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# Fully integrated modelling for sustainability assessment of resource recovery from waste

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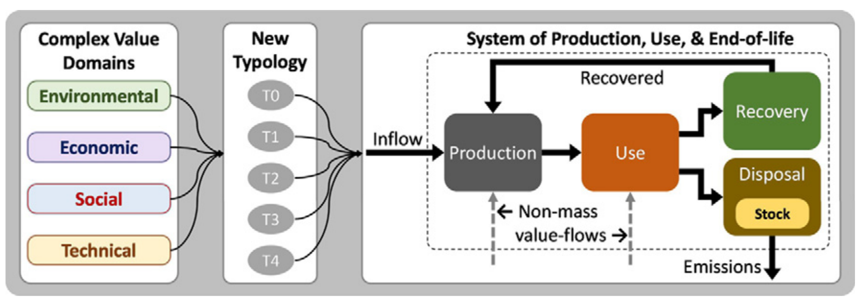
## HIGHLIGHTS

- We develop a multidimensional model for assessing resource recovery systems.
- Social, environmental, technical and economic domains of value are fully integrated.
- We propose a new typology for metrics better suited to integrated modelling.
- We apply the model to a case linking electricity and concrete/cement production.
- Interdependencies between domains and temporal dynamics can be modelled.

## GRAPHICAL ABSTRACT

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## ABSTRACT

This paper presents an integrated modelling approach for value assessments, focusing on resource recovery from waste. The method tracks and forecasts a range of values across *environmental*, *social*, *economic* and *technical* domains by attaching these to material-flows, thus building upon and integrating unidimensional models such as material flow analysis (MFA) and lifecycle assessment (LCA). We argue that the usual classification of metrics into these separate domains is useful for interpreting the outputs of multidimensional assessments, but unnecessary for modelling. We thus suggest that multidimensional assessments can be better performed by integrating the *calculation methods* of unidimensional models rather than their *outputs*. To achieve this, we propose a new metric typology that forms the foundation of a multidimensional model. This enables dynamic simulations to be performed with material-flows (or values in any domain) driven by changes in value in other domains. We then apply the model in an *illustrative* case highlighting links between the UK coal-based electricity-production and concrete/cement industries, investigating potential impacts that may follow the increased use of low-carbon fuels (biomass and solid recovered fuels; SRF) in the former. We explore synergies and trade-offs in value across domains and regions, e.g. how changes in carbon emissions in one part of the system may affect mortality elsewhere. This highlights the advantages of recognising complex system dynamics and making high-level inferences of their effects, even when rigorous analysis is not possible. We also indicate how changes in social, environmental and economic '*values*' can be understood as being driven by changes in the *technical value* of resources. Our work thus emphasises the advantages of building fully integrated models to inform conventional sustainability assessments, rather than applying hybrid approaches that integrate outputs from parallel models. The approach we present demonstrates that this is feasible and lays the foundations for such an integrated model.



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## 1. Introduction

The increasing recognition that human activities are seriously impacting on the planets capacity to support civilisation (Rockstrom et al., 2009) has led to a wide range of strategies to decarbonise and dematerialise the global economy. Achieving such significant changes will require (IPCC, 2014) more efficient production processes, more sustainable consumption patterns, radical reductions in energy and material use and waste generation, enhanced recovery of resources, and a socio-political environment amenable to such a transition (Bailey and Wilson, 2009).

The limited remaining scope for improvements in technological-efficiency of individual production-processes (Allwood et al., 2012) makes it essential for environmental impacts and material demands of production and consumption to be considered systemically. This *lifecycle* thinking is central to concepts such as *Sustainable Consumption and Production* (SCP) (Lebel and Lorek, 2008) and *Circular Economy* (Gregson et al., 2015), and to various established methods for environmental impact assessments such as *Life Cycle Assessment* (LCA) (Guinée et al., 2011), *Material Flow Analysis* (MFA) (Cencic and Rechberger, 2008), *Environmental Cost-Benefit Analysis* (ECBA) (Atkinson and Mourato, 2008) and *Environmentally Extended Input-Output Analysis* (EEIOA) (Barrett et al., 2013).

However, it has always been recognised that sustainability assessments must look beyond environmental impacts to consider a concept of sustainability encompassing all three primary domains of value: environmental, social and economic (UNCED, 1992; Zamagni et al., 2013). Accordingly, a number of methods have been developed (Sala et al., 2013b) that typically apply techniques similar to LCA in other domains, such as *Social Life Cycle Assessment* (sLCA) and *Life Cycle Costing* (LCC). Over the past decade, researchers and practitioners have worked to unify these into *Lifecycle Sustainability Assessment* (LCSA) (Guinée et al., 2011; Kloepffer, 2008; Sala et al., 2015). The purpose of these developments has been to create a robust and comprehensive sustainability assessment methodology that addresses three key challenges:

1. the need for systemic approaches that combine a lifecycle perspective with a triple bottom line accounting of impacts;
2. a recognition of the interdependencies between environmental, economic and social domains of value; and
3. the ability to capture the disparate and potentially conflicting perspectives of stakeholders required for transparent decision support.

The operationalisation of LCSA frameworks remains an ongoing project with relatively few practical implementations (Onat et al., 2017; Sala et al., 2013a, 2013b).

In contexts of Resource Recovery from Waste (RRfW), integrated social, economic and environmental assessments of different system configurations are rare (Chong et al., 2016). However, to fully understand the impacts and benefits of maximising resource recovery, it is essential that systemic assessment methodologies are developed that consider interdependencies, synergies and trade-offs between different domains.[1] As in sustainability assessments more generally, attempts to maximise environmental and/or economic outcomes are not always compatible with desirable social outcomes (Velis, 2015). Developing such methods will allow systems to be designed that help us (i) move away from end-of-pipe solutions and look upstream to consider how production and consumption can be reconfigured such that materials are more easily recoverable, (ii) minimise detrimental impacts and maximise positive ones, i.e. ensuring diverse sets of values are optimised, and (iii) build resilience in the context of the social, political and economic forces and actors motivations that shape the dynamics of such systems.

[1] Indeed, the very concept of *waste* relies upon a unidimensional mode of evaluation: i.e. a zero or negative economic value, within the contemporary political economy.

The *CVORR* project (complex value optimisation for resource recovery) aims to develop such an assessment framework for RRfW systems. We consider *complex value* to be a multidimensional variable, comprising potentially incommensurable sets of individual values. These can display diverse behaviours during modelling and analysis, including complex interdependencies (as described later in Section 3), and they may be quantitative or qualitative. The framework under development is composed of three sequential processes: selection of appropriate metrics, integrated modelling, and a multi-criteria decision analysis of outputs. These are all grounded in a political economy narrative to gain insight into the socio-political context of the system being studied (Brown and Robertson, 2014). The wider framework and metric selection are presented elsewhere (Iacovidou et al., 2017a, 2017b). Here we focus upon the development and conceptualisation of the integrated model. The primary novel contribution of the model is that it offers a typology that brings values across all domains into a common framework. This, in turn, allows for an integrated assessment of complex value in which interdependencies between domains are considered.

In this article, we first outline the broader context of sustainability assessments methods and introduce a simple case study analysing links between the UK electricity-production and concrete industries. Second, we describe the structure of the model, discussing its conceptual and mathematical foundations, the required input data, and our new typology for classifying metrics of complex values. Third, we apply the model to the case study to demonstrate the value of the approach. We then draw our conclusions.

## 2. Background and methodology

### 2.1. Conceptual background

The concept of sustainable consumption and production (SCP) has climbed up the global political agenda in recent decades. It now forms the twelfth of the United Nations *Sustainable Development Goals* for 2030 and a crucial aspect is to drastically cut the generation of waste via prevention, reduction, recycling and reuse (UN, 2015). SCP aims to address sustainability in a comprehensive and holistic manner, going beyond engineering and technological solutions to also look at issues such as the dependence of consumption patterns on collective vs. individual psychology and their impacts upon wellbeing (Jackson, 2005).

The goals and value judgements of SCP analyses are particularly explicit, depending broadly upon whether researchers' tend more towards reform, revolution, or reconfiguration of current social, economic and political structures (Geels et al., 2015). Such values are highly relevant to strategies for resource recovery and waste management, as they may determine to what degree intellectual and political resources are directed towards, for example, upstream demand management or downstream waste processing systems. Moreover, they may determine where in the lifecycle of products waste reduction interventions are applied and at what actors they are aimed (households, businesses, etc.). Such values may also affect the design of sustainability assessments more broadly via the choices made when selecting methods, metrics, system boundaries, and allocation coefficients for secondary products (Hanes et al., 2015; Sala et al., 2013b). For example, the monetising of environmental and social impacts in ECBA (Kallis et al., 2013; McCauley, 2006; Millward-Hopkins, 2016) is a contentious approach that opponents have argued is fundamentally incompatible with sustainability science (Anderson et al., 2015).

For environmental assessments in contexts of resource recovery from waste, methods such as MFA and LCA are widely applied (Allesch and Brunner, 2014). Reviews of MFA applied to RRfW have indicated that it is valuable for observing how waste management systems function and understanding the pathways hazardous substances take through systems (Allesch and Brunner, 2015). Bespoke LCA tools have been developed for waste management (easetech, 2017) and reviews of LCA applied to RRfW have highlighted the additional advantages of including the impacts of capital equipment and maintenance in assessments (Cleary, 2009).

However, there remain a number of challenges relevant even for these (one-dimensional) environmental assessments. For example, in resource recovery contexts, only a small proportion (≈10%) of MFA studies and almost no LCA studies are temporally-dynamic (Allesch and Brunner, 2015; Laurent et al., 2014). System boundaries in LCA studies are also frequently ambiguous, even though their choice can be more significant than uncertainties in input data (Laurent et al., 2014). A further important issue relates to the scope of analysis and, specifically, the bias towards assessments of downstream processing systems rather than waste prevention strategies (Laurent et al., 2014). Examples of applications of LCA to reuse of resources do exist (Ardente and Mathieux, 2014; Castellani et al., 2015). Nonetheless, this bias reflects the political apprehension to challenging the unsustainable increasing demand for materials and energy in a growing economy, and alternative focus on meeting and mitigating this demand (Jackson and Senker, 2011).

The additional complexity of LCSA approaches that attempt to integrate social, economic and environmental domains into assessments of RRfW systems further exacerbates these challenges. As for MFA and LCA, there is an issue with the scope of analysis. Current LCSA developments are directed mostly towards broadening the domains of value analysed rather than the *level* of analysis. Thus, while studies are increasingly including environmental, social and economic impacts together, many analyses are still focused upon a single product, rather than a whole industrial sector or economy (Guinée, 2016; Onat et al., 2017). This is problematic in contexts of RRfW, for obvious reasons. Many other issues arise from the uneven development of assessment models in different domains and, relatedly, the problems that can arise when aggregating outputs (issues that apply to LCSA in contexts beyond RRfW). For example, for LCA, environmental databases are well developed and the metrics to be considered are standardised. But for sLCA, data is much more difficult to obtain and even choosing appropriate metrics can be problematic (Guinée, 2016; Valdivia et al., 2013; Wu et al., 2014). And even where consistent data is available across all domains for an assessment, weighing incommensurable outputs to make meaningful decisions still represents a challenge (Bachmann, 2013; Chong et al., 2016). Aggregating outputs from different models for an assessment leads to a further two specific issues that we intend to address here, namely (i) the need to maintain consistency across the models (or to account for inevitable, potentially opaque, inconsistencies) and (ii) the problem of neglecting interdependencies between domains (Sala et al., 2013a; Valdivia et al., 2013; Zamagni et al., 2013).

Due to these difficulties in implementation, LCSA of RRfW systems remain rare (Onat et al., 2017). Of those that have been reported, notable examples include analyses of different management strategies for bottom ash in Macao (Sou et al., 2016), alternative wastewater treatment technologies (Kalbar et al., 2016), the reuse of mobile phones in China (Lu et al., 2014), disposal scenarios for PET bottles in Mauritius (Foolmaun and Ramjeawon, 2013), the management of used cooking oil (Vinyes et al., 2013), and municipal waste management in Thailand (Menikpura et al., 2012). But while each of these is consistent in applying standard LCA to their case studies, the coverage of economic and social sustainability issues varies widely. Sou et al. (2016) apply cost benefit analysis and use a measure of public acceptance as the only social indicator and Kalbar et al. (2016) use only acceptability and participation as social indicators; both lack a lifecycle approach across all three sustainability domains. The other case studies are more consistent in applying lifecycle methods across all three domains, with social indicators informed by United Nations Environment Programme/The Society for Environmental Toxicology and Chemistry guidelines [UNEP/SETAC 2009]. But each still acknowledges the challenges of LCSA outlined above. Valuable steps have also been made recently by Vadenbo et al. (2014). Their approach involved extending the boundaries of their waste management analysis to include production processes and integrated environmental and economic metrics into a single model to optimise outcomes via systemic reconfigurations. However, it does not represent a full LCSA as it excluded social metrics and, further, they did not develop a detailed political narrative to understand the socio-political barriers to how such systemic changes may be achieved.

We aim to resolve a number of these shortcomings both via the model developed here and the wider CVORR framework within which the model sits (Iacovidou et al., 2017a). Here, our primary contribution is to address the issue of performing multidimensional assessments by presenting the foundations for a temporally dynamic, fully integrated model that can consider complex values across all domains: environmental, social, technical and economic. We also aim to address the issue of bias towards downstream processing, rather than waste prevention strategies, by extending the boundaries of our analysis to include production processes. Finally, via the case study, we intend to highlight the advantages of making high-level inferences of interactions between a system under investigation and interlinked systems, even when rigorous analysis is not possible. In addition, the wider CVORR framework intends to (i) understand how complex sets of value may be optimised by using the current model in combination with multicriteria decision analysis tools to examine trade-offs and synergies and (ii) understand not only what optimised system reconfigurations may look like, but how they may be achieved given existing socio-political constrains and tendencies and the *provisioning systems* involved (Brown and Robertson, 2014).

In short, we aim to make steps towards developing the necessary tools – focusing on the context of resource recovery – for implementing holistic sustainability assessments that have existed in concept for over three decades, but that have yet been put into practice.

### 2.2. Case study: background

After describing our model in Section 3, we apply it in Section 4 to an exploratory case study that considers the UK coal-based electricity production sector and its links to the UK concrete and cement industries (*EP* and *CCI* herein). Here we examine the inputs (e.g. fuels), products (e.g. electricity) and secondary products or wastes (e.g. combustion ash) and hence consider the advantages and potential trade-offs that may occur alongside increased use of low-carbon fuels. The UK energy mix is undergoing a transition to low carbon fuels led by the major power station Drax (who provide ≈10% of UK power) where biomass has replaced 50% of the coal input (Drax, 2017). There is much debate over the upstream, supply-chain environmental impacts of sourcing large quantities of forest-based biofuels (Creutzig et al., 2015), but less widely discussed are the downstream impacts on the CCI. High-quality pulverised fly ash (PFA), a by-product of coal combustion, is often used as a low-carbon cement replacement for concrete production. But biomass – and also solid recovered fuel (SRF), a waste-derived substitute for coal obtained from mechanical biological treatment (Séverin et al., 2010; Velis et al., 2013) – have critical impacts on fly ash characteristics and thus its suitability for concrete production (Iacovidou et al., 2017b).

We therefore chose this case as it is clear that the drive to lower the carbon emissions of electricity production has impacts on other connected production systems, and potentially in other domains of value beyond the environmental. Further, these wider impacts are driven largely by the ability (or not) to recover high-volume resources such as fly ash, which would otherwise be disposed of as waste (with associated direct impacts) and require replacement by other primary resources. These potential impacts are qualitatively outlined in Fig. 1 and can be summarised by asking:

- How will UK CCI industries respond to a shortage of domestic PFA, e.g., by increasing domestic cement production, or importing PFA from other coal-burning economies?

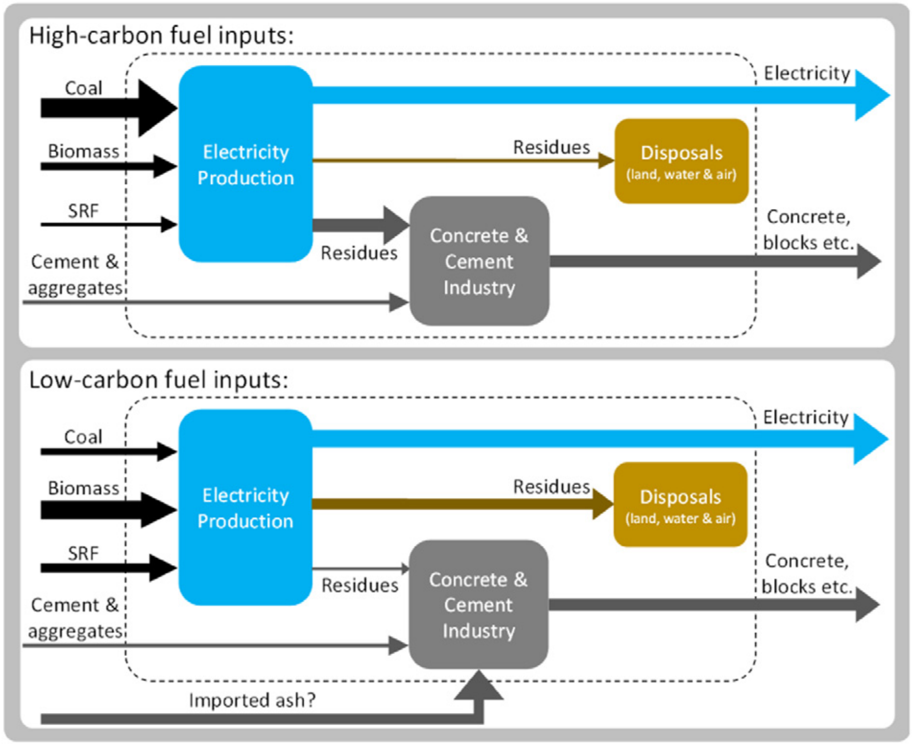


**Fig. 1.** Schematic diagram of the interlinkages between UK electricity production and the concrete and cement industry.

- In each case, what are the wider impacts upon carbon emissions beyond the point in the system (i.e. electricity generation) at which the low-carbon intervention is being made? In other words, what are the impacts both upstream *and* downstream on carbon emissions?
- Again in each case, what are the wider impacts in other domains – environmental, economic and social – and in other geographical regions?
- How might changes in the *technical values*[2] of flows drive the system as a whole?

Asking such questions guides the selection of metrics and system boundaries appropriate for exploring expected and hidden changes in complex values under these environmentally-motivated interventions, thus allowing this simple case study to clearly demonstrate the benefits of the CVORR approach.

## 3. Methodology

### 3.1. Foundations

The model developed tracks and forecasts mass flows and a range of metrics across social, environmental, economic and technical domains in contexts of RRfW. As we describe elsewhere, hundreds of potential metrics exist and an appropriate selection must be tailored to the system under investigation (Iacovidou et al., 2017c). We extend the boundaries of the systems we study upstream to production processes and downstream to disposal. A system within these boundaries is referred to as the *foreground system*, while connected systems laying outside these boundaries are referred to as *background systems*.

The foundation of our model is MFA and this is applied to the *foreground system*. Here, materials comprised of *substances* (uniform molecular arrays) and chemical *elements* (1) enter into the system as *inflows*, (2) flow between the system's *processes* as *intermediate* flows and (3) either exit the system as *outflows*, or accumulate as *stocks* within processes (Brunner and Rechberger, 2004). Processes may transform materials and substances into different forms (specified by *transfer coefficients*), but overall flows' behaviour is assumed to follow the principle of mass conservation.

In general, we aim to first build a mass-balanced representation of the materials flowing through the system. We then build upon these layers of metrics to investigate the transformation, creation, and destruction of complex values across the four domains. However, when interrelations between mass-flows and other metrics are expected to occur, the calculation process for each time-step must be run in a hierarchical manner, ordered according to the direction of the causal relationship of these interactions. For example, a substance concentration in the outflow of a process may determine the technical value of a flow, which may in turn determine how much material a process diverts to disposal and how much to recovery. This substance concentration must therefore be determined before the mass-flow layer can be resolved. The mathematics required to account for such interrelations depends entirely upon the complexity of the interrelations and systems under study; simple effects may be modelled easily by combining established approaches.

The impacts of inflows into (or outflows from) *foreground systems* can be accounted for via embodied values that describe cumulative impacts of (or from) *background systems*. These values represent impacts of national/international supply chains as comprehensively as available data allows. For the environmental domain, values can be taken from LCA databases describing standard input processes, or addressed via hybrid approaches that couple LCA to EEIOA tables (Bush et al., 2014).

However, using embodied values to represent background systems assumes (normally incorrectly) that they function independently of the foreground system. A *consequential* LCA approach is sometimes used to address this, which typically involves assessing the consequences of changes in the foreground system on the background system using partial- or general-equilibrium models to estimate market-

[2] Technical value here refers to a qualitative description of a material to fulfil a particular role given its physical and/or chemical characteristics and/or related regulations.

mediated consequences of model LCA outputs (Earles and Halog, 2011). Although such approaches potentially provide more holistic insights, we argue—as others have (Pelletier and Tyedmers, 2011)—that relying on market-mechanisms as the sole mediator is insufficient to fully account for such consequences in contexts of complex value assessments of RRfW. CVORR thus recommends that researchers/decision-makers expand system boundaries to include all important processes or, where this is not feasible, include (at least) qualitative assessments of background-foreground interactions, with a particular focus on non-market-forces.

### *3.2. Typology for modelling metrics*

We argue that classifying metrics into environmental, social, economic and technical domains is not relevant to the *modelling stage* of multidimensional assessments. These four domains are only useful for understanding the real-world implications of model outputs.[3] But when describing the behaviour of metrics *within* a model, an entirely different typology is required. Consider that sLCA is mathematically identical to LCA, while LCC demands a different calculation process (Heijungs et al., 2013). In this case it is not the domain that determines the calculation process, but the particular behaviour of each metric. Instead, when operationalising the behaviour of a metric in a mathematical model, the following questions must be asked (Fig. 2):

1) Is the metric a conserved value or can it be created/destroyed (e.g. capital)?
2) Is value transferred into, across, and out of the system only via material-flows (e.g. coal), or is it also associated with processes or non-material flows (e.g. money)?
3) Are the values of intermediate flows and outflows determined:
   a. Endogenously, as model outputs: by the functioning of the system itself and embodied value flowing into it (e.g. substance concentration), or
   b. Exogenously, as model inputs: via activities occurring outside the system (e.g. market prices)?

4) Are transfers (allocations) of value across multifunctional processes determined by physical and chemical processes (e.g. pathways of contamination) or by accounting conventions (e.g. assignment of embodied GHGs to process outputs)?

We propose a typology of five metric-types (see Table 1) based on these behavioural characteristics. The typology is intended to encompass metrics that are *transferred*, *transformed*, *created* and/or *destroyed* across systems. We expect it to encompass the majority of such metrics across the four domains of interest, although there are important metrics that do not fit into the typology as they either form model inputs[4] or are derived from systemic outputs.[5] The typology determines (i) what model-inputs are required and (ii) how each metric is mathematically modelled:

*Type 0* is a fundamental type of metric as they include chemical *elements*. These are (i) *conserved* (assuming no nuclear reactions occur), (ii) associated only with *material-flows* (iii) *endogenous* and (iv) modelled with transfer coefficients determined by *physical and chemical laws*.

*Type 1* metrics include embodied carbon-emissions and working hours. These are (i) *conserved*, (ii) associated with *material*- and *non-material* flows and *processes*, (iii) *endogenous* and (iv) modelled with transfer coefficients determined by *accounting conventions*. For example, embodied carbon can flow into processes' via materials, while processes themselves will also have emissions associated with energy use or fixed capital etc. Further, all embodied carbon flowing into a process should be assigned to out-flows, with the proportions attributed to each out-flow determined by accounting conventions.

*Type 2* metrics include many economic metrics (prices; wages) and various social metrics (noise pollution; social acceptance). These are (i) *not conserved* (economically-valueless 'waste' flowing into a resource-recovery process may acquire a market value), (ii) associated with *material*- and *non-material* flows and *processes* (communities may be reluctant to live near a particular plant, or subject to the transportation of a particular material); (iii) *exogenous*, (iv) modelled *without* transfer coefficients, as value is *not* transferred through systems via flows (for example, social acceptance of a process is typically unrelated to upstream activities). They are mathematically simple, but as flow values are exogenous they can require extensive data inputs.[6]

*Type 3* metrics are (as for type 0 metrics) fundamental as they include *substances*. They also include metrics that are cumulative, weighted sums of multiple substances concentrations, such as human toxicity potential (*HTP*) or ozone depletion potential (*ODP*). These metrics are (i) *not conserved*, (ii) associated only with *material-flows*, (iii) *endogenous* and (iv) modelled with transfer coefficients determined by *physical and chemical laws*.

*Type 4* metrics, as we define them, are broad in scope. They are (i) *not conserved* (a flow may become 'low-quality' following a particular process), (ii) associated only with *material-flows*, (iii) at least *partially exogenous* (they may be based upon endogenous substance concentrations relative to exogenous regulations), and (iv) modelled *without* transfer coefficients. Well-developed metrics may fit this type, such as water quality metrics. But it also includes system-specific technical values that may be qualitative (*low*, *medium* or *high*) and dependent upon other metrics, such as the level of contamination relative to government or industry standards. For example, contamination levels may be benign at one point in a system, but downstream they may have detrimental impacts (a mixture of plastic types in a bottle may have no impact on its technical value when in use, while this mixture may be considered contamination when the bottle is to be recycled).

The significant advantage of this typology is that it can form the basis for an integrated approach that can consider metrics irrespective of their domain of value within the same modelling framework. This leads to the benefits of (i) consistency of analysis across domains and (ii) technical capability to consider dynamic interdependencies.

### *3.3. Compiling metric input data*

In order to run the model, various data inputs are required. These relate metrics to the flows ($i$) and processes ($j$). The elements and characteristics of these inputs differ for each type of metric in the typology, but in each case they are a combination of the four distinct sets shown in Fig. 3. It must be noted that, as for MFA and sustainability assessments

[3] Although even this can be debated, as the domain of many metrics is highly ambiguous, e.g. employment and poverty could be seen as economic and social at the same time.
[4] E.g. processes' direct carbon emissions may form model inputs to calculate embodied carbon emissions (as in LCA).
[5] E.g. *recycled material fraction* would typically be derived from model outputs, i.e. the ratio of recycled to primary material passing through a full system.
[6] Further, this definition of *exogenous* depends upon modelling assumptions regarding background-foreground system interactions. For example, market prices of materials may be assumed to be determined exogenously by economic and political forces outside the system *only* if the system being studied is sufficiently small. In reality, prices are dependent upon upstream material, capital, and labour costs and profit margins etc.

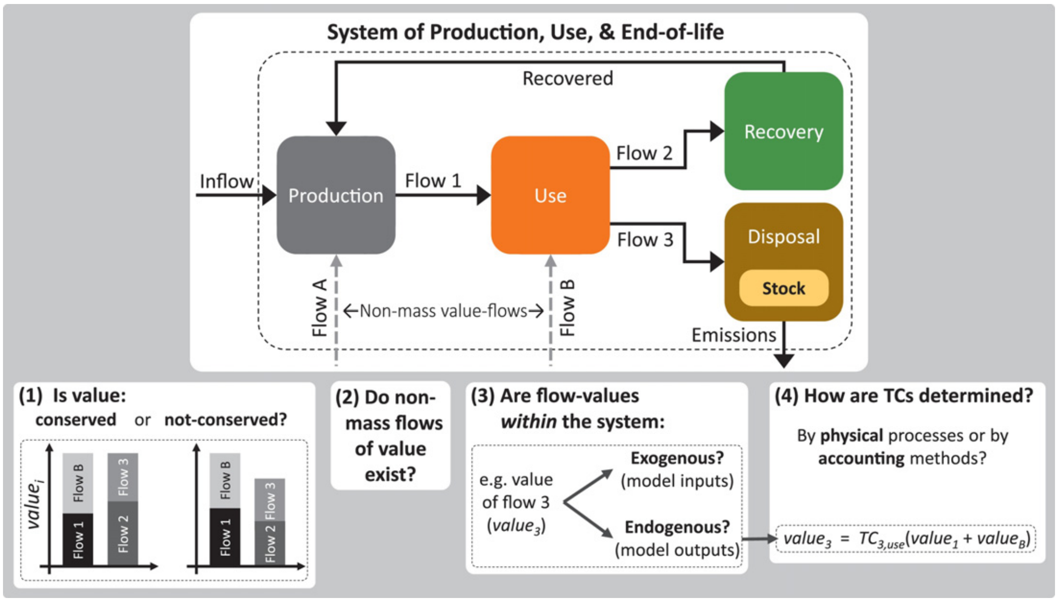


**Fig. 2.** Visualisation of the 4 characteristics underlying the metric typology.

of the social domain in general, the gathering and estimation of input data for our model is more difficult and time consuming than running the model itself (Brunner and Rechberger, 2004; Guinée, 2016).

Input set $A$ specifies relationships ($r_{i,n}$) of the metric $n$ to each mass-*inflow* to the system. This is typically a linear *intensity* or *concentration*, but non-linear relations or qualitative criteria can also be specified. For most metrics, the relationships specified will account for the embodied impacts due to background systems—for example, the relationship of a job creation metric to a mass-inflow should reflect the jobs created by producing and supplying it to the system (e.g. hrs/t coal).

Input sets $B$ and $C$ are similar: $B$ are the metric's relations to the mass-flows *within* the system and those *exported* from the system (also $r_{i,n}$); $C$ are relations to each *process* ($r_{j,P,n}$), *stock change* ($r_{j,\Delta s,n}$), and *existing stock* ($r_{j,Ts,n}$). These could again be linear scaling factors, such as intensities or concentrations. For example, for a metric of aggregated monetary costs, the costs associated with processes could include (among others) operational, maintenance and labour costs per unit-mass output, and those associated with stock changes and existing stock could include (for example) gate fees at a landfill site and the site's running costs, respectively.

The fourth input is a matrix of *transfer* (or *allocation*) *coefficients* ($TC_{ij,n}$) coefficients specifying how value is transferred from process inputs to outputs. Their mathematics are discussed further below. It should be noted that allocations present major issues when $TC$s are not physically-based, and results of LCA are highly sensitive to practitioners' assumptions (Reap et al., 2008), particularly when system expansion is used to estimate avoided impacts due to processes' secondary products (Hanes et al., 2015). In our cases, avoidable issues around assigning responsibility remain. However, where possible we track the technical properties of secondary products and bring the industrial or end-user destination processes into our foreground system, thus allowing us to examine in detail what flows of materials, energy, or values secondary products may be replaced and hence more accurately determine the magnitudes of avoided/increased impacts.

It is useful now to follow an example. If we consider the metric of embodied GHG emissions, we can first determine from Table 1 that this is type 1. Accordingly, assuming linearity, Fig. 3 indicates that type 1 metrics require (i) input $A$, embodied carbon per unit mass for the inflows, (ii) input $C$, carbon emissions per unit-flow for each process, stock change, and existing stock, and (iii) input $D$, a matrix of transfer coefficients specifying how carbon is allocated to each processes' outflows.

It is important to note that inputs are open to interpretation: for processes, carbon intensities may be used that only relate to direct energy use (imported heat or electricity, etc.) or they could be extended to reflect emissions embodied in, say, process's fixed capital (buildings,

**Table 1**
Metric typology, based upon the four characteristics outlines in Fig. 2.

| Opposing characteristics | | Type 0 | Type 1 | Type 2 | Type 3 | Type 4 |
|---|---|---|---|---|---|---|
| Conserved (embodied) | Not conserved | | | | | |
| Associated *only* with material-flows | Associated with material- *and* non-material flows or processes | | | | | |
| *Intermediate and outflow values endogenous* | *Intermediate and outflow values exogenous* | | | | | |
| Transfer coefficients determined by *physics* | Transfer coefficients determined by *accounting* | | | *N/A* | | *N/A* |
| | Examples: | Element concentration; Critical metals | Embodied carbon; Required working hours | Net profits; Working hours wage; Social acceptance | Substance concentration; Ozone depletion potential | Technical value of a flow; Water quality |

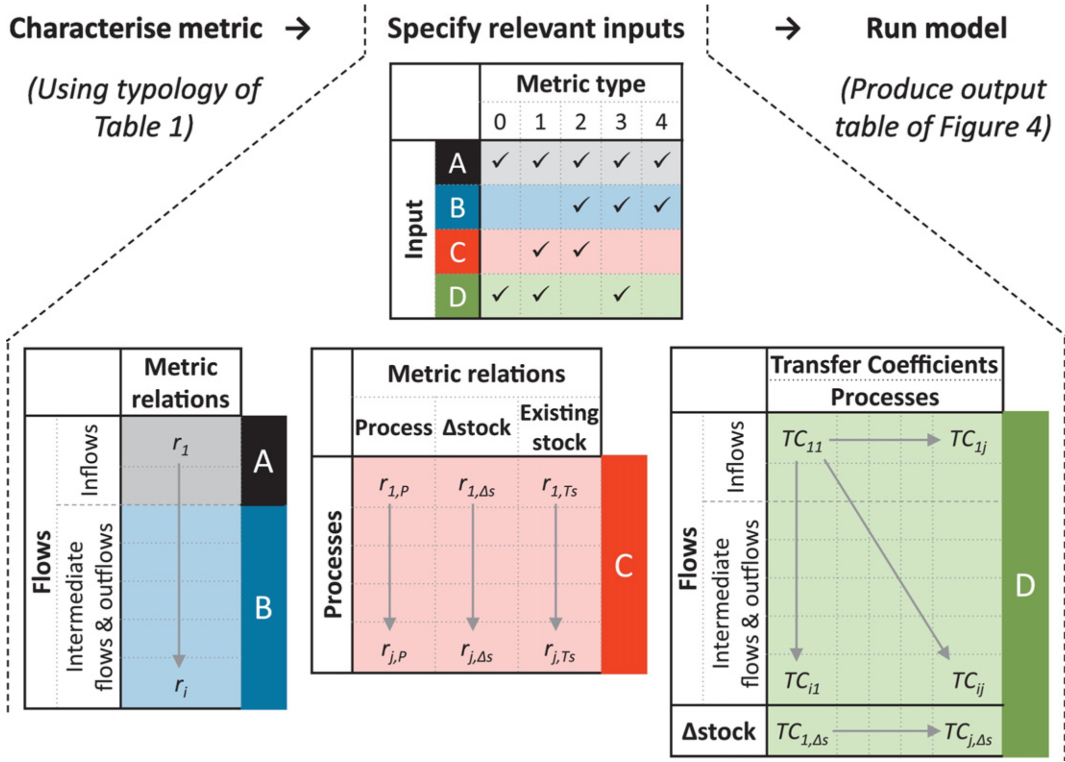


**Fig. 3.** Input data requirements for each metric type.

plants, etc., accounting for equipment lifetimes) (Pauliuk et al., 2015). It is crucial to be completely transparent regarding the extent to which model inputs capture such background-system impacts.

*3.4. Modelling complex value*

We can now describe the full modelling process: (i) the case-study is set-up as a bounded system with inflows, outflows, intermediate flows, processes and stocks (ii) transfer coefficients and the flows driving the mass-flow layer are specified, (iii) suitable metrics are chosen across the domains, are categorised in the typology, and the relevant input parameters and interrelations are set, and (iv) the system is mass-balanced and changes in complex values across it are evaluated to obtain initial output matrices (Fig. 4).

Standard MFA techniques are used to mass-balance such systems (Cencic and Rechberger, 2008) with three primary equations, written

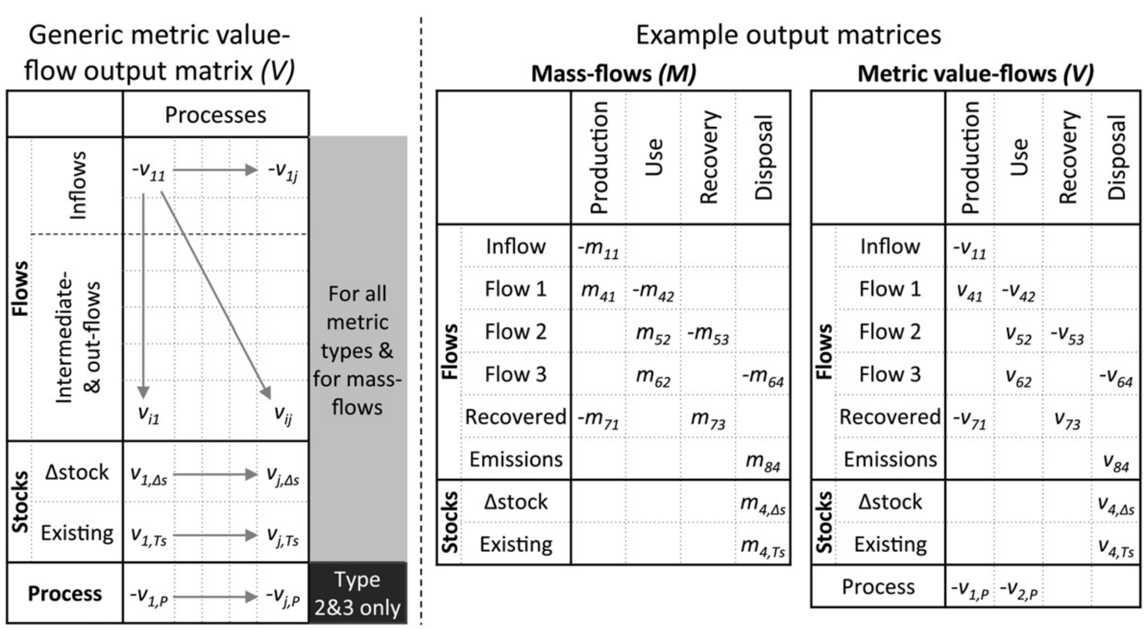


**Fig. 4.** Left—the generic output matrix of value-flows for a single metric (and time-step if dynamic simulations are performed). Right—specific examples of this value-flow matrix and a mass-flow matrix for the system sketched in Fig. 2 (right). Negative values denote inputs to processes, as is standard in LCA (Heijungs and Suh, 2002). Note here that $v_{1,P}$ and $v_{2,P}$ correspond to Flow A and Flow B from Fig. 2, respectively.

in a generic form as (Brunner and Rechberger, 2004):

$$\sum_i inputs_{ij} = \sum_i \left(outputs_{ij} + \Delta stock_{ij}\right) \tag{1}$$

$$output_{ij} = TC_{ij} \times \sum_i inputs_{ij} \tag{2}$$

$$stock_{j,t+1} = \Delta stock_{j,t} + stock_{j,t}. \tag{3}$$

Mass conservation is ensured provided:

$$1 = \sum_i TC_{ij} + TC_{j,\Delta s}. \tag{4}$$

here, each $i$ represents each mass-flow into process $j$, $t$ is the time-period, and $TC_{ij}$ and $TC_{j,\Delta s}$ are transfer coefficients indicating the proportion of the inputs into process $j$ that are transferred to $output_{ij}$ and $stock_j$, respectively. More detailed nomenclature is described in Table 2.

The mass-layer transfer coefficients can be static and set by the user at the beginning of a model run. But dynamic simulations can also be performed by specifying relationships between the outputs of one time-period (mass flows, $M_t$, or metrics, $V_{n,t}$) and the transfer coefficients or other parameters (inflows, etc.) of the subsequent period, for example:

$$TC_{ij,m,t+1} = f\left(TC_{ij,m,t}, M_t, V_{1,t}, \ldots, V_{N,t}\right). \tag{5}$$

This is done without encountering mathematical difficulties, as mass-balancing within each time-step remains the same (Buchner et al., 2015). In contrast, interdependencies within a time-step, if:

$$TC_{ij,m,t} = f\left(V_{1,t}, \ldots, V_{N,t}\right), \tag{6}$$

dictate that changes in relevant metrics' values across the system must be considered prior to resolving the mass flow layer. Simple interrelations may be modelled without difficulties (as in our case study). However, modelling complex interrelations in large systems can be conceptually and mathematically difficult. In LCA, such issues are far from being resolved (Zamagni et al., 2012).

Type 0 metrics are also governed by Eqs. (1)–(4) as they are conserved elements of the mass-flows and thus behave in the same way as masses. Further, if the relations between metric $n$'s value ($v_{ij,n}$) and the mass-inflows are linear, then the value-inflows driving the system are obtained via a simple multiplication ($v_{ij,n} = m_{ij} \times r_{i,n}$).

For type 1 metrics the important mathematical difference is that each process has an additional non-mass inflow and important conceptual difference is that $TC_{ij}$ is no longer determined physically or chemically. Consequently, the value-inflows driving the system are obtained as for type 0 metrics and Eqs. (1)–(4) still apply. Eqs. (1) and (2) however, acquire an additional term due to the non-mass value-flow, denoted here as $process_j$:

$$process_j + \sum_i inputs_{ij} = \sum_i \left(outputs_{ij} + \Delta stock_{ij}\right) \tag{7}$$

$$output_{ij} = TC_{ij} \times \left(process_j + \sum_i inputs_{ij}\right). \tag{8}$$

A useful aspect of the typology becomes evident here. The transfer coefficients for type 1 metrics can be based upon metric values from any another layer—i.e. a partition allocation (Hanes et al., 2015)—provided that value is conserved over the process in the other layer. This highlights a problem with allocating, say, carbon emissions by (relative) market prices, as the latter can be destroyed by a process, while the former should be conserved.

For type 2 metrics there are no transfers or allocations. Thus, instead of embodying value through the system, metric values associated with each mass-flow are obtained directly via the input relations between the metric and the mass-flows. If these relations are linear, then values are obtained via simple multiplications ($v_{ij,n} = m_{ij} \times r_{i,n}$) and the same calculation gives the value associated with process and stocks ($v_{j,\Delta s} = m_{j,\Delta s} \times r_{j,\Delta s}$, etc.).

Type 3 metrics are more complicated. Across some processes value may be conserved, while across others it may be created or destroyed. Eqs. (1) and (4) therefore no longer apply, while Eqs. (2) and (3) do. Creation and destruction of value is easily captured via Eq. (2), the latter by setting transfer coefficients to zero (or their sum over a single process to <1), and the former by using inputs from a different metric layer than that being modelled (for example, a substance concentration flowing out of a process may be derived from an elemental concentration flowing into it).

Type 4 metrics follow equations that are specific to particular flows and processes. For example, the relation of a technical value of flow $i$ to metric $n$ could be:

$$v_{ij,n} = r_{i,n}(V_1, \ldots, V_N) = \begin{cases} Low, & x < f(V_1, \ldots, V_N) \\ Med, & y \le f(V_1, \ldots, V_N) \le x \\ High, & f(V_1, \ldots, V_N) < y \end{cases} \tag{9}$$

where the function $f$ defines a quality measure based upon other metrics, and $x$ and $y$ are thresholds for this measure determining if the flow is *high*, *medium*, or *low* quality. In contrast to other metric types, changes in type 4 metric values across process are not necessarily easy to interpret (flows of varying mass will also vary in technical values in such a way that is not always easy to aggregate into an overall change in value) and hence these changes should be examined alongside wider changes in value across the system.

**Table 2**
Table of the nomenclature used in the model.

| Variable | Description |
|---|---|
| $xx_{P,\Delta s,Ts}$ | Subscripts for *processes*, *stock changes*, and *existing stocks*, respectively |
| $r_{i,n,t}$ ($r_{j,P,n,t}$) | Relation between metric $n$ and flow $i$ (or process $j$) at timestep $t$ |
| $r_{j,\Delta s,n,t}$ ($r_{j,Ts,n,t}$) | Relation between metric $n$ and the stock change (or existing stock) in process $j$ at timestep $t$ |
| $TC_{ij,m,t}$ ($TC_{j,\Delta s,m,t}$) | Transfer coefficient for *mass-flows*, determining the proportion of inputs into process $j$ transferred to mass-flow $i$ (or to the stock) at timestep $t$ |
| $TC_{ij,n,t}$ ($TC_{j,\Delta s,n,t}$) | Transfer coefficient for metric $n$, determining the proportion of inputs into process $j$ transferred to value $i$ (or to the stock) at timestep $t$ |
| $m_{ij,t}$ | Mass of flow $i$ flowing in (or out when negative) of process $j$ at timestep $t$ |
| $m_{j,\Delta s,t}$ ($m_{j,Ts,t}$) | Mass of stock change (or total existing stock) associated with process $j$ at timestep $t$ |
| $M_t$ | Matrix of mass-flows at timestep $t$ |
| $v_{ij,n,t}$ | Value of metric $n$ corresponding to mass-flow $i$ flowing in, or out, of process $j$ at timestep $t$ |
| $v_{j,P,n,t}$ | Value of metric $n$ corresponding to process $j$ at timestep $t$ |
| $v_{j,\Delta s,n,t}$ ($v_{j,Ts,n,t}$) | Value of metric $n$ corresponding to the stock change (or total existing stock) associated with process $j$ at timestep $t$ |
| $V_{n,t}$ | Matrix of metric $n$ values at timestep $t$ |

### 3.5. Case study: scenario design

To demonstrate the value of the approach, we explore it with the illustrative case study introduced above. We include UK electricity-production plants, concrete industry operations and disposal sites within the foreground system; design four scenarios and investigate associated systemic impacts; and select a small range of metrics suitable for capturing characteristic changes. Below we briefly describe our scenarios, input parameters, and system configuration; more thorough descriptions are included in the supplementary material (SM).

We investigate a hypothetical 5-year time horizon in which low-carbon fuel inputs of biomass and SRF are progressively increased (biomass from 10% to 50% and SRF from 2.5% to 12.5%; by mass). We use fixed production of electricity (100 TWh) and construction products (100 Mt concrete; 10 Mt concrete-blocks), which reflect current UK production. Other parameters remain static, thus a simple *business-as-usual* scenario can be represented via fixing the 1st year's outputs over the full 5-year time horizon. We then consider four other scenarios, two relating to biomass, and two to ash:

*Biomass scenarios*: Estimates of embodied GHGs associated with biomass supply-chains are uncertain and complex, thus we consider *low-* and *high-GHG biomass* scenarios. Our value for the former (25 g $CO_2$e/MJ primary biomass energy) sits within the wide range reported in the literature (Creutzig et al., 2015). Our *high* value (75 g $CO_2$e/MJ) reflects the argument that wood-based biofuels may increase or decrease emissions relative to conventional fossil fuels, depending on the forest type and the impacts on carbon sequestration (Hudiburg et al., 2011).

*Ash scenarios*: We consider two different responses of the UK construction industry to a reduction of high-value PFA from UK electricity production: (i) increasing domestic cement production and (ii) importing PFA; the *domestic production* and *imported ash* scenarios, respectively. The *imported ash* scenario appears feasible: coal-burning economies in Eastern Europe and South-East Asia are building links with construction industries in countries moving away from coal such as the USA, Western Europe and South-West Asia.[7] Interestingly, this represents a reversal of trends, in that newly industrialised countries are beginning to export wastes to more mature industrial (or even post-industrial) countries. For demonstration purposes we investigate a scenario where the UK imports PFA from Turkey, which has the largest coal-power development plans outside of India and China, a notoriously dangerous mining sector, and a prevalence of low-quality, high-ash content coal (IEA, 2016). Turkey thus represents a realistic, but worst-case scenario.

In this *illustrative* analysis, we select one metric from each domain in order to demonstrate significant expected trade-offs:

*Greenhouse gas emissions* (*Type 1*): For the environmental domain, this metric is selected as it is GHGs which interventions in the foreground system are intended to reduce.

*UK profits* (*Type 2*): To evaluate the economic domain, we consider the total revenues minus costs for each process i.e. those relating to material inputs, plant operations, labour, taxes, etc.

*Mortality* (*Type 1*): For the social domain we consider mortality that attempts to capture occupational hazards and air-pollution related public health impacts (although climate change related impacts are not included). We select this metric due to ongoing, heated debates regarding the safety of coal-based power production relating to mining and air pollution.

*Technical value of fly-ash* (*Type 4; based upon Type 0*): We assign a technical value to the fly ash output from UK power production: PFA suitable for cement substitution is considered *high-quality*, FBA (furnace bottom ash) to replace aggregates *medium-quality*, and landfilled ash *low-quality*. The PFA values are, in turn, based upon chlorine concentrations and related standards (BS EN-450) specifying the concentration of chlorine permitted in PFA when used as a cement substitute (0.1%).

The value of the typology becomes clear here as, while these metrics cover the four domains of value, their mathematical treatment is instead aligned with the typology. Note that the *low-* and *high-*GHG biomass scenarios affect *only* the GHGs outputs, while the *domestic production* and *imported ash* scenarios affect *only* mortality. UK profits and technical values remain independent of these different scenarios.

## 4. Results and discussion

Fig. 5 shows our results across all the scenarios for each metric, aggregated into electricity production, concrete industry and disposal sectors. But first we emphasise again that this is an *illustrative* case study not a comprehensive analysis, and hence we make no assessment of uncertainty; this will be a central focus of future work.

When embodied emissions in the imported biomass are low, decreases in emissions from electricity production are more than sufficient to offset increases in emissions elsewhere in the system. However, total GHG emissions rise in our *high GHG biomass* scenario as the small GHG reductions from electricity production are outweighed by (i) emissions at the disposal stage and (ii) the response of the concrete industry to the falling UK supply of high-quality PFA and the related accounting conventions. Note that our GHG data inputs are reasonable accurate (significantly better than for profits and mortality) and therefore we highlight here a rough value that embodied GHGs in biomass must undercut if total GHGs are to reduce.

Profits for electricity production are assumed to fall due to increasing fuel costs (biomass) and disposal fees for low-quality ash, which is reflected in the increasing profits of the disposal sector. But these data are particularly illustrative as they are highly contingent upon many assumptions, as outlined in the SMs.

Whether there is a significant change in modelled GHGs between the *domestic production* and *imported ash* scenarios depends on the accounting convention applied to PFA from Turkey. Typically, secondary products such as PFA are not allocated any combustion-related GHGs, but here we use UK prices for cement (and electricity) to allocate GHGs to Turkish PFA, leading to embodied GHGs (per t) very similar to that of UK cement (see SMs Section 3.1). We recognise that these allocation decisions are highly contentious and that different assumptions would show results counter to ours. We discuss this thoroughly in the SMs (Section 4).

Total mortality is shown in Fig. 5, aggregated into electricity production, concrete industry and disposal sectors, but also disaggregated into UK and non-UK mortalities (assuming *non*-UK fatalities are *only* embodied in inflows of biomass and PFA; see SMs Section 3). Mortalities are dominated by the impacts of electricity production: these are an order of magnitude larger than those from the concrete industry, which are in turn two orders of magnitude larger than those due to disposal. In all cases the health impacts of air pollution are far more significant than occupational fatalities (see SMs Section 3.4). Mortalities due to air pollution from landfill sites are likely to be more significant than our results suggest, as increasing disposal of UK fly-ash would lead to increases in airborne particulates. But currently, there is limited research into these impacts (Mataloni et al., 2016).

Notable is that our total mortality estimates are higher in the *imported ash* case, as we estimate mortality per-unit mass to be higher for imported PFA than for UK cement. This is using the same assumptions and allocation procedure used for GHGs in conjunction with the substantial mortality rate estimates for lignite-based electricity generation (Markandya and Wilkinson, 2007) (see SMs). This could be interpreted as an *offshoring* of social impacts similar to the offshoring of carbon emissions that typically accompanies deindustrialisation. Such analysis appears to support the argument that environmental

[7] For example, see industry conferences on emerging ash markets (www.ashtrans.eu)—particularly relating to Turkey—and analysis from an engineering perspective (www.newcivilengineer.com/technical-excellence/the-dash-for-ash/10015486.article; accessed 7/12/2016).

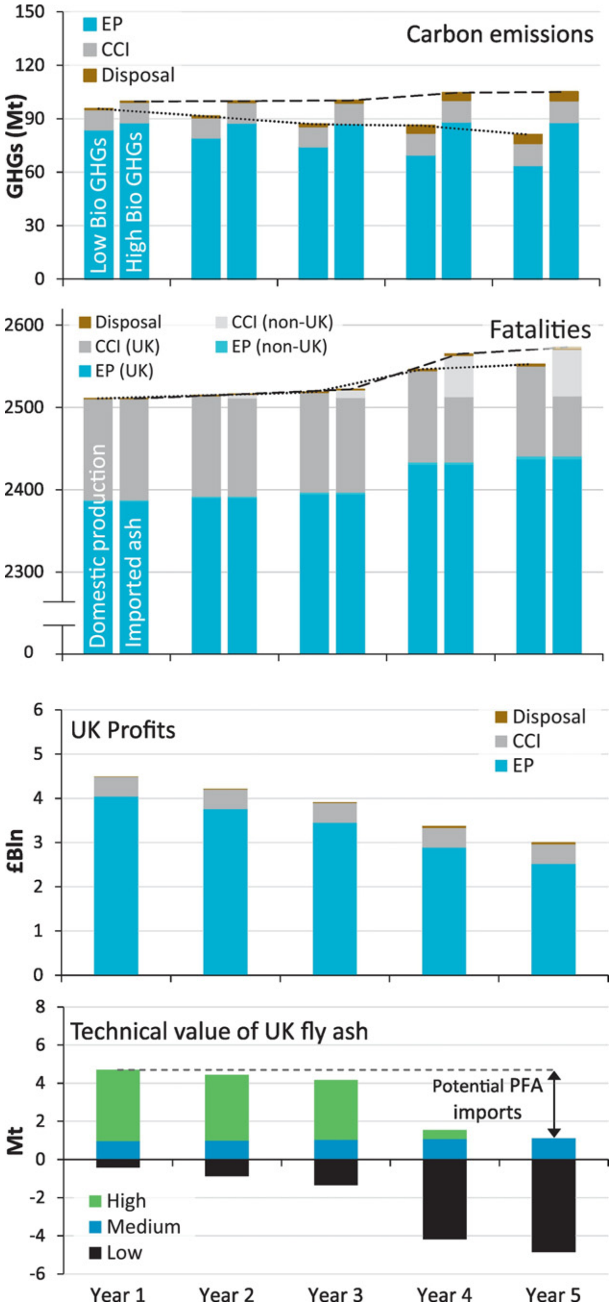


**Fig. 5.** Carbon emissions (both in the *low*- and *high-carbon* cases); total fatalities (in the *domestic production* and *imported ash* scenarios); and UK profits aggregated over the *EP*, *CCI*, and disposal sectors. Below these are amounts of UK fly ash by technical value. Fatalities are disaggregated also into those occurring within and outside the UK.

interventions in the UK electricity-production sector can have complex, geographically distant, but potentially foreseeable, impacts across multiple domains of value. However, the picture is more complex. Total mortalities shown in Fig. 5 are those *assigned* to the (UK) foreground system, but there are also associated changes in moralities in the (Turkish) background system that remain unknown. Unlike normal offshoring of environmental or social impacts, in our case it is not clear that any change in the Turkish electricity production system would actually be provoked by changes in the UK, particularly within the 5 year timeframe of our scenario. Indeed, by exporting ash, Turkey may actually be offshoring their own environmental and social issues (see SMs, Section 4). The strength of the systemic and socio-political CVORR approach when applied more thoroughly than in this illustrative case is to uncover such effects.

Finally, Fig. 5 shows how these changes in multidimensional value are driven by changes in technical values. An abrupt decrease in the quantity of *high-value* fly-ash occurs when the chlorine concentration in the ash obtained from co-fired coal and SRF exceeds the threshold of 0.1%. This is assumed to occur from year 3 to 4[8] and results in the diversion of fly ash from a useful industrial destination to landfill. This abrupt shift is reflected across all the other domains, due to various factors such as increased cement production, landfill gate-fees, and the addition of new imported materials. For this illustrative case-study, this technical value captures a relatively straightforward effect. But when thoroughly analysing larger systems, tracking technical values of flows may offer insights into system dynamics that are more complicated and/or intangible than what we have explored here.

However, it is also important to note the limitations of our analysis. The (hypothetical) international trade links that we assume between the UK and Turkey would be motivated by many factors aside from market forces. And hence our decision to base allocation decisions on market prices could be questioned. UK concrete producers have environmental policy imperatives to produce low-carbon cement and it is essentially the allocation decision (i.e. is fly ash carbon neutral or not?) that determines if PFA satisfies this requirement. Coal burning countries such as Turkey may wish to export fly ash largely to avoid public health impacts of disposals. This also highlights the value in conceptualising foreground-background interactions from a broader perspective than market-centric methods such as consequential LCA typically adopt.

It must also be noted that other construction industry responses to a domestic PFA shortfall are possible, for example, its replacement by steel slags. Indeed, this may be more likely than the *imported-ash* scenario we consider here as future policies may be designed to foster mutual economic benefits for both domestic concrete and steel producers and/or discourage industry imports from regions with questionable health and safety records (which, in our case, may include Turkey). Including such alternatives as scenarios would be essential for a more complete analysis than we undertake here.

## 5. Conclusions

The primary contribution of this work is that it aims to address the argument that comprehensive sustainability assessments of RRfW systems must take a systemic and consistent approach with respect to system boundaries and consider all domains of value. Assessments must be capable of recognising trade-offs, synergies and the incommensurability of values. We have thus presented the foundations of a fully integrated modelling approach applicable across all domains of value. This allows for an alternative approach to the current tendency for parallel development of unidimensional models, thus avoiding the problematic integration of their results. It does not aim to replace multidimensional sustainability assessment methodologies such as LCSA, rather it offers an alternative modelling approach to support them.

### 5.1. Methodology

We argue that when performing assessments across multiple domains of value, classification of metrics into *environmental*, *technical*, *social* and *economic* domains is not relevant from a modelling perspective and, therefore, the dichotomy between LCA, sLCA, LCC, etc., is a false one. In other words, the categorisation of a metric as *environmental* or *social* has no mathematical significance. Instead, the relevant questions are (among others): is value *conserved* or can it be *created and destroyed*? Is it attached *only* to material-flows, or does it also enter the system via other mechanisms? Our new typology for classifying metrics enables us to develop an integrated model covering all domains

[8] This 'tipping point' may have occurred earlier or later in the scenario had different chlorine concentrations been chosen for the inflows (see SMs).

of complex value, removing the need to integrate results from multiple unidimensional models in order to perform multidimensional assessments. This allows for the mass-flow layer to be driven by temporal dynamics and interdependencies based upon multidimensional sets of values. A particularly important interaction—usually not endogenised in models—is between the technical values of resources and their flows: including this interaction enables the technical reasons for tipping points observed across other dimensions of value to be easily identified.

We also argue that, rather than ignoring complex (often globalised) social and political dynamics, such assessments should be guided by a socio-political narrative to understand how systemic changes may be achieved and why current systems function as they do. An extension of this reasoning is that we recognise the incommensurability of changes in value across different domains and, consequently, the inherent socio-political factors that determine the criteria by which to 'optimise' such systems. It follows that, rather than employing black-box mathematical techniques, it is more appropriate to design transparent tools to support decision making.

### 5.2. Case study

We apply the model to an illustrative case study linking the UK coal-based electricity-production sector to the UK concrete and cement industries, examining (some of the) aggregate impacts that may follow an increased use of low carbon fuels. We show how the model may investigate tipping points; in this case, the upstream conditions under which total GHG emissions rise due to impacts downstream of electricity-production. The case also highlights the contentious nature of allocation decisions and the need to examine socio-political imperatives; together, this framing of the case study leads us to investigate how potential new international trade links may induce further offshoring of environmental and social impacts. Finally, we indicate how these systemic, multidimensional changes may be understood as being driven by changes in the *technical value* of resource flows.

More broadly, the results highlight the advantages of approaching such analysis with an intention to make high-level inferences of complex system dynamics—including important interactions between background and foreground systems and distributional effects etc.—rather than taking market-centric approaches and devoting disproportionate attention to optimising incommensurable sets of outputs using limited (and subjective) constraints. In our case study, we only began to probe these issues, but any sustainability analysis wishing to be comprehensive will require a comprehensive treatment of such effects.

### 5.3. Limitations and unaddressed challenges

Our approach aims to address a number of shortcomings that we have argued are inherent to current assessment methods, but various challenges remain. First, our approach is no simpler than other approaches – or sustainability assessments in general – and it is equally demanding in terms of data input requirements. Also of importance is our omission of any treatment of uncertainty, of which a robust analysis (complemented by sensitivity analysis) is necessary for any comprehensive sustainability analysis.

Other modelling challenges include questions regarding how to value fixed capital (e.g. long-lasting plants, infrastructure) or account for discrete (perhaps disruptive) technical transitions. There are also issues relating to the wider framework within which our model is intended to sit—how to select appropriate metrics for the valuation assessment and, crucially, how to integrate the results across the various domains in such a way that outcomes can be optimised via a set of criteria that remain totally transparent.

Clearly there are many difficult conceptual and technical challenges to overcome, and these are an ongoing focus of our research. But the challenge of adopting such transparent and provocative approaches into political decision making—and having these displace those such as cost-benefit-analysis that spit out a single (seemingly objective) number—may be even greater.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2017.08.211.

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