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# Highlighting the need to embed circular economy in low carbon infrastructure decommissioning: The case of offshore wind

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ABSTRACT

Development and deployment of low carbon infrastructure (LCI) is essential in a period of accelerated climate change. The deployment of LCI is, however, not taking place with any obvious long term or joined up thinking in respect of life-cycle material extraction, usage and recovery across technologies or otherwise. This proposition is demonstrated through empirical quantification of selected infrastructure and a review of decommissioning plans, as exemplified by offshore wind in the United Kingdom. There is wide acknowledgement that offshore wind and other LCI are dependant on the production and use of many composite and critical materials that can and regularly do inflict high impacts on the environment and society during their extraction and manufacturing. To optimise resource use from a whole system perspective, it is thus essential that the components of LCI and the materials they share and are comprised of, are designed with a circular economy in mind. As such, LCI must be designed for durability, reuse and remanufacturing, rather than committing them to sub-optimal waste management and energy recovery pathways. Beyond a promise to remove installed components, end-of-life decommissioning plans do not however provide any insight into a given operators' awareness of the nuances of their proposed material management methods or indeed current or future management capacities. Decommissioning plans for offshore wind are at best formulaic and at worst perfunctory and provide no value to the growing movement toward a circular economy. At this time, millions of tonnes of composites, precious and rare earth materials are being extracted, processed and deployed in infrastructure with nothing in place that suggests that these materials can be sustainably recovered, managed and returned to productive use at the potential scales required to meet accelerating LCI deployment. Academic and industry literature, or lack thereof, suggest that this statement is largely reflected throughout LCI deployment and not just within the deployment of offshore wind in the UK.



## 1. Introduction and background

Low Carbon Infrastructure (LCI) can be defined as the physical structures and components of any system that facilitates the generation, supply and distribution of renewable energy and/or use of energy efficient technologies. Examples of LCI range from solar photovoltaic panels (PV) and wind turbines, through to energy storage (including batteries and heat networks), fuel cells (including hydrogen production), electrified transport (including electric cars and trains), and each of their respective production, distribution, fuelling and charging networks.

Largely in response to growing climate change concerns and the consequent need to decarbonise power production and wider societal activities, the installation and use of LCI has grown significantly and continues to grow on a global scale. For example, in 2018 globally installed solar PV capacity stood at 488 GW with on/offshore wind energy capacity collectively standing at 564 GW (BP, 2019). Over the previous decade (i.e. 2008–2017), the annual average growth in installed capacity of PV and on/offshore wind was 47% and 19% respectively (BP, 2019). In 2018 alone, more than 49 GW of wind energy capacity and more than 94 GW of solar PV capacity was installed (IRENA, 2019). Beyond power production, the International Energy Agency report that in 2017 more than three million electric vehicles (EV) were in use on public roads, with this figure markedly growing by 65% to 5.1 million in 2018 (IEA, 2019). Though currently representing only 2% of all globally registered vehicles, this figure is expected to grow rapidly with numerous major global cities promising bans on the use of petrol and

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diesel vehicles over the next decade (Deloitte, 2019). The growth in capacity of other sources of renewable energy, such as hydro, geothermal, bioenergy and tidal power, has not been as significant over recent years; however, their installed capacity has also continued to grow on a global scale (CF: IRENA, 2019). The COP21 Paris Agreement commitment to limit average global temperature rise in the 21st century to well below 2 °C (UNFCCC, 2019), largely through reducing greenhouse gas production via continued adoption of renewable energy and energy efficient technologies, suggests that the use and accelerated deployment of LCI will continue.

Facilitating LCI growth will increase demand on a myriad of environmentally impactful rare earth, precious and other highly processed materials (Stamford and Azapagic, 2012; EC, 2017; POST, 2019a; Watari et al., 2019). By definition of being components of a low carbon economy, it is essential that these increasingly in-demand materials are extracted, deployed and indeed managed and reintegrated into society at their respective End-of-Life (EoL) in the most socially, environmentally and materially efficient manner possible (e.g., see Gislev and Grohol, 2018). This, in effect, is the objective and increasing practice of Circular Economy (CE) (e.g. Velenturf et al., 2019a). The long-standing 'Take-Make-Use-Dispose' mode of production and societal behaviour is increasingly unacceptable. Insightful environmental product declarations from LCI manufacturers, dedicated life-cycle impact assessments (LCA), and discussions on recycling of all manner of LCI, all acknowledge this failing model of resource use within their rationale (e.g., Vestas, 2006; Cherrington et al., 2012; Latunussa et al., 2016; Komoto and Lee, 2018; Tesla, 2018; Siemens Gamesa, 2019). CE, however, goes well beyond understanding both direct and consequential LCA impacts or the recycling of materials, and is more technically and ethically nuanced than observing minimum waste management protocol.

CE brings together and builds on several existing and nascent innovative resource efficiency, resource productivity and other strategies for clean equitable growth and sustainable development (e.g., dematerialisation, design for environment, industrial ecology, environmental justice) (e.g., Geissdoerfer et al., 2017; Korhonen et al., 2018; Suarez-Eiroa et al., 2019; Johansson and Henriksson, 2020). Importantly, CE is a systems-based approach to resource management that considers not only the product and its components in question, but the infrastructure and products it shares materials with, and how their continued intra and inter-active use, reuse or disposal, can repeatedly create value for society, the environment and economy (e.g. Busch et al., 2017, Purnell, 2017; O'Dwyer et al., 2020; Lag-Brotons et al., 2020). Better understanding of the long-term durability, performance, recyclability, and the most energetically and environmentally low impact direct reuse possibilities of LCI and its component materials, within and across LCI technologies, is thus essential for low carbon CE planning. The significant volumetric growth in LCI material use that accompanies its past, current and future operational growth, necessitates that such understanding is developed at the earliest opportunity.

Despite CE being actively promoted as a strategy for sustainable development by influential organisations such as the United Nations and the European Union, and indeed being an operational requirement in several regions (UNIDO, 2017), there is little practical evidence to suggest that the development and deployment of LCI is indeed happening with CE or any other long term resource conservation in mind. Evidence from the literature and author engagement with industry, through personal communication and research agenda co-creation workshops (e.g., Velenturf et al., 2017; Purnell et al., 2018), would suggest that resource recovery strategies and recycling of LCI components is outwardly being undertaken as an afterthought, or problem to be dealt with in the future with little forethought for issues relating to waste management capacity or, from a strategic development perspective, cross-technology resource security (i.e. a perception that LCI "*decommissioning is far away*" and not a current issue). This is concerning because such an attitude to LCI development arguably reflects the past myopic deployment of nuclear, oil, coal and gas infrastructure that left current generations with large clean-up bills that impact on public finances and the environment (Invernizzi et al., 2019, 2020; NAO, 2019; Vaughan, 2019). There is no evidence that lessons have been learned from such failures within the emergence of LCI deployed for the purpose of sustainable development. Indeed, despite the presence of producer/operator responsibility frameworks, recent studies on decommissioning of a variety of LCI, within several regions, provides evidence that the shifting of the financial and environmental clean-up burden to the public purse and wider environment could be repeated (e.g., within offshore wind in the UK, Topham and McMillan, 2017; Velenturf et al., 2017; Purnell et al., 2018; for PV and storage batteries in Europe and Australia, Sica et al., 2018; Salim et al., 2019; and for specific LCI components, such as wind turbine blades in the United States, Martin, 2019; and, without development of a CE, more globally, Jensen and Skelton, 2018).

As will be demonstrated, discussions on the scale of waste that will be generated by LCI are not new, neither are studies on the demand for resources required to facilitate growth in a range of LCI technologies. In general, these discussions are based on modelling and projections derived from best evidence available at that time; the value of this body of work is not questioned. The originality of this article, however, lies in its empirical grounding: the assessment of material stocks and probable resource flow draws on evidence and long term market and technology data from the rapidly maturing offshore wind (OSW) industry in the United Kingdom. Uniquely, it places the necessary management of these material stocks in the context of the mandatory EoL management plans of wind farm operators, for which an in-depth critique in respect of embedding CE is provided. Based on an appraisal of key LCI materials, namely copper, rare earth elements (REE) and man-made composites, this article duly continues by further discussing general LCI development and the extent of its deployment and management at its EoL (Section 2). The appraisal and scale of the issue at hand is then refined through the exemplar lens of OSW in the United Kingdom (UK), an LCI for which the UK sees itself as a world leader in all aspects of its development and deployment (Section 3). Assessing strategies for OSW EoL management documented within pertinent literature and decommissioning plans, the article highlights limitations to proposed material recovery and disposal methods before proposing new approaches for integrating CE within OSW deployment (Section 4). The article concludes by providing areas for further research and recommendations for integrating the myriad resource conservation aims of CE into wider LCI development (Section 5).

## 2. The materials of low carbon infrastructure

A search of Scopus indicates that academic literature on a CE for LCI is minimal and at best nascent, with only two articles specifically framing their work in the context of LCI, cross technology material flow and CE. Focussing on the shared use of permanent magnets in EVs and wind turbines, Busch et al. (2014, 2017) specifically discuss resource demands across LCI and, notably, the potential for a LCI CE to significantly reduce demand for the extraction of raw materials if components are designed for reuse and remanufacturing alongside the timely establishment of recovery and recycling infrastructure. This work was in part based on historical LCI deployment data and in part based on future projections of LCI growth scenarios. Narrowing the perspective to CE and low carbon electricity, Boubault and Maizi's (2019) study on the impacts of

LCI material demand was loosely based on aspirational deployment by projecting the cumulative demand for materials across the LCI system. From a more specific technology perspective, several governmental and academic studies have been conducted on the material demands of respective nations LCI, particularly with regards to wind energy and its use of critical and other rare earth elements (e.g., in the UK, Griffiths and Easton, 2011 and AMEC, 2014; across Europe; Lacal-Arántegui, 2015; in the USA, Wilburn, 2011; Imholte et al., 2018; and in "Fairytale Country", Cao et al., 2019). Though not within the context of a CE, competition for these resources across a range of technologies, particularly EV, PV and energy storage, has also been addressed in an absolute supply sense (e.g., in the USA, USDoE, 2011; in Europe, Janssen et al., 2012; Speirs et al., 2013; Viebahn et al., 2015).These studies all highlight the challenges faced by a growth in LCI regards ensuring continued access to materials, including in respect of their cross-technology material usage. However, most studies, CE focussed or otherwise, also acknowledge the difficulties involved in estimating the amounts of material that are already 'locked-up' within existing LCI and when and how these materials can be can be recovered in a sustainable and resource conserving manner. In respect of future EoL planning within a CE, this lack of robust data could be deemed problematic.

For wind power, however, significant levels of information exist on the specifics of developments, particularly the specific turbine used, their exact location and their probable time of repowering or removal (i.e., 20 – 25 years from commissioning). As such, estimations of material stocks and flows can be accurately made. The availability of this information is largely due to the number of affected stakeholders and level of planning control involved in the development of a typical wind farm. From such planning records, it is possible to compile a relatively accurate picture of specific material use to-date and probable material demand in the immediate future (see Section 3), which is not necessarily the case for LCI such as PV and EV that is deployed in a less centralised and highly dispersed manner. The potential for producing such an inventory, particularly for OSW, provides a good example for impressing the need for forward thinking in respect of incorporating CE into LCI deployment. It is not possible within the scope of this article, however, to cover the myriad of issues that follow the global extraction, manufacturing and use of the materials employed within OSW in respect of embedding the resource conservation ethos of CE into LCI planning. Within this article a focus has thus been deliberately placed on three key materials that are critical not only to the development of OSW, but also across other forms of LCI (particularly EVs, solar PV and Energy Storage). By focussing on a select range of materials, namely Rare Earth Elements (REE), copper and composites (i.e., largely reinforced plastics), with high cross technology demand, it is possible to gain greater insight into why more emphasis on design for durability, recovery and reuse within a CE is required at LCI development and deployment. Herein, the production, use and recyclability of composites, REE and copper are summarised.

### 2.1. Composites

At its most basic, a composite can be defined as the combination of two (or more) materials with complimentary properties to produce a new material. This new material can be designed and analysed as a single material in its own right - in contrast to the likes of reinforced concrete where the concrete and steel are still considered separate components - but the components remain, in principle, separable (Purnell, 2017). A composite is more than the sum of its parts and is generally produced for strength, durability and other desirable performance characteristics that are superior to their component materials. Given these enhanced characteristics, composites have become prominent constituent parts of a variety of modern products where such attributes as light weight strength are critical to performance (e.g., within the automotive, aerospace and medical industries). Fibre Reinforced Polymers (FRP) are one of the most prominent composite materials found within modern products.

The environmental impact of producing FRPs can be significant, with the prominent production impact emanating from the significant levels of energy expended in the production of the polymers (Hammond and Jones, 2011) and, to a lesser but still notable extent, the melting of glass and pyrolysis of carbon fibres (Duflou et al., 2012). Emphasising this point, it was estimated that the embodied energy just within UK produced composite production waste, *not end-products*, would equate to more than 5 TWh by 2015 (Shuaib et al., 2015).

Within LCI, composites are employed in numerous areas, particularly within the Wind Turbine Generators (WTGs) of on/offshore wind. With FRPs being relatively light and strong, they are an especially ideal material for producing the modern blades of almost all WTG rotors. Indeed, though some early onshore wind blades were made from aluminium and even timber, WTG blades have been predominately made from glass-FRPs and more recently, for the bigger offshore turbines, carbon-FRPs. The evolving choice and use of composite for WTG blades has been largely dictated by stiffness to weight ratio, i.e. the need to keep weight down whilst retaining strength and rigidity as blades have increasingly grown in length.

In respect of EoL management and CE, the material characteristics that make composites so suited to the production of blades that can increasingly exceed 80 m in length and 30–35 t (see Section 3), are the same characteristics which are creating an impending issue for decommissioners, i.e., how to recycle physically bulky and logistically awkward materials that are designed to be strong, resistant to degradation within harsh environments, and made of multiple intimately joined materials with low specific cash value (in contrast to copper or steel).

Despite several decades of onshore wind development and decommissioning which has already produced thousands of tonnes of waste (WindEurope, 2017; Veolia, 2020), there continues to be little in the way of the development of environmentally congruent management methods that could, currently, be transposed to future OSW EoL management. Indeed, aside from some innovative architectural uses (e.g., as noise barriers, play frames, public art: see, *Re*-Wind, 2020), onshore blade management has and continues to primarily involve shredding prior to environmentally sub-optimal incineration or, worse, dumping in landfill (e.g. Shuaib et al., 2015; Jensen, 2018). There are claims that as blade waste has grown, fibre-reinforced composites recycling has progressed (Wind Europe, 2017). However, there is little evidence of such innovation or emergence of sustainable blade recycling on an industrial scale, with industry openly possessing concerns over the scale of blade waste that will increasingly appear with *no established recycling solution* beyond the existing options of incineration or use as fuel and raw material within cement making (e.g., Veolia, 2018; Nagle et al., 2020). Innovations that are being explored (see, e.g. Jensen and Skelton, 2018), particularly in respect of pyrolysis aimed at producing valuable chemicals (Port Esbjerg, 2020), and chemical decomposition of blades through processes such as solvolysis, have so far not proven to be economically viable or produce suitably reusable fibres (Leahy, 2019).

Moreover, as a long-term solution, the wider sustainability of such resource and energy intensive management options is as questionable as incineration of any waste. Indeed, it has to be remembered that composites are pervasive in modern society and have been a long term issue for waste managers globally, hence suggestions that increasing blade waste and consequent economies

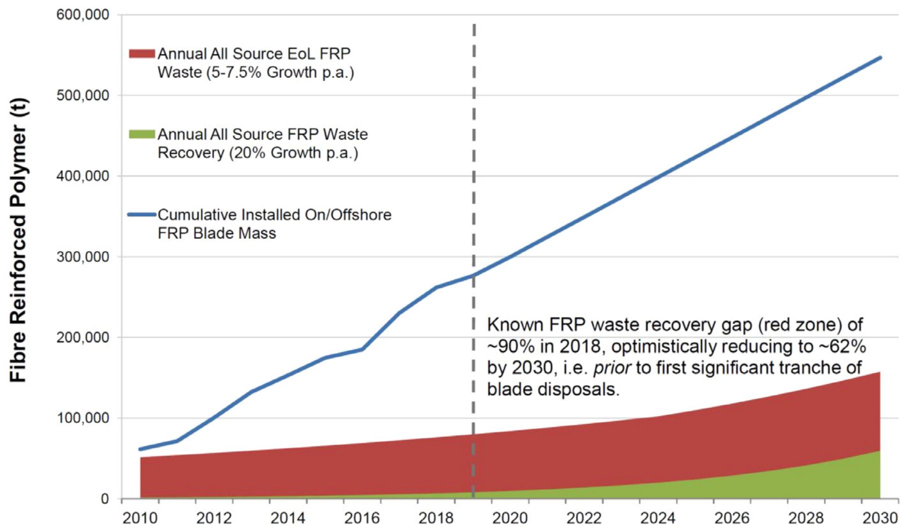


**Fig. 1.** In-use FRP blade mass and all source EoL FRP recycling capacity gap in the UK. *Note:* blade mass (blue series) represents the known mass of all on/offshore blades in-use in the UK as of 2019 (see Section 3), with projected growth to 2030 based on a UK Government commitment to 30 GW of offshore wind capacity and Norris's (2019) prediction of a minimum addition of 4.5 GW of onshore wind capacity. Annual FRP production and EoL waste growth from all sources (red area) is based on the ~80,000t produced in 2018–2019 (i.e., Brown et al., 2018), with projected 5–7.5% FRP market growth to 2030 shown to the right of the hatched line.. FRP waste recovery (green area) shows the 10% of FRP that was diverted from landfill in 2018–2019 (Tyrrell, 2019), with optimistic 20% growth in recovery management capacity to 2030 shown to the right of the hatched line.

of scale will promote rapid industrial scale EoL innovation, do not reflect any current reality. Pertinently, given the nuanced challenges of managing composite EoL WTG blades, CE focussed calls have already been made on the need for sustainable materials to be used in blade production, rather than attempting to manage waste at EoL (Jensen and Skelton, 2018). Such arguments are emphasised by the UK's current and projected figures for blade mass shown in Fig. 1, compared to known rates of EoL blade diversions from landfill in 2018–2019 where only 10% of *all* source FRP EoL waste was diverted from landfill (Brown et al., 2018; Tyrrell, 2019). Based on an annual 5–7.5% growth in the UK FRP market, including increased production of blades in the UK, and an *optimistic* growth in waste recovery capacity of 20% per annum, a 67% recovery capacity gap would still exist in 2030 when the country's first tranche of EoL blades will require management (i.e., ~60 kt deployed before 2010). Importantly, the above narrative and evidence from the literature suggests that such a recovery management gap for composites will be present in numerous regions, not just the UK.

### 2.2. Copper

Though a common and highly visible element within modern society, from an economic and development perspective copper (Cu) is increasingly recognised as a critical metal. Currently, most copper is mined in South America, particularly Chile. As of 2015, global consumption of Cu was estimated to be 20 million tonne per year, with consumption expected to increase for the foreseeable future (e.g. Hammarstrom et al., 2019). Despite new mines being constructed and to be opened over the next five years, demand for the metal will continue to outstrip supply (Lombrana and Farchy, 2019). Geologically, Cu is deemed to be one of the "*scarcest industrial commodities*", and the amount of Cu produced from each tonne of mined ore almost halved between 2010 and 2016 (Livsey, 2017). Though there are discussions around the changing economics of continued ore extraction and its consequent availability (e.g., Rötzer and Schmidt, 2018), it is clear that freely available Cu ore will be ever lower grade, i.e. potentially exasperating the already significant impacts of Cu extraction and processing (Martínez et al., 2009; Rötzer and Schmidt, 2018).

Indeed, Cu extraction and processing, of any grade, has a significant impact on the environment. Operational mining standards vary between regions, nevertheless extraction, depending on methods employed, regularly results in habitat destruction, wider land degradation and pollution of water sources, whilst processing requires significant inputs of energy and water (see Rötzer and Schmidt, 2018; Sonter et al., 2018; Chen et al., 2019; Greenspec, 2020). For example, though variations exist between processes and types of ore, LCA studies have estimated that between 33 MJ and 64 MJ of energy per processed kilogram of copper sulphide ore is required with likewise significant quantities of water required at all stages of processing (Northey et al., 2013). Notably, energy processing demand is impacted by reduced ore grades (Norgate and Rankin, 2000; Northey et al., 2013).

Cu is found within many areas of the operational infrastructure of low carbon technologies. For example, within wind turbines Cu is a prominent material within several components housed within the nacelle, including the electrical equipment of the control system and extensively within the primary windings of both conventional geared and direct drive generators. Cu is, however, most conspicuously found within the internal, inter-array and export cables of OSW farms. Notably, for OSW farms increasingly installed further from shore, the reduced comparative cost and weight of aluminium compared to Cu has led to its increased use an export cable core (though needing more overall material due to reduced conductivity, i.e. larger core and consequent insulating material). For the vast majority of existing installations, however, several thousand kilometres of Cu core has been deployed within export cables and Cu is used almost exclusively within inter-array cables (see Section 3.2).

In respect of EoL management of Cu, unlike composites, there is a strong market and well-established recycling methods in place in most regions for those Cu wastes that can be easily recovered, i.e. within nacelles. Indeed, the recovery and recycling of many sources of Cu has played a significant role in some of the largest consumer economies, such as the United States and China (e.g., Goonan, 2009; Brininstool and Flanagan, 2017), with recycled Cu accounting for approximately 30% of production in the latter (Chen et al., 2019). Much of this production of Cu from secondary sources, amounting to almost 4Mt of scrap imports in 2015

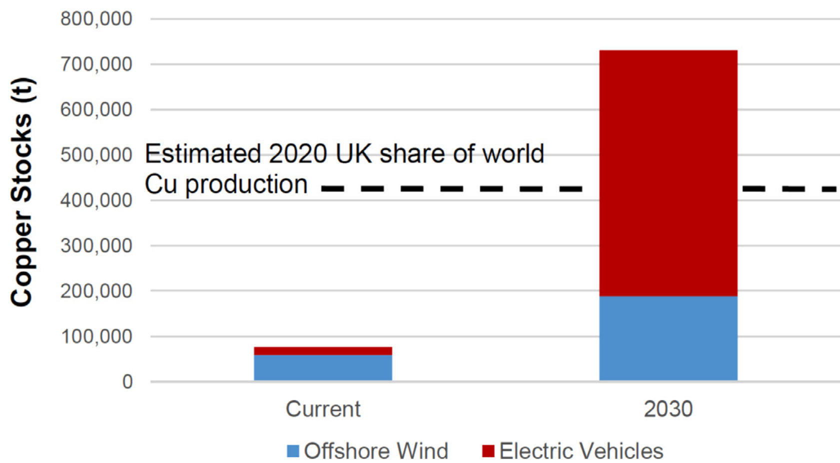


**Fig. 2.** Example of shared UK copper demand for selected LCI technologies. *Note:* based on the example of OSW and EV deployment in the UK, Fig. 2 demonstrates the potential need for circular economy driven resource security to maintain expected growth in LCI technologies. Data and calculations of current and future stocks of Cu in OSW can be found in Section 3. Projections of current and future stocks of Cu in EVs are derived from the CDA (2017), Dft (2018) and Lilly (2020).The hatched line represents an estimation of the UK's *all* sources demand of global Cu production scaled by UK/Global GDP in 2020, i.e. $2.8 Trillion/$142 Trillion.

(Brininstool and Flanagan, 2017), has derived from China's active pursuit of such waste to feed their own growth demands. Thus making the country a significant market stakeholder in a source of Cu whose environmental (re)processing impact can be a fraction of that of primary sources (Chen et al., 2019) – assuming, that is, it can be recovered in an environmentally sympathetic manner (see Section 4). Even with China's recently implemented 'Green Fence' (e.g., Earley, 2013, d'Escury, 2014; The Economist, 2017), aimed at reducing the import of scrap wastes that are substandard and involve highly polluting activities, it still holds a key role in the future of sustainable Cu recycling and reuse capacity. Risks relating to an increasing lack of control over higher grade increasingly costly stocks of Cu, are perhaps placed in greater context when it is recognised that a common practice for EoL cable management, across technologies and industries, is to abandon them. This 'resource management' approach has largely been adopted under the guise of a concern for the environment in respect of the impacts of recovery on land and the seabed and potentially deprives markets of an increasingly in-demand and critical LCI resource (Fig. 2).

### 2.3. Rare earth elements

REEs are used widely in modern society, particularly as process catalysts and as conductors and magnets within a growing range of electrical and electronic devices. Indeed, there is said to have been an "*explosion*" in their use in modern technologies (Balaram, 2019: 1286). Since the 1980s, the production of REEs prominent in LCI, namely neodymium (Nd) and dysprosium (Dy), has been heavily concentrated in China where reserves are greatest (Van Gosen et al., 2014), with minimal production also taking place in the likes of Australia, India and the United States (largely due to Chinese restrictions on production for economic and environmental reasons). Many REEs, including Nd and Dy, are formally recognised as critical materials in respect of economic development importance and ongoing supply and price volatility.

Given their criticality to modern economies, prominent organisations, such as the USGS and EU, have attempted to quantify recoverable reserves and other sources of available critical materials such as REEs; however, estimates are not necessarily reliable due, in part, to the disparate nature of data collection and methods for reporting of reserves (Lusty and Gunn, 2015). Hence, as indicated in the introduction to this section, calls have been made for more robust techniques for estimating availability of these economically critical materials (e.g., Graedel and Nassar, 2013). More pertinently in the context of this article, questions over wider REE deployment and availability emphasise the need to be more protective, from a CE perspective, of known quantities, location and form of critical materials deployed within LCI. Given the environmental and social impact of REE extraction and processing, such questions are important from a wider systems and impact perspective.

Indeed, in respect of assigning an outright environmental impact to their extraction and primary processing, it should be noted that many REEs are extracted as a co-product with other minerals (Elshkaki and Graedel, 2014), e.g. iron ore. As with Cu and many other forms of mining, however, it can be categorically said that REE extraction, as a co-product or otherwise, is energy and water intensive at both extraction and processing. Likewise, REE mining can also be the source of myriad land and wider habitat degradation (e.g., Balaram, 2019). Indeed, it has been recognised that more sustainable extraction technologies are required to meet the growing demand for minerals in a low carbon economy (Lusty and Gunn., 2015). Moreover, these assertions further emphasise the need to be more aware of our existing (cross-technology) use and location of REE sinks within the technosphere.

LCI, particularly OSW, is a clear and obvious REE sink. Most notably, Nd is an increasingly important, in fact critical in the widest sense, material to the development of OSW and EVs who are both reliant on its desirable magnetic qualities for the operation of their electrical motors (Fig. 3). Indeed, within OSW, there is a distinct and almost blanket move toward the use of motors containing REE based magnets, particularly in the form of NdFeB (Neodymium-Iron-Boron) (see Section 3). Given this, concerns over the environmental impact and security of the materials should be high on the agenda of OSW stakeholders, particularly in the face of competition for these materials from the EV industry (USDoE, 2011).

However, methods of recovery for REEs, from OSW or otherwise, has until recently received little study and it is clear that the logistical and technical challenges involved in recovering these materials require further investigation. What has been acknowledged by influential organisations such as the European Commission, however, is that, despite the knowledge of possible supply issues and criticality, recovery and recycling of REE is low (EC, 2014). Beyond LCI, it has been stated that only 12.5% of *all* metals are recovered from all WEEE sources with approximately 50 Mt of WEEE still being sent to landfill on an annual basis (Balaram, 2019). Though innovative NdFeB recovery and reprocessing is being explored (Yang et al., 2017), the lack of wider WEEE processing is reflected in less than 1% of REEs being recovered from EoL mag-

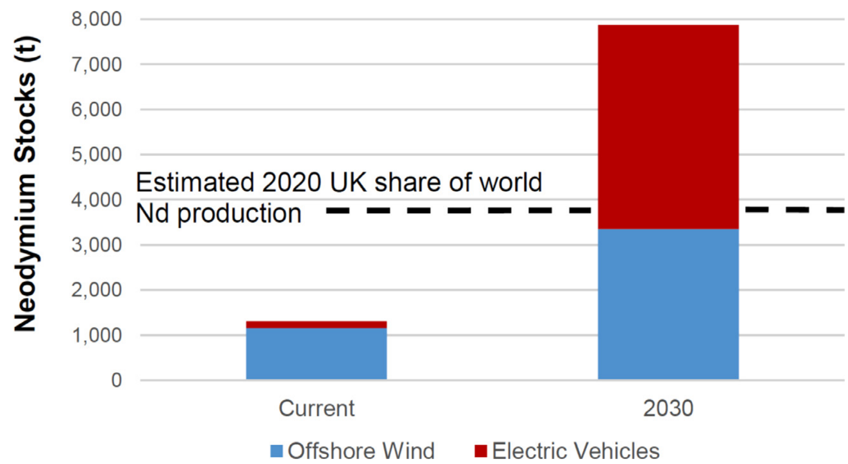


**Fig. 3.** Example of shared UK neodymium demand for selected LCI technologies. *Note:* based on the example of OSW and EV deployment in the UK, Fig. 3 demonstrates the potential need for circular economy driven resource security to maintain expected growth in LCI technologies. Data and calculations for current and future stocks of Nd in OSW are presented in Section 3. Projections of current and future stocks of Nd in EVs are derived from conservative author calculations of EV Nd content and vehicle deployment data (i.e., see DfT, 2018; Lilly 2020).The hatched line represents an estimation of the UK's *all* sources Nd demand based on global REE $Nd_2O_3$ production (see USGS, 2020) and UK/global GDP in 2020, i.e. $2.8 Trillion/$142 Trillion.

nets (e.g., AMEC, 2014). Indeed, due to their brittleness, it is noted that wasteful magnet production methods lead to significant losses of NdFeB into generic waste streams with it not proving, to date, economically viable to retain and reprocess these wastes (Ibid). As such, industrial scale recovery of Nd from scrap industrial magnet resources is not currently taking place, and despite suggestions that its substitution as a material in OSW generators could be a solution to supply issues (e.g., POST, 2019b), this will not be an option any time in the foreseeable future. With no critical material strategy in place within the UK (POST, 2019a) - the supposed OSW market leader - and significant stocks of REE being 'locked-up' in OSW for approximately 25 years and in highly dispersed EVs for approximately a decade, the lack of action on REE recovery will place undesirable greater demands on primary stocks extracted from politically sensitive areas.

## 3. Low carbon infrastructure and offshore wind in the United kingdom

The UK has in the past been behind many of its nearest European neighbours in the adoption of renewable energy technologies and other LCI (Jensen and Gibbs, 2018); nevertheless, it is home to a significant and wide range of LCI:

- Wind: based on author analysis of public records, the UK and its territorial waters are currently home to close to 10,000 wind turbines (see RUK, 2019a). Approximately three quarters of this figure, 7476, represents fully commissioned and operational onshore turbines and their associated distribution infrastructure, including more than 21,000 largely composite blades. 58% of the wind farms these onshore turbines are operating within have been in use for at least five years. Of these, more than 2300 have been in operation for more than ten years. Though a recent negative political attitude to onshore wind has seen its adoption in the UK stall, and its deployment is minimal compared to the likes of the United States or, closer to home, Germany, UK onshore wind infrastructure can be seen from a potential material management perspective to be significant and ageing.
- Solar PV: though not intuitively associated with the UK due to its low average irradiance (compared to many of its European neighbours), by April 2019 > 13 GW of residential and commercial PV solar panels had been installed (BEIS, 2019a), amounting to 40 – 50 million panels, notably containing in the region of 180t of Ag which is critical to the operation of a variety of modern electronic devices. Notably, in 2014 this highly distributed technology was included in the reformed WEEE Directive with the specific aim of guaranteeing the finances required to collect and treat impending EoL PV in a sustainable manner (PV CYCLE, 2020); questions still however remain over the ability to do this on a sustainable industrial scale (Latunussa et al., 2016; Sica et al., 2018; Heath et al., 2020).
- EV: by the first quarter of 2019, almost 200,000 'plug-in' vehicles which share materials with PV and OSW generators were driving on UK roads (DfT, 2019). This represents a 38% increase over one year from the first quarter of 2018, with more than 50,000 of the total number of plug in vehicles being on the road for five years or more. Notably, over the next ten years to 2030, the UK government has pledged that 50–70% of all car and 40% of all van sales will be made up of Ultra Low Emission Vehicles, primarily in form of fully electric or plug-in hybrid vehicles (DfT, 2018).

Such significant levels of LCI deployment, which for some technologies is minimal compared to many countries (e.g., *on*shore wind and PV within Germany, the United States and China), goes some way to highlighting the scale of LCI material demand and EoL management that society will be increasingly faced with. It is with the planning and deployment of OSW, however, that this proposition is exemplified.

### *3.1. UK offshore wind context and technology*

Of the myriad of LCI being installed globally, the UK is the current leader in deployment of OSW (The Crown Estate, 2020). As of 2019 almost 10 GW of turbines were operational in UK waters, which represents more than a third of the ~29 GW globally installed OSW capacity (Lee and Zhao, 2020) and almost half of Europe's ~22 GW of installed capacity (Wind Europe, 2020). By 2022, when all currently under construction OSW is commissioned, the total UK installed capacity will be more than 13 GW. Following a commitment by the UK government to the ongoing development of OSW within its Industrial Strategy, it is expected that the UK will be home to a minimum 30 GW of turbines by 2030 (see

HMG, 2019; N.B. following the 2019 UK general election, the UK informally aim to achieve 40 GW of installed OSW capacity by 2030 and achieve 75 GW of capacity by 2050).

OSW has proven to be a rapidly evolving industry with the size and consequent generation capacity of turbines growing significantly over a relatively short period of time. Analysis of public planning records (TPI, 2019) and UKWED (RUK, 2019a), show that between 2003 (the commissioning of the first UK OSW farm still operational) and 2013, the average size of a WTG was 3.6 MW with the blades averaging 52.5 m in length. By 2018, the average rating of commissioned WTGs in UK waters had risen to 5.6 MW with blades averaging 68.9 m in length. Based on OSW commissioned in 2019, and those under construction (that will be fully operational by 2022), the average rating of a turbine will rise to 7.7 MW with blades measuring at least 77.7 m in length. Given the proposed capacity and location of consented OSW, and the WTGs being developed by Siemens Gamesa and MHI Vestas (i.e., dominant European WTG market share leaders, e.g. Jensen and Gibbs, 2018; WindEurope, 2020), it is reasonable to assume that in the next decade turbines, within Europe, will largely contain >8 MW generators and employ >80 m blades.

In respect of type and scale of future material demand, it is important to note that increases in turbine size and the greater distances they are being installed has coincided with the switch from conventional high speed induction generators and drive trains, to lower speed and direct drive permanent magnet generators (PMG). Indeed, in addition to European market leaders Siemens Gamesa and Vestas moving toward the dedicated use of REE PMGs in offshore WTGs, the largest WTG currently on the market - the GE Haliade-X - incorporates a 12 MW direct drive PMG with a rotor measuring 220 m in diameter (each blade measuring 107 m). Moreover, though development of larger WTGs are limited by current logistical restraints, and indicating that the business case for larger turbines would require careful future technology risk assessment, Siemens Gamesa intend to offer a 14–15 MW PMG turbine to the market by 2024–2025 (de Vries, 2020).

The switch in drive technologies, from long established SC/DHIGs to PMGs and direct drives, has been partly driven by the reduced number of moving parts and the consequent reduction in costly offshore maintenance required. The reduced maintenance needs of PMGs and growing overall size and capacity of wind farms that require fewer more efficiently installed and supported turbines is partly credited for the drastic and rapid reduction in UK OSW Contracts for Difference (CfD). Indeed, a strike price of £114.39/MW in 2015 was halved in 2017 to £57.50 for the Hornsea Two and Moray East projects and was reduced again to £39.65 in September 2019 (BEIS 2019 b). The reduction in Levelised Cost of Energy (LCOE) and consequent strike price has occurred much quicker than the £100/MW by 2020 the UK government anticipated and had planned for (see DECC, 2011). Though some developers may be concerned by the rapid competiveness of OSW, the rapid fall in strike prices arguably hints toward the increasing maturity of the industry and the current technologies in use and almost ensures that the (now minimum) commitment of 30 GW of installed capacity by 2030 will be achieved (N.B. total installed global capacity is expected to reach 55 GW by 2024). As such, it is important to know, from a planning and wider materials perspective, that the increase in growth of distributed energy will also require an upgrade in grid capacity and its 'smart' integration (Siemens 2014). More pertinently, the growth in size of rotors, capacity of generators and widespread switch to PMGs, has led and will continue to lead to an increase in the use of the target discussion materials of this article, namely composites (largely in blades) and increasingly valuable metals such as copper and REEs (within PMG motors/magnets).

### 3.2. UK offshore wind inventory

Though the use of larger WTGs potentially means that fewer overall turbines will be required to meet planned growth in total installed capacity, the stock and types of materials that will be deployed in UK waters will still be significant in terms of total material demand and eventual EoL management. Exploring this premise, projections were and have been made of growing LCI itinerary within UK waters. Several of these projections, some of which contained significant over and underestimations of total OSW plant, were used to estimate future demands for critical LCI materials (e.g., Griffiths and Easton, 2011; Speirs et al., 2013; AMEC, 2014). Few projections, however, were made in respect of the management of OSW farms at the point of repowering and/or complete decommissioning or the role of EoL management in replenishing LCI materials, beyond what has transpired to be a probable under-costing of EoL management (see Section 4). With more than 15 years of OSW construction having now taken place, however, it is possible to derive a more empirical idea of the scale of OSW EoL management tasks facing decommissioners and those trying to build toward a low carbon CE.

Based on an interrogation of the UK National Infrastructure Planning portal and the RenewableUK Project Intelligence database (RUK, 2019a), in addition to a review of Crown Estate and OSW operator websites (e.g., Ørsted, Vatenfall, SSE, ENGIE, E.ON, Innogy), an assessment was made of the UK's growing OSW assets in regards of the use of rare elements, copper and composites. The assessment and resultant statistics represent all currently operational OSW and those currently undertaking offshore construction activities but does not include the Blyth wind farm, decommissioned in 2019, or those that were under construction but had not yet 'broken water' and commenced offshore installation activities at the time of writing (i.e. the 857 MW Triton Knoll wind farm).

Table 1 details the headline findings of the UK OSW inventory assessment and provides calculations of material dimension and mass for a range of components. Also provided is an estimate of selected pertinent additions to this inventory based on industry trends and the assumption that the UK Government will make good on its commitment to UK waters being home to (a now minimum) 30 GW capacity by 2030 (i.e., HMG, 2019). In summary, based on documented planning and WTG specifications, the assessment found that the 13,403 GW of installed and currently under construction wind farms in UK waters equated to some 2555 WTG's, laying a combined distance of 734 km offshore. From a future CE logistics perspective, the combined length of the 7655 blades attached to these WTGs will stand at 476.6 km, further than London to Dublin 'as the crow flies'. More importantly in terms of waste management, these blades have a combined mass of more than 151 kt of which more than 85% is comprised of composite materials. In total, based on the sum of calculations for each *specific WTG model*, there is almost 550 kt of nacelle installed or being installed in UK waters that house ~12.7 kt of Cu within the generators and 1.0–1.3kt of neodymium (Nd) and 0.15–0.2. kt of dysprosium (Dy) within the magnets of PMG drives (Fig. 4). Regarding the veracity of Nd figures, the Nd per MW intensity multipliers for each WTG were derived from extensive stakeholder discussion and analysis of literature exploring NdFeB content of different generators (e.g., Griffiths and Easton, 2011; USDoE, 2011; Wilburn, 2011; Constantinides, 2012; Hoenderdaal et al., 2013; Speirs et al., 2013; AMEC, 2014; Lacal-Arántegui, 2015; Imholte et al., 2018). The Nd multipliers employed in this article reflect those derived from and verified by industry operators and notably incorporate one of the lowest of stated average Nd NdFeB contents, i.e. 27% (Griffiths and Easton, 2011) (see Fishman and Graedel, 2019, for reference to Nd NdFeB intensity ranges used within articles).

**Table 1**
Selected UK offshore wind component and material inventory.

| **Pertinent Metrics and Cumulative Figures for Installed and Under Construction Offshore Wind Farms in the United Kingdom (as of Autumn 2019)** | | |
|---|---|---|
| **Capacity (MW)** | 13,403.5 | Based on all WTGs *currently* in or being installed in UK waters |
| **Number of Turbines** | 2555 | As above, i.e. does not include decommissioned Blyth or Triton Knoll |
| **Number of Blades** | 7655 | |
| **Blade Length (km)** | 476.6 | i.e. combined length of the 7655 blades |
| **Blade Mass (kt)** | ***151.6*** | |
| **Blade Fibre/Resin Mass (kt)** | 128.9 | i.e. based on 85% of blade mass consisting of composites |
| **Nacelle Mass (kt)** | 549.9 | |
| **Proportion of PMG WTGs (%)** | 42 | |
| **Proportion of DD WTGs (%)** | 32 | |
| **Nacelles Cu Mass (kt)**[1] | 12.7 | |
| **Nd Mass in PMG WTGs (kt)**[1] | ***1.0–1.3*** | i.e. DDPMG = 165.6 – 216.2 kg/MW; MSPMG = 37.4 – 46 kg/MW |
| **Dy Mass in PMG WTGs (kt)** | 0.15–0.20 | i.e. based on 4% of NdFeB magnet being Dy |
| **Distance to Shore (km)** | 734 | N.B. distance to shore is 'as the crow flies' |
| **Length of Subsea Export Cable (km)** | 3113 | **–** |
| **Cu Mass of Subsea Export Cable (kt)** | ***23*** | N.B. 55.5 kt if Hornsea 1 and 2 use Cu, rather than Al, export cables |
| **Length of Array Cable (km)** | 3123 | – |
| **Cu Mass of Array Cable (kt)**[2] | ***22.8*** | Based on the average of known cables specifications |
| **Conservative Estimate of Pertinent Additions to Total UK OSW Inventory by 2030** | | |
| **Capacity (MW)** | 16,600 | – |
| **Number of Turbines** | 2075 | Based on 8 MW turbines, i.e. ~8 MW turbines are the current norm. |
| **Number of Blades** | 5532 | – |
| **Blade Length (km)** | 498 | i.e. based on (at least) 80 m blades |
| **Blade Mass (kt)** | ***186.8*** | i.e. based on (at least) 80 m blades weighing (at least) 30 t |
| **Blade Fibre/Resin Mass (kt)** | 158.7 | i.e. based on 85% of blade mass consisting of composites |
| **PMG WTG NdFeB Mass (kt)** | ***8.3 – 10.8*** | i.e. based on range of NdFeB content range for MSG/DD WTGs |
| **PMG WTG Nd Mass (kt)** | ***2.2–2.9*** | i.e. based on a conservative 27% Nd NdFeB content |

[1] · A vast array of figures for Cu within nacelles was offered by industry stakeholders and within the literature and mostly derived from experience of onshore wind. As such, a range of figures were used from several sources (see: Frost and Sullivan, 2012; Broehl and Guantlett, 2018). [2]. The figure provided for the mass of export cables is based on reported cable specifications. The figure provided for the array cable, for which almost all are Cu, is based on a calculated mean cable core specification of 3 × 323mm^2.

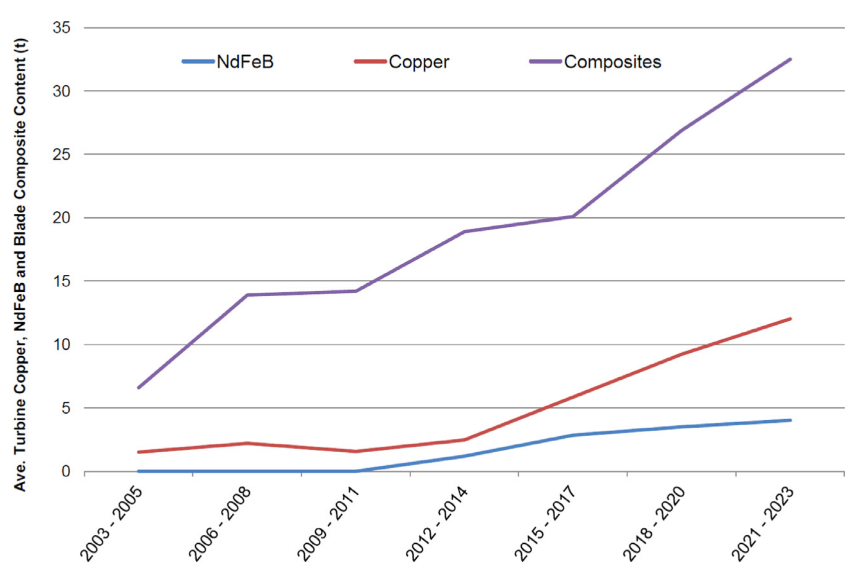


**Fig. 4.** Average growth in content of selected wind turbine nacelle and blade materials.

Connecting WTGs, the total length of subsea export and array cables amounts to 3113 km and 3123 km respectively. For the wind farms using Cu cabling, in the region of 22.8 kt of Cu will be present in array cables and 23 kt in export cables with a significant remaining balance of mass consisting of polyethylene insulation material and metallic armour (e.g., lead). In respect of building an accurate estimate of the UK's OSW materials itinerary, it should be noted that the afore figure given for Cu in cabling excludes the material present within the Hornsea One and Two wind farms due (at the time of writing) to not being able to confirm whether aluminium or Cu cables were employed. If the Hornsea sites employed Cu, the *Cu content* for these two farms alone would be in the region of 22.5 kt (based on the reported combined use of 847 km of XLPE 3 × 1000 mm^2 export cables). As a matter of significance, this would result in a doubling of what is increasingly deemed a precious metal present in OSW export cables before 2022. In respect of the wider context of potential demands on Cu, RenewableUK notably predict that up to 2023–2024 16,000 km of export and array cables will be installed, largely within the UK, United States, Taiwan and Germany (RUK, 2019b).

Based on a reasonable (*and probably conservative*) assumption that the turbines that will be installed over the next decade will be in the 8 MW range, carrying blades of at least 80 m in length, the amount of increasing strategic materials such as Nd present within these WTGs will be in the region of 2.2 kt – 2.9 kt (within 8.3 – 10.8 kt of NdFeB magnets). The total Nd figure would be highly dependant on the ratio of direct to medium speed drives employed, which in UK waters currently stands at ~3:1 in favour of direct drives that require significantly more Nd than their medium speed alternatives (e.g., Table 1). Notably, however, regards anticipated continued growth in turbine capacity and the accuracy of projections, if a similar ratio of direct to medium speed drives (i.e., ~3:1) were installed up to 2030, the total NdFeB demand of 2075 8 MW WTGs would be the same as 1660 10 MW turbines (see Table 1).

Minimum additional blade mass by 2030 will total 186.8 kt with a combined tip-to-end length of 498 km (i.e. despite less total WTGs, greater than all presently operational blades). The very possible use of larger than 8 MW WTGs and associated >80 m blades, as reflected in GE's 12 MW direct drive Haliade-X and Siemens Gamesa's 14–15 MW unit, would clearly see these additions of key materials increase significantly (i.e., the proposed Haliade-X blades weigh 20–25t more than SGRE's current market leading 81.5 m blade [i.e., ~30 t] and *up to* an additional ~216.2 kg of Nd could potentially be required for each additional MW of generator capacity. Notably, however, to highlight issues with assessing future resource management needs, overall Nd per MW employed within larger later generation turbines was expected by PMG manufacturers to decrease (Griffiths and Elston, 2011); whilst use of Dy, deemed critical to the performance of PMGs at high temperatures in WTGs and EV batteries (i.e., Hoenderdaal et al., 2013), is expected to fall due to a chronic shortage and consequent design changes to NdFeB magnets. This, arguably, gives some hope that demand for some problematic or rare materials will not increase in a linear manner to turbine capacity increases, hence in part the use of conservative generator specific Nd figures in this article compared to some published studies. However, despite the conservatism in material demands and potential waste production shown here, it remains a fact that LCI demands for critical and/or environmentally difficult to manage materials have been and will continue to be significant well into the future.

As discussed in Section 2, within the confines of an article it is not possible to fully detail the UK's OSW inventory and future waste management scenarios, hence the decision to focus on the three target resources of composites and rare and precious materials. Indeed, it is acknowledged that towers and foundations, and the balance of Cu not found within the subsea cabling or within the nacelle generator are not covered within this appraisal. However, dependant on long term concrete foundation and steel fatigue characteristics, and any marked changes to the physical proportions of future nacelle and rotors, there is an argument for the WTG support structures to be retained and employed in the repowering of a site (e.g., OWIH, 2019). Moreover, compared to recovery of increasingly valuable Cu within cabling, rare Nd and Dy, and the management of composites, the recovery and management of any concrete and/or steel employed within the most prevalently used foundations and towers is a relatively straightforward task and recycling capacity is strong (i.e. steel reprocessing). Also, regarding environmental impacts, whether the foundations of an array are completely repowered or not, there are strong arguments that their contribution to development of subsea biodiversity is too great to justify complete removal (e.g., Langhamer, 2012; Smyth et al., 2015). Indeed, in the light of wider EoL management considerations, given that most existing OSW was placed in areas deemed to be the 'windiest' and most accessible, and considering the rapid advances in turbine technology that reliably deliver greater energy yields, it is likely that early full or partial repowering of some wind farms will take place (see OWIH, 2019), thus clouding 25 yr WTG lifespan predictions and related EoL management plans.

## 4. Managing the end-of-life of offshore wind infrastructure

Sections 2 summarised the cross-technology use of select materials in LCI, the environmental impacts deriving from their extraction and processing and the current capacity for their recovery and EoL management. The section was constructed around highlighting the need to be aware of stocks and flows of materials and why such knowledge is important with regard to the need and benefits of developing a LCI CE. Section 3 placed this summary and arguments in the empirical context of the deployment of OSW in the UK and known industry developments over the last two decades, highlighting the scale and variety of considerations that need to be made in the pursuit of CE. Herein, the article explores and critiques what EoL measures have been, and could be, put in place to ensure that the materials used in LCI development can be conserved within a CE and/or, if absolutely necessary, disposed of in a sustainable manner.

### *4.1. Offshore wind end-of-life obligations*

In the process of seeking consent for the development of an OSW farm, in addition to following standard planning consent processes that include community stakeholder engagement and environmental impact assessments, a Decommissioning Plan (DP) must be produced in line with OSPAR (Convention for the Protection of the Marine Environment of the North-East Atlantic) and other local/national planning commitments and legislation (The Crown Estate, 2016). DPs are a direct requirement of the UK Energy Act 2004 (*as amended*) (HMG, 2004). The requirements of the Act largely focus on the financial cost of managing the EoL of OSW: "…*taking into account our international obligations – that a person who constructs, extends, operates or uses an installation should be responsible for ensuring that the installation is decommissioned at the end of its useful life, and should be responsible for meeting the costs of decommissioning (the "polluter pays" principle)*" (DECC, 2011; *recently superseded by* BEIS 2019c: 6).

The Act makes clear a prospective operators' obligation regards the installation and eventual removal of OSW infrastructure. Even given allowances for development of future 'best practice' and, limiting long term environmental impact, Offshore Renewable Energy Installations "*should be designed with full removal in mind, and full removal will be the default position for OREIs unless there are strong reasons for any exception*" (BEIS 2019c: 8). For components and infrastructure that is removed from site, it is expected that developers will manage the balance of removed materials according to the Waste Hierarchy. Specifically, OSW waste: "…*from decommissioning should be reused, recycled or incinerated with energy recovery in line with the waste hierarchy, with disposal on land as the last option. BEIS does not consider disposal of waste at sea to be acceptable. Waste management must be carried out in accordance with all relevant legislation at the time, including control of any hazardous wastes*" (BEIS 2019c: 28).

Comparing these development obligations, in respect of the UK learning any lessons from its European neighbours, it is worth noting that demands placed on UK based OSW are at least on a par with the two countries that arguably pioneered the development and deployment of OSW (i.e., Denmark and Germany). Denmark for instance has relatively strict controls on OSW developments and their EoL management which are controlled by a construction and, in time, decommissioning permit that similarly to the UK consenting process is underpinned by an Environmental Impact Assessment (EIA). Though constrained by the same international regulations as the UK and Denmark, there is no definitive decommissioning obligation within Germany as there is an expectation that, as with their onshore wind farms, German OSW will be subject to relevant federal state law rather than national policy that does not currently exist. Notably, regardless of future regulation within Germany, it has been highlighted that many wind farms were not designed to be recycled (Kruse et al., 2019).

### 4.2. Lessons learned to date

As a point of reference, several relatively near shore farms have, to date, been decommissioned. Decommissioned farms include:

- Hooksiel (Germany), 1 × 5 MW turbine (2008–2016)
- Lely (Netherlands), 4 × 0.5 MW turbines (1997–2016) (notably one turbine had already failed/collapsed due to metal fatigue in 2014)
- Blyth (UK), 2 × 2 MW turbines (2000–2019) (albeit this was a demonstration/pilot site)
- Yttre Stengrund (Sweden), 5 × 2 MW turbines (2001–2015) (only one of the five turbines where still functional when decommissioning was announced in 2014)
- Vindeby (Denmark), 11 × 450 kW turbines (1991–2017)
- Utgrunden (Sweden), 7 × 1.5 MW turbines (2006–2018)

Lessons learned from decommissioning these farms are not freely available and do not seem to have been incorporated into any recently produced EoL plans (see Section 4.3). Moreover, it is noted that repowering of the majority of these older sites was not given serious consideration due to sub-economic benefits, for instance they would not provide the increased efficiencies and yield that can be drawn from the increasingly larger WTGs that are constructed further from shore in areas with more productive and reliable wind resources. Thus suggesting that much of the existing OSW that have been in operation since the late 2000s, will be brought back in their entirety in the near future, which would fit with published expectations for a first peak in decommissioning activities in 2030 (e.g. Kruse et al., 2019).

In a perfect world, this would suggest that there is in the region of ten years to bring industrial scale solutions online that can ensure a sustainable and resource conserving future for any EoL wind farm. However, it is notable that not all of the OSW decommissioned to date, largely operating within mainland European waters, reached the expected 20–25 years of operation (albeit some were always intended to be shorter lived demonstration installations); and it is worth noting that several hundred offshore WTGs have already required costly repairs, particularly to blades that have failed to perform as expected in harsh offshore conditions (Constable, 2018).

### 4.3. The current reality of offshore wind end-of-life commitments

Decommissioning programmes for 20 UK OSW farms were reviewed in respect of the given operators' intentions for EoL management of their LCI (Table 2). The plans were reviewed, where possible, alongside each wind farm's respective EIA. Aside from the DPs for the Hornsea I and II sites, which were directly requested and supplied to the authors, the reviewed DPs were for wind farms whose operators have made plans freely available online or which can be viewed via the UK government's National Planning Infrastructure portal. Notably, despite being for public consumption (as per the Energy Act 2004), 15 or so DPs for UK wind farms could not be appraised due to their removal from operators' websites during the period of this study and/or their lack of response to requests for copies of programmes. As demonstrated in Table 2, the reviewed DPs were at several stages of development which highlights the 'live' nature of these documents.

It is notable that throughout the DPs the format and content is largely similar (as would perhaps be expected given the presence of preferred DP frameworks, i.e. as shown in BEIS 2019c). More interestingly, however, they show little change or improvement in terms of specifics of material recovery or management from one iteration to the next, nor do they incorporate any reference to the industry's capacity to follow through with proposed programmes of decommissioning (i.e., at scales that far exceed that of the sites that have been deconstructed to date or within the confines of existing EoL management technologies, i.e. Section 2.1-3). The review of DPs found that - on a technical level - they all met the content demands as described within Section 4.1. Throughout all DPs, a commitment is made to removing infrastructure, albeit almost all invoke the caveat of fulfilling this obligation where such action is economically viable and not environmentally punitive. This caveat, in all DPs, is placed in the context of adhering to BPEO (Best Practicable Environmental Option), as requested within development guidelines, and in respect of the traditional Waste Hierarchy. In essence, the reviewed DPs commit operators to meeting their legal and technical obligations regards decommissioning of OSW infrastructure. However, that is all they do.

From the perspective of a nascent CE and long-term sustainable waste management, it can be said that the review of available DPs found that, regardless of iteration, their content is generic and stated management strategies are formulaic. Indeed, in places DPs could be deemed to be perfunctory. This feeds into existing narratives of decommissioning being "*poorly understood*" and published concerns over a significant undercosting of OSW decommissioning (e.g., Freeman, 2015). Indeed, in regards to the focus-materials of this article, the content of DPs provides no reassurance over their management at EoL or recovery for sustainable reuse. Despite their critical role in the ongoing development of the industry, and other emerging LCI, it was observed that none of the twenty DPs provide any reference of any kind to either the presence or specific recovery of multiply valuable REEs. This was also the case for EIAs that accompanied DPs. With a primary focus on the OSW farm, pre, during and post operation, EIAs provide no specific reference to REEs or any other EoL element of the employed material. The lack of acknowledgement of such critical materials or commitments to their recovery does not seem to fit with the sustainable technology narrative of OSW. Indeed, such a stance is exemplified by one of the other focus materials of this article, i.e. copper, with all reviewed DPs being clear that it is expected that export and array cables, containing several thousand tonnes of valuable material, will remain in-situ at the wind farms' EoL. It is acknowledged that

**Table 2**
List of reviewed publically available UK offshore wind farm decommissioning plans.

| Decommissioning Plan | Stated Version |
|---|---|
| Beatrice Alpha* | 5th Draft, June 2018 |
| Beatrice Offshore Windfarm | Draft, September 2015 |
| Burbo Bank Extension | 1st Issue, June 2018 |
| West of Duddon Sands | 3rd Draft, July 2016 |
| Dudgeon | Draft, No Date (Pre 2015) |
| Forthwind Demonstration | 1st Consultation, October 2017 |
| Greater Gabbard | Final Version for Approval, August 2007 |
| Gunfleet Sands III | Version A., March 2012 |
| Gwynt y Môr | Final Version for Approval, September 2010 |
| Hornsea One | Draft for Consultation, September 2016 |
| Hornsea Two | Draft for Consultation, September 2016 |
| Lincs | No Version Number, December 2010 |
| London Array | V.3 Approved, October 2013 |
| Moray (East) | Preliminary Programme, July 2012 |
| Moray (East) | Final Version, May 2018 |
| Neart na Gaoithe | For Approval, August 2019 |
| Ormonde | V.2 Approved, May 2013 |
| Sheringham Shoal | V.5 for Approval, April 2010 |
| Triton Knoll | V.3 Draft, December 2018 |
| Walney One | No Version Number, September 2011 |

* Reviewed plan is for the Beatrice Alpha O&G platform that is connected to two WTGs.

there are environmental arguments against disturbing the seabed with attempted extractions; however, setting aside the effects on the future material flow and availability of this increasingly valuable metal, such a current blanket stance to abandon these materials is contrary to at least the spirit of the permitting regime that demands that all structures will be removed (i.e., BEIS 2019c). This brings much of the value and purpose of DPs into question.

Indeed, for what recovery and recycling is covered in DPs, all operator plans make numerous assumptions about the management capacity and reuse of materials at eventual repowering and/or decommissioning that ignore the fact that, for example, composite recycling solutions do not exist in any meaningful manner in the UK, or globally. As such, blanket recommendations to reassess DP commitments to abandon multi-valuable materials (i.e. Cu) and commit resources to "*sustainable incineration*", or push sub-standard or exhausted components overseas, is questionable from a wider systems management and resource conservation perspective. As such it is of no surprise that there is no reference to CE within any DP. Encouragingly, as discussed above, the Waste Hierarchy (i.e., in simple terms: reduce, reuse, recycle, energy recovery, dispose) is referenced throughout the reviewed DPs. However, it is noticeable that there is a distinct focus on the lower inferior reaches of the hierarchy. Indeed, there is blanket reference to incineration of blades, but notably, once again, no acknowledgement whatsoever of the lack of capacity for this method of material 'management'. At this time, such a commitment could only be facilitated by ignoring supposed restrictions on the export of wastes, thus potentially creating another international waste merry-go-round akin to that seen for WEEE and plastics, or ignoring arguments relating to the undesirable technology lock-in effects of incineration as a preferred waste management tool.

What is perhaps more concerning within the range of reviewed DPs, and their lack of obvious improvement over time, is that they all roundly translate as effectively waiting for somebody else to take the lead in EoL management efforts, e.g.: "*The decommissioning plan and programme would be updated during the lifespan of the wind farm to take account of changing best practice and new technology*". (Vatenfall, 2018) - "*Once larger-scale wind farms start to be decommissioned, it will provide valuable insight into the timing, costs and operational challenges to be faced*" (Ørsted, 2018). Such statements, which could have been referenced from any of the operators DPs, may be acceptable in respect of a genuine wish to be able to adopt future BPEO, however the development of CE requires pro-activity and forward thinking at the point of project development, not at its point of removal (as noted for blades by Jensen and Skelton, 2018). Moreover, such statements on BPEO provide the assessor of the DP (i.e. the State) no insight of value into how the operator intends to meet their obligations beyond vague promises to do whatever others are doing. In some ways, it could be argued that this is an irresponsible permitting condition given the risks it leaves the State open to, not least from a 'decommissioner of last resort' perspective (e.g., widespread environmental and financial oil and gas infrastructure clean up bills left to tax payers). Accepting this risk is *contrary* to the demands of the decommissioning framework: "*The Government's approach is to seek decommissioning solutions which are consistent with relevant international obligations, as well as UK legislation, and which have a proper regard for safety, the environment, other legitimate uses of the sea and economic considerations including protection of the taxpayer from liabilities relating to decommissioning. The Government will act in line with the principles of sustainable development*" (BEIS 2019c: 7).

It is important to acknowledge that OSW DPs have been critically reviewed by other authors. Indeed, Freeman (2015) was also clear in their assertion that the critical detail of offshore renewables decommissioning is missing. This is true both in terms of the financial burden and longer term environmental impact of the OSW sector. Indeed, the lack of detail within DPs was the basis for claims that decommissioning of OSW has been significantly undercosted (e.g., Freeman, 2015; Topham and McMillan, 2017; Purnell et al., 2018). Notably, following these publications, the arguments they make were augmented and largely confirmed by a duly commissioned UK Government (re)appraisal of OSW decommissioning (see Arup, 2018). The commissioned report found that undercosting potentially ran into the £billions and was partly due to the impact of changing legislation, uncertainty over the availability of specialist and expensive vessels and, in regard this article (and Purnell et al., 2018), partly due to distinct vagaries around waste management in respect of the many statements on what is, and is not, recovered and how. It is fully acknowledged that it is difficult to foresee how - exactly - a given decommissioning programme will take place. Indeed, Ørsted state: "*…the decommissioning phase is not expected to commence before a timeframe of at least 27 years. Therefore, it is not possible to describe the precise technology and methods of decommissioning works. These will develop over the operational lifetime of the wind farm*" (DONG, 2016: 23). However, the review of publically available DPs suggest that this has not happened in any meaningful manner to date and highlights the perfunctory nature of DPs, which translate as little more than an admission from the operator that they installed infrastructure and promise to remove it (or some of it in the case of cabling). To their credit, some manufacturers, such as MHI Vestas (2006), provide publically available LCA reports for their WTGs which highlight the presence and impact of the likes of much discussed REEs. Siemens Gamesa (2019) do similarly within Environmental Product Declaration's for their most recent iteration of their direct drive PMG turbines. Notably, such documents emphasise the role decommissioning and resource recovery holds in reducing the life-cycle environmental impact of WTG materials, which as demonstrated in Section 2 can be significant. If manufacturers have any confidence in such documents and their claimed material life-cycle scenarios, it is reasonable to assume that operators should be aware of the waste management capacity limitations (or non-existence) highlighted within Section 2 and have more nuanced plans in place to deal with OSW infrastructure – plans that should be incorporated into DPs if an LCI CE is to be encouraged.

### *4.4. Integrating circular economy approaches into decommissioning plans*

Placing the critique of DPs into the CE context, CE, as a whole system approach, aims to minimise resource extraction from the natural environment, maximise waste prevention measures and optimise the use of materials, components and products throughout their life-cycle (Velenturf et al., 2019a). Optimisation is guided by values on enhancing environmental quality, social well-being and economic prosperity. These high level aspirations are often translated into practice with 'R-Ladders' i.e. reduce, reuse, recycle, or a variation thereon and are interpreted in the UK governance system as the 'Waste Hierarchy' (and mentioned in DPs). These whole life-cycle principles are, however, still to be fully converted to the OSW sector.

As evidenced above, applying CE thinking to OSW reveals important gaps in the whole life-cycle management, particularly in respect of 'decommissioning'. Decommissioning should be seen as a point of system regeneration, not an end-point. It includes steps that can be embedded in OSW development and considered throughout the life-cycle of OSW turbines and infrastructure, from design for balance durability, reparability, disassembly and recyclability (i.e. design for circularity), to extending component lifetime with better O&M and repair, reuse and refurbishment and remanufacturing before recycling, energy recovery, and controlled storage (Fig. 5). While author engagement suggests that manufactur-

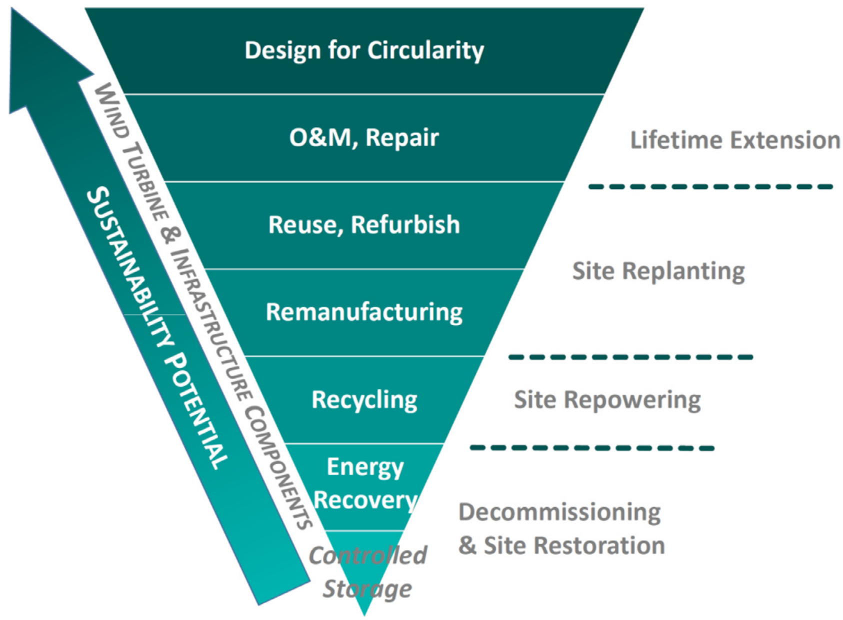


**Fig. 5.** Whole system overview of circular economy strategies for offshore wind. *Note:* a rethinking of the traditional waste hierarchy, the CE hierarchy shows a path to resource conservation and system regeneration, with approaches for wind turbine materials and components shown within the inverted triangle and hierarchical approaches for wind farm sites shown to the right of the figure. Current Decommissioning Plans indicate that the OSW industry is almost exclusively planning the EoL of wind farms within the lower reaches of the hierarchy (i.e., recycle, energy recovery and landfill).

ers and operators are aware of CE, the review of DPs shows that their focus is firmly on recycling (albeit with limitations in this area), energy-from-waste and landfill. Design with the full spectrum of CE approaches in mind is, generally, not on the agenda. It is important however that the industry adopts more CE practices, both for its own resource security purposes and because OSW has a relatively high environmental impact due to its material usage (e.g. water, human and eco-toxicity of metal processing) compared to other power generation technologies (Stamford and Azapagic 2012). Many of these impacts could be offset in part by CE practices that avoid the re-processing of materials for the next generation of OSW farms.

In general, solutions higher up the extended Waste Hierarchy require innovation but can generate more environmental benefits, such as greater carbon and water savings owing to reduced processing and waste diversion, more social benefits in the form of job creation and new skills and more financial benefits in the form of reduced raw material and waste management costs that coincide with new business opportunities (e.g. Laybourn and Morrissey, 2009; Green Alliance, 2019). For OSW, CE could offer new business opportunities in the form of design for circularity services, expanded O&M services (e.g. component repair and/or refurbishment; data systems for live and long term component monitoring), and EoL management services that promote maximum functionality of components via extended reuse, refurbishing and remanufacturing (Velenturf et al., 2019b).

At the level of whole wind farms, the lifetime of sites can be extended. This would be the most effective way to increase the resource productivity of the materials used in OSW turbines and, indeed, wider LCI. Where lifetime extension is no longer an option, sites could be replanted with similar turbines, theoretically creating a ready market for reused, refurbished and remanufactured components. However, technological advances enabling greater economies of scale may swing current preference to the repowering of sites with larger turbines. Using different technology may initially limit the potential for reuse, refurbishing and remanufacturing of components, but more components could be made from recycled materials – within the limits of EoL technologies currently available (Section 2). Assuming most current OSW site leases can and will be extended, the initial configuration of OSW farms could be designed with the future in mind, for example by developing more durable foundations that can be potentially used for multiple generations of turbines. Finally, sites should always be designed to enable the full decommissioning and restoration of a location to similar environmental conditions as prior to its development. However, with no clear resource security benefits to be derived from the recovery of concrete, a discussion is to be had around full decommissioning of foundations and the potential destruction of habitats that have promoted biodiversity and improved the quality of the marine environment. Though such man-made habitats may create welcome synergies between OSW and nature conservation, other users of marine resources and space, such as the fishing and marine transportation and logistics industries, will have to be managed proactively given the significant expansion of OSW farms within a shared space over the next 10–30 years.

Most importantly, the integration of CE approaches into OSW decommissioning has to be supported by governance changes. In the UK, Offshore Renewable Energy Installations decommissioning guidance – including OSW – is based on North Sea oil and gas decommissioning, but these are not comparable (Velenturf et al., 2020). As suggested above, the operational life of many OSW farms is likely to be extended rather than being fully decommissioned like oil and gas. North Sea oil and gas infrastructure reuse is, however, notably low (1–2%) for various resource, technical and economic reasons. A supposed sustainable industry like OSW can and should strive for better. As demonstrated within the critique of DPs, State approved guidance on waste management is insufficiently challenging OSW operators to aim for sustainable EoL solutions and, due to the claimed unavailability of waste management solutions for some components, companies do not have to provide comprehensive EoL costings. This creates financial risks for industry and Government. Government is currently accepting this risk, contrary to the demands of decommissioning guidance (Section 4.1). Government could oblige industry to build on the overview of materials used in a wind development to include a gap analysis in their DPs that would, as a minimum, identify:

1) Current scarcities in EoL management infrastructure, including the availability of vessels that are required to undertake decommissioning;
2) The current capacity and limitations of the waste management technologies expected to be used; and,
3) The impact of these current limitations and their own efforts, or awareness of others' efforts, to address these limitations.

A whole system approach should be adopted in order to access the benefits of a CE as aspired to by numerous regions (e.g., in the UK, under the Industrial and Resources and Waste Strategy). This will require expanding the minimum stakeholders that need to be engaged in DP preparation consultations, including organisations with knowledge of decommissioning logistics, project management, and waste management solutions and costs (Velenturf et al., 2020). This will better safeguard the quality and value of DPs and produce more realistic cost estimates. Current decommissioning guidance is based within marine navigation and energy legislation, but surprisingly lacks a grounding in resource management and sustainability. DPs should include evidence on how the deployment of offshore renewable energy infrastructure has been designed to optimise economic, social, technical and environmental values at every stage of the infrastructure's life-cycle including at EoL. This will require iterative feedback to the design of the offshore infrastructure itself, and not just to the DP. The timing of DP preparation and submission has to be adapted to accommodate for this.

## 5. Summary and conclusions

A significant and rapid growth in low carbon infrastructure (LCI) has occurred in recent years. This growth needs to continue in order to enable low carbon development, but it should do so in a responsible manner. Based on the example of OSW, it is clear that there is little in place in the way of managing the EoL stage of our LCI and ensuring material value is retained and returned to society in the most environmentally and energetically efficient manner as possible. Despite the scale of their deployment and use, OSW DPs lack clarity in respect of how critical, rare and/or extremely difficult to recycle materials will be recovered and reintegrated into a low carbon economy. OSW EoL plans, both in the form of DPs and their accompanying EIAs (and relevant Environmental Product Declarations), have been found to be formulaic at best and perfunctory at worst, with many OSW DPs seemingly absolving themselves of EoL responsibility by placing an overreliance on the development of future 'best practicable environmental options', which they fail to characterise in any meaningful way. There is little evidence to suggest that hazy and non-holistic visions of OSW EoL management are different to other LCI technologies (i.e. PV, onshore wind, EV). This article, however, should not be interpreted as a rebuke of OSW or other LCI operators, or indeed the UK's deployment of LCI, particularly OSW which is making great strides toward meeting the countries renewable energy obligations and tackling the 'climate crisis'. Nevertheless, in terms of ensuring a move toward a low carbon CE, it is necessary for LCI to be designed not only for durability and longevity, but also for low impact recovery, direct reuse and, when necessary, recycling and remanufacturing. Thinking about the EoL of our LCI, within and across technologies, is thus required at its inception.

Artificial arguments relating to LCI decommissioning being too far away to be currently paid any serious consideration are unacceptable, particularly with regards to component materials that possess high environmental and/or societal impacts at the point of extraction. Development of low carbon management and recovery of the materials embedded in all manner of LCI is needed now. It is fully acknowledged that proactive waste-to-resource innovation is a timely and costly exercise. Within the waste management sector, waste data is notorious for being out of date almost as soon as it is produced, thus hindering exploration and indeed investment in potential recovery and reuse innovations. Specifically, one of the biggest limitations to waste-to-resource innovation is regularly argued to be the variability and uncertainty of material forms, quantity and ultimately their continued or long-term availability. For much LCI, and particularly on/offshore wind, this limitation to resource innovation does not exist. For all intents and purposes, it should be known exactly where and what materials are deployed, in what quantities and, most importantly, when it will be recovered and require EoL management. This is a relatively unique scenario in waste-to-resource innovation and provides a distinct opportunity for LCI developers to fully embed LCI in CE in an environmentally, socially and financially sound manner. It is accepted that sound arguments will exist around economies of scale and financial viability, but this is why a CE perspective is required at project development - thus allowing the exploration of synergies with other LCI whose deployment is also gathering pace and will share EoL material management needs. This would require proactive and joined up thinking by all key stakeholders. Indeed, it has been noted that the European WTG market is and has been dominated by a small number of large manufacturers who hold a highly influential position in OSW development. Given the cross-technology demand and use of materials, the burden of developing CE EoL solutions should arguably be spread across the LCI community, with lessons learned from the likes of oil and gas and nuclear decommissioning, and not be allowed to develop in narrow industry silos. As a minimum, operators should be compelled to provide a bill of materials within DPs to provide more clarity over specific EoL management needs and potential options for innovation and at what scales.

In summary, as the world leader in installed OSW capacity and power generation, this article's case study focussed on OSW development in the UK but the overriding discussions and findings appear to be relevant to LCI development and deployment policy globally. To truly be a low environmental impact solution, negate technology lock-in and facilitate cross technology critical resource security, it is essential that LCI components are designed for durability, recovery, modularity, and resource reuse, in line with proven CE principles and practice. Though focussed on UK LCI, there is little evidence to suggest that proposals for EoL management are any better in other countries with similar or greater scales of LCI, both currently and in the development pipeline. It is thus recommended that similar studies, relating to the readiness of industries and regions to incorporate their LCI into a CE, be conducted.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Supplementary materials

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