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Resource recovery and low carbon transitions: The hidden impacts of substituting cement with imported 'waste' materials from coal and steel production

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ABSTRACT

Here we investigate the increasingly complex relationship between the resource recovery practices of the UK concrete industry and ongoing low-carbon transitions taking place in electricity and steel. Reductions in UK coal-based electricity and primary steel production are reducing domestic availability of residues – coal ash and steel slag – that are used to replace cement in concrete; for decarbonisation purposes and to increase concrete quality. This is leading to an unusual mass-transportation of 'wastes' from the Global South to Global North. Focusing closely upon the mitigation pathways of concrete producers, we develop an inter-industry model of material flows, and a diversity of scenarios and sensitivity tests, to consider how resource recovery practices and carbon emissions of the three sectors may evolve. A continuation of domestic shortages in waste-derived cement substitutes appears inevitable and future international shortages possible. But even if foreign producers supplied enough cement substitutes to meet UK demand, the broader carbon implications of such trade may be far from benign. Using a revenue-based approach to allocate emissions to coal ash leads to a wide range of embodied carbon estimates – from relatively low (0.15 t.CO_2/t.ash) to exceeding that of traditional Portland cement (1 t.CO_2/t.ash). However, the carbon associated with internationally traded recovered resources currently stands behind a 'double-blind' system of accounting: emissions do not register in the conventional territorial accounts of the importing country and they may be hidden from its consumption-based accounts as well. The impacts of such trade and related carbon accounting conventions are unclear and we emphasise the need for further investigation. To this end, our results demonstrate the importance of incorporating highly interconnected sectors and international trade into analyses of low-carbon transitions, and highlight the challenges this presents for designing appropriate policies, accounting frameworks, and interdisciplinary impact assessment methods that look beyond sectorial and national horizons.

1. Introduction

Electricity, steel and concrete are used extensively in modern society. The magnitude of their impacts can be perceived unambiguously almost everywhere that humans are found. Electricity production now accounts for nearly 40% of global primary energy consumption and this is projected to rise over the coming decades (IEA, 2016), concrete accounts for at least 50% (by weight) of all global materials manufactured (Purnell and Roelich, 2015) and steel is widely used throughout the technosphere and is often difficult to replace (Allwood et al., 2012).

Unsurprisingly, these sectors are also massive sources of greenhouse gases (GHG; 'carbon emissions' herein). Electricity accounts for > 20% of global emissions (IPCC, 2014), whereas steel and cement account for 5–10% each (Allwood et al., 2012). These substantial impacts place them at the forefront of global efforts to reduce emissions to safe levels. But even accounting for the ambitious actions agreed at the UN's Paris climate conference, anthropogenic emissions are expected to raise temperatures well above safe levels (Rogelj et al., 2016). Achieving reductions in emissions consistent with international ambitions is thus set to remain a daunting task, particularly while economic growth

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remains a higher political imperative than reductions in energy and material demand (van den Bergh, 2017).

Further, when implementing low-carbon initiatives, it is important to consider trade-offs and unintended consequences (Konadu et al., 2015). Unintended consequences are increasingly likely and difficult to foresee in an increasingly interconnected global economy, particularly where regulations – environmental and otherwise – vary dramatically between sectors, regions and nations. Trade-offs of low-carbon initiatives may be environmental ('environmental problem shifting'; van den Bergh et al., 2015), such as when policies for low-carbon power increase biofuel demand that, in turn, reduces carbon sequestration in some forest types (Hudiburg et al., 2011) and intensifies conflicts over land (Scheidel and Sorman, 2012). Trade-offs can also occur in other 'domains of value', namely the social, economic or technical (Iacovidou et al., 2017a,b, Iacovidou et al., 2018), such as when recycling targets of 'developed' countries result in high waste exports that impact upon rates of child labour in importing countries (Velis, 2015). A consideration of such complexities adds to the already complex decision-making processes that underpin the planning of power- and industrial-production facilities, which have lifetimes spanning many decades. Cases for investment and policy decisions must look decades ahead into a highly uncertain future, while taking a holistic perspective (Brown and Robertson, 2014); a conceptually mundane proposition, which remains notoriously difficult to implement.

A first step towards capturing the broader impacts – environmental, social, or otherwise – in complex systems is to trace the impacts of consumption up the supply chain. Well-established methods such as Life Cycle Analysis (LCA) address precisely this issue (Guinée et al., 2011), their core aim being to estimate environmental impacts throughout supply chains – extraction, manufacture, transport, etc. – and reallocate these to final materials or products (Heijungs et al., 2010; Kloepffer, 2008). Methods like LCA have various limitations (Iacovidou et al., 2017a), but they remain valuable for understanding the impacts of consumption. The advantages are particularly clear when considered alongside conventional carbon accounting frameworks. Territorial carbon accounting considers only the emissions emitted within the borders of a country or region, and are the standard method implemented in the United Nations Framework Convention on Climate Change. By contrast, consumption-based accounts reallocate supply chain emissions to goods and services, thus basing the carbon footprint of a country or region on the consumption of these goods and services (Millward-Hopkins et al., 2017; Peters, 2008; Scott and Barrett, 2015); an approach conceptually similar to LCA but methodologically very different (Barrett et al., 2013). A common problem shared by these lifecycle or supply-chain based approaches is the question of how to allocate emissions and other impacts to multiple products for processes that produce multiple outputs (Hanes et al., 2015; Majeau-Bettez et al., 2014; Weidema and Schmidt, 2010).

The above-mentioned issues are particularly pressing for the system that we study here. At the centre of our analysis are the electricity, steel and concrete sectors in the UK. The interlinkages between these three sectors are substantial and continually evolving. Over 50% of global steel production is currently used in infrastructure and buildings and a substantial proportion of this reinforces concrete (Allwood et al., 2012; Purnell, 2013). Most importantly for our work, cement and concrete producers across the world are increasingly utilising residues from coal-fired power generation and primary steel production (Yao et al., 2015).

When producing concrete, pulverised fly ash (PFA) from coal combustion or ground granulated blast-furnace slag (GGBS) from primary steel production can replace up to 55% wt. or 80% wt., respectively, of the required cement (Leese and Casey, 2015), which is by far the most carbon intensive ingredient of concrete (MPA, 2008). Coal-PFA and GGBS are currently considered low- or even zero-carbon materials, and hence important means to lower the carbon impacts of construction (IEA, 2017a). Further, substituting a fraction of cement with coal-PFA or GGBS actually improves the performance of concrete, increasing strength and durability while decreasing permeability (Thomas and Matthews, 2004). Currently, PFA is one of the most abundant anthropogenic materials produced and its use as a cement replacement is by far its most significant resource recovery route (Yao et al., 2015). But the availability of PFA and GGBS is falling due to global transitions away from coal towards renewables and biomass and waste-based fuels, and from primary steel production towards secondary production from scrap metal (IEA, 2016; New Civil Engineer, 2016). Further, the different chemical composition of PFA obtained from combustion of biomass or waste-derived fuels, such as solid recovered fuels (SRF), means it does not normally have the chemical composition required to substitute large fractions of cement (Iacovidou et al., 2018; Kalembkiewicz and Chmielarz, 2012; Sarabèr, 2014; Velis et al., 2012).

Reliance upon the residues of high-carbon processes – which are undergoing necessary decarbonisation transitions that make these residues increasingly unavailable or unsuitable –presents obvious issues for concrete and cement producers. This raises the question as to how the impacts of industrial processes should be allocated to multiple outputs: specifically in our case, should the residues of high-carbon processes such as coal-power and primary steel production be allocated some of the primary processes' GHGs or not? If so under what circumstances? Such questions must be answered in order to design appropriate accounting frameworks for distributing carbon emissions throughout supply chains to final products. Further, they are increasingly important to ask as PFA and GGBS are rapidly becoming economically valuable, internationally traded commodities (Alberici et al., 2017).

Here, we develop a simple inter-industry model integrating the electricity, steel and concrete sectors (referred to herein as the ESC sectors) to investigate the interactions between resource recovery from waste issues and industrial low-carbon transitions. We consider production taking place within the UK and that occurring globally in regions that produce materials imported into the UK. We thus formulate various scenarios – informed by historical patterns of consumption and production across the ESC sectors and projections of various industrial and independent research bodies – to explore potential future developments. We emphasise that our results are not forecasts, but rather visioning exercises, the results of which may inform the development of appropriate strategies of future action; governmental, industrial, or otherwise.

In other work, we have laid a foundation for how holistic sustainability assessments may be undertaken for resource recovery from waste systems, reviewing the metrics available (Iacovidou et al., 2017b), conceptually grounding a modelling approach (Millward-Hopkins et al., 2018) and developing an assessment framework (Iacovidou et al., 2017a). Here, using a single metric (GHGs), we aim to demonstrate the importance of widening system boundaries to incorporate complex, highly interconnected industrial sectors and international trade into analyses of low-carbon transitions and resource recovery systems, in order to guide appropriate policies and effective mitigation strategies.

2. Materials and methods

2.1. Material flow analysis

The backbone of our method is material flow analysis, which is based upon the principle of mass conservation (Cencic and Rechberger, 2008). Our system is thus comprised of various processes with material inflows and outflows and stocks. Inflows are transformed into outflows using transfer coefficients for each process, which we base upon a large number of sources as detailed in the Supplementary Materials (SM). For each of the scenarios considered we make projections of UK consumption and production for the ESC sectors, and we consider how global production systems may influence the availability of materials to

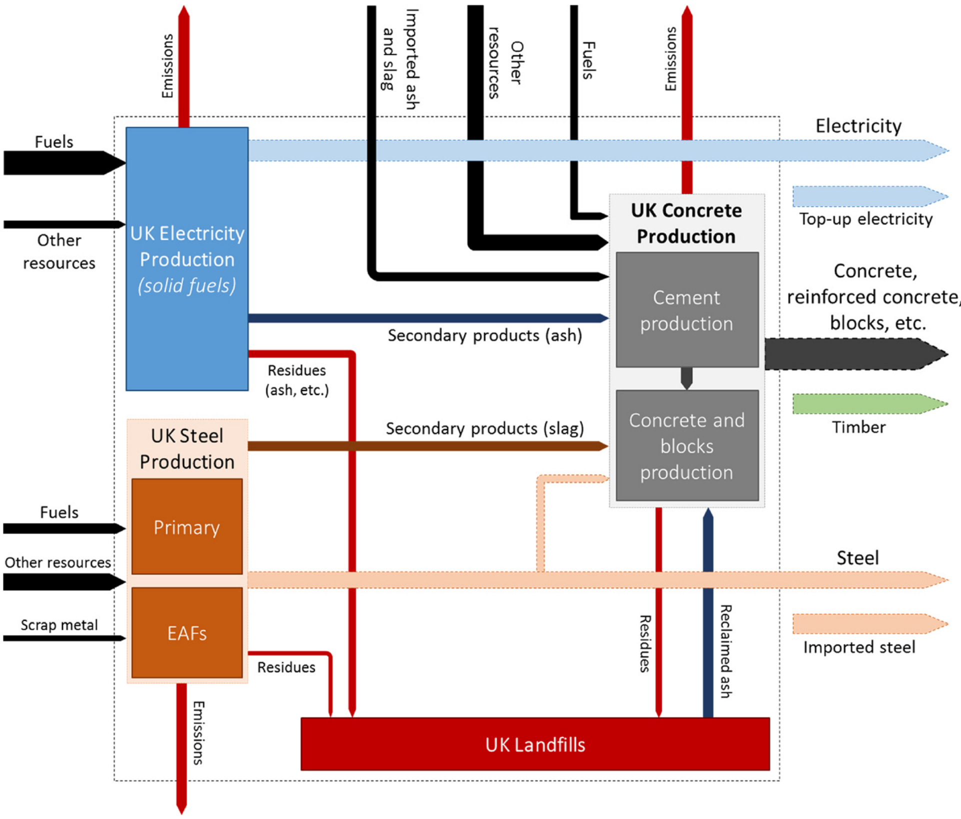

Fig. 1. Schematic diagram of the UK electricity, steel and concrete production sectors and the flows of materials and products between them. Steel production is split into primary production and secondary production from electric arc furnaces (EAFs). A more detailed version of this figure is included in the supplementary materials (also as Fig. 1).

the UK via our scenarios. Note that material flow analysis has its limitations, which we discuss in detail elsewhere (Iacovidou et al., 2017a). Nonetheless, it serves as a valuable foundation for the current work.

2.2. System Design

A schematic of the ESC sectors and their interlinkages (Fig. 1) shows the main processes and flows, with the latter grouped for clarity (*fuels*, *residues*, etc.). A more detailed schematic is included in the SM (Section 1). For electricity production, the inflows include fuels – coal, biomass and SRF – and other resources – limestone, aggregates, etc. The outflows include electricity, ashes utilised in concrete and elsewhere, and waste residues – emissions to air and solids sent to landfill disposal. The *co*-products utilised in concrete include bottom ash and PFA. Coal-PFA has various potential uses, but cement replacement is the highest value recovery route (Yao et al., 2015). Bottom ash from either coal or biomass can replace aggregates in various construction and infrastructure applications, but cannot be used as a cement replacement (Cabrera et al., 2014; Kalembkiewicz and Chmielarz, 2012). Similarly, PFA from biomass and SRF combustion cannot be used to replace cement in high-quality concretes (Kalembkiewicz and Chmielarz, 2012; Sarabèr, 2014), unless co-fired at low ratios with coal, or used to substitute only small proportions of cement (up to 10%) (Cheah and Ramli, 2011; Wang et al., 2008). This PFA is thus either disposed or used for lower value applications such as aggregates (UKQAA, 2016). UK utilisation rates for bottom ash and PFA have been ≈100% and 50–70%, respectively in recent years (UKQAA, 2016), but global utilisation rates have been much lower (Yao et al., 2015). Further, around five times more PFA is produced than bottom ash (UKQAA, 2016), making it a major pollution and waste issue. Therefore, our primary focus is PFA as a cement replacement for concrete production.

For UK steel we consider both primary and secondary (electric arc furnace; EAF) production routes. Secondary production currently forms only ≈17% of the total UK output of 11 Mt (EEF, 2016). Inflows to primary production are mostly imported iron ore and coal, with a small amount of scrap steel (DECC, 2015). Both production routes generate residues (Worldsteel, 2016), including blast furnace slag, basic oxygen furnace slag and EAF slag. A proportion of blast furnace slag can be converted to GGBS and thus used as a cement replacement, whereas other slags may only be used for lower value recovery routes, such as aggregates in road construction (Yi et al., 2012). There is thus a parallel with the ash residues of electricity production: residues from the lower-carbon processes (biomass combustion and EAFs) have lower value than those from related high-carbon processes. UK utilisation rates for steel slag are around 90% (Alberici et al., 2017), but international rates are lower (Yi et al., 2012).

We consider production of concrete, reinforced concrete and concrete blocks, via intermediate production of cement. Cement production requires inflows of fuels (coal, SRF, etc.) and resources (limestone, clay, etc.; MPA, 2016). Concrete requires cement, or cement substitutes, plus large amounts of aggregate materials. Historically, in the UK, cement substitutes such as ash and slag were sourced domestically, but they are now increasingly imported (Alberici et al., 2017). In contrast, aggregates are abundant, and also cheap and low-carbon (Hammond and Jones, 2011). Cement itself may be imported and the quantities are becoming significant, having recently reached ≈10% of UK consumption (BGS, 2015).

Residues that are not utilised must be disposed. In the UK disposals of steel slag are relatively small, but disposals of PFA have been 1.5–3 Mt/yr since 2000. It is estimated that ≈50 Mt of accessible stockpiled PFA exists around the country, but only a small proportion is likely to be recoverable (Alberici et al., 2017). We incorporate this into our results, but the impacts are small.

There are three flows in Fig. 1 that do not enter the system boundaries, namely *timber*, *top-up electricity* and *imported steel*. When UK production falls more than consumption in our scenarios, these flows allow UK consumption to be met and scenarios to be comparable. For example, when, in some scenarios, we assume a large decline in UK steel output, but only a modest fall in consumption, imports of steel increase to meet UK consumption.

Table 1
Basic characteristics of the electricity, steel and concrete sectors for each background scenario, summarised separately for the UK and globally (with rest-of-Europe and rest-of-Word referred to as RoE and RoW, respectively). Full details on the scenarios are in the supplementary materials, Section 3.3.

		BAU	GG	NSP	FW	BLB
Electricity	*UK*	**Medium** consumption & production	**High** consumption & production	**Low** consumption & production	**Medium** consumption & production	**Low** consumption & production
		Quick phaseout of coal	**Rapid phaseout** of coal	**Rapid phaseout** of coal	**Slow phaseout** of coal	**Slow phaseout** of coal
		Medium phasein of Biomass	**High phasein** of Biomass	**High phasein** of Biomass	**Low phasein** of Biomass	**Low phasein** of Biomass
	Global	**Quick phaseout** of coal in RoE **slower** in RoW	***As for BAU***	**Rapid phaseout** of coal in both RoE and RoW	**Slow phaseout** of coal in both RoE and RoW	***As for NSP***
Steel	*UK*	**Medium** consumption & production	**High** consumption & production	**Low** consumption & production	**Medium** consumption, **low** production	**Low** consumption, **v.low** production
		Medium fraction of secondary production	**High** fraction of secondary production	**V.high** fraction of secondary production	**Low** fraction of secondary production	**Low** fraction of secondary production
	Global	**Medium** production	**High** production	**Low** production	***As for BAU***	**Medium** production
		Medium fraction from secondary production	**High** fraction from secondary production	**High** fraction from secondary production	***As for BAU***	**High** fraction from secondary production
Concrete	*UK*	**Medium** production & consumption	**High** production & consumption	**Low** production & consumption	***As for BAU***	***As for NSP***
	Global	PFA & GGBS availability **medium**	PFA & GGBS availability **low**	PFA & GGBS availability **low**	***As for BAU***	***As for NSP & GG***

Regarding electricity, in 2017, electricity from solid-based fuels (coal, biomass and SRF) contributed $\approx$22% to total UK production. Coal is planned to be phased-out and biomass (and SRF) phased-in (EC, 2017), but the latter are not expected to fully replace the former (Beis, 2017; Stephenson and MacKay, 2015). Total UK electricity consumption is also expected to fall over the next decade due to efficiency gains, but if we only considered electricity from solid-based fuels this would (erroneously) exaggerate this fall in consumption. Therefore, as the electricity production of our system drops below 22% of the total projected UK consumption, we consider the *top-up electricity* needed to meet this 22%. Projected UK consumption is taken from government data (Beis, 2017) and we assume *top-up electricity* comes from the average of all other generation systems supplying the national grid (see SM, Section 3). Note that the substantial grid decarbonisation that all UK Government scenarios assume will be achieved over the next 10–20 years may or may not happen, but we nonetheless restrict our analysis to these scenarios.

2.3. Carbon emissions estimates

Using the material flow analysis as a foundation, we estimate GHGs for the ESC system. GHGs are normally divided into *direct*, *indirect* and *embodied* sources: *scope 1, 2 and 3*, respectively (Fong et al., 2015). *Direct* are those emitted on-site from combustion and industrial processes, *indirect* those associated with electricity and heat consumption, and *embodied* those associated with all upstream activities required to produce intermediate goods, fixed capital and materials used for production. For our system, *direct* emissions can be obtained from our material flow analysis, while *indirect* are obtained by estimating electricity usages for each process. *Embodied* values for the inflows to the system are obtained from the scientific literature (see SM Tables 3&4) or estimated as part of our analysis (e.g., our estimates for PFA and GGBS).

Estimating *embodied* emissions requires allocation decisions to be made where processes produce multiple outputs. European standards (EN 15804:2012) for assessing the environmental performance of construction materials – themselves based upon standard LCA procedures (ISO 14044:2006) – state that when *co*-products of processes contribute less than 1% to total revenues, no allocation of primary process emissions to these *co*-products is necessary. Above this 1% cut-off, an appropriate physical property or revenue-share approach should be used for allocations, even in the case of process wastes. Revenue- and mass-based methods are most widely used to assign environmental impacts to *co*-products in standard LCA, although other methods based upon energy or exergy can also be applied (Hanes et al., 2015). As the mass-based allocation is nonsensical for electricity production, and an energy/exergy approach appears inappropriate for PFA and GGBS, we use the revenue-based approach here for allocating GHGs to PFA and GGBS. Specifically, by multiplying the amounts sold with average prices, the respective revenues from electricity (or steel) and PFA (or GGBS) are estimated. The contribution of the *co*-product's revenue to the total process revenue is taken as the *allocation* coefficient, which is used to allocate upstream emissions to the *co*-product, with the remainder allocated to the primary product. Data sources supporting these calculations are described in SM, Sections 3.2.

2.4. Scenarios

The timeframe of our analysis is 2017–2050 and we produce material flows and GHG emissions trajectories over this period for a variety of scenarios. We develop what we refer to as *background scenarios* and *response scenarios* (the former has similarities to the 'frameworks of development' of Messner et al., 2006). The former refer to developments relating to (i) UK production and consumption of electricity, steel and concrete and (ii) UK and global transitions away from coal and primary steel towards biomass and secondary steel. We consider these transitions separately for the rest-of-Europe (excluding the UK) and rest-of-World. We consider five *background scenarios*: (i) Business As Usual (BAU), (ii) Green Growth (GG), (iii) New Sustainability Paradigm (NSP), (iv) Fragmented World (FW), and (v) Britain Left Behind (BLB). These are characterised roughly in Table 1 and in detail in SM, Section 3.3. The role of rest-of-Europe and rest-of-World trends is to determine how much useful PFA and GGBS may be available for UK imports. GG, for example, is a high-consumption scenario with very low coal use, high secondary steel production, and thus low global and UK availability of GGBS and PFA. BLB assumes UK consumption and production fall significantly due to unintended economic impacts (which may follow developments such as the exit from the EU), with steel production falling particularly rapidly, but the rest of the world is assumed to continue on a relatively successful decarbonisation pathway, so GGBS and PFA availability remains low.

The *response scenarios* describe three specific ways in which UK cement/concrete producers may react to shortages in PFA and GGBS, referred to as: *Minimal Imports* (MI), *Medium Supply* (MS) and *High Timber* (HT). MI assumes imports of GGBS and PFA are seriously restricted (which could occur for many reasons; geopolitical issues, waste legislation, prohibitive prices, etc.), while the MS scenario assumes imports are more easily available (see SM, Section 3.3). HT retains the same mass of imports of GGBS and PFA as the MS scenario, but assumes a higher level of timber use to reduce concrete demand. MI and MS response scenarios also assume some timber use, but this is much lower than in HT.

Two important points should be made here. First, options exist for reducing the carbon emissions of cement production that we have not incorporated into our scenarios, most significantly carbon capture and storage and novel low-carbon cements (which include those made with substitute materials other than GGBS and PFA). Our reason for excluding the former is simply that it is yet to proven on a large-scale. Regarding the latter, the official UK cement industry strategy assumes that novel low(er)-carbon cements will only contribute 5% to total cement output by 2050 and less before then (MPA, 2013). As these cements may still have over 50% of the embodied carbon of Portland cement (Barcelo et al., 2014), overlooking such options may have only a negligible impact upon our results. However, clearly it is possible that such novel cements will be developed and deployed much more rapidly than the industry foresees. Second, although we have chosen timber as a nominal replacement material due to its (perceived, but disputed) sustainability credentials, we recognise that it cannot always replace the functionality of concrete. For example, building foundations require large amounts of concrete that cannot be replaced by timber, concrete in infrastructure may often be replaced by steel, timber is (in practice) limited to medium rise buildings, and other materials (such as clay bricks) have a role to play. Assessing how much of UK concrete consumption could realistically be replaced by timber is itself a phenomenally big research question. We thus make a supply-based assumption, namely that current timber consumption for construction roughly doubles from 2017 to 2035 and that all this extra timber can replace a fraction of UK concrete consumption, and this could prove generous. We discuss these assumptions further in the SM, Sections 3.3 and 3.4.

Our *background* and *response* scenarios must be combined to form final scenarios and associated model outputs. For each final scenario we use notation such as GG-MI, which refers to a combination of GG *background* conditions and the MI *response* scenario. The fifteen final scenarios allow us to consider a wide range of future industrial pathways, while simultaneously observing the sensitivity of the model to different inputs. In addition, we undertake a sensitivity analysis where we consider variations and uncertainties in a number of our other inputs.

3. Results and discussion

3.1. Production and consumption of recovered resources

We first consider resource recovery trends across our scenarios by examining UK consumption of cementitious materials and limits placed upon the latter by production in UK, rest-of-Europe and -world. Cementitious materials include cement and supplementary cementitious materials ('cement substitutes' herein), PFA and GGBS.

Fig. 2 (top) shows projected consumption of cement and cement substitutes for the GG-MS scenario, next to the cement industry's target for substitution (30% of cementitious materials to be substituted by 2050; MPA, 2013). Domestic supplies of GGBS and PFA appear significantly less than needed to meet the target, leaving the UK increasingly reliant upon imports. However, beyond 2040 assumed limits upon the accessibility of GGBS and PFA by the UK mean that even imports are not enough to meet UK industry targets for cement substitutes. The consequence is a 'shortfall' in cement substitutes relative to this target and an increase in cement consumption.

Fig. 2 (bottom left) shows this is exacerbated when supplies of cement substitutes in the GG-MS scenario are lower than our central assumptions dictate (the *low supply* case shown), or when imports are seriously restricted (GG-MI scenario). Fig. 2 (bottom right) shows shortfalls occurring across the majority of scenarios. Further, shortfalls would be much greater if industry targets for cement substitutes were

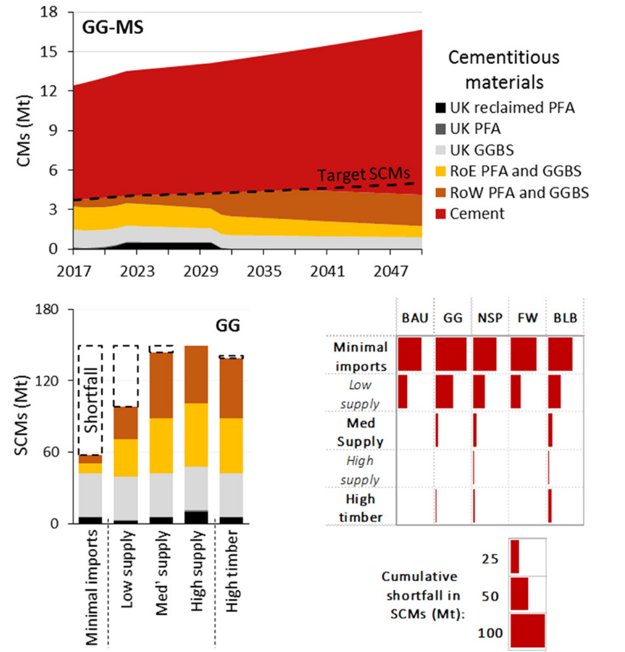

Fig. 2. *Top*: Annual UK consumption of cementitious materials (CMs) and supplementary cementitious materials (SCMs) in the *GG-MI* scenario. *Bottom left*: Cumulative consumption of SCMs from 2017 to 2050 in the *GG* scenario across all *response* scenarios, with a sensitivity to SCMs supplies performed on the MS scenario (which results in the *low* and *high supply* cases shown) and showing also the shortfall behind the (30%) industry target. *Bottom right*: shortfalls in SCMs across all combinations of background and response scenarios, again with a sensitivity test on the MS scenario. Note, rest-of-Europe and rest-of-Word are referred to as RoE and RoW, respectively.

more ambitious and reflected technical limits for substitution. It is also notable that while high timber usage lowers cement demand the effect is small, and hence shortfalls are mitigated only slightly by this response scenario.

Irrespective of these shortfalls, it is clear that substantial quantities of recovered resources are expected to be utilised as cement substitutes in the UK and, in the future, the majority of these may be imported. It is thus useful to ask how recovered resources are integrated into carbon accounting: what is, or should be, the emissions allocated to recovered resources like PFA and GGBS and how do these compare to those of the primary resources that they replace?

3.2. Recovered resources and embodied carbon

Processes outputs considered to be 'wastes' are typically not allocated any of the GHGs occurring upstream, but when wastes are recovered as resources and the revenues they generate become significant this can change, as explained in Section 2.3. Here we consider the potential embodied carbon of GGBS and PFA in terms of *secondary-processing* and *allocated* emissions (note that we do not include transport emissions; see SMs). *Secondary-processing* emissions are the direct and indirect emissions related to the processing of GGBS or PFA for substituting cement; the processing occurring after primary processes – here steel and electricity production – have taken place. For PFA, these emissions are negligible (approx. 4 kg.CO_2e/t.PFA), while the grinding processes involved in producing GGBS from steel slag are more significant (approx. 67 kg.CO_2e/t.GGBS; Leese and Casey, 2015). *Allocated* refers to the proportion of supply chain emissions – from primary production processes themselves and further upstream – that could arguably be allocated to recovered resources.

Consequently, the uncertainties in estimates of embodied carbon of recovered resources can be substantial and arise from various sources:

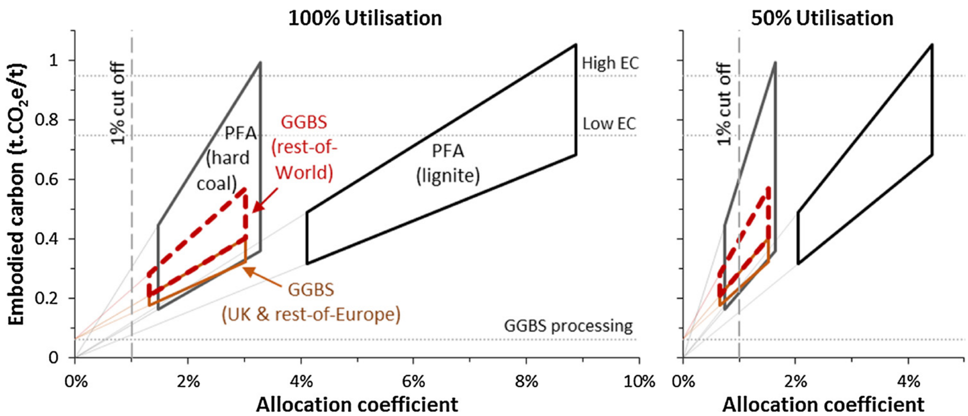

Fig. 3. The embodied carbon that may be allocated to GGBS and PFA under various assumptions, as described in the text. Also shown are horizontal lines indicating high and low embodied carbon cements and the emissions relating to processing steel slag into GGBS. A vertical line is also shown for the revenue-share-based cut off of 1%, below which allocation to *co*-products need not be considered under European standard EN 15804:2012.

(i) **accounting** uncertainties as to how (if at all) carbon will be allocated to *co*-products in accounting frameworks, (ii) **economic** influences when revenue-share allocation is applied, due to price and exchange rate volatility and (iii) **physical** or technical factors relating to plant efficiencies and resource yields, etc.

In Fig. 3 (left) we estimate the amount of embodied carbon that may be allocated to GGBS or PFA, including both secondary-processing and allocations from primary production via the revenue-share approach. The left hand figure assumes a 100% utilisation rate and the right a 50% rate – this has no effect on the embodied carbon of PFA and GGBS, but is important for reasons explained below. Horizontal variations in estimates in Fig. 3 depict economic uncertainties as they come from varying *only* the prices of PFA and GGBS. (For practical purposes, described in the SM Section 3.2, we hold steel and electricity prices constant across scenarios and in time, so variations in allocation coefficients arise from the range of assumed PFA and GGBS prices only). Vertical variations come from differences in LCA GHG data found in the literature for coal-fired power and primary steel production. We show separately PFA from high (hard coal) and low (lignite) quality coal, and GGBS from high and low efficiency primary steel production, which we categorise (crudely) here as rest-of-Europe/UK and rest-of-World, respectively. Further information and sources supporting these calculations are given in the SM Section 3.2.

The calculations shown in Fig. 3 suggest that low (thermal) plant efficiencies and high prices for cement substitutes could lead to PFA being assigned higher embodied carbon than even the highest-carbon cements. This is a more likely outcome for lower quality coals with high ash and low heat content, such as lignite (see SM Section 3). In contrast, the (revenue-share based) embodied carbon of GGBS is always lower than cement, and is lower than PFA under most conditions and assumptions. This broader perspective gained by considering *allocated* emissions thus reverses the picture seen when only *secondary-processing* emissions are considered – as noted above, from the narrower perspective of the latter, GGBS appears of higher embodied carbon than PFA due to the energy required to grind steel slags.

Consequently, under revenue-based allocation, importing PFA from countries with old, inefficient coal-plants and/or poor quality coal would not necessarily lower the embodied carbon of concrete production any more effectively than increasing cement production would. In contrast, utilising PFA from more efficient plants, or utilising GGBS, would most likely lower the embodied carbon of UK concrete.

In practice, however, recovered resources are assumed low-carbon, perhaps largely as revenues are considered insignificant. More specifically, UK industry studies have concluded that the 1% cut off described above (EN 15804:2012) applies to PFA and GGBS, i.e. the revenues they contribute are insufficient to necessitate any allocation of upstream or primary process emissions (Leese and Casey, 2015). In contrast, variations in our estimates suggest that such a conclusion depends upon the particular assumptions made.

Firstly, this is because the revenue share contribution of PFA or GGBS varies substantially with the assumed prices of primary- and *co*-products. Our estimates in Fig. 3 are wide and, as described in the SM (Section 3.2), allocation coefficients may be higher/lower than these, due to geographical and temporal variations in electricity and steel prices that we have not captured. Secondly, we have not considered avoided disposal costs, as they are not strictly revenues – but they do contribute to the financial benefit of recovering resources and it could thus be argued they should be added to revenues when assessing both the revenue-share and the 1% cut-off criteria. Finally, a comparison between the two graphs of Fig. 3 shows the influence of assumptions regarding utilisation rates. The revenue share contribution of PFA or GGBS must be evaluated over a particular geographical and temporal scale – for example, for a single plant over one week, or for the whole UK over a full year. Results vary, as a plant close to a major cement producer may sell almost all the residues they produce while another may dispose the majority. In the latter case the revenue-share will likely be well under 1%, such that no allocation of upstream emissions to *co*-products would seem necessary under EN 15804. However, estimates of the embodied carbon per tonne for PFA and GGBS are the same irrespective of this utilisation rate, as Fig. 3 shows for rates of 100% and 50% – while the revenue shares (and allocation coefficients) in the latter case drop 50%, the mass of materials receiving an allocation of GHGs falls by 50% as well (as disposed resources do not contribute revenues and should not be allocated GHGs). In summary, assumptions can easily be made (unintentionally or otherwise) that make revenues from recovered resources appear insignificant, such that no allocation of upstream emissions is undertaken.

Against this backdrop, we now consider how the revenue share approach we have taken influences the total GHGs assigned to the production of concrete across our scenarios.

3.3. Carbon emissions for concrete production

Our projections of the carbon emissions related to UK concrete from 2017 to 2050 – including *direct*, *indirect* and *embodied* sources and timber replacements – vary significantly across the scenarios. Fig. 4 (left) shows these projections across all combinations of background and response scenarios, each with and without an allocation of emissions to GGBS and PFA (using our central estimate; see SM Section 3.2). They range from indicating emissions increases of over 15% to decreases of just over 40% by 2050; few scenarios achieve significant reductions.

Fig. 4 (left) shows the BAU background scenario with a breakdown of emissions into the three sources – *direct*, *indirect* and *embodied* – and with emissions *allocated* to cement substitutes by revenue-share shown separately. (Note, however, that our results do not allow an estimation of where, geographically, these embodied emissions occur and thus what proportion of them would be subject to UK Government targets.)

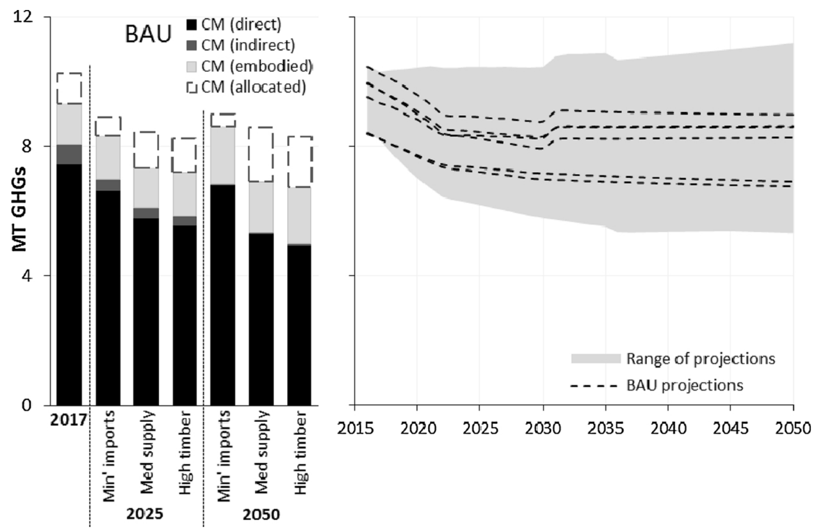

Fig. 4. *Left*: breakdown of greenhouse gases (GHGs) into various sources for BAU scenarios. *Right*: annual GHGs for concrete production (and timber substitutions) across all combinations of background & response scenarios, repeated with and without allocation of upstream GHGs to cement substitutes.

Direct and indirect emissions together make up about 80% of 2016 emissions. But their relative contribution falls over time, most significantly for the *high timber* response scenario where they drop to 60% by 2050. Direct and indirect emissions are thus replaced by embodied emissions. This trend occurs for many other scenarios as well and, further, it is reported across the UK economy more broadly (Barrett et al., 2013). UK climate targets, however, remain focused upon direct and indirect emissions (Scott and Barrett, 2015).

There are of course large uncertainties in these estimates. Fig. 5 examines uncertainties as they arise from (i) *background* scenarios, (ii) *response* scenarios, (iii) data uncertainties and (iv) *accounting* uncertainties (i.e. how, if at all, a revenue-based allocation of upstream GHGs to cement substitutes is applied). Data uncertainties appear as the largest source of uncertainty, and may be larger than those shown here (see SM Section 3.4). Different *background* and *response* scenarios are also significant, the former slightly more so. The uncertainties arising from accounting conventions are as significant as the *response* scenarios; indeed, this simple decision to allocate GHGs has as much influence on the emissions assigned to concrete as the variety of ways that we assume the sector may respond and evolve.

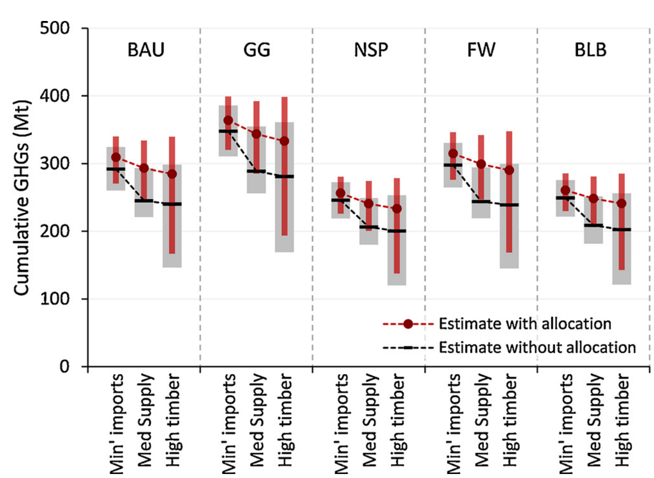

Fig. 5. Cumulative greenhouse gases (GHGs) for concrete production (and timber substitutions) for years 2017–2050. Central estimates for each combination of scenarios are given by the horizontal lines and red dots, with and without allocation of impacts to cement substitutes, respectively. Data uncertainty ranges are captured by the bars (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

There are obvious interrelations as well. For example, accounting decisions are substantially more important when consumption of cement substitutes is high, which, in turn, occurs when imports are higher. When timber usage is high, data uncertainties increase due to large uncertainties in the embodied carbon of timber. While low-carbon timber products undoubtedly exist, timber per se cannot always be considered a low carbon material and debates around this issue are heated (Purnell, 2012; Sathre et al., 2012; Sathre and O'Connor, 2010). Accordingly, timber's embodied carbon in our inputs ranges substantially, from negative to positive (-0.9 to + 0.6 t.CO_2e/t.timber, respectively). Taking into account these uncertainties, the important question to ask is how these changes in assigned emissions compare to those occurring upstream in steel and electricity production?

3.4. Carbon emissions across the sectors

Fig. 6 shows GHGs across the ESC sectors for three different final scenarios: GG-HT, NSP-MS and BLB-MI. These are again split into *direct*, *indirect* and *embodied* emissions, but we separate some emissions sources out for visualisation purposes. For concrete/cement production, *allocated* emissions are shown separately, for electricity *top-up* emissions are shown separately, and for steel *net-imports* are shown separately.

Perhaps the most significant point to recognise from Fig. 6 is that emissions reductions for UK electricity are substantial, even for scenarios that are slow to phase-out coal, i.e. FW & BLB. This occurs despite the large increases in embodied carbon due to biomass, which fade out themselves after 2040 due to our assumption that biomass is eventually scaled-down. Emissions reductions for UK steel are also large across all scenarios, although where UK production slows-down substantially (e.g. BLB) there are significant increases in embodied carbon from imported steel.

In contrast, emissions relating to concrete/cement and timber are almost flat. They begin at approximately a third of electricity sector emissions and half of those from steel, but by 2050 all ESC sectors are relatively similar. And the small reductions in direct and indirect emissions that are achieved for concrete and cement can be wiped out when embodied carbon is considered (NSP-MS and BLB-MI; Fig. 6). This conclusion is in line with other work into the sector's decarbonisation potential, which generally find that the potential for cement production to be decarbonised is restricted due to unavoidable process emissions and theoretical limits regarding efficiency (Allwood et al., 2012). Even

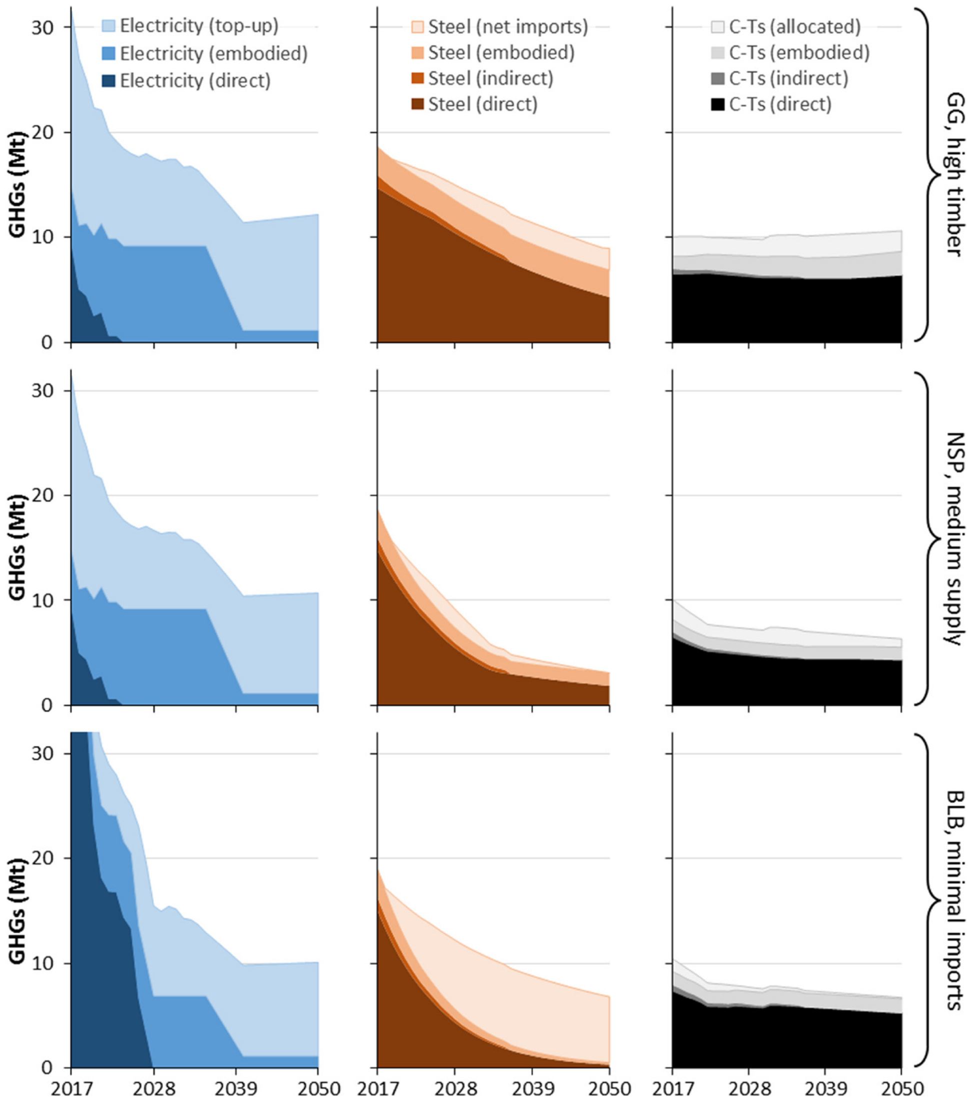

Fig. 6. Total greenhouse gases (GHGs) across the electricity, steel and concrete sectors, split by emissions source (direct, indirect, etc., as explained in the text), for a select three scenarios: GG-HT, NSP-MS and BLB-MI. 'C-Ts' refers to concrete and timber substitutions. Note also that the vertical axis on the bottom row is cut for visualisation purposes – coal-based electricity here leads to approx. 50 Mt of GHGs in 2017.

the industry's official low-carbon strategy promises less than a 30% reduction in (*direct* and *indirect*) emissions from now to 2050 in the absence of carbon capture and storage (MPA, 2013), which, we reiterate, is not included in our scenarios. Further, this reduction assumes that demand stays flat, while other reports by the same industrial body make it clear they are hoping for growth.

We emphasise that our results cannot be compared directly with the official industry strategy, as we consider embodied as well as direct and indirect emissions. Nonetheless, what we add to these debates is that allocating even a very small proportion of upstream emissions to PFA and GGBS

leaves GHG trajectories for concrete/cement production almost flat, and may even cause the associated emissions to rise (e.g. GG-HT scenario; Fig. 6). Increases in cement production in the face of shortfalls in these recovered resources could make the situation still worse.

3.5. The potential consequences of allocation decisions

So far, we have discussed the emissions that may be allocated to concrete production via the recovery of cement substitutes under a particular (revenue-based) allocation convention. But we have not discussed i) whether this (re)allocation reflects real changes in systemic GHG emissions, or ii) whether allocation decisions themselves may influence these systemic emissions. Could resource recovery and the associated allocation decisions have systemic feedbacks that influence industrial transitions, or are these decisions simply accounting exercises that shift (inevitable) emissions between sectors' balance sheets?

In the case of GGBS and PFA and the ESC sectors, arguments could be made both ways. On one hand, it could be argued that GHGs allocated to recovered resources reflect increases in system-wide emissions, compared to if these residues were disposed (Millward-Hopkins et al., 2018). For steel and coal-based electricity producers, trade in recovered resources is offering increasingly non-negligible financial benefits. Fig. 3 indicated that revenues from PFA and GGBS could potentially range from approx. 1% to 9% of these producers' incomes and, as we discussed above, the range could be much larger given additional variations in prices. This could provide a non-negligible incentive to continue these activities, thus delaying the transition to lower-carbon activities. Further, given their potential scarcity internationally, current low-carbon credentials, and high technical value – i.e. the fact that they improve concrete quality – prices of GGBS and PFA could potentially exceed that of cement, thus rising to the upper end of our assumed range of prices. One could thus speculate that if 100 coal plants across India, China, Turkey, etc. were in the stages of planning, but were at the margins of financial viability, the extra revenues expected from the sale of residues to international buyers could tip a handful of these towards being financially viable, thus nudging planning decisions towards a green light. From this perspective, by purchasing PFA, UK cement and concrete producers would have a real influence on the magnitude of coal-based electricity produced elsewhere – albeit a small one. However, these arguments are perhaps weaker for GGBS than for PFA, as coal-based electricity can be replaced by other forms of electricity with no loss in usefulness, but high-quality primary steel cannot (currently) be easily replaced by secondary materials (Reck and Graedel, 2012).

However, financial considerations are clearly not the only factors involved. Factors such as energy security, domestic job creation and decarbonisation plans may dominate the investment decisions of coal-based power generators and primary steel producers, leaving the financial benefits of recovering PFA and GGBS with no perceptible influence. The changes in value are complex and go far beyond the economic (Iacovidou et al., 2017a). From this perspective, anything that may disincentivise the recovery of resources that are (inevitably) going to be produced should be avoided. Allocating emissions to PFA and GGBS may create precisely such a disincentive and may result in more waste being disposed of, more primary resource use, and more emissions. In-depth analysis of the technical, environmental, political and economic conditions of the ESC system – including qualitative research investigating the factors influencing the decisions of actors and stakeholders involved in coal-based electricity and primary steel industries – would be hugely valuable areas of future research. Such research could explore which (if any) of our above speculations most closely resemble reality, and under what circumstances.

3.6. Beyond greenhouse gases

While our analysis so far has focused exclusively upon GHGs, this reference to waste disposal and primary resource use hints at the broader issues at stake. The idea that assessments of sustainability must look beyond environmental impacts to also consider broader changes in social, technical and economic domains of value – changes in what we may call complex value (Iacovidou et al., 2017b) – has long been held (UNCED, 1992). A vast literature has developed assessment methods that take a lifecycle perspective to assess changes in value across domains, but these ideas remain notoriously difficult to put into practice (Sala et al., 2013; Valdivia et al., 2013; Zamagni et al., 2013). Applying the framework we have previously developed (Iacovidou et al., 2017a) to the current case study would open up a huge range of environmental, social, technical and economic factors to be considered beyond GHGs, and we discuss some of these below.

We have focused upon the potential GHG issues of the UK importing PFA or GGBS from other countries, but there are obvious benefits as well. First, adding PFA or GGBS to concrete not only reduces cement consumption, but improves concrete quality, increasing strength and decreasing permeability (which in turn greatly improves structural durability; Thomas and Matthews, 2004). Hence even if the embodied carbon of domestic cement and imported PFA were deemed similar, the latter retains a performance advantage. However, perhaps more important are the benefits in the exporting country. Ash disposal sites present well-known local environmental and health hazards (Yao et al., 2015), with risks for local populations ranging from ongoing releases of airborne particulates (Guttikunda and Jawahar, 2014) to the catastrophic impacts that follow when disposal sites fail (Ruhl et al., 2010). And those most vulnerable to such risks are typically the less well off (D-waste, 2014). But by exporting coal ash, countries such as India, China or Turkey – who have much lower ash utilisation rates than those achieved in Europe (Yao et al., 2015) – may be able to minimise waste disposal issues that would otherwise accumulate in their own back yards. These countries may therefore offshore their own environmental issues, just as economies like the UK have been doing for decades.

On the other hand, there are questions regarding risks that may be offshored completely, such as the risk that, at some point, a ship carrying thousands of tonnes of ash will sink in transit. One could also conjecture that the offshoring of social and environmental issues is precisely what has allowed countries like the UK to maintain such unmitigated high-consumption lifestyles – if social and environmental issues of consumption had not been so effectively externalised, high-impact consumption patterns may have received greater attention. This again raises the question of whether, by importing recovered resources, a country such as the UK may have a tangible (negative) impact on the industrial activity in the exporting country.

Many other broader impacts could be discussed, but we restrict our remaining discussion to those relating to timber, as these are particularly diverse. Considering our results – despite the substantial uncertainties – increasing the replacement rate of concrete by timber certainly appears far from a silver bullet for reducing the emissions associated with the former: even a massive increase in timber use above current levels could only displace a relatively small quantity of total concrete use. Timber's environmental and technical credentials for domestic buildings are strong (Sathre and O'Connor, 2010), but concrete will likely dominate buildings' foundations, infrastructure and large commercial buildings (Purnell and Roelich, 2015). However, timber could have many more advantages for the UK if well managed, and particularly if grown domestically. The UK Government have put a figure on the value of the UK's current woodlands that reaches billions of pounds, once forests benefits for recreation, carbon sequestration and air pollution removal are added to the direct benefits of selling timber (ONS, 2017). And absent from this list are the potentially huge benefits for flood management (Bradshaw, 2009), if dialogues with British farmers over land-use can be depolarised (Wynne-Jones, 2016). There may be further benefits if waste from the timber industry was used to help meet the UK power sector's rapidly increasing demand for biomass pellets, which are currently sourced largely from the USA (Drax, 2013).

These broader questions have, however, remained beyond the scope of our main analysis. Answering such questions would be essential for any holistic sustainability analysis of resource recovery from waste and a consideration of such '*complex value*' would be the natural direction for future research into this system to take (Iacovidou et al., 2017a).

4. Conclusions

We have developed a simple inter-industry model integrating the UK electricity, steel and concrete sectors in order to investigate interactions between resource recovery practices and industrial low-carbon transitions. Across a broad range of scenarios and assumptions, our results show that a continuation of domestic shortages in cement replacements (GGBS and PFA) is inevitable and future global shortages quite possible. This may present problems for cement and concrete producers relating to their GHG reduction targets, resource security, and the technical value of their outputs. But clearly our results don't suggest that it makes sense to keep burning coal and producing primary steel simply to supply cement/concrete producers with useful co-products, as the potential emissions savings in the electricity and steel sectors dominate (questions of the sustainability of biomass supply chains aside). Nonetheless, our results emphasise the significance of the trade-off that is occurring, which must be recognised and managed.

But we have also argued that even if international markets succeed in minimising shortfalls in UK cement replacements, the implications may not be entirely benign. Some allocation of the impacts of high-carbon activities to resources such as GGBS and PFA may be reasonable, given the potential for interactions with low-carbon transitions, and this may leave the embodied carbon of some types of coal ash not too dissimilar to that of cement. Currently though, internationally traded PFA and GGBS may stand behind a *double-blind* system of accounting. This is because, firstly, official UK carbon accounts, based upon territorial carbon accounting, are (by definition) blind to the emissions embodied in trade (Millward-Hopkins et al., 2017) – as are all national accounts following the standard United Nations framework (Scott and Barrett, 2015). Consumption-based accounting addresses this blindness by considering the full supply chains of goods and services consumed by the population of a geographical area (Peters, 2008). However, when *co*-products such as PFA and GGBS are considered to offer negligible revenues (perhaps incorrectly, as data can be chosen selectively) and are thus allocated none of the impacts of the processes that produce them, they will not register even in consumption-based emissions accounts. In this sense, conventional territorial emissions accounting may be doubly blind to them. The only way to address this is to combine full supply-chain analyses with allocation methods designed in a way that they support effective low-carbon transitions. As a first step, this may involve applying allocation procedures even when the revenues generated by recovered resources appear, on first glance, insignificant, as their embodied carbon may turn out to be surprisingly high.

The USA's situation is particularly strange. In 2016, over 10% of the coal exported by the USA went to India (IEA, 2017b). The same year, journalists reported that ships full of ash from Indian power plants were received by ports on the East Coast of the USA. It's quite possible that there are now blocks of concrete sat in the USA that contain ashes from coal that was extracted from an USA mine, but burnt in an Indian power station. Yet the emissions from burning that coal will appear nowhere in the USA's official carbon accounts, despite them digging the coal up and reclaiming a significant share of its value via both the sale of the coal and recovery of the high technical value of the ash.

The conclusions that we have drawn above have been made possible by expanding the assessment of a domestic (UK) resource recovery issue into a multi-regional and inter-industrial analysis, which incorporates highly interconnected sectors, while considering a diversity of scenarios that capture an equally diverse range of potential socio-political developments. Future research could elaborate upon ours in various ways. For example, future analysis may refine the global component of our model with more dynamic estimates of global developments in electricity and steel production (in particular, by including how prices vary geographically and temporally). Such research could also integrate the various novel technical options for lowering the carbon emissions of concrete production that are currently in the development stage (Barcelo et al., 2014). Further, qualitative research approaches investigating the link between the increasingly significant revenues generated by the resources recovered by high-carbon production processes and the decisions of actors and stakeholders directly involved in these processes, would be hugely valuable. This may offer insights into the consequences of accounting conventions for decision-making, thus allowing recommendations to be made regarding what conventions would support effective low-carbon transitions.

A broad conclusion that we can draw from our present work is that significantly reducing the GHGs of concrete production will require reducing consumption; an argument frequently directed at the modern industrial economy more broadly (CIE-MAP, 2015). As others have highlighted, reduced consumption can be achieved via material efficiency at the production stage (e.g. prefabricated concrete components, lightweight structural steel elements), effective maintenance and repair practices, improvements in construction design, and end-of-life management of infrastructure assets that promotes reusability and longevity (Allwood et al., 2012; Iacovidou and Purnell, 2016). For concrete, this does not necessarily even require innovation, but simply a return to production methods that were used when the ratio of labour to material costs was lower, such as the tapering of concrete beams. It is crucial that the increasing trade in recovered resources does not become a distraction from this necessary challenge.

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