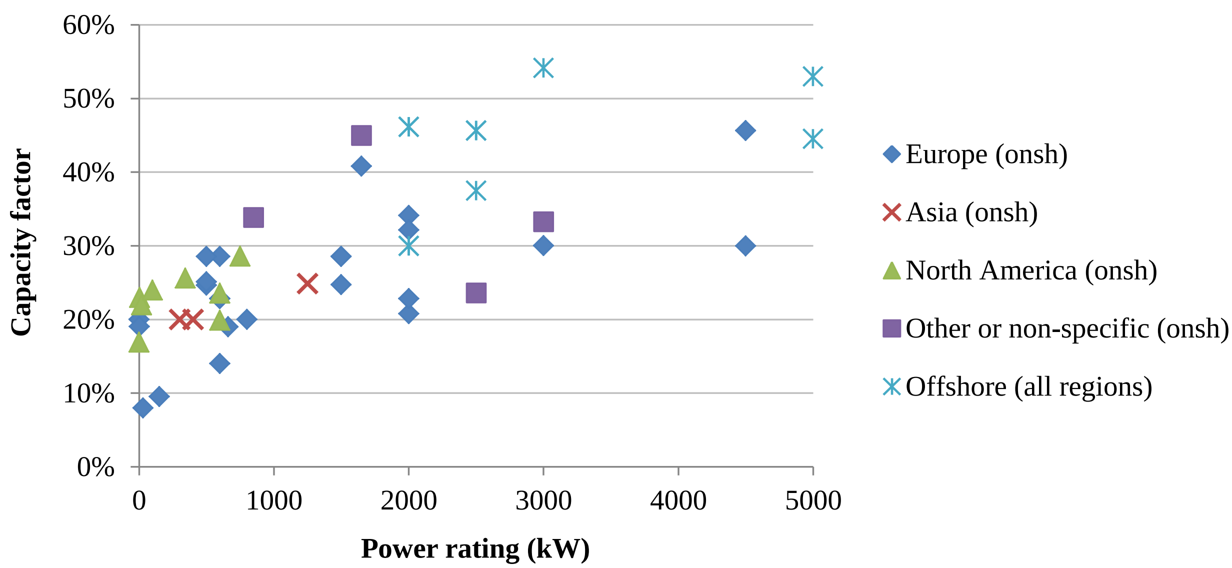

Fig. 1. Capacity factor by location as a function of power rating.

Energy demand and GHG emissions have historically been the main focus of attention for LCA research on wind power [5], and still dominate the impact assessments in recent literature (Table 1; also, compare the sample sizes of energy, GHG and CO_2 versus other air pollutants in

Fig. 2). Estimates of climate change indicator values are often comprised of contributions from CO_2, CH_4 and N_2O, but in some cases (e.g., [23, 41]) fluorinated GHGs (SF_6, HFC, PFC) are also taken into account. Of the studies cited in Table 1, more than half include impact categories other than energy and GHGs. In general, environmental stressors of high coverage are air pollutants associated with production and combustion of fossil energy carriers: CO_2, CH_4, CO, NH_3, NMVOC, N_2O, NOx, particulates and SO_2. Such a set of pollutants facilitates meaningful impact assessments in the categories climate change, acidification, eutrophication and photochemical oxidation (smog). In comparison to fossil fuel-related air emissions, other kinds of pollution have received little attention; only 10 studies cited in Table 1 quantify characterized toxicity indicator results. Apart from fossil energy carriers, resource requirements and non-renewable resource depletion are scarcely addressed in detail. A handful of studies [21, 24, 31, 35] address non-renewable resource depletion; others [26, 46] display life cycle inventories for individual mineral resources without applying any impact assessment. One publication [29] was identified that examines in some detail direct and indirect land use of power generation technologies, including wind. Some studies quantify life cycle water use, but water use is generally not highlighted or discussed in detail. Fthenakis and Kim [62] review previous studies and evaluate life cycle use of water in electricity supply by different technologies.

As is evident from Table 1, process-LCA studies dominate the wind power LCA literature, and few studies employ hybrid LCA methodologies. As a final point regarding methodology, we note that different kinds of future-oriented LCAs of wind energy have started to emerge in the literature, but are yet to gain widespread employment (cf. 'temporal scope' column in Table 1). Methodological approaches and results of future-oriented LCAs are discussed in Section 5.3.

5 Stressor and impact indicator results

Fig. 2 presents literature survey results with respect to total emissions and impact indicator values, and the numbers of estimates and studies that were surveyed; numerical results in tabulated form are provided in the supplementary information. For onshore and offshore wind power respectively, the mean energy intensity value is 0.063 (±0.061 standard deviation on either side of the mean) and 0.055 (±0.037) kWh/kWh; mean GHG emissions are 20 (±14) and 16 (±9.6) g CO_2e/kWh; and mean CO_2 emissions 16 (±14) and 12 (±7.3) g/kWh. These relatively large standard deviations, and the broad ranges that can be observed for all categories displayed in Fig. 2, illustrate that results vary considerably. For example, reported energy intensity values across all wind power system categories form an interval of 0.014-0.333 kWh/kWh. If analyses of wind turbines with nameplate capacity less than 100 kW are excluded, however, the interval narrows to 0.014-0.137 kWh/kWh – this exemplifies a general pattern that the by far highest emissions and indicator values are observed for small wind turbine sizes (< 100 kW). Offshore wind power systems show comparable or slightly higher emissions than onshore systems comprised of large wind turbines (Fig. 2), despite the systematically higher wind capacity factors assumed for offshore systems (Fig. 1). This is due to the higher resource requirements of wind power systems located offshore. Another observation that can be made from Fig. 2 is a tendency for estimates to concentrate in the lower part of the observed intervals (note from Fig. 2, for example, that the mean values lie systematically above median values).

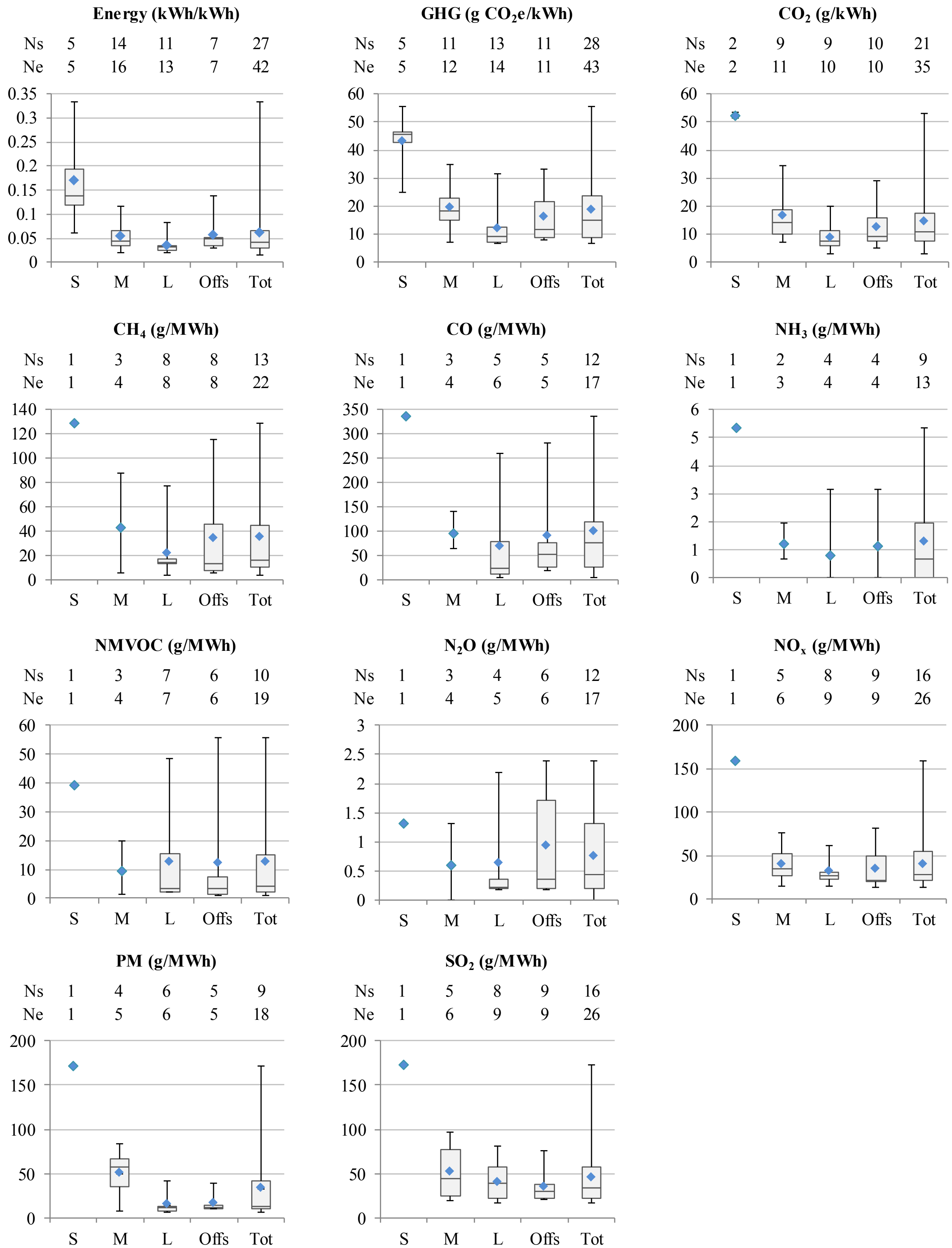

Fig. 2. Stressor and impact indicator results by 5 wind power system categories and 11 impact categories. Box: range from first to third quartile; Horizontal bar within box: median value; Diamond: mean value; Upper and lower fences (whiskers): maximum and minimum values. 'S' means small wind turbine (< 100 kW) at onshore site; 'M' means medium wind turbine (100 kW - 1 MW) at onshore site; 'L' means large wind turbine (> 1 MW) at onshore site; 'Offs' means offshore wind power (any wind turbine size); 'Tot' denotes total sample. Ns = Number of studies; Ne = Number of estimates. Ne > Ns if more than one estimate was surveyed from one study. In some cases total sample size slightly exceeds the sum of the sub-sample sizes; this is because estimates for wind farm portfolios were not assigned a system type, but were included in the total sample. If Ne < 5, interquartile ranges (boxes) are not shown; If Ne = 1, the one value is shown as a diamond. Energy indicator value refers to the ratio between life cycle energy demand and electricity generated over the lifetime. GHG = Greenhouse gases; CO_2 = Carbon dioxide; CH_4 = Methane; CO = Carbon monoxide; NH_3 = Ammonia; NMVOC = Non-methane volatile organic compounds; N_2O = Nitrous oxide; NO_x = Mono-nitrogen oxides; SO_2 = Sulfur oxides.

Releases of individual toxic substances in the life cycle of wind power systems are in some cases reported, but to synthesize these findings is difficult due to differences in what chemicals are reported and a lack of transparency on calculation methods and assumptions. Table 2 compares human toxicity and freshwater and terrestrial eco-toxicity indicator results from five studies. Marine aquatic eco-toxicity is not included due to weaknesses in current impact assessment methods [63], and because two of the cited studies [22, 35] do not address this impact category. One of the publications [21] cited in Table 2 report results that are up to three orders of magnitude smaller than those from the other studies. The reason for this discrepancy is unknown, but could possibly be a consequence of different impact characterization methods.

Table 2. Overview toxicity indicator results by three impact categories, as quantified by five studies. HT = Human toxicity. FET = Freshwater eco-toxicity. TET = Terrestrial eco-toxicity. DCBe = 1,4-dichlorobenzene equivalents.

Citation	Wind turbine size, site	Stated impact characterization method	Results (g 1,4-DCBe/kWh)		
			HT	FET	TET
[21]	1.85 MW, onshore	USETox (2008)	0.83	0.03	0.03
[22]	5 MW, offshore	-	69	-	-
[31]	2 MW, onshore	CML (2000)	16	2.8	0.16
[35]	5 MW, offshore	CML (2000)	83	12	0.23
[41]	800 kW, onshore	CML (2001)	54	10	0.16
[41]	2 MW, offshore	CML (2001)	53	10	0.18

5.1 Contribution analysis

Looking at the relative contribution from different life cycle stages to total energy use and climate change indicator result, manufacturing of components dominates, and is sometimes of the order 90% of total impact indicator values (Fig. 3; see also discussion in previous LCA reviews [5, 7]). Fig. 3 compares breakdowns of energy use and GHG emissions by components and life cycle stages. It should be noted that ambiguity exists in the categories shown in Fig. 3; for example, some studies separate transportation as an individual category, while other studies subsume transportation activities within other categories. Nevertheless, it is clear from Fig. 3 that for onshore wind power systems, the wind turbine is the most important single component with regards to energy use and GHG emissions, followed by the substructure (i.e., the foundation). The tower may hold a share of 30-70% of total wind turbine indicator values. For offshore wind farms, the substructure becomes relatively more important.

Generally, emissions associated with transportation are found to be negligible or of minor importance, though they sometimes are relatively more important for NMVOC and NOx emissions. The results of [34] (not included in Fig. 3) stand out with large relative contributions from transportation (34% of GHG emissions are due to transportation) – this could possibly be related to the choice of concrete as tower material in [34], as opposed to (lighter) tubular steel towers modeled in most other studies. Emissions of heavy metals in manufacturing processes is the primary cause of toxicity indicator results [21, 22, 35].

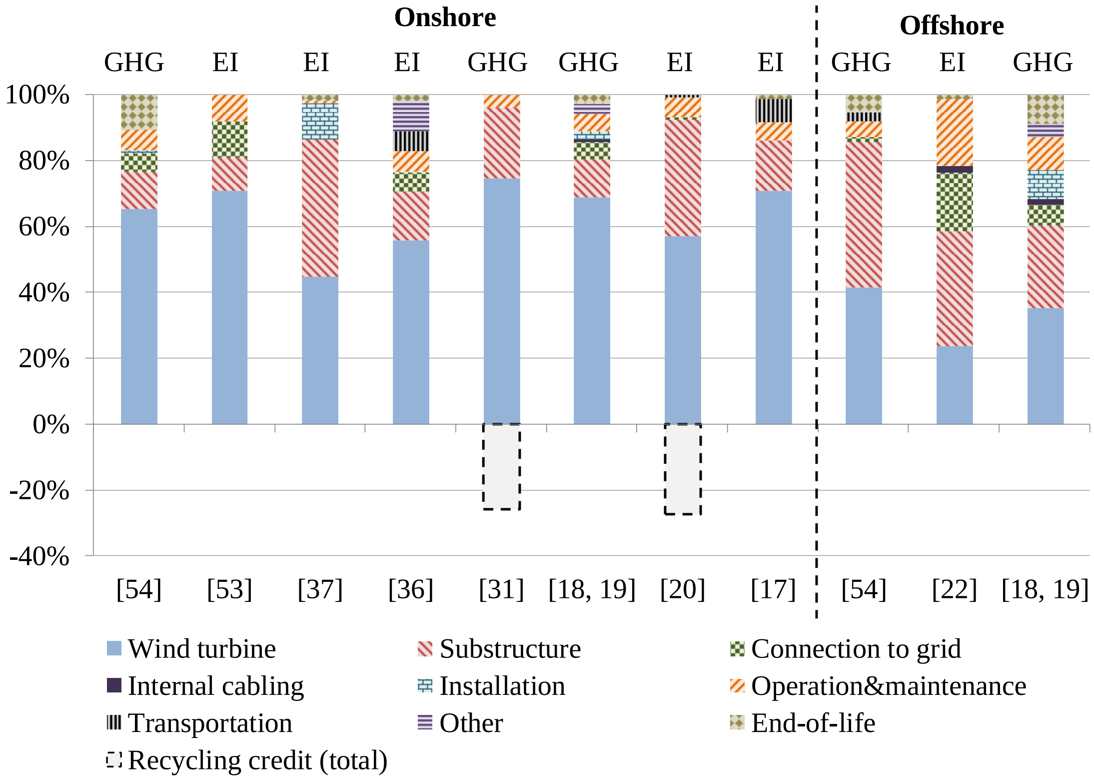

Fig. 3. Breakdown of energy intensity (EI) or greenhouse gas emission intensity (GHG) by main components or life cycle stages according to 8 onshore and 3 offshore estimates. In some cases interpretation of results and/or reading off charts in the cited publications was necessary. Shown positive indicator shares from [20, 31] do not include recycling credits.

If the avoided burden method is applied, the end-of-life phase typically yields considerable emissions reductions: Recycling credits approximately halve the energy or GHG emissions embodied in the wind turbine and lower total indicator values by 26-27% in [20, 31] (Fig. 3). In another study, recycling credits lead to around 20% (4.5 MW wind turbine) and 40% (250 W) reductions in GHG emissions [34]. In total, the end-of-life phase contributes -19% to GHG emissions in [35].

5.2 *Effects of wind turbine size and method for life cycle inventory*

Previous reviews of wind power LCA studies maintain economies of scale in the life cycle environmental impacts of wind power systems. Lenzen and Munksgaard [5] report that a 1 MW wind turbine appears to require only one third of the life cycle energy per unit output needed for a 1 kW sized unit. Kubiszewski and colleagues [6] and Raadal and colleagues [7] show evidence of energy use and GHG emissions decreasing with growing wind turbine size, but in the former case it remains unanswered to what extent the trend continues when moving into the MW size spectrum, and in both cases it appears that the practice of surveying old and, arguably, outdated analyses (going all the way back to the late 70s) on a par with recent analyses obscures the picture. Moving on to the results of the present survey, Fig. 4 depicts GHG emissions with increasing wind turbine nameplate capacity. The figure confirms the presence of strong economies of scale for power ratings up to 1 MW or so, but a downward trend is not readily discernible for larger turbine sizes.

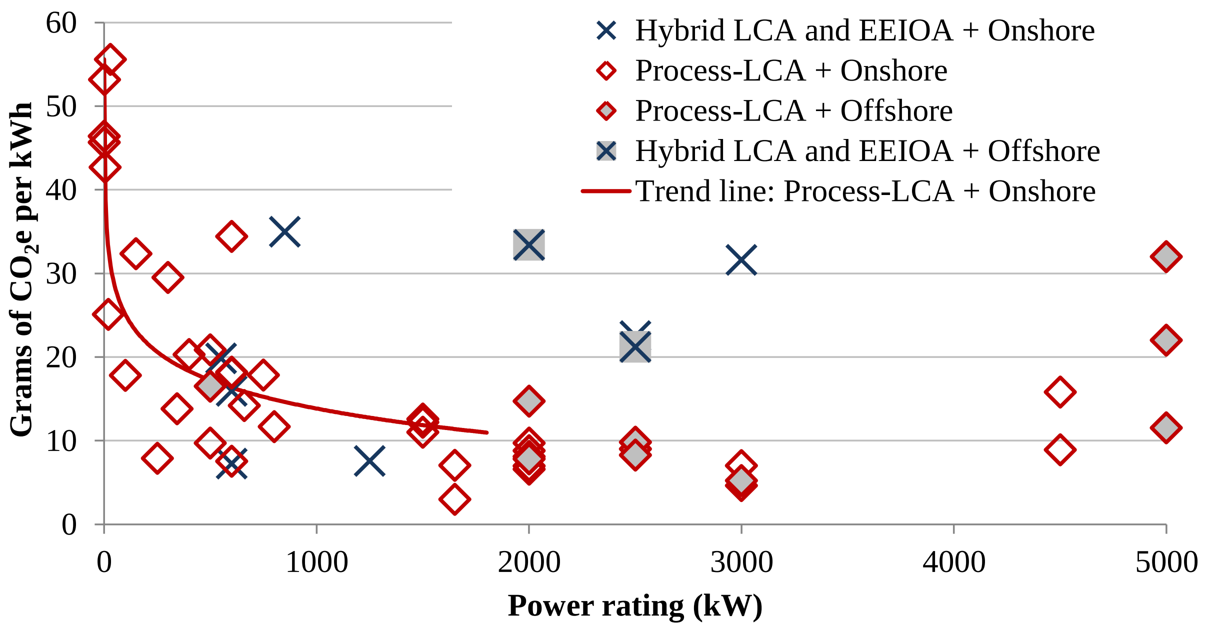

Fig. 4. Total GHG or CO_2 emissions as a function of wind turbine power rating by 4 combinations of methods (hybrid LCA and EEIOA versus process-LCA) and sites (onshore versus offshore). When available, total GHG emissions estimates were included in the figure. If GHG emissions estimates were not available, CO_2 emissions estimates were included. For estimates for offshore wind farms, markers are filled with grey. The trend line represents the sample characterized by a process-LCA method and onshore site, and power rating ≤ 1800 kW. Trend line equation: $y = -4.9ln(x) + 47.7$. $R^2 = 0.72$.

Theory and empirical evidence from the broader LCA literature foretell that hybrid LCA and EEIO-based assessments give systematically higher impacts (cf. Section 2), and Fig. 4 gives some confirmation of this. Total contributions from economic input-output sectors amount to 23-26 g CO_2e/kWh (74% of totals) in Crawford [27], 19 g CO_2e/kWh (57% of totals) in Wiedmann et al. [23] and 10-13 g CO_2e/kWh (45-61% of totals) in Arvesen and Hertwich [18, 19]. The size of the sample representing hybrid LCA in Fig. 4 is too small to admit a robust assessment comparing results of hybrid LCA and process-LCA, however.

5.3 Future-oriented assessments

In a forward-looking study for Germany, Pehnt et al. [39] couple life cycle inventories with a stochastic electricity market model to study the life cycle CO_2 emissions of wind power, grid expansion, energy storage by means of compression of air, and balancing requirements, in an integrated framework. Results for the year 2020 show only negligible emissions from storage and grid upgrades, but a relatively large emission penalty of 18-70 g CO_2/kWh arising from the balancing of variable wind electricity by fossil-fueled power stations. A global scenario-based assessment is presented by Arvesen and Hertwich [18, 19], who estimate 3.5 Gt CO_2e emitted due to the act of building and operating wind power plants in the time period 2007-2050 to supply 22% of worldwide electricity in 2050. The same study includes an integrated life cycle modeling of cumulative avoided emissions; results suggest emissions avoided by wind power grossly exceed emissions caused by wind power. Lenzen and Schaeffer [15] analyze caused and avoided climate change impacts of eight energy technologies towards 2100, the primary objective being to illustrate differences between emissions and temperature-based indicators for climate change mitigation potential (the authors argue that indicators of avoided temperature are more relevant for decision-making than avoided emissions). In yet another study, Gonçalves da Silva [25] proposes a mathematical framework for simulating the time dynamics in net and gross energy balances of renewable energy technology deployments; computational results are favorable for wind power. Finally, the report on offshore wind technology in the NEEDS project [38] makes assumptions on design changes and economies of scale in wind electricity technologies to establish life cycle inventories for future offshore wind power systems. For all scenario assessments cited above, there are important simplifying assumptions and thus careful interpretations of results are required – indeed, this point is also emphasized by the authors of the original publications.

6 Current state of knowledge and research needs: a discussion

6.1 Capacity factor and lifetime assumptions

The strong influence of assumed capacity factors and lifetimes on results is obvious, as emissions per unit of electricity (in units of grams of CO_2 per kWh, or similar) scale in inverse proportion with the amount of electricity generated over the lifetime – this is analogous to calculations of generation costs (in units of Euro per kWh, or similar).

With respect to capacity factor, one interesting comparison to make is that of assumptions made in LCAs (Fig. 1) versus real-world experiences. The average realized capacity factor in EU15 in 2003-2007 is reported at 20.8%, with country-level averages ranging from a low 18.3% (Germany) to a high 26.1% (UK) [60]; these real-world performances are significantly lower than the overall picture emerging from the assumed values shown in Fig. 1 for onshore wind turbines in the range of 1 MW and above, but relatively more consistent with assumptions for smaller turbine sizes. As regards capacity factors in the US, there are conflicting reports of real-world average values of around 26% [60] and 30% [61], while average capacity factors for China are reported at 16-17% in [64] and 23% in [65]. Data points representing North America and Asia in Fig. 1 are mostly in the lower end of the turbine size spectrum, however, not providing a good basis for comparison. Turning to the offshore case, LCA studies often assume capacity factors above 40% and even 50% (Fig. 1), which also appears somewhat optimistic in comparison with currently available measurements data: One study [66] concludes from a survey that "a typical offshore installation has an utilization time of 3000 hours or more" (i.e., capacity factor 34% or more); while, based on experiences from early Danish and Dutch wind farms, [67] generally expects a 35% capacity factor value for UK offshore wind farms, but finds that the real average value for UK round 1 offshore wind farms is 29.5%. Finally, we note that [66] proposes a constant 37.5% average capacity factor for offshore wind power to be used in scenario analysis towards 2050, while more optimistic scenarios are derived in [18] from IEA data [4], leading to an average offshore load 43% in 2050.

Based on the above information, there appears to be a general tendency of wind power LCAs to assume higher capacity factors than current averages from real-world experiences. At the same time, it needs to be emphasized that many LCAs make assumptions that are specific to one technology or wind farm site and as such not intended to be representative for overall trends.

Unlike capacity factor, real-world empirical evidence on the lifetimes of modern wind turbines is lacking; assumptions on lifetimes thus need to be guided by wind turbine specialists' evaluations and design lifetimes set by manufacturers. While LCAs typically assume lifetimes of 20 years for onshore and offshore systems alike (Table 1), Blanco [68] find that current economic assessments of wind power generally set lifetimes to 20 years onshore and 25-30 years offshore. On this basis, it appears that assumptions regarding lifetimes in the LCA literature are less favorable for offshore wind power, compared with equivalent assumptions underlying economic assessments.

6.2 Impact category coverage

The current survey shows that life cycle energy demand and GHG emissions of wind power are extensively covered in the extant literature. Also, there is a fairly large set of quantifications on air pollutants typically connected with the burning of fossil fuels (e.g., NOx, SO2) and associated impact categories (acidification, eutrophication, photochemical oxidant formation, and to a lesser extent, particulate matter). In our view, given the material intensive nature of wind power compared to fossil alternatives [69], and that toxic releases to the environment are known to originate from materials manufacturing [21, 22, 35], the most serious gap in knowledge is the insufficient understanding of toxic emissions generated in the life cycle of wind power systems. From the viewpoint of the LCA practitioner, assessing toxic effects may be difficult because: i) emissions data on toxic substances is missing or is incomplete, and ii) current impact assessment methods for toxicity produce contradictory results and hence lack robustness [13, 70]. The neglect or incomplete modeling of toxicity is not a problem specific to wind power LCAs, however, but applies to the LCA literature in general [13]. For marine ecological impacts from emissions to water, robust impact assessment methods are in the early stage of development (in the general case, unresolved issues may be exemplified by contradictory results for toxic effects of long-term metal releases, as discussed in [63], and effects of particle emissions [71]) – this is unfortunate for LCA research on offshore wind power, for which there are operations taking place in ocean waters which need to be modeled.

Another significant gap in knowledge is that represented by a lack of comprehensive evaluations of non-renewable (abiotic) resource demands. As with toxicity, this is a much debated impact category in the LCA community for which there is no consensus on impact assessment methods [13]. In the broader literature (e.g., [72, 73]), concerns have been raised about future shortage of supply of neodymium, a metal belonging to the group of rare-earth elements that is increasingly employed in permanent magnets in wind turbine generators. At the same time, the in-use stocks of neodymium have been found to be significant, suggesting that recycling may to some extent alleviate future constraints on primary resource supply [74].

Sustainability assessments of wind power need also adequately consider site-specific impacts, such as visual impacts, habitat change, and bird and bat collisions (see, e.g., Sections 7.6.2 and 7.6.3 in [1] for a summary). There is, however, little tradition for including such impact categories in LCA, and they are more frequently assessed using other environmental impact assessment methods (e.g., cost-benefit analysis, as in [75]).

6.3 Life cycle phases: research coverage, research agreement and quality of knowledge

Table 3 summarizes our overall judgments of the current knowledge about potential environmental burdens associated with four life cycle phases of wind power systems. The evaluation builds on previous sections of this paper and discussions provided in Sections 6.3.1-6.3.3. Extended life cycles (e.g., with respect to network integration or re-powering of systems) are discussed in Section 6.5.

6.3.1 Production of components

Production of system components forms a natural part of any wind power LCA. Discrepancies concerning values for embodied energy and emissions in materials contribute to differences in impact indicator results. In some cases, values for the emissions embodied in materials are meant to be different across studies; this may be, for example, because the assumed energy mix in production is different (see, e.g., [17, 52]). In other cases, discrepancies may be due to different LCA databases being utilized, or arise because, in the face of uncertainty about the exact types of materials that go into the components, analysts make different assumptions about material types (e.g., steel alloys).

Table 3. Authors' overall judgments regarding research coverage of life cycle phases in existing studies (the number of asterisks indicates the degree to which we find studies include the life cycle phase in their scope), the degree to which research results are in agreement (low number of asterisks indicates research do not agree and that the reasons for the disagreements are hard to establish; more asterisks indicates higher level of agreement or

that reasons for disagreements are well understood), and quality of knowledge (the number of asterisks indicates the degree to which we judge current knowledge to be sound and transparent). The latter indicator (quality of knowledge) depends on the research coverage and agreement, but also our (qualitative) evaluations of level of uncertainty and transparency.

Life cycle phase	**Coverage**	**Agreement**	**Quality**	**Remarks**
Production of components	******	****	****	Complete coverage (Section 4). Uncertainty about emissions embodied in materials. Detailed material compositions are often not known. Toxic emissions from manufacturing are poorly understood; issues of mineral resource pressures are not well understood (Section 6.2). Studies assuming European energy systems dominate. Few studies of very large wind turbines and offshore wind turbines in deep waters and/or far from shore (Section 4).
Transportation to site, on-site construction	****	***	***	Coverage is variable (Section 4). Onshore: not important according to most studies (results of [34] disagree; Section 5.1). Offshore: possibly important; modeling appears simplistic; NO_x from fuel oil-burning may be significant. Few studies of wind turbines in deep waters and/or far from shore (Section 4).
Operation and maintenance	****	***	***	Coverage is variable (Section 4). Offshore transportation and on-site activities: modeling appears simplistic; NO_x from fuel oil-burning may be significant. Empirical basis for assumptions about replacement of parts seems to be lacking. Few studies of wind turbines in deep waters and/or far from shore (Section 4).
End-of-life	***	****	**	Scarcely assessed in detail (Section 4). Future waste handling practices for rotor blades are unknown. Assessments using the avoided burden method are often lacking in transparency and may be inconsistent.

A general problem is that detailed material compositions of components are typically not available (by detailed we mean specifications of exact material type, e.g. steel alloy), and furthermore, that LCA databases provide life cycle inventories for only a limited selection of generic materials. This creates uncertainty, which we illustrate here using a simple calculation exercise: In one study [40], ferrous metal content in the wind turbine (800 kW onshore; foundation is excluded here) is comprised by 7% cast iron, 78% low-alloy steel and 15% high-alloy (chromium) steel; while in a second study [31], the corresponding shares are 16% cast iron and 84% reinforcing steel (2 MW onshore wind turbine). Both studies utilize the Ecoinvent LCA database to model materials manufacturing; we find the relevant GHG emission intensities in Ecoinvent are 1.48 kg CO_2e/kg (cast iron), 1.45 kg CO_2e/kg (reinforcing steel), 1.72 kg CO_2e/kg (low-alloy steel), and 4.50 kg CO_2e/kg (chromium steel) [41]. Hypothetically, assuming a 2 MW wind turbine contains 200 tonnes ferrous metals, has a lifetime of 20 years and capacity factor 25%, these values translate to either 4.9 g CO_2e/kWh (if adopting the ferrous metal shares of [40]) or 3.3 g CO_2e/kWh (if adopting the shares of [31]) caused by the production of ferrous metals for the wind turbine. This exemplifies how modeling choices concerning material types – choices that are often not justified and scarcely discussed in LCA studies – may significantly influence total impact indicator values. Another potentially important, poorly understood factor is the composite materials used in the rotor blades and nacelle.

6.3.2 Transportation, on-site construction, and operation and maintenance

The overall picture emerging from the current LCA literature is that emissions associated with transportation and on-site construction are small or negligible (cf. Section 5.1). While this conclusion appears to be fairly well documented with respect to the energy use and GHG emissions for onshore wind farms, one could question to what extent it is valid also for offshore projects (for which installation is more complicated than onshore), and perhaps especially for NOx emissions (largely as a result of NOx, transportation and construction activities are dominant contributors to marine eutrophication and photochemical oxidant formation impact indicator values for the offshore wind farm modeled by [18]). The same argument may apply to transportation and construction activities associated with maintenance. To our understanding, existing LCAs of offshore wind farms rely on rather simplistic and theoretical calculations for modeling on-site operations, and consistency with real-world conditions has not yet been demonstrated.

LCA studies either neglect replacement of parts (e.g., [37, 40]) or variably assume that certain shares of components must be replaced (e.g., [27] assumes 50% gearbox replacement during lifetime, [36] one blade and 15% generator replacement, and [35] 5% complete wind turbine replacement). One study develops a high-maintenance scenario in which 1 generator, 1 gearbox and 1 set of blades requires replacement [32]. While assumptions are not uniform across studies, one can discern that gearboxes, generators and rotor blades are expected to be most susceptible to failure and replacement. An empirical basis for assumptions about replacement seems to be lacking, however (a similar point is made by [32]). One central question is how the assumed replacement rates relate to past experiences from operational wind farms [76]; another question is how to extrapolate information from past experiences to modern wind turbines and more immature application areas (e.g., wind farms in marine environments [77]). In our judgment, these questions are not adequately addressed in the LCA literature.

6.3.3 End-of-life

Since LCAs typically assume the bulk of materials contained in wind power systems will either remain in situ or be recycled to be returned to usage as raw materials, waste disposal is generally not an important contributor to emissions. Excluding ‘new’ lifecycles that are created when materials are recycled is common practice in LCA (cf. the cut-off allocation principle in open-loop recycling; see, e.g., [78]).

There is considerable uncertainty surrounding the fate of fiber-reinforced plastic materials used in the rotor blades: Unlike the well-established processes of recycling basic metals, recycling fiber-reinforced plastic composites represents a technological challenge, and little practical experience exists [79, 80]. While there is a consensus that the traditional practice of landfilling reinforced plastics is unsatisfactory, and regulatory measures to phase out landfilling of these materials are coming into place [80, 81], which waste treatment strategies that are viable and should be chosen remains an open question [79]. There are concerns about toxic emissions occurring in cutting the blades (which may be needed to ease transport) [81], from waste treatment if the materials are landfilled [24], and from flue gas and ashes if the materials are incinerated [79, 80]. Future LCA research may have to address waste handling of rotor blades in order to ensure environmentally sound end-of-life phase for wind turbines.

A significant number of studies credit the system with perceived emissions reductions from end-of-life recycling (avoided burden method; Table 1; Sections 4 and 5.1). However, applications of the avoided burden method sometimes use inappropriate methodologies and are generally lacking in transparency. The root of the problems appears to be that it is not widely recognized that the two issues of 1) including recycled content as input materials in the production phase and 2) crediting the system with prevented environmental burdens at the end-of-life cannot be viewed independently. The share of secondary inputs in the production phase should always be zero for the materials for which avoided burden is calculated; otherwise one would use one perspective to model benefits of recycling in the production phase, and a different (and inconsistent) perspective to model benefits of recycling at the end-of-life – effectively, one would double-count benefits of recycling. The crux of the issue is that analysts must decide whether benefits of recycling should belong to systems that use recycled materials (as is the implicit assumption if secondary materials are used as inputs in an LCA) or make available recyclable materials (as is the assumption if avoided burden method is applied), and not mix these two perspectives.

We are aware of one study [35] that uses the avoided burden method appropriately, assuming no secondary resources as inputs in production when the avoided burden method is applied. (Another study [21] in which the avoided burden method is used also assumes only virgin resource inputs in production, but the stated reason is lack of data on recycled content, and the assumption is inappropriately described as "very conservative".) One apparently inconsistent assessment is [31], where materials containing significant amounts of recycled content (i.e., cast iron, reinforcing steel and copper Ecoinvent processes [82]) are stated to be used in the production phase, while simultaneously, recycling credits are given for avoided production. Other LCAs use the avoided burden method while not specifying that only virgin resources are used in production.

6.4 Method for life cycle inventory and system boundary issues

In 2002, Lenzen and Munksgaard [5] recommended that future wind power LCA research employs hybrid LCA methodologies "in order to achieve system completeness while dispensing with the problem of selecting of a boundary for the production system". However, the current survey demonstrates that hybrid LCA studies on wind power are still relatively scarce – this fits into a general trend that despite its acknowledged advantages, hybrid techniques have not yet become standard practice in LCA [10]. Hybrid LCA is more challenging to conduct and requires additional data, which may be an explanation for its lack of use. Moreover, it is interesting to note that Wiedmann et al. [23] employ two hybrid LCA calculation techniques separately, and find that while the total emission estimates obtained by the two techniques are comparable, there are considerable differences in the relative contribution from IO sectors. This points to yet unresolved issues with IO-based calculations techniques.

Notwithstanding the data and methodological challenges of hybrid methods, hybrid LCA is the only technique that offers both process-level detail and a nearly complete coverage of the entire product system. While there is no consensus in the LCA community on how to measure the truncation bias of process-LCA, in all explorations into this issue surveyed by Majeau-Bettez et al. [10] it is found that process-LCA fails to account for 30% or more of total indicator values. This predicates that the employment of hybrid LCA methodologies should be a goal of future LCA research on wind power; and that if hybrid techniques on the other hand are not applied, the problem of cut-off errors should at the least be recognized – in existing literature this is not the case.

6.5 Aspects of scale, temporal evolutions and network integration

In recent years, analysts have remarked on the insufficiency of static, unit-based analyses for evaluating implications of future wind energy developments [18, 25, 39, 44]. One shortcoming of existing research is the general failure to address the magnitudes of aggregated impacts: A transition away from conventional and towards lower-carbon energy systems in coming decades – as envisaged for example by contemporary climate change mitigation scenarios [3, 4] – will in itself cause harmful emissions. Due to the sheer scale of the transition, total emissions and resource use brought about by 'clean' energy technologies may be significant in the aggregate, even if unit-based assessments (i.e., assessments where indicator values are measured per kWh) indicate low impacts. In the literature, climate change mitigation scenario analyses explore energy transitions at the economy-wide level [3, 4], but do not consider emissions arising from building and operating non-fossil power plants; while conversely, LCAs of power generation predominantly have a purely micro-level focus. The integration of these two perspectives could potentially provide valuable new insights on the economy-wide effects of large-scale energy transitions (a similar point is made by [8]). Ideally, such scenario calculations incorporate some projections of future technological changes, as discussed below.

Inventories for wind power systems are not static, but change over time as new technological configurations are adopted, and due to economies of scale and changes in background economies. Projections of impacts of research and scientific developments on future technological designs – based on technology forecasting studies or learning curve studies [38, 83] – may provide LCA analysts with a basis for modeling future inventory changes, as demonstrated by Viebahn et al. [84] for concentrated solar power. Besides changes in wind power technology configurations, impact indicator values are influenced by the characteristics of background economies through relatively clean or dirty manufacturing; indeed, it is the current economies' preoccupation with fossil fuels which is the very reason why electricity from wind is not CO_2-free. The importance of background energy system characteristics is illustrated by the results of [52], where the embodied CO_2 is a factor of five lower for a wind turbine produced in Brazil compared to Germany; the difference stems entirely from the higher portion of renewable sources (hydro, biomass) in Brazil's energy supply. It is not just the energy mix as such which is important, however, but also the energy efficiency. Another important factor are the environmental impacts of metals supply which will change due to combined effects of technological advances in mining and manufacturing, changes in the portion of secondary to primary materials used, and reduction in ore grade [85-87]. Future research may address the effects of such changes through scenario analyses.

The final type of scaling or temporal aspect discussed here relates to the variable and (partly) unpredictable nature of wind power. Higher shares of intermittent electricity supply, such as electricity from wind, increase the overall costs of short-term balancing in the system (i.e., matching electricity supply with demand over seconds to days), reduce overall peak-load system adequacy (because the contribution of a wind power plant to peak-load capacity adequacy is smaller than for conventional technologies), and may require upgrades in the electricity transmission infrastructure to admit transfer of electric power to the load centers (see, e.g., Section 7.5.4 in [1] and p. 321-326 in [88]). In the literature, life cycle emissions of wind power and emission penalties due to the variability of wind power [89-91] are generally analyzed separately and lead to separate evaluations of emissions connected with wind power deployment; in a sense, these two areas of research form two independent departures from the notion that wind power is 'emissions-free', both aiming to provide a more complete picture. The potential exists to combine the assessments in these two research fields, as exemplified by the study by Pehnt et al. [39] discussed in Section 5.3 – this would indeed be congruent with the often stated goal of LCA to provide holistic assessments, but on the other hand it involves substantial methodological and data challenges. In any case, when interpreting results of current LCA studies it is important to bear in mind the failure of LCA research to account for emission penalties due to intermittency.

6.6 Comparison with competing technologies

A detailed exploration of how life cycle emissions for wind power compare with that of other power-generation technologies falls outside the scope of this paper, although a few points are noted here. In the LCA survey presented in Sathaye et al. [8], the interquartile range (i.e., the range between 25th and 75th percentile levels) for life cycle GHG emissions for wind power are 8-20 g CO_2e/kWh (median value 12 g CO_2e/kWh). The corresponding ranges (median values) for competing technologies are 8-45 (16) g/kWh for nuclear, 3-7 (4) g/kWh for hydro, 14-32 (22) g/kWh for concentrating solar, and 29-80 (46) g/kWh for solar photovoltaic power. Life cycle GHG emissions of electricity from coal and natural gas with carbon capture and storage (CCS) are estimated to 180-220 g CO_2e/kWh and 140-160 g/kWh, respectively, in [92]; the corresponding numbers without CCS are around 1000 g/kWh for coal and 500-600 g/kWh for natural gas [8, 92]. Judging from these figures, the carbon footprint of wind power is significantly lower than that of fossil-based power with CCS, and is comparable or lower than that of other important non-fossil power generation technologies. Likewise, comparisons of life cycle emissions of NOx, SO_2, NMVOC and particles of multiple power generation technologies in Sathaye et al. [8] suggest good environmental performance for wind power.

Some research suggests that toxicity impacts may be of relatively high importance. Two studies of offshore wind power find, respectively, that a wind farm scores 2-6 times worse in toxicity impact categories than a natural gas combined cycle plant [35], and that wind electricity is slightly worse than the average German electricity with respect to human toxicity [22]. A different picture is presented by other publications, whose findings suggest that wind power grossly outperforms European [46] and Spanish [31] average electricity mixes with respect to human toxicity.

7 Final remarks and recommendations

Despite the considerable variability in results, and the limitations of current knowledge that have been mentioned, we conclude that existing LCA research provides many insights into and gives a fairly good overall understanding of the life cycle environmental impacts of wind power in terms of cumulative fossil energy demand and associated pollution. Discrepancies between studies can likely be explained by a combination of actual differences in the systems studied (e.g., small versus large wind turbines), key assumptions (e.g., capacity factor and lifetime), data inconsistencies (e.g., emission intensities of materials), and differences in methodologies and approaches (e.g., process-LCA or hybrid IO-LCA, accounting of recycling benefits). Previous LCA reviews [1, 5, 7] have duly noted that the large gap between low and high values limit the usefulness of results to decision-makers, and that compliance with some standardized sets of methods and assumptions in future analyses would be advantageous.

The problems of confusion and uncertainty due to variability in results, and incomprehensibility due to the complex networks of operations that are studied and many assumptions that are made, need to be given due attention. One measure that can be taken to alleviate these problems – in conformity with the guiding principle that LCAs should be transparent [9] – is to make process-level inventory input data available together with LCA publications: Such a step would increase the transparency as to how results are obtained and help give clarity on why results differ across studies, and allow for proper meta-analyses of wind power LCAs [93]. Furthermore, making inventory input data at the level of unit processes available can contribute to a cumulative build-up of knowledge, rather than having efforts going into repetitions of sometimes cumbersome data collection processes.

This review has shown that to date, the largest research efforts have been devoted to studying typical onshore wind turbines or wind farms in European locations, placing most emphasis on the production life cycle stage. Future research may focus attention on system types or life cycle phases for which research is still relatively scarce or robust assessments are lacking. This may include:

- Systems that are produced and operated under conditions of other regions than Europe.
- Large wind turbines (> 3 MW) and offshore systems in deep waters and/or far from shore.
- Installation and operation and maintenance phases, in particular for offshore systems.

Wind power LCAs have traditionally had their domain in assessing potential environmental impacts caused by one small reference unit (1 kWh of electricity), have primarily focused on fossil energy-related emissions, and have predominantly employed a process-LCA methodology. Such assessments have proved valuable in the past and are likely to continue to play a role in future research. At the same time, given the sizeable number of published studies that are similar with regards to goal and scope, one could wish that research had made further strides in analyses with different or broader scopes, or more sophisticated methodologies. In this respect, we call for future research efforts to be directed into:

- The employment of hybrid LCA methodologies.
- Broadening the scope with regards to environmental impacts, as far as available impact assessment methods allow it. In particular, we call for more detailed explorations of toxicity and mineral resource depletion.
- Exploring technology evolution through scenario analyses, addressing for example the scale of environmental burdens at regional or global levels, changes in life cycle inventories as key technologies or background economies change, or emission penalties due to intermittency.

In all cases, future studies should avoid inconsistent modeling of recycling benefits.

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