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Review

Metrics for optimising the multi-dimensional value of resources recovered from waste in a circular economy: A critical review

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A R T I C L E I N F O

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Established assessment methods focusing on resource recovery from waste within a circular economy context consider few or even a single domain/s of value, i.e. environmental, economic, social and technical domains. This partial approach often delivers misleading messages for policy- and decision-makers. It fails to accurately represent systems complexity, and obscures impacts, trade-offs and problem shifting that resource recovery processes or systems intended to promote circular economy may cause. Here, we challenge such partial approaches by critically reviewing the existing suite of environmental, economic, social and technical metrics that have been regularly observed and used in waste management and resource recovery systems' assessment studies, upstream and downstream of the point where waste is generated. We assess the potential of those metrics to evaluate 'complex value' of materials, components and products, i.e., the holistic sum of their environmental, economic, social and technical benefits and impacts across the system. Findings suggest that the way resource recovery systems are assessed and evaluated require simplicity, yet must retain a suitable minimum level of detail across all domains of value, which is pivotal for enabling sound decision-making processes. Criteria for defining a suitable set of metrics for assessing resource recovery from waste require them to be simple, transparent and easy to measure, and be both system- and stakeholder-specific. Future developments must focus on providing a framework for the selection of metrics that accurately describe (or at least reliably proxy for) benefits and impacts across all domains of value, enabling effective and transparent analysis of resource recovery form waste in circular economy systems.

Contents

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Abbreviations

AC	Avoided carbon emissions
AIChE	American Institute of Chemical Engineers
AP	Acidification potential
BAT	Best available technology
BC	Black carbon emissions
BC_e	Biogenic carbon emissions
BoL	Beginning of life
CBA	Cost-benefit analysis
CBO	Community-based organisation
CCS	Carbon capture and sequestration
CEA	Cost-effectiveness analysis
CED	Cumulative energy demand
CFC	Chlorofluorocarbon
CHP	Combined heat and power
CRM	Critical raw material
DC	Direct carbon emissions
DECC	Department of Energy and Climate Change (UK)
EC	Embodied carbon emissions
E_{eff}	Energy efficiency
EE-IOA	Environmentally extended Input-Output Analysis
EEA	European Environment Agency
EF	Ecological footprint
EfW	Energy from waste
EIA	Environmental impact assessment
EIR	Environmental investment ratio
EIRI	Environmental impact recoverability indicator
ELR	Environmental load ratio
EMC	Environmentally weighted material consumption
EOC	Emerging organic contaminant
EoL	End of life
EoU	End of use
EpE	Entreprises pour l' environnement
ERA	Environmental risk assessment
ERI	Energy recovery indicator
ESG	Environmental, social, and governance
ESI	Environmental sustainability index
ETP	Ecotoxicy potential
EuP	Eutrophication potential
EUROSTAT	Statistical office of the European Union
EYR	Environmental yield ratio
GER	Gross energy requirement
GHG	Greenhouse gas
GIS	Geographical information system
GRI	Global Reporting Initiative
GWP	Global warming potential
HCFC	Hydrochlorofluorocarbon
HFC	Hydrofluorocarbon
HTP	Human toxicity potential
IAEA	International Atomic Energy Agency
IChemE	Institution of Chemical Engineers
IEA	International Energy Agency
IOA	Input-output analysis
IRS	Informal recycling sector
ISWM	Integrated sustainable waste management
LCA	Life cycle assessment
LCC	Life cycle costing
LCSA	Life cycle sustainability assessment
LHV	Lower heating value
LPG	Liquefied petroleum gas
LULU	Locally unwanted land use
MCDA	Multi-criteria decision analysis
MCDM	Multi-criteria decision making
MCPs	Materials, components and products
MFA	Material flow analysis
MIPS	Material input per service unit
MRF	Material recovery facility
MSW	Municipal solid waste
NIMBY	Not in my backyard
NIR	Near infrared
NPV	Net present value
ODP	Ozone depletion potential
OEF	Organisation Environmental Footprint
PAH	Polycyclic aromatic hydrocarbon
PEC	Primary energy consumption
PEF	Product environmental footprint
PFC	Perfluorocarbon
PM	Particulate matter
POFP	Photochemical ozone formation potential
POP	Persistent organic pollutant
PTE	Potentially toxic element
RCE	Resource conservation efficiency
RRfW	Resource recovery from waste
SEA	Strategic environmental assessment
SF_6	Sulphur hexafluoride
SFA	Substance flow analysis
SI	Sustainability index
sLCA	Social life cycle assessment
SRF	Solid recovered fuels
SWM	Solid waste management
TBL	Triple bottom line
TRI	Technical recovery indicator
UNCED	United Nations Conference on Environment and Development
UNDESA	United Nations Department of Economic and Social Affairs
UNEP	United Nations Environment Programme
VOC	Volatile organic compound
WBCSD	World Business Council for Sustainable Development
WRI	World resources institute
ws_x	MSW management self-sufficiency indicator
wt	weight

1. Introduction

Current initiatives promoting a 'circular economy' build upon preceding research into resource efficiency (Ashby, 2016; Butterworth and Bleriot, 2014; EC, 2015; Ghisellini et al., 2016; Gregson et al., 2015; Haas et al., 2015; Murray et al., 2015), and provide an imperative to reconsider our approach to resource recovery from waste (RRfW). This should aim to resolve RRfW system inefficiencies, and transform waste management practices into systems that 'manufacture' secondary resources of high value. There is a need both to remove structural barriers within the industry and reform existing policy and legislation, in order to empower interventions that transform currently unsustainable practices (Gregson et al., 2015; Silva et al., 2016; UNEP and ISWA, 2015). Transformation requires a shift in thinking such that RRfW is conceptualised and operationalised on the basis of preserving the value of materials, components and products (MCPs) by retaining their functionality for as long as possible, as underpinned by the rationale of a circular economy (Ellen MacArthur Foundation, 2012; Ghisellini et al., 2016).

Ideally, this concentrates on the direct reuse of products and components; but often the degree to which this can be achieved is limited owing to aging, design, performance (including environmental and resource efficiency performance), or recovery constraints. In such cases, repair, reconditioning, remanufacturing, recycling (closed- and open-loop recycling) or energy recovery from MCPs are considered to be the next best option for recovering the value embedded in and/or associated with MCPs (Benton and Hazell, 2013; Huysman et al., 2015; Thormark, 2000). The established EU guidance for recovering resources from waste mandates – via the "waste hierarchy" of the Waste Framework Directive (European Union, 2008) – that in principle reuse is better for the environment than materials recycling, recycling is better than energy recovery, and energy recovery is better than disposal. In reality, efficient and environmentally sound recovery of value from waste is far more complex than just following a ranked description of generically preferred management options. Hence, the option to modify the waste hierarchy, e.g. by taking a case-by-case life cycle assessment (LCA), is endorsed in the Waste Framework Directive (European Union, 2008). However, even these slightly more sophisticated options have little to say about the prevention of dissipation of value into waste; the transition to a resource efficient circular economy requires approaches that allow a more holistic analysis and evaluation of value creation, appropriation and dissipation within the systems in question.

The term 'value' herein has a wide meaning, referring to measurable benefits (creation of positive value) and impacts (creation of negative value, or loss of value) in the environmental, economic, social and technical domains (Iacovidou et al., under review). Considering all these domains – and potentially more e.g. governance – in the evaluation of interventions, allows for a more holistic analysis of options needed towards the overarching objective of sustainable development, as required by the "Agenda 21" (UNCED, 1992). Nested within these four generic domains of value are multiple specific dimensions of value that are associated with the production, use, recovery and disposal of MCPs, from their beginning of life (BoL) towards their end-of-use (EoU) and end-of-life (EoL) stage, and subsequent redistribution (circulation, looping, cascading) back into the anthropogenic system or disposal into the biosphere (final sinks). At present, established assessments focusing on the recovery of value from waste are based on dimensions of value from few – in fact typically a single – domain/s of value; for example, the waste hierarchy or LCA are preoccupied only with dimensions of value from the environmental domain. This partial approach often delivers misleading messages for policy and decision-makers, failing to cut through the systemic complexity, poorly accounting for undesirable effects in other sectors and/or domains of value, and obscuring as such impacts, trade-offs and problem shifting that some RRfW processes or systems may cause (Lyytimäki et al., 2013; Ulgiati et al., 2011). In addition, this partial approach might hinder the exploitation of hidden beneficial synergies along the supply chain.

To elaborate, studies have shown that 46% (wt.) of the post-consumer plastic waste collected for recycling in the EU is exported, the bulk of it to the Far East (Velis, 2014, 2015), where in the near past it may have been reprocessed in facilities with deficient environmental protection, by poorly-paid workers in unhealthy conditions (Puckett et al., 2002); a hidden social and health impact. Other studies have shown that co-firing biomass and/or partly biogenic solid recovered fuels (SRF) with coal in power plants, while beneficial in reducing the use of fossil fuels and mitigating climate change, may influence the operation and performance of the boilers used, increase trace element emissions, and render certain by-products (e.g. pulverised fly ash) chemically unsuitable for previously established applications (e.g. concrete manufacture); all leading to unassessed hidden technical, economic and environmental impacts (Iacovidou et al., 2017a). A meaningful way of measuring the multiple dimensions of value embedded in and associated with all the MCPs in a system, would allow investigators to concurrently analyse and weigh up all these factors.

In such holistic evaluations, the selection of appropriate metrics (quantitative or semi quantitative descriptors) (Tanzil and Beloff, 2006) that accurately describe (or at least reliably proxy for) benefits and impacts, is critical. Simple ways of measuring value can facilitate a transparent assessment process, and allow for comparisons between various options for recovering resources from waste under different scenarios to be made. Meanwhile, a balance between simplification in measurement and comprehensiveness in addressing systemic complexity has to be reached. As such, the metrics selected for optimising the value of recovered MCPs should be useful and informative, but at the same time simple, transparent and measurable based on characteristics common to all processes, MCPs and services (Allegrini et al., 2015; Atlee and Kirchain, 2006; Ingwersen et al., 2014; Schmidt-Bleek, 2008), and amenable to evaluation. Prior to selecting metrics, consideration of the suite of metrics that currently exist and of the way these can be used in ensuring an effective and transparent analysis of entire systems, while retaining simplicity and comprehensiveness, is a gap that needs to be addressed in order to facilitate sound decision-making processes.

Therefore, this paper aims to address this gap by providing a critical review of the existing metrics suggested and used by the literature on RRfW and sustainable resource management for measuring the benefits and impacts in environmental, economic, social and technical dimensions of value associated with MCPs lifecycle management. A methodology as to how this literature was processed is presented in Section 2, where we provide an overview of the evolution of metrics used in supporting decision-making in this field with particular reference to MCPs production, consumption and management, and of how we examined all the methods and tools and the metrics used therein in order to aid evaluation and optimisation of the RRfW systems. Also in Section 2, we explain the importance of mass balance analysis in enabling a comprehensive appraisal of the system under examination, the problem that is to be solved, and as a consequence of the importance of assigning metrics to mass flows and stocks. In Section 3 we outline the metrics that have been widely used for evaluating environmental aspects associated with the use of MCPs and their recovery from waste. Given the relatively large number of environmental metrics that have been developed over the past decades, it was considered prudent that these were distinguished by the scope they aim to serve, including amongst others, carbon emissions, pollutant emissions, energy and non-energy related categories. The metrics used for evaluating economic, social and technical aspects are presented in Sections 4, 5 and 6, respectively. The identified metrics and their ability to support systems analysis and decision-making for promoting the complex value recovery from resources/wastes and optimisation, are then discussed in Section 7. Section 8 presents the final conclusions of this study.

2. Methodology

Metrics can be established as standalone items or defined within frameworks, methods, complex tools and composite metrics. To identify the metrics that are regularly observed and used in waste management and resource recovery systems' assessment studies a considerable array of publications from peer reviewed journals and other sources (mainly from industry and public sector guidance) that catered mainly to the assessment and evaluation of RRfW systems and the evolution of these frameworks, methods and tools over the years were identified and selected for analysis. These publications included an important number of decision-support frameworks that differed widely depending on their:

- **Scope**: whether they are aimed at the optimisation of a given waste management system, such as an energy from waste (EfW) plant, or at comparing different alternatives (e.g. EfW vs. materials recycling);
- **Scale**: whether they define their system boundaries at a single unit operation (e.g. a near infrared (NIR) technology), an entire plant (e.g. a material recovery facility), or a waste management system at a wider geographic unit (e.g. local, regional or national level);
- **Focus:** whether they are focused on traditional solid waste management (SWM) (Björklund et al., 1999; Clift et al., 2000; Finnveden, 1999; Powell, 2000); integrated sustainable waste management (ISWM) (Ness et al., 2007; Seadon, 2010; Wilson et al., 2015); or the so called 'zero waste' management (Zaman, 2014); and,
- **Specific method or tool:** whether they are referring to the practical choices in translating the framework into an operational act of measurement.

Review and analysis of the above frameworks, reported and/or used in both theoretical and empirical literature, revealed that the metrics used or proposed for the assessment and evaluation procedures were often repeated in the different decision-making frameworks, or were very similar, pointing to their potential usefulness. The metrics selected in this paper ideally meet three generic criteria as follows:

(1) Have the potential to provide evidence for, or to support, evaluate and optimise RRfW systems;
(2) Are relevant for the environmental, economic, social and technical evaluation of RRfW processes and systems, with specific reference to MCPs production, consumption and EoL management;
(3) Have the potential to be measurable (quantitatively or qualitatively).

Metrics that fulfil these criteria were included in the analysis. Many of these metrics emerged from environmental analysis techniques such as LCA; whereas others derived from studies in economics, engineering and social science (Chong et al., 2016; Ingwersen et al., 2014; Morrissey and Browne, 2004; Ness et al., 2007; Pires et al., 2011; Zurbrügg et al., 2014). Here, we focused at the more complex analytical and decision support frameworks that contained metrics mainly associated with resource production, consumption and EoL management. A comprehensive review of such complex entities in terms of their analytical, evaluation or decision-support power was not intended for this study and can be found elsewhere – indicatively at Ness et al. (2007), Allesch and Brunner (2014), Morrissey and Browne (2004), Zurbrügg et al. (2014), and Pires et al. (2011).

2.1. Development and use of metrics in emerged and emerging decision-support frameworks, methods and tools for SWM and/or RRfW evaluation

Frameworks, methods and tools commonly used to support decision-makers when evaluating SWM and/or RRfW processes and systems include, amongst others, the LCA, cost-benefit analysis (CBA) (Begum et al., 2006; Clift et al., 2000; da Cruz et al., 2014; Djukic et al., 2016; Varouchakis et al., 2016; Wang et al., 2016), lifecycle costing (LCC) (Gluch and Baumann, 2004; Woodward, 1997), social life cycle assessment (sLCA) (Dreyer et al., 2006; Guinée et al., 2011), input-output analysis (IOA), environmentally extended IOA (EE-IOA), strategic environmental assessment (SEA), environmental impact assessment (EIA), environmental risk assessment (ERA), multi-criteria decision making (MCDM), cost-effectiveness analysis (CEA) (Allesch and Brunner, 2014; Gasparatos et al., 2009a, b; Morrissey and Browne, 2004; Ness et al., 2007; Singh et al., 2012; Wilson et al., 2015); and purpose-built optimisation models.

Whereas many approaches customarily include an optimisation stage, a distinct category of optimisation tools stand for the mathematical modelling techniques originally developed to deal with the cost-effectiveness of municipal solid waste (MSW) collection, treatment and disposal infrastructure and operation. These models focused on technical application aspects: for example, on the vehicle routing network used for the collection and transportation of MSW (Nuortio et al., 2006; Sonesson, 2000; Tavares et al., 2009; Truitt et al., 1969); on the selection of the type, size and location of waste facilities; and on the distribution of waste streams (municipal, commercial, etc.) to the treatment facilities within a specific geographical region (Badran and El-Haggar, 2006; Chang and Davila, 2007; Esmaili, 1972; Wilson, 1977). The strength of these optimisation models is that they can optimise aspects of technical performance against minimising the overall system cost, taking, for instance, into account transportation costs to transfer stations, landfills, incinerators, composting facilities, material recovery facilities, and the operational and fixed costs of these facilities (Chang and Davila, 2007; Wilson, 1977).

Such optimisation approaches, however, may not account for other important considerations. For example, disposal in sanitary landfills may be the preferred waste management option regarding minimisation of hazards, but may also result in high environmental impacts, and might conflict with adopted policies (Najm et al., 2002). Subsequently, as environmental and socio-economic concerns around SWM and the need to promote RRfW have gained importance, new assessment frameworks were developed, capable of including environmental and socio-economic metrics into the decision-making of SWM systems (e.g. waste recycling, facilities siting, and system operation), promoting a more sustainable management of waste (Chang and Davila, 2007; Pires et al., 2011). For instance, Fuertes et al. (1974) adapted an optimisation model to take into account the trade-offs between system costs and social aspects, such as aesthetics, size and number of regional facilities (Fuertes et al., 1974). In a number of other studies (Chalkias and Lasaridi, 2009; Chang et al., 2008), geographical information system (GIS) modelling was used in conjunction with environmental, biophysical, ecological, and socio-economic variables to provide an advanced modelling framework for decision-makers to simulate and analyse spatial waste management challenges. In other works, linear programming was integrated with a life cycle perspective to assess economic, environmental and other associated impacts (e.g. solid waste generation rate, solid waste composition and characteristics, time and transport distance, generation sources, capacity) (Chalkias and Lasaridi, 2009; Ekvall et al., 2007; Eriksson et al., 2003; Najm et al., 2002; Sudhir et al., 1996), all of which are important in long-term planning, and suitable in providing a realistic representation of SWM practices (Kondili, 2005; Morrissey and Browne, 2004; Najm et al., 2002; Pires et al., 2011).

Because it is now well established that sustainable waste management requires a comprehensive analytical approach, combinations of assessment methods and tools that combine metrics from different domains of value are also increasingly endeavoured (Chong et al., 2016; Finnveden et al., 2005). For example, amalgamations of environmental with technical (eco-design) (Knight and Jenkins, 2009) or economic aspects (eco-efficiency analysis, optimisation models), and of methods such as LCC with LCA (Carlsson Reich, 2005; Gu et al., 2008; Heijungs et al., 2012; Norris, 2001; Ristimäki et al., 2013) or IOA with LCA (Joshi, 1999; Junnila, 2008; Ochoa et al., 2002) have gained recognition. Even the development of new methods that incorporate metrics previously coined for use in LCA, LCC and sLCA, such as the newly developed life cycle sustainability assessment (LCSA) have emerged (Chong et al., 2016; Finkbeiner et al., 2010; Gencturk et al., 2016; Giannakis and Papadopoulos, 2016; Guinée et al., 2011; Klöpffer, 2003).

A method used to assess the sustainability of organisations is the accounting framework called the triple bottom line (TBL). TBL goes beyond the traditional measures that organisations use to assess their profits to also include environmental and social elements (Dao et al., 2011; Elkington, 2004; Saavalainen et al., 2015; Slaper and Hall, 2011). However, no universal standard method, or an accepted standard for the metrics that comprise each of the three TBL categories currently exists, while the metrics included are difficult to measure (Slaper and Hall, 2011). As such, this method is not considered further herein. A number of sustainability assessment methods that evaluate the performance of industrial facilities have been developed, e.g. by the World Business Council for Sustainable Development (WBCSD, 2000), the Global Reporting Initiative (GRI, 2013), the American Institute of Chemical Engineers (AIChE) and the Institution of Chemical Engineers (IChemE) (IChemE, 2001). These approaches provide the metrics organisations need to use to measure and report their economic, environmental, and social performance (Saavalainen et al., 2015).

More explicitly, GRI has developed an environmental, social, and governance (ESG) reporting framework used by many industries worldwide (GRI, 2013; Saavalainen et al., 2015), whereas WBCSD have developed the eco-efficiency indicators successfully used in many studies (WBCSD, 2000). The AIChE has developed the *sustainability index* (*SI*) to measure the sustainability performance of representative companies in the chemical industry, using seven key metrics including: environmental performance that measures the 'greenness' of the companies through assessing material intensity, energy intensity, water consumption, toxics release, and pollutant effects (Saavalainen et al., 2015; Tanzil and Beloff, 2006), product stewardship, sustainability innovation, value chain management, social responsibility, safety performance and strategic commitment (Saavalainen et al., 2015). Similarly, in the IChemE methodology the *sustainable development progress metrics* (environmental, economic, and social) have been fashioned to measure the sustainability of operations within the process industry for enhancing their sustainability performance (IChemE, 2001; Labuschagne et al., 2005; Saavalainen et al., 2015).

The strength of the above methods and tools in supporting decision-making in complex, interdependent systems, such as RRfW, lies on their ability to adopt a whole system approach that reflects their complexity (Blengini et al., 2012; Turner et al., 2016). To handle the increased complexity of RRfW systems, Turner et al. (2016) proposed to combine assessment methods and tools with material flow analysis (MFA). In these combinations, MFA provides valuable information about the flows and transformations of MCPs as they move through the economy at different system levels (e.g. regional, national or economy-wide systems) (Brunner and Rechberger, 2004; Hotta and Visvanathan, 2014), while the assessment methods and tools provide the valuable information on the performance of the RRfW systems via the use of metrics. The combination of MFA with metrics for 'value' analysis and assessment tools, can help in evaluating existing RRfW processes, and most importantly support stakeholders in identifying optimal future RRfW strategies. Details about the use of MFA are outlined in the following section.

2.2. The importance of MFA in systemic analysis

Material flow analysis is a tool that has been widely used for analysing the flows and stocks of economic entities ('goods'). It looks specifically into the flows in and out of a system (Ness et al., 2007), and provides an insight into the fate of 'goods' (in the form of MCPs and residues) from their BoL towards their EoU and EoL stage, and their subsequent management and treatment as presented in Fig. 1. MFA is often represented in the form of detailed flow diagrams that invokes the mass balance principle, in line with the law of conservation of matter, to get an integrated view of resource flows, comparing all inputs, outputs, stock growth or sinks, and hidden flows (e.g. mining overburden, harvest losses, waste generated upstream).

In contrast to MFA, substance flow analysis (SFA) is widely used for analysing the substances that flow in a system; where substances are defined as uniform entities consisting of uniform units (e.g. chemical elements (atoms) or chemical compounds (molecules)) (Brunner and Rechberger, 2004; Stanisavljevic and Brunner, 2014). SFA provides an essential insight into the characteristics of MCPs, with a focus on their hazardousness, technical performance, lifecycle transformation and exchanges with the environment. MFA and SFA are linked by the fact that SFA can be defined as a detailed level application of the basic MFA concept tracing the flow of selected chemical substances or compounds - e.g. potentially toxic elements (PTEs) such as, mercury, lead, chromium, arsenic, etc.; nitrogen; and phosphorous – that are contained in the goods analysed by MFA (Stanisavljevic and Brunner, 2014). This said, SFA can be extremely useful at various levels of the RRfW system (Antikainen et al., 2005; Asmala and Saikku, 2010; Ness et al., 2007), offering important metrics that can describe technical dimensions of value. This level of information, which is essential for the optimal recovery of resources at their EoU and EoL stage, can be used as the backbone onto which to 'attach' the multi-dimensional values. In addition, there is inherently a certain degree of uncertainty in the data used to calculate the metrics. This will be explicitly and transparently tackled, inter alia carried through as metadata.

The following sections provide a critical analysis of the usefulness, robustness and strength of environmental, economic, social and technical metrics discussed in the literature for SWM and/or RRfW assessments that fall under the four domains of value.

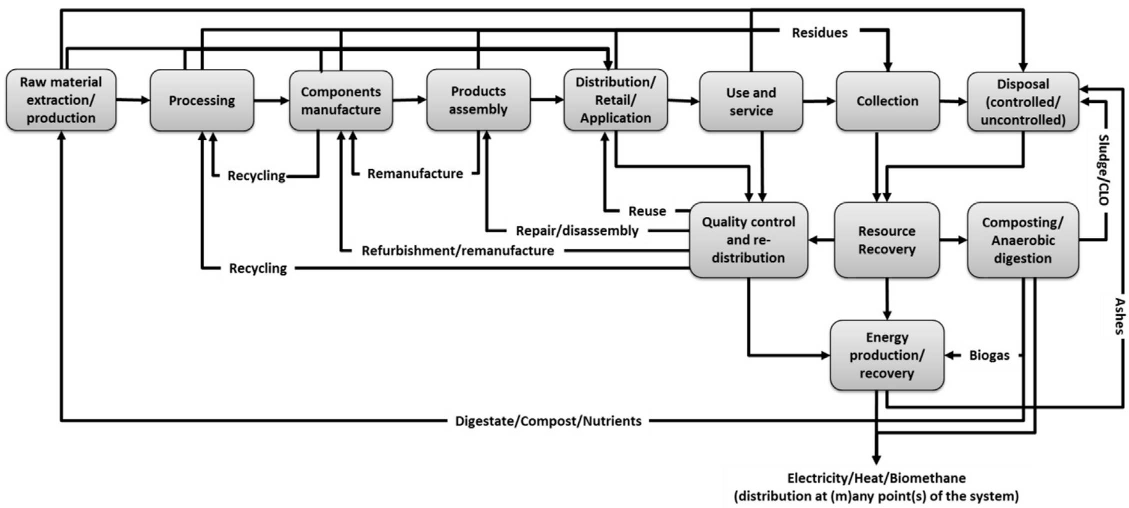

Fig. 1. Simplified RRfW system representation. Releases to biosphere (air, water, soil) at all stages of supply chain are for sake of simplicity not included in the figure.

3. Environmental metrics

Understanding the environmental benefits and impacts of all processes, including those associated with RRfW, is important for ensuring the protection of human health and ecosystems. LCA is the best known and commonly used tool for assessing the environmental impacts of a product's life from raw material extraction to EoU, disposal and EoL management, making capable and useful comparisons between products, processes and systems (Allegrini et al., 2015; Finnveden et al., 2009; Guinée et al., 2011; Hellweg and Milà i Canals, 2014; Laurent et al., 2014; Parkes et al., 2015; Rigamonti et al., 2013a, 2013b). In essence, LCA creates a model of the flow of MCPs through processes in a system and examines the environmental impact of each one of the processes and how they should be allocated to products and co-products. LCA is similar to MFA, but in practice often simplified especially with respect to system boundaries and the functional complexity of many processes in the RRfW systems and sub-systems (Turner et al., 2016). The wide and versatile nature of LCA in capturing the environmental benefits and impacts of RRfW systems is exemplified in its use to:

- assess the environmental and energetic performance of waste management systems (Al-Salem et al., 2009; Antonopoulos et al., 2012; Arena et al., 2003; Blengini et al., 2012; Bovea et al., 2010; Buttol et al., 2007; Eriksson et al., 2005; Finnveden et al., 2005; Giugliano et al., 2011; Kirkeby et al., 2006; Rigamonti et al., 2013a, 2013b);
- assess the environmental and energetic performance of industrial processes (Azapagic and Clift, 1999; Brentner et al., 2011; Burgess and Brennan, 2001; Jacquemin et al., 2012; Sonnemann et al., 2003; World Steel Association, 2010);
- compare different waste management processes and/or energy recovery strategies (Abduli et al., 2010; Astrup et al., 2009; Blengini, 2008a, 2008b; Bovea and Powell, 2006; Cherubini et al., 2009; Chong et al., 2016; Christensen et al., 2009; den Boer et al., 2007; Eriksson et al., 2005; Finnveden et al., 2005); and
- evaluate component and product performance used in different applications (Azapagic, 1999; Joshi, 1999; Junnila, 2008).

LCA classifies environmental impacts into a number of impact categories (in our terminology: dimensions of value from the environmental domain) of which number varies depending on the LCA methodology framework used (Stranddorf et al., 2005). Amongst the different frameworks, the impact categories that are most widely used, and for which there is international consensus, include *global warming*; *stratospheric ozone depletion*; *acidification*; *terrestrial eutrophication*; *aquatic eutrophication*; *photochemical ozone formation*; *human toxicity*; *ecotoxicity;* and *resource depletion* (Acero et al., 2015). The first one is the topic of sub-section 3.1 while the others are presented and discussed in sub-sections 3.2, 3.3 and 3.4. All these metrics are aggregates composed of more specific ones, e.g. the acidification metric aggregates metrics on the emissions of various substances with an acidifying property.

Furthermore, metrics can be distinguished between *direct* and *indirect* metrics based on the way they are measured. Direct metrics refer to the on-site and/or internal measurements that occur during a specific process whether this is the production, use, collection and management of a functional unit (i.e. a material, component or product); whereas indirect metrics refer to the off-site, external, upstream or downstream measurements that are not physically related to the functional unit, but are associated with it (Lee, 2011; Peters, 2010; Wiedmann and Minx, 2008; WRI/WBCSD, 2011). LCA methods aim to account for both direct and indirect environmental impacts, and this is especially prevalent in the case of emissions and resource consumption (Zhang et al., 2010). There is no established term to label the sum of direct and indirect emissions/ resource consumption. The term 'total' should be avoided as full completeness is impossible to achieve. Most of the metrics used in LCA have also been used in other assessment methods or as sustainability indicators, demonstrating further their usefulness, robustness and informative character. Other metrics, widely used in a number of assessment methods including GRI, WBCSD eco-efficiency analysis, green design, eco-design, and sustainability assessment methods include, amongst others, the *recycled content* or *renewability feedstock*, *energy efficiency, landfill use,* which are presented in Section 3.5 (GRI, 2013; UNEP and SETAC, 2011; WBCSD, 2000; Zurbrügg et al., 2014).

Section 3 is organised as follows: the first 4 sub-sections look at environmental metrics from a thematic perspective (carbon, pollution, resource depletion); sub-section 3.5 looks transversal at all those environmental dimensions from an *efficiency* perspective, while 3.6 presents metrics that integrate several dimensions of environmental benefits and impacts.

3.1. Carbon emission metrics

Perhaps the most widely known and used metric in environmental assessment of RRfW systems is that related to greenhouse gas (GHG) emissions (e.g. carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O)). GHGs are substances which absorb and re-emit heat, thereby warming up the globe's atmosphere; hence the *global warming potential (GWP)* metric (Fuglestvedt et al., 2001). *GWP* is known by many names including GHG emissions, carbon emissions, or carbon footprint (Christensen et al., 2009). Carbon footprint has emerged from ecological economics and has been widely used in EE-IOA (Fang et al., 2014; Heijungs, 2011; Minx et al., 2009; Ridoutt et al., 2015; Wiedmann, 2009). Nonetheless, some uncertainty still governs its definition, meaning and measurement (Peters, 2010; Wiedmann and Minx, 2008). For example, in a large volume of studies carbon footprint is used interchangeably with the *GWP* or carbon emissions, as a measure of the 'total' amount of GHG emissions that are directly and indirectly caused by an activity or accumulated over the lifecycle of a product (Fang et al., 2014; Heijungs, 2011; Hertwich and Peters, 2009; Hoekstra and Wiedmann, 2014; Schulz, 2010). In other studies carbon footprint is used to account for only some of the GHGs, most often CO_2 emissions, generated by an activity or accumulated over the lifecycle of MCPs (Matthews et al., 2008; Wiedmann and Minx, 2008). Evidently, the term 'carbon' is used in a variety of ways to express GHG emissions, which tends to be ambiguous and potentially confusing.

The World Resources Institute (WRI) and WBCSD, in an effort to provide guidance to businesses for measuring direct and indirect carbon emissions associated with the entire lifecycle of MCPs, have developed the GHG protocol. In this protocol carbon emissions include only the Kyoto Protocol GHGs (i.e. CO_2, N_2O, CH_4, hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulphur hexafluoride (SF_6)) (Lee, 2011; Matthews et al., 2008; Peters, 2010; WRI/WBCSD, 2011). However, in the waste sector, carbon emissions are reported based on the Entreprises pour l'Environnement (EpE) protocol that was developed to provide guidance for waste management activities, and includes only the gases most relevant to the sector which are CO_2, CH_4 and N_2O (EpE, 2013; Gentil et al., 2009; UNEP, 2010). Essentially this means that so far no approach has looked at the full range for GHGs when it comes to RRfW systems, let alone other impacts of resource use and recovery on climate change, such as forest clearance or albedo change (Gentil et al., 2009).

The literature contains a considerable body of work on how to account for carbon emissions. Carbon emissions can be directly and/or indirectly generated at each process of the RRfW system, including emissions from energy (e.g. electricity and/or fuel use) and non-energy related activities (Gentil et al., 2009; Machado et al., 2001; Nishimura et al., 1997; US EPA, 2006) (e.g. process and fugitive emissions, such as CH_4 released from digesters and composting technologies used in mechanical biological treatment plants) (Amlinger et al., 2008; Flesch et al., 2011). Based on the GHG protocol direct carbon (or GHG) emissions are those arising on-site by the process considered as 'central' and fall into scope 1 (direct GHG emissions generated on-site) and 2 (GHG emissions from consumption of purchased electricity, heat or steam on-site) (WRI/WBCSD, 2011).

Indirect carbon emissions refer to the off-site, external, upstream or downstream emissions and are also known as *embodied carbon (EC)*. These emissions are categorised into scope 3 and include emissions associated with the extraction and production of purchased materials and fuels, transport-related activities, electricity-related activities not covered in Scope 2, outsourced activities, waste disposal, etc. (WRI/WBCSD, 2011). This metric has gained increase popularity over the last decades due to its potential to account for the carbon embedded in MCPs through their whole lifecycle (Cabeza et al., 2013; Ecorys, 2014; Lee, 2011; Peters, 2010; Purnell, 2012; Schulz, 2010) (Table 1). *EC* is heavily dependent on the system boundaries applied, which can lack transparency. Moreover, conversion factors for many MCPs are only very roughly estimated by both LCA and EE-IOA.

In waste management systems *direct carbon (DC)* emissions are usually related to the processes of collection, transportation, management (i.e. incineration, reprocessing, composting, remanufacturing) and transboundary movement of waste resources, whereas carbon emitted during the upstream processes of manufacturing, including extraction and processing, transportation, use and international trade of MCPs, are accounted for as indirect or *embodied carbon (EC)* (Table 1) (Bernstad and la Cour Jansen, 2012; Clift et al., 2000; Nässén et al., 2007; Schulz, 2010). This points to an important aspect concerning the system boundaries of an assessment study: direct or indirect emissions depends on the 'central' system looked at, and on what are direct emissions from the producer's perspective and indirect emissions from the waste manager's perspective.

The emissions of the various GHGs (whether direct or embodied) are aggregated to carbon emissions based on the warming potential of each single GHG over a given period of time (normally a time-horizon of 100 years is adopted) as specified in the Kyoto Protocol, using CO_2 as the reference gas (IPCC, 2007; Peters, 2010; UNEP, 2010); hence the measurement unit in tonnes CO_2 equivalent (tCO_2e). For example, 1 kg of CH_4 causes 25 times more warming over a 100 year period than 1 kg of CO_2.

The *avoided carbon (AC)* emissions, presented in Table 1, are important in carbon accounting strategies. These may represent the: i) emissions saved from the avoided landfilling of waste; ii) the reduced input of raw materials and other resources when these are replaced by reusable, repaired, or recycled materials (e.g. reuse of construction components in new buildings, redistribution of edible food, replacement of fertiliser by compost, plastic bottles recycled into plastic bottles, use of solid recovered fuels for energy generation) (EpE, 2013; Gentil et al., 2009; Scheutz et al., 2009; Smith et al., 2001; UNEP, 2010; US EPA, 2006); or iii) those that occur when energy is produced as a co-product in waste treatment processes (e.g. electricity and/or heat produced out of landfill gas, biogas, incinerator), that replaces partially or fully the energy generated from fossil fuels (UNEP, 2010). In the latter case *AC* emissions are difficult to estimate due to the uncertainty related to what is being replaced (e.g. operating patterns, energy content, energy mix) (Smith et al., 2001). For material resources the energy content is usually determined based on their net calorific value. Differences in *AC* emissions are thus expected, especially when taking into account local circumstances (US EPA, 2006). Usually, assumed emission factors per unit of energy recovered are used, which for the European context can be estimated using a number of variables including energy mix and heat generation efficiencies (Smith et al., 2001). Conceptually, *AC* emissions are significantly different from *DC* and *EC* emissions. For *DC* and *EC* emissions one estimates the emissions from one process (with different system boundaries), while for *AC* emissions one must estimate the emissions for two alternative processes and calculate the difference (for the same system boundaries).

The real hurdle when it comes to accounting for carbon emissions is in regards to *biogenic carbon* (BC_e) emissions (see def. in Table 1) (US EPA, 2014). In a vast number of studies it is implicitly assumed that the BC_e of biodegradable materials (e.g. organic waste contained in food and garden waste, paper and cardboard) released in the atmosphere after combustion, is in equilibrium to that absorbed by the biogenic pool (i.e. during the growth of plants);

Table 1
Key carbon emission metrics used in RRfW assessment methods.

Metric	Description	Unit	References
Direct carbon emissions (DC)	Sum of carbon emissions released on site or from a specified process, due to direct consumption of energy (fuels or electricity) by the process considered as 'central' (independent of whether generated on-site or purchased, e.g. engines, motors, power plants, buildings, appliances, electronics, lighting), and direct use of GHGs and/or products that contain or form GHGs that are emitted during use (e.g. refrigeration and air-conditioning equipment, industrial gases, fertilizers).	tCO_2e	(WRI/WBCSD, 2011)
Indirect or embodied carbon emissions (EC)	Emissions associated with upstream and/or downstream (off-site) processes associated with MCPs. They may also include emissions from the generation of purchased electricity, steam, heating and cooling consumed by any process not considered as 'central', measured based on the sum of units input upstream and/or units output downstream (e.g. in kg), multiplied by the process-specific unit emission factor (e.g. $kgCO_2e/kg$).	tCO_2e	(WRI/WBCSD, 2011; Schwarz et al., 2002)
Avoided carbon emissions (AC)	For MCPs: EC savings accruing from reductions in quantity and quality of primary materials, measured based on the sum of avoided units input (in kg), multiplied by the process-specific input emission factor (e.g. $kgCO_2e/kg$). For energy: EC savings from energy substitution by renewable energy, measured based on the difference of the emissions calculated from e.g. the amount of gas recovered, calorific value, conversion efficiency and assumed emission factors for the replaced alternative energy source.	tCO_2e	(Smith et al., 2001)
Biogenic carbon emissions (BC)	Emissions related to the natural carbon cycle, as well as those resulting from the production, harvest, combustion, digestion, fermentation, decomposition, and processing of biologically based materials by humans. A common measurement method has not yet been agreed.	tCO_2bioe	(US EPA, 2014)
Carbon capture and sequestration (CCS)	It is the carbon stored in a processed material output that is not released in the atmosphere. A common measurement method has not yet been agreed.	tCO_2e	(Gentil et al., 2009; UNEP, 2010; WRAP, 2010a,b; Levasseur et al., 2012)

hence it is purported that it should not be accounted for as contributing to the global warming effect (Boldrin et al., 2009; Giugliano et al., 2011; Gunn et al., 2012; Smith et al., 2001). Contrariwise, other studies suggest that BC_e released from activities such as permanent deforestation, burning of a tropical forest or combustion of forest biomass for energy, is not entirely absorbed by biomass systems (Gunn et al., 2012; Rabl et al., 2007). The basis of their argument is that the *lower heating value (LHV)* (or net calorific value) of carbon is the same regardless of its source, and as such BC_e that is released in the atmosphere can also contribute to the global warming effect, measured using a unit-based index called GWP_{bio} or $CO_{2biogenic}$ emissions (Blengini, 2008a; Cherubini et al., 2011; Christensen et al., 2009; Gentil et al., 2009; Parkes et al., 2015).

Nonetheless, there is not yet a common measurement method for BC_e and this is also due to the fact that not all of the carbon from organic materials entering treatment, is returned to the atmosphere. In fact some of it remains stored in the material after the treatment process, reducing BC_e emissions. This carbon is accounted using the *carbon capture and sequestration (CCS)* metric (see def. in Table 1) (Rabl et al., 2007; Smith et al., 2001; WRAP, 2010b). If the *CCS* is in a form unavailable to the natural carbon cycle over a sufficiently long time period, then it could be argued that a 'sink' for carbon has been created. The two main routes for carbon storage in waste management systems are via landfills (e.g. where anaerobic conditions inhibit the decomposition of lignin based materials) and compost applications to soil (e.g. where carbon is converted to stable humic substances that may persist for hundreds of years) (Smith et al., 2001; WRAP, 2010a).

The stability of such sinks is difficult to assess, and may have different time scales between *CCS* and BC_e for different MCPs due to degradation rates (Rabl et al., 2007; Smith et al., 2001). For example, wood used in buildings, furniture and wood-based materials can have *CCS* for decades or centuries, but eventually much, or all of it, will be re-emitted to the atmosphere (Rabl et al., 2007). Similarly, the rate by which the compost will re-emit *CCS* depends largely on how the soil is managed (e.g. cropping, tillage, irrigation, compost application rate), the climate, the composition of the soil, and the time-period that the compost is applied to land, as carbon releases (in the form of N_2O) are likely if vegetation is not taking up the nitrogen at the time of application (Boldrin et al., 2009; Smith et al., 2001). What makes BC_e accounting different from the accounting of other metrics is the time dimension. To measure BC_e adequately one must consider the distribution of GHG emissions, storage and sequestration along the time axis, as any delay in the release of GHG emissions by temporal storage in products or waste, is important in terms of slowing down climate change (Cherubini et al., 2011; Hellweg and Milà i Canals, 2014).

This raises the issue of how this carbon should be accounted especially when comparing the treatment options of different MCPs. Given the complex dynamics of *CCS*, and the high degree of uncertainty related to its measurement, discussions on how to measure *CCS* in landfills and soils amended with compost, or exchanges with the energy industry and the wood, pulp and paper industries, are still ongoing (Gentil et al., 2009; Levasseur et al., 2012; UNEP, 2010). In the case of BC_e the U.S. Environmental Protection Agency (US EPA) has developed a framework for accounting BC_e from stationary sources (US EPA, 2014). Notwithstanding its potential to account for carbon emissions, this framework measures *EC*, *AC* and *CCS* emissions all in the same formula, thus does not provide insights into specific carbon emissions. In spite of that, the controversy around the carbon neutrality of BC_e is ongoing, and further clarifications on how to measure it are highly desirable in the resource and waste management sectors, as this would shed some light on the environmental assessment process of waste management options. For example, if BC_e is disregarded, emissions from the incineration of organic wastes are unaccounted for, making incineration to always look better than landfill as CH_4 emissions from landfill are in fact accounted for (WRAP, 2010a).

3.2. Pollutant emissions to air, water and soil

Other gases and compounds critical to the environment and as a consequence to human health are also emitted during RRfW processes. These include (Acero et al., 2015; Azapagic et al., 2003) the release of:

- ozone-depleting gases such as chlorofluorocarbons (CFCs), hydrochlorofluorocarbons (HCFCs) and halons that contribute to

the damage of the stratospheric ozone layer, measured by the *ozone depletion potential* (*ODP*) metric;
- volatile organic compounds (VOCs) and other substances that contribute to photochemical ground-level ozone formation, measured by the *photochemical ozone formation potential* (*POFP*) metric;
- gases that contribute to air, water and soil acidification, measured based on their acid formation ability (ability to form H^+ ions), measured by the *acidification potential (AP)* metric;
- ammonia, nitrates, nitrogen oxides and phosphorous that contribute to the eutrophication of marine, freshwater and terrestrial ecosystems, measured by the *eutrophication potential (EuP)* metric; and
- other substances, such as PTEs, particulate matter (PM), polycyclic aromatic hydrocarbons (PAHs) and persistent organic pollutants (POPs) that contribute to human and ecotoxicity, measured by the *human toxicity potential (HTP)* and *ecotoxicity potential (ETP)* metrics.

These metrics, which are perhaps the most widely known metrics for assessing the environmental impact of various pollutants, are described in detail in *Supplementary Material* (Table S1). All these metrics can be summed in a single metric called the *pollutant emissions* metric (Allegrini et al., 2015; Schwarz et al., 2002). However, such an aggregation can provide no insights into the specific pollutants nor their effects on the environment and human health; hence, this metric is not further considered herein. Air pollution control technologies and processes, such as wet-scrubbing, desulphurization, ammonia removal, acid dry neutralization, fabric fly ash filtration, tar cracking and dioxin absorption are often implemented to meet emission requirements (Chong et al., 2016). These measures aim at reducing the release of these pollutants to the atmosphere and their subsequent impacts to the environment and human health, linking the social (Section 5) and technical (see Section 6) with the environmental domain of value.

Some additional forms of pollution that have not yet been (extensively) used to assess the environmental damage caused by RRfW processes, have been identified. Because of the little discussion around these forms of pollutions, there are currently no specific metrics used to describe them, regardless the fact that are widely accepted by the science community. Even so, these forms of pollution are gaining increased momentum and could not be ignored. These include the:

- *emerging organic contaminants (EOCs)* such as pharmaceuticals, hormones, and bisphenol A (e.g. those found in consumer products or over the counter prescription medications) (Boxall, 2004; Kleywegt et al., 2011; Kümmerer, 2003);
- *black carbon (BC)* emissions, which constitute the main component of soot produced due to incomplete combustion of fossil fuels, biomass and/or SRF;
- *waste heat* losses to the environment that occur during most industrial processes and in power plants, which can significantly increase the temperature of the environment, leading to waste heat pollution or thermal pollution; and
- *nanoparticle emissions.*

The latter (i.e. nanoparticle emissions) is relatively new (Acero et al., 2015; Hellweg and Milà i Canals, 2014). Although nano-sized particles are common in nature (e.g. proteins, enzymes, DNA), this form of pollution in the RRfW context refers particularly to the engineered nano-sized particles that are intentionally designed to serve a specific purpose (e.g. carbon black and fumed silica for applications in plastic fillers and car tyres, silver nanoparticles coated onto polymers like polyurethane) (Albrecht et al., 2006; Hoet et al., 2004; Jain and Pradeep, 2005). A detailed description of this, and of the other three metrics can be found in *Supplementary Material.*

What is worth noticing herein is that discussions around the potential consequences of these forms of pollution in the environment and associated measurements are still unclear. Further research is thus required into accounting for their effects on human health and ecosystems.

3.3. Resource depletion: energy related metrics

The resource depletion metric used in LCA studies includes the consumption of resources such as fossil fuels, metals and minerals. This sub-section concentrates mostly on fossil fuels, but nuclear and biogenic fuels, wind, water, geothermal and solar energy are also considered as they are gaining increased momentum as energy-related resources.

Fossil and other fuels are used for energy generation (e.g. electricity, motion and heat) of which consumption is one of the most widely documented metrics in the literature (alongside carbon emissions with which it is associated). The list of energy metrics in use is extensive, and can be found in the reports of IAEA, UNDESA, IEA, EUROSTAT and EEA, 2005 (Kemmler and Spreng, 2007), EEA, 2006, and DECC 2015 (DECC, 2015). LCA, IOA and sustainability assessment indicators include energy use as one of the most basic metrics to describe a production or recycling system. The basic energy consumption metric known as *direct energy consumption*,

Table 2
Key energy related metrics used in RRfW assessment methods.

Metric	Description	Unit	References
Primary energy consumption (PEC)	Sum of 'raw' or 'gross' energy input (per process), based on the calorific value of fuels used. For energy not based on 'burnable' fuels a variety of – partly inconsistent – conventions exist.	kWh	(Nässén et al., 2007; IEA, 2014)
Specific energy consumption	Sum of final energy consumed per unit of output[a].	kWh/t output	(Siitonen et al., 2010)
Cumulative energy demand (CED); or Gross energy requirement (GER)	Sum of all the material and energy inputs to the analysed system multiplied by appropriate oil equivalent factors (g oil eq./unit of inputs) and then converted to SI energy units by multiplying by the standard crude oil equivalency factor of 41.860 J/g.	kWh/unit of inputs	(Cherubini et al., 2009)
Renewable energy generation	Sum of (potential) renewable energy generation from recovered MCPs per unit input.	kWh/t input	(Bernstad and la Cour Jansen 2012; Perkoulidis et al., 2010)
Exergy	Sum of the η_{Ex} of useful exergy from the process to the total exergy to the process based on the exergetic value of all characterized resources in the process.	MJ	(Gundersen, 2009; Ayres et al., 1998)
Emergy	Sum of the emergy value of all characterized resources in the product system, using the solar emjoules (sej).	sej/t	(Ingwersen et al., 2014; Giannetti et al., 2010)

[a] If energy in different forms (e.g. fuels, steam and electricity) is used, then these are accounted for in the final energy consumption.

refers to the energy extracted or purchased and consumed directly by any sector of the economy (e.g. transport fuels, electricity, gas for heating). It includes both primary and secondary energy consumption. *Primary energy consumption* (Table 2) is the energy produced from coal, crude oil, natural gas, nuclear materials and renewable sources (including solar energy, wind energy, bioenergy, hydropower, marine energy, geothermal) (Liu et al., 2014; Nässén et al., 2007), of which conversion to electricity, heat and other human induced transformation (e.g. refinery gas, diesel, naphtha, ethane, gasoline) represents the *secondary energy produced* (OECD/IEA, 2005; Øvergaard, 2008). As opposed to direct energy consumption, indirect energy consumption refers to the energy consumed to produce the energy, goods or services used in a specific process in the RRfW system called as 'central' (Baynes et al., 2011). In an IO consumption analysis, indirect energy refers to energy embodied in the production, storage and transport of goods and services consumed (Baynes et al., 2011), for which more details are provided further down in this section.

To gain some scrutiny into the energy used per unit of MCP produced, used, collected, sorted, repaired and reprocessed the *energy intensity* metric was proposed in the studies of Székely and Knirsch (2005), Schwarz et al. 2002, Bernard and Côté 2005. This metric can be used to measure the energy consumed to provide the heat and power requirements for the process per tonne of product produced, and thus it can provide information on the process operations, focusing on environmental performance and process improvements. However, *energy intensity* has a dual function as besides its ability to express the energy use per mass unit of output, it can also be used to express the energy use per unit of monetary value (Schwarz et al., 2002; Székely and Knirsch, 2005). In the study of Bernard and Côté (2005) this was indicated as a more meaningful way of measuring industry outputs and assessing process improvements, such as enhanced energy recovery or higher production capacity (Bernard and Côté, 2005).

This is in line with other studies, indicatively Liu et al. (2014), Krajnc and Glavic (2003), where the *energy intensity* metric was only associated with economic terms (e.g. the prices of components/products sold or value added), whereas Reddy and Ray (2011) and Nässén et al., 2007) made a distinction between physical and economic *energy intensity*, with the first accounting for the final energy use per physical unit of output, and the latter for the final energy use per monetary value of unit of output sold (Nässén et al., 2007; Reddy and Ray, 2011). Reddy and Ray (2011) state that implications in economy (e.g. fluctuations in the price of materials, monopsonistic industries that control a significant share of the price of specific materials, and monopolistic industries that control a significant share of the produced goods and/or have a better image), as well as a number of other factors (e.g. energy mix changes, energy input mix changes and energy-for-labour substitution processes) (Patterson, 1996), can result in variations in the energy intensity which may not reflect or even negatively affect changes in (technical) *energy efficiency* (more on energy efficiency on Section 3.5).

As such, for defining physical *energy intensity*, Reddy and Ray (2011) have used the *specific energy consumption* and the *adjusted energy consumption* metrics. The latter was defined as the total energy consumption adjusted to the weight of unit process output, whereas *specific energy consumption* was defined as the ratio of total energy consumption to total unit (MCPs) production. The *specific energy consumption* metric was also used in the study of Krajnc and Glavic (2003) and Siitonen et al. (2010) to express the amount of energy used per unit output. As such, to retain consistency, the *specific energy consumption* metric was used herein to express the energy used per unit output (Table 2).

It is important to keep the energy *consumption* metrics and energy *efficiency* metrics distinct; while the consumption compares energy used to produce a unit of MCP, energy efficiency (gain) compares energy input and output of a process. Such a comparison provides the ability to assess the effectiveness of a process in recovering resources from waste as for example, by reducing the amount of resources used in production – consumption processes or by increasing the amount of resources recovered (in a variety of forms) during collection, reuse, recycling, recovery after their disposal as wastes. Further details on the use of efficiency metrics are found in Section 3.5.

Cumulative energy demand (CED), also known as *lifecycle embodied energy (LEE)* or *gross energy requirements (GER)* are metrics originating from the Embodied Energy Analysis method that deals with the direct and indirect energy consumption required across the entire lifecycle of MCPs (Table 2) (Cherubini et al., 2009; Patel et al., 2000; Ulgiati et al., 2011; Worrell et al., 1994). The different forms of energy (renewable or non-renewable) consumed throughout the system are converted back to their primary energy sources including crude oil, natural gas, anthracite, lignite, uranium ore, hydropower, biomass and others, taking into account conversion losses from electricity and heat generation (Arena et al., 2003; Bengtsson, 2004; Kaufman et al., 2010; WBCSD, 2000). As such, when using these metrics, different system boundaries and geographic location can pose a constraint in getting a certain value for a certain product and as such, deviations can be relatively high. A prerequisite when using *CED* or *GER* is to retain consistency in the way data are collected and used. These metrics have been widely used in the evaluation of the environmental and energy impacts of several sectors. The *CED* is frequently employed to determine energy payback periods for alternative generation technologies such as solar and wind; and to evaluate the efficacy of efforts to produce energy from biomass (Kaufman et al., 2010).

Energy recovered from material landfilling (i.e. biogas) and from the digestion and combustion of bio- and other wastes with energy recovery (EfW), can be accounted for as contributing to renewable energy generation measured using the *renewable energy generation* metric (Table 2). This metric is a measure of the renewable energy generated from MCPs in the form of biogas and syngas that are often combusted to produce electrical energy and heat via a CHP engine, or alternatively are converted to liquid fuel such as, gasoline and liquefied biomethane (Belgiorno et al., 2003).

Meanwhile, to account for all the energy flows in an economy based on the first law of thermodynamics, the energy analysis tool has emerged. The key principle of this tool is that energy is constant and cannot be created nor destroyed, but it can only be converted into different types or 'qualities' of energy measures, such as *exergy* and *emergy* (Finnveden and Moberg, 2005; Hovelius, 1997). Both the exergy and the emergy form of analyses are more advanced than the previously mentioned energy related metrics as they consider both the quality and quantity of energy consumed (Dincer and Rosen, 2012). *Exergy*, or else 'useful' energy, is the maximal amount of mechanical or ideal work (or the work content of a variety of streams, e.g. mass, heat, work, that flow through a system) that can be obtained from a system that moves from a particular state to a state of equilibrium with the environment, based on the second law of thermodynamics (Table 2) (Bejan, 2002; Gundersen, 2009; Wall, 1977; Wall et al., 1986, 1994). It is not only a measure of inputs, but also a measure of outputs (Ayres et al., 1998).

As such an *exergy* analysis gives an overview of the effectiveness of resource utilisation, indicating where losses occur (i.e. where exergy is destroyed), and where technological improvements can be made to increase energy efficiency. Losses can be in the form of low temperature heat, but also in the form of chemically or physically reactive materials that are dissipated into the environment (Ayres et al., 1998). These losses (i.e. waste heat and waste products) can drive undesired environmental impacts, as for example the insertion of nano- and micro-scale chemical species (e.g. toxins, nanoparticles, etc.), increasing the entropy of the system that has the potential to disrupt delicately balanced ecosystems and life processes far from equilibrium (Ayres et al., 1998).

Exergy, based on the second law of thermodynamics, is always destroyed when energy is converted, either partially or totally, and its destruction is proportional to entropy production and the reduction of products quality. This is the reason why *exergy* has been suggested as a measure of assessing the thermodynamic efficiency and resource depletion of a system (Finnveden and Östlund, 1997). In order to perform an exergy analysis the exergetic values of any occurring energy form (e.g. electrical or mechanical work, heat and material streams) have to be calculated (Fonyó et al., 1999). A detailed list of thermodynamic properties of materials and a mathematical apparatus associated with defining and calculating the exergies of various products can be found elsewhere (Bejan, 2002; Gundersen, 2009). Examples of using the *exergy* metric include the regional exergy analyses for Japan (Wall, 1990), Norway (Ertesvåg and Mielnik, 2000), Brazil (Schaeffer and Wirtshafter, 1992), Italy (Wall et al., 1994), Sweden (Wall, 1977), and the United States (Ayres et al., 2003). In the context of RRfW, exergy analysis has been reported to be a suitable way of contributing to measuring the sustainability of industrial metabolisms of MCPs (Amini et al., 2007; Dewulf and Van Langenhove, 2002).

Emergy is defined as the quantity of direct and indirect solar energy required to obtain all resources and goods used and produced by a given process, which is estimated by converting energy inputs and other flows into their solar equivalent, using the solar transformities (Table 2) (Giannetti et al., 2010; Odum and Peterson, 1996). Odum and Peterson (1996) have created a methodology for Regional Emergy Analysis where the environmental and economic values associated with the use of MCPs are represented in a common energy unit (Giannetti et al., 2010). Specifically, they have created four indices for *emergy* that aim to capture the environmental, economic and social aspects associated with the use of resources, including the *environmental yield ratio (EYR)*; the *environmental investment ratio (EIR)*; the *environmental load ratio (ELR)*, and the *environmental sustainability indices (ESI)*, for which further information can be found elsewhere (Daley, 2013; Giannetti et al., 2010).

3.4. Resource depletion: non-energy related metrics

Depletion of resources other than energy based ones is also of concern in RRfW systems. One could consider this to be the most important aspect as all efforts to recover, reuse, repair, recycle materials from wastes aims at reducing the depletion of primary natural material resources, and the environmental degradation as a consequence thereof. Whereas the importance of these metrics in assessing RRfW systems is outlined herein, a description of how these metrics are measured can be found in Table S2 *Supplementary Material*.

Water consumption, or *water footprint* or *blue water footprint* as it is known in the IOA, is a measure of the volume of surface and groundwater consumed by the production of, or incorporated into a MCP. It is one of the most widely used environmental metrics in many assessment methods and tools related to RRfW systems (Fang et al., 2015b). Mining, oil refineries, manufacturing industries (e.g. chemical producers, pulp and paper industry, food and beverage industry) and power plants use large amounts of water either as a cooling/heating medium, a cleaning agent, or a reaction solvent (Krajnc and Glavic, 2003). As such, the *water consumption* metric includes evaporation and misting losses from cooling water, water vapour vented to the atmosphere, water lost through waste treatment or disposal, and water lost through deep-well injection (Schwarz et al., 2002). *Water consumption* is important not only because its abstraction can create problems associated with low riverine flows, lowering of ground water tables, and salt-water intrusion in coastal areas, but also due to water scarcity, soil quality and biodiversity impacts in the long term in many areas around the globe (EEA, 2008; FAO, 2009; Thames Water, 2016). In areas where water scarcity constitutes a problem, production and waste treatment processes that require large amounts of water (e.g. food production, incineration, etc.) might be unsustainable.

The loss of *water quality* due to its use in RRfW systems is especially important because it reduces the availability of water as a resource, leading to water resource depletion. For example, the use of water for the removal of hazardous substances from the flue gas stream during incineration (Oppelt, 1987), or the leachate produced during composting and anaerobic digestion requires special treatment in order to ensure its secure discharge, and avoidance of uncontrolled leaks of leachate (Krogmann and Woyczechowski, 2000). The use of water in industrial processes results in the generation of wastewater, the recovery and recycling of which is important in ensuring the removal of nutrients, metals, and POPs for meeting environmental protection regulatory standards (Renou et al., 2008). This is critical not only for the protection of water, soil and air from pollutants emission, but also of human health, of which metrics are discussed in Section 3.2. In this section, the description of the loss of *water quality* metric is not straightforward. It requires a series of tests that measure the physical properties of the waste water including its temperature and turbidity, as well as its chemical (e.g. pH, salinity, oxygen dissolved, total dissolved solids) and biological (e.g. algae, bacteria, etc.) parameters (APHA, 1992; UNEP/WHO, 1996).

Non-energy raw materials (e.g. minerals and metals) are essential inputs to all industries across all supply chain stages. For example, 50 different kinds of metals are used to produce a smartphone, all of which are needed to give it its light weight and user-friendly small size (Benton and Hazell, 2014). Evidently, the amount of raw materials used for the production of components and products is growing in importance due to pressures related to their future availability, the environmental damage caused by their extraction, as well as their wastage per MCP output. In the study of Schwarz et al. (2002) the amount of material used in a component/product was measured by the amount of material wasted (not converted to desirable product) per unit output, using the *material intensity* metric (Schwarz et al., 2002). However, the use of this metric in the study of Krajnc and Glavic (2003), has been associated with economic terms (e.g. value of component or value added per product sold), and the *specific material consumption* metric was used instead to express the amount of material wasted per unit output. To retain consistency with the terminology used, the *specific material consumption* metric was also used herein.

Special reference must also be made to critical raw materials (CRMs) (e.g. rare earth elements, cobalt, niobium, scandium, etc.) which in the EU have gained increased momentum over the last decade due to the need to secure reliable and unhindered access to these critical reservoirs in the coming years (European Commission, 2016). Therefore, *critical raw materials use* is an important metric in capturing the amount of these materials used and disposed during the entire lifecycle of MCPs. Criticality is an important element to be accounted for because it may vary between the different levels of the infrastructure system (e.g., materials, component, technology), as well as due to the implementation of more technically specific engineering solutions (Dawson et al., 2014).

Other metrics related to the measurement of resource depletion include the *recycled/reused content* or *feedstock renewability*, which are widely used in the GRI, WBCSD eco-efficiency analysis, green design, eco-design, and sustainability assessment methods amongst others (GRI, 2013; UNEP and SETAC, 2011; WBCSD, 2000; Zurbrügg et al., 2014). *Recycled/reused content,* or *feedstock renewability,* is an important metric that indicates the circularity of materials, components or parts of a product that are being repaired, reused, returned and recycled, displacing primary raw material use in new products, and providing a measure of waste diversion from incineration and/or landfill (Atherton, 2007; Broadbent, 2016; Steenkamer and Sullivan, 1997).

Land use is increasingly thought to be important over recent years, and is appearing in the metrics list of many assessment methods and tools (Tanzil and Beloff, 2006). This is because in RRfW systems, land has been widely used for the construction of landfill sites and other facilities that produce and manage resources, taking up space that could be used for other applications (agriculture, house building, etc.) or natural land/wilderness (Acero et al., 2015; Bengtsson, 2004; Stranddorf et al., 2006; US EPA, 2006). In addition, recovery of biotic resources can reduce the pressure on land (and sea) areas to produce ever more crops and other materials. Demolition and decommissioning of facilities and landfill sites on the one hand, will have implications to the physical characteristics and land use of the site. By their nature, such projects have the potential to change the land and landscape completely; soils may be compacted or may become contaminated with toxic materials (e.g. leachate) or they may be reduced in quality by mixing with demolition waste, such as bricks and concrete (EA, 2002); hence the *land use* metric. On the other hand, the construction of new plants will also change *land use* impact on agricultural production, and as a result in the societal opportunities provided by the specific piece of land. From an environmental point of view, land degradation as a result of land-use change constitutes an important threat to ecosystems, and may lead to natural habitat loss and alterations of the landscape (EEA, 2010). However, land degradation cannot be assessed by a single metric, but requires a series of observations and measurements related to the land condition, including soil erosion by water and wind, soil fertility decline, loss of vegetation cover and increased desertification, amongst others (Dent et al., 2007).

3.5. Efficiency metrics

This category includes a number of important metrics that aim to assess the efficiency of a process or system based on information concerning the same resource or energy flow from two specific points of the system (i.e. usually the input and output sides of a process) (Brunner and Ernst, 1986), or for two different resources (usually a secondary and primary resource) that flow at the same specific process.

Based on the second law of thermodynamics, energy conversion is never 100% efficient; it always leads to energy ‘losses’ (more correctly, e.g. downgrade of high-value mechanical or chemical energy into unusable low-temperature waste heat). As such, the *energy efficiency gain (or loss)* is one of the most widely used efficiency metrics in the literature. It is used to express the reduction in the energy consumption of a process, while delivering same unit output, or the use of same energy input for a higher output (OECD/IEA, 2005).

Although, there is no one unequivocal quantitative measure of energy efficiency (gain/improvement) (Patterson, 1996), in general, *energy efficiency* is the ratio of energy unit output to energy input (Herring, 2006), and thereby influences the amount of useful energy required to produce a unit of MCPs (Table 3). It is also used as a proxy for the energy lost to the environment as waste energy (e.g. waste heat; see Section 3.2). In a number of studies it is indicated that the use of *energy efficiency* is not representative of the unit

Table 3
Key efficiency metrics used in RRfW assessment methods.

Metric	Description	Unit	References
Energy efficiency (E_{eff})	Estimated by converting energy from one form into another, using the ratio of the sum of the useful energy output of a processto the sum of energy input for a specified energy conversion process or system. Expressed using the Greek letter η.	% (on kWh)	(Patterson, 1996; Giacone and Mancò, 2012)
Energy efficiency index	Measured using the specific energy consumption of a process using a reference value based on the best available technology (BAT), a benchmark value of the product in question, or a specified reference period divided by the actual specific energy consumption of the process or system.	% (on kWh/t)	(Siitonen et al., 2010)
R1 formula	Measured based on the ratio of the energy content of output (e.g. output of electricity and heat) to the energy content (caloric value) of the material input that is used for energy recovery.	% (on J)	(CIWM, 2017)
Resource conservation efficiency (RCE)	Measured based on the energy savings of different waste management options per material input in each option, divided by the energy savings achieved by the best management practice.	% (on MJ/t)	(Kaufman et al., 2010)
Upstream material efficiency	Sum of material output of full upstream production process divided by the sum of material input in the full production process for a specific MCP. *NOTE: this metric can also be based on the difference in the density of the product made with less materials over the density of the same product that was made with more material providing the same physical and functional characteristics.*	% wt.	(Tabone et al., 2010)
Downstream material efficiency	Sum of material output of recycling process divided by the sum of material input in the recycling process.	% wt.	(Bartl, 2015; Graedel and Allenby, 2003)
Recycled material fraction	Sum of recycled material input divided by the raw material input for a specific MCP.	% wt.	(Krajnc and Glavic, 2003; Azapagic and Perdan, 2000)
Weight recovery (for product recovered)	Measured based on the difference between the mass of the product and the sum of material waste as output from recovery process x, divided by the mass of the product.	% wt.	(Mathieux et al., 2008)
Weight recovery (for product recycled)	Measured based on the difference between the mass of the product and the sum of material waste generated and quantities of materials diverted into energy recovery from recovery process x, divided by the mass of the product.	% wt.	(Mathieux et al., 2008)

output per energy input. In these studies, it is reported that thermodynamic or thermal efficiency indicators, physical-thermodynamic indicators, economic-thermodynamic indicators (sometimes also referred to as energy intensity index, i.e. energy input in thermodynamic terms and output enumerated in monetary terms), and economic indicators (i.e. where both the energy input and unit/service output are enumerated in monetary terms) could be used instead to measure gross energy efficiency of a process or system (Bunse et al., 2011; Giacone and Mancò, 2012; Patterson, 1996).

Thermodynamic or thermal efficiency indicators measure efficiency in terms of the heat content of the inputs and outputs of a process. They make no distinction between high- (e.g. electricity) and low-quality (e.g. solar energy) energy sources, and as such, these efficiency metrics are often limited in their use because they do not allow for the comparison of *energy efficiency* across processes that have different energy inputs and outputs (Patterson, 1996). The quality aspect is a fundamental problem across all energy efficiency indicators, especially when trying to compare processes with different quality inputs and outputs. Despite the quality aspect limitation, the physical-thermodynamic indicators are considered to be good general measures of *energy efficiency*. Physical-thermodynamic indicators measure the energy input in thermodynamic units, but the output is measured in physical units in terms of mass (e.g. tonnes of butter, bricks, aluminium) or volume (e.g. litres of milk, cubic metres of timber) (Patterson, 1996). However, due to the variety of types of products produced in the same industry, allocating one energy input to several outputs in an industry (e.g. steel and slag in steel production) can make the measurement of *energy efficiency* a challenging and uncertain process.

The use of an *energy efficiency index* has been presented in the study of Siitonen et al. (2010). This metric is expressed as the ratio between the optimum specific energy consumption of a process based on the best available technology (BAT) (e.g. EfW with advanced flue gas treatment, landfills with methane collection, etc.) (Table 3) (Kaufman et al., 2010). The benchmark value of the unit output or a specific period of time and the *specific energy consumption* (see def. in Table 2) of the process, were developed to monitor and evaluate the progress in achieving energy efficiency (Siitonen et al., 2010). A yet another energy related efficiency metric of great importance in the waste management sector the so called '*R1 formula*' (Table 3). This formula is used to assess the efficiency of EfW recovery processes. For being accepted as 'high efficiency' the *R1 formula* outcome must be equal to or higher than 65% for recently constructed plants (European Commission, 2008; European Union, 2008). This formula is only dedicated to the treatment of MSW in EfW plants and its use must be based on the functional incineration unit. Use of this formula enables the classification of EfW plants treating MSW to be considered as recovery rather than disposal operations (CIWM, 2017).

A metric that focuses specifically on evaluating the energy balance of RRfW systems has been developed by Kaufman et al. (2010), in order to assess and compare different waste management options based on their energy savings. This metric, called the *resource conservation efficiency (RCE)* metric, uses the *CED* and energy savings from the recycling of different materials and applies them to identify which management option gives the higher energy savings (see definition of *CED* in Table 2) (Kaufman et al., 2010). The development of this metric is based on the fact that any material being considered for resource recovery can result to an amount of energy savings under current resource recovery technologies, such as the energy recovered from material landfilling (i.e. landfill gas); from energy recovery options (e.g. from EfW); and from the energy saved by recycling, in order to indicate which process gives the maximum (energy) savings (Kaufman et al., 2010; Klinghoffer and Castaldi, 2014). For example, energy savings from recycling is the difference between the energy required to manufacture a new product minus all of the energy required to transport and reprocess the product in the recycling phase (Klinghoffer and Castaldi, 2014). The use of technical metrics (Section 6) can provide clarity to the use of *RCE*, indicating the strong linkages between different domains of value.

In some studies, the substitution effect in the energy use, has been accounted using the *RCE* metric (Bernstad and la Cour Jansen, 2012; Perkoulidis et al., 2010). In the study of Kaufman et al. (2010), it was suggested that if the difference in energy savings for a recyclable material is greater than the energy that would be attained from the combustion of that material with energy recovery, or the recovery of methane from its landfilling, then recycling is considered the best option. For non-recyclable materials or those materials that have energy recovery (either from EfW or landfilling) values higher than their energy saving value, then the best option is considered to be the one that offers the highest energy recovery value (Kaufman et al., 2010; Klinghoffer and Castaldi, 2014). The major drawback of this metric is that it indicates the best management option from a pure energy conservation perspective, but not from a wider environmental or even sustainability perspective (Table 3).

In regards to the *material efficiency* metric we distinguished this into *upstream* and *downstream material efficiency*. *Upstream material efficiency* provides an insight into how efficient the design processes can be when it comes to raw material use at the upstream part of the RRfW (i.e. production of MCPs) (Tabone et al., 2010). *Downstream material efficiency,* also known as *material recyclability* (Azapagic and Perdan, 2000), denotes the amount of material recovered by the recycling facilities for reprocessing into new materials (i.e. at the downstream part of the RRfW system) (Bartl, 2015; Graedel and Allenby, 2003). This is one of the most important metrics for measuring the efficiency of waste management systems, but it generates variable results, based upon the definition of the system, i.e. what counts as input to be recycled and what is actually recycled. Whereas recycling of many materials is certainly preferable, recycling rate alone is not suitable as a measure of the overall waste management quality, efficiency or even sustainability.

There are several reasons for this, but a couple stand out as being the most significant. First, it does not account for the differences between landfilling and EfW for non-recycled wastes as the potential alternative EoL management options. Second and perhaps more importantly, it omits materials that are not recyclable by the contemporary technologies (e.g. many plastics, contaminated wood, novel composites), and therefore not even collected for recycling, and does not accurately account for the inevitable 'rejects' during processing and reprocessing for recycling. In other words, by default, the maximum possible recycling rate, based on all possible definitions of 'recycling', is well below 100%. Intrinsically, *downstream material efficiency* should be interpreted with care, as it only allows to get an estimate of the potential for recycling, and not of the actual amount of material that will be recycled. Especially, for multiple-material products, of which dismantling and subsequent material separation may present numerous difficulties, this metric must be used in combination with technical metrics (Section 6) in order to avoid any major misestimating, and/or double counting.

An additional metric that could be included to show the actual proportion of the recycled materials used as input in MCP production, is the *recycled material fraction*, presented in the study of Krajnc and Glavic (2003). The *recycled material fraction* metric, also known as *recycled content*, attempts to provide information on the intensity of secondary (recycled) material used in the production of new MCPs, by using the ratio of recycled material over raw material content (Table 3). The *recycled material fraction* of new MCPs has been reported in many studies (Allacker et al., 2014; Ardente and Mathieux, 2014; Ardente et al., 2013). In the study of Allacker et al. (2014), it was reported that there are different allocation approaches related to *recycled material fractions* including the market-based approached suggested by Ekvall (2000), the eco-cost and value-ratio (EVR) model suggested by Vogtländer et al. (2001) and the material-quality-based approach suggested by Kim et al. (1997), for which more information can be found in the respective studies. What must be highlighted herein, is that the *recycled material fraction* of a MCP is independent of its original use, as this might differ widely, especially when materials and components fall into the cascade recycling system (more on cascading on Section 6) (Allacker et al., 2014; Kim et al., 1997); therefore, this metric does not provide any insights into the so called 'closed' vs. 'open' loop or 'upcycling' vs. 'downcycling' debate.

Focusing on the EoL of components and products and their recoverability potential, Mathieux et al. (2008) have developed a method to enable designers to produce better recoverable products. This method, using the results of multi-criteria and multi-scenario recoverability assessments, has generated the *weight recovery indicator* which differentiates between the weight of product recovered and the weight of product that is actually recycled in proportions as shown in Table 3 (Mathieux et al., 2001).

3.6. Integrated metrics

Integrated metrics, often in the form of composite metrics, combine multiple aspects of system performance into a single measurement, based on a common scientific or economic principle, with the aim to address a broad number of aspects (Atlee and Kirchain, 2006; Ingwersen et al., 2014). Over the last two decades, a large number of integrated metrics have made their appearance as a way to measure sustainability in a single measurement.

A specific aspect that must be emphasised in assessing the sustainability of a process, and of RRfW systems in general, is the geographic and socio-demographic environment in which a manufacturing or a waste management plant is located. Thus, a pulp and paper plant isolated in the Canadian forest, making extensive use of forest products and discharging waste water into a large river, may turn out to be sustainable because ecosystems are only locally affected and within their ecological limits, and as such long-term equilibria remain intact. On the contrary, the same plant, using the same resources and with the same polluting activities but located in a heavily populated area, will not be as sustainable; it would deplete the limited local natural resources (forest, water) posing negative impacts on the nearby communities and ecosystems.

A metric that can account for these spatial characteristics is critical. The concept of *environmental space* has evolved to account for the environmental pressures that a given space can handle without incurring any damages to the existing ecosystems, and affecting the biota they support. In essence *environmental space* refers to the space available for the water, energy, land, materials (renewable and non-renewable) that can be exploited, including stocks and sinks (i.e. capacity to absorb waste and pollution) (Callens and Tyteca, 1999; Hille, 1997). The per capita *environmental space* available in EU has been calculated and reported in an EEA report (1997) as a way to promote reduction in the use of primary resources (Hille, 1997). But controversy over its potential to properly account for the resources available per capita, and the differing needs according to spatial, economic and socio-political aspects, has undermined its use to promote sustainability.

A metric developed in the same lines as the *environmental space* is the *ecological footprint (EF)*. *EF* aims to measure how much of the biosphere's annual regenerative capacity is required to renew the natural resource demand of a defined population in a given year (see Table S3 in *Supplementary Material*) (Monfreda et al., 2004; Venetoulis and Talberth, 2008). It intends to cover all relevant components of a population's resource consumption and waste production (Monfreda et al., 2004; Wackernagel and Rees, 1998), however in its current version it only captures renewable material resources (e.g. food, fuel, fiber) and carbon emissions. This is done by identifying all the individual items, and amounts thereof, and then assessing their *EF* using lifecycle data (Monfreda et al., 2004).

The *EF* - although used widely as an integrated metric for assessing environmental sustainability (Barrett and Scott, 2001; Wiedmann and Barrett, 2010) - is also used as a tool for benchmarking environmental performance and monitoring progress towards sustainability (Monfreda et al., 2004; Venetoulis and Talberth, 2008). However, there is still controversy when it comes to its usefulness in decision- and policy-making (Giampietro and Saltelli, 2014; Wiedmann and Barrett, 2010), despite its ability to be used as a stand-alone tool for communicating over-consumption and its wider issues. Nevertheless, research around its potential use is ongoing with recent investigations suggesting that the *EF* does not provide a meaningful modelling of sustainability (Giampietro and Saltelli, 2014).

Other integrated metrics that are directly related to MCPs production, distribution, use, EoU and EoL management, include the:

- *Environmentally weighted material consumption (EMC);*
- *Environmental impact recoverability indicator (EIRI);*
- *Cleaner treatment index;*
- *Material input per service unit (MIPS);*
- *Material recovery indicator (MRI);*
- *Energy recovery indicator (ERI);*
- *MSW management self-sufficiency indicator* (ws_x);
- *Net recovery index; and*
- *Transport intensity index.*

A description of these metrics can be found in *Supplementary Material*. The last five metrics have so far only been used to assess waste management performance at a regional or country level, and have largely to do with demographics, collection and management practices used for specific waste streams, e.g. MSW, construction and demolition waste (CDW), or SRF. It must be emphasised that the use of integrated metrics must come with some caution by taking the specific strengths and limitations of this type of metrics into account. For instance, the significant differences in the resource requirements of a variety of components and products, and the way these are produced, distributed, used, disposed and recovered at their EoU and EoL stage needs to be kept in mind. This often makes integrated metrics less versatile and useful in making comparisons across products, industries or processes. To deal with this the *product environmental footprint (PEF)* and the *organisation environmental footprint (OEF)* have been established by the European Commission (European Commission, 2013), to measure the environmental performance of products, services, and organisations (e.g. from extraction of raw materials, through production and use, to final waste management) based on a LCA-based multi-criteria measure (Finkbeiner, 2014; Lehmann et al., 2015; Manfredi et al., 2012). However, due to "*applicability of the data quality assessment scheme, the suitability of the provided allocation approach for recycling, and weighting or the identification of appropriate measures to communicate PEF/OEF results [..]*" the use of *PEF/OEF* is currently unclear (Lehmann et al., 2015). In addition, benchmarks and performance classes that enable comparisons between products are not yet clearly defined practically, nor theoretically, and as such the use of *PEF/OEF* has not been fully established (Lehmann et al., 2015).

Nonetheless, the process of detecting all factors of a composite metric and analysing their respective demands has heuristic value, judging from the hundreds of projects replicating this approach worldwide (Schwarz et al., 2002). Essentially, one could see integrated metrics already as a formalised evaluation method, especially when these metrics are used to rank RRfW options in terms of environmental impact or overall net benefit.

4. Economic metrics

The consideration of the economic performance of a RRfW system is critical not only for ensuring its financial viability, but also for fulfilling wider economic objectives at local, regional or national levels such as the provision of employment or contribution to sustainable prosperity (Jackson, 2016). Economic impacts have to be considered alongside environmental, social and technical aspects in order to ensure the overall viability of any proposed resource recovery strategy or intervention, and to justify the decisions made to support them via investments, subsidies and taxation.

Cost-benefit analysis (CBA) that collapses all costs and benefits into monetary terms, is a well-known tool for assessing positive and negative impacts of RRfW projects in monetary terms (Begum et al., 2006; da Cruz et al., 2014; Djukic et al., 2016; Finnveden et al., 2005; Varouchakis et al., 2016; Wang et al., 2016). Proponents of CBA argue that this procedure provides a transparent, clear and systematic assessment, whereas critics say that this method can create false comparability due to: methodological bias towards recognising only what can be monetised; (often) inconsistent results (Pickin, 2008); ethically questionable monetisation of human life (Ackerman and Heinzerling, 2002) and life-sustaining ecosystem services (Gómez-Baggethun and Ruiz-Pérez, 2011). Others point to some principle conceptual incompatibilities of CBA and sustainability (Anderson et al., 2015). In conclusion it seems that the core result of any CBA, the *net present value (NPV)* metric, is not very suitable for complex value assessment of RRfW systems.

CBA in the waste management literature largely falls into two categories: those which try to capture ‘externalities’ and those which do not; externalities being *“the effect(s) of production or consumption of goods and services imposes costs or benefits on others which are not reflected in the prices charged for the goods and services being provided”* (OECD, 2003). Cost-benefit studies dealing mainly with prices, which are reflected in market transactions, can be found throughout the waste management literature (Farel et al., 2013; Keeler and Renkow, 1994; Leu and H. Lin, 1998). More recent studies have endeavoured to price externalities (Dijkgraaf and Vollebergh, 2004; Yuan et al., 2011). CBA without regard to externalities can map to purely commercial or financial analysis (an analysis of cost and benefits from the perspective of one private commercial entity), whereas CBA taking fully into account externalities involves taking the perspective of society as a whole (the original intent of CBA as developed for Government), albeit subject to the criticisms of CBA listed above.

Despite ignoring externalities, even determining the financial or ‘internal’ costs and benefits of waste management options is a complex process. The financial costs and benefits of a RRfW system in a given area may include regulatory tax, investment, capital and operational costs related to waste management practices (e.g. collection, transportation, processing), and revenues created from the sale of secondary resources. For recovered MCPs to be viable from a business perspective, there must be a demand for them in the market place, and their price must be competitive with the price of the primary (raw) materials. This can be achieved via subsidies, as is common with energy generation systems, or via environmental taxation (in which case the justification requires the relevant externalities to be included in the CBA). Additional aspects related to the operational economic viability of resource recovery systems are:

- the ability to acquire feedstock at their full capacity (Choy et al., 2004; Kothari et al., 2010), as well as the availability of existing waste infrastructure to meet capacity demands (Najm et al., 2002; Ristimäki et al., 2013);
- the strategic location of recovery facilities that can render the recovery of resources economically viable (Chong et al., 2016; Ghose et al., 2006; Najm et al., 2002; Tavares et al., 2009; Wu et al., 2002); and
- the longevity of assets, land and building (Longden et al., 2007; Ristimäki et al., 2013).

Measures that can be taken to improve the economic viability of RRfW include technical innovation and optimal configuration avenues (Chong et al., 2016). These can improve the quality of secondary MCPs recovered, which is critical in building confidence in the remanufacturing/reprocessing industry and in creating the space for higher demand (Iacovidou and Velenturf, 2016); leading to expanding and upgrading process operations that can take in advantage the amount of secondary materials available. Until now, metrics capturing those aspects are rarely used or proposed in RRfW system assessments (Table 4).

The need for economic, social and environmental externalities to be included in CBA is an aspiration across infrastructure assessment; an aspiration that has been the norm in transport scheme assessment for some time, albeit the fact that difficulties of such incorporation remain formidable (Hanley and Spash, 1993; MacKie, 2010). However, a recent review found very few international examples of socially and environmentally extended CBA in the resource recovery sector that attempt to cost externalities for which there is not already a secondary market (Allesch and Brunner, 2014). The externalities that can be included in CBA of RRfW systems are varied, but fall into the following categories: direct health impacts, local and global pollutants (see Section 3), ecosystem services, and local economic impacts. Reporting each measure in monetary terms requires a related valuation method. There is substantial debate over the selection and refinement of monetisation methods across ecological and environmental economics (Gómez-Baggethun and Ruiz-Pérez, 2011). Importantly, whether the externalities associated with resource recovery projects are assigned to the category of ‘economic metric’ depends on whether or not an attempt at monetisation has been made. For instance, metrics in health, environmental quality or climate stability can be assigned to various domains of value.

In Table 4 metrics used in different valuation methods are included with example studies which have deployed these approaches in resource recovery or infrastructure related projects (Chong et al., 2016; Davis et al., 2005; den Boer et al., 2007; Ferrão et al., 2014; Hofstetter and Müller-Wenk, 2005; Quiggin, 1997; Rodrigues et al., 2016; Strobant, 2015; WBCSD, 2000; Weinstein, 2006).

5. Social metrics

Materials, components and products (MCPs) may have been largely linked with environmental and economic impacts, but in reality their impacts are far-reaching, affecting all levels of the society, not only within the physical scope of their manufacturing processes and the impacts thereof, but over the entire lifecycle of

Table 4
Key economic metrics used in RRfW assessment methods.

Metric	Description	Unit	References
Cost of raw materials and intermediates	Sum of costs of material input per unit output (MCP).	% (£ on £) or £/t	(WBCSD, 2000; Chong et al., 2016; IChemE 2001)
Net sales	Sum of recorded sales minus the sales discounts and sales returns and allowances per unit output (MCP).	£/t	(WBCSD, 2000)
Net profit/loss	Estimated based on *net sales* minus all expenses for the period including: • cost of goods sold; • general and administrative expenses; • technology expenses; • R&D costs; • amortization and adjustment of intangible assets; • restructuring and special charges; • interest expenses; • other expenses; • income tax.	£	(WBCSD, 2000; Chong et al., 2016)
Net present value (NPV)	The difference between the present value of cash inflows (and the positive monetary value of externalities) and the present value of cash outflows (and the negative monetary value of externalities).	£	(Weinstein, 2006; Strobant, 2015)
Capital cost	Sum of costs including: • planning costs - costs for planning activities, e.g. engineering design and environmental impact assessment; • investment costs – costs for providing the infrastructure or services, e.g. bins • land costs - land acquisition and site footprints; • equipment purchase - cost of equipment needed in the different waste/resource treatment facilities; • setup - site development and equipment installation costs, e.g. civil work, access roads, electrical distribution network, piping and assembly work.	£	(Chong et al., 2016)
Operational & maintenance cost	Sum of costs including: • depreciation of installations, based on expected lifespan (using manufacturing data); • running and extraordinary repairs, and inspection costs - may be calculated as percentage of equipment cost; • unit cost of working hour requirement e.g. in operation, supervision, training; • insurance costs calculated as a percentage of fixed capital cost; • transportation cost.	£	(Chong et al., 2016)
Utilities costs	*Energy costs:* sum of the cost of energy consumed per unit output (MCP). *Non-energy costs*: sum of the cost of: • domestic water and sewer service; • collection of residuals, and/or landfilling or transport elsewhere.	£/t	(Davis et al., 2005; Chong et al., 2016)
Revenue from secondary resource sale	Sum of the cash inflow made from the sale of: • electricity/heat generated from the treatment of waste; • biofuels generated from the treatment of waste; • earth conditioner and fertiliser produced in organic waste treatment facilities; • heat captured from energy from waste activities that can also be used on onsite applications (e.g. in dehumidification and heating); • secondary materials (e.g. paper and cardboard, glass, metals and plastic), components (e.g. steel and timber beams, concrete slabs) and products (e.g. electrical and electronic equipment, furniture).	£	(Chong et al., 2016)
Taxation	Sum of the tax based on • national/local tax rate (e.g. on profit); • energy sales tax (electricity and biofuel); • landfill tax; • carbon tax.	£	(Chong et al., 2016)
Subsidy and incentives	Sum of the amount received e.g. for the: • planning and operating new and existing facilities; • generating and/or using renewable energy; • carbon credit offset; • renewable energy tax credits; • other price support.	£	(Chong et al., 2016; den Boer et al., 2007)
Health costs	Measured by various methods.	£	(Hofstetter and Müller-Wenk, 2005)
Ecosystem services	Measured by various methods.	£	(Peh et al., 2013)
Economic spillover effects	Measured by various methods, including gross value added uplift calculation, IOA modelling, or net jobs additions.	£ or jobs	(Ferrao et al., 2014; Rodrigues et al., 2016)

components and products distributed into the supply chain (UNEP and SETAC, 2009). These impacts are often a result of the social interactions and relationships created in the context of resource extraction, processing, manufacturing, assembly, marketing, sale, use, recycling, and disposal, amongst others, as well as between the key stakeholder groups involved in the lifecycle of MCPs (e.g. workers, employees, consumers, local communities, waste managers, policy makers, value chain actors) (UNEP and SETAC, 2009).

This, as suggested by UNEP and SETAC (2009), indicates that social impacts of the lifecycle of MCPs are a result of three basic aspects, presented in Fig. 2.

The elements shown in Fig. 2 are also interlinked, which means that the roots of social impacts can be more complex than it seems, as often socio-economic processes can affect behaviours, and behaviours (and decisions made by individuals or groups at sectoral level) can affect socio-economic processes and alter the cultural and human capital (UNEP and SETAC, 2009). For example, "*pressure for low prices (socio-economic processes) may draw suppliers to allow illegal child labour (behaviour), a practice that may be accepted in a given society because of systemic poverty (capital)*" (UNEP and SETAC, 2009).

The study of these interrelationships is necessary in order to get some clarity on the way social impacts are created and evolved based on the geographic locations, the impact of processes carried out (mines, factories, roads, rails, harbours, shops, offices, recycling firms, disposal sites), and the shared, separate or conflicting interests of all stakeholders involved in the supply chain, and how these impact the environment and economic landscape. To that end, it is recommended to start the assessment of social impacts with the identification and analysis of stakeholders (UNEP and SETAC, 2009; Zurbrügg et al., 2014).

Based on the individual behaviour sphere (Fig. 2), it becomes clear that the acceptance and participation of citizens or professionals to processes and measures that aim at resource recovery at source, is critical in creating of positive human, social and cultural capital, but also in enabling socio-economic decision-making (Chong et al., 2016). Acceptance of a resource recovery practice by stakeholders is increasing the likelihood of its implementation and thus to changes in social-cultural considerations (Balkema et al., 2002). *Social participation* can be invigorated by education and awareness campaigns, which on turn can enable behaviour and mind-set change into realising the benefits participation can bring in the community, but it can also be supported and enforced by legislation. For example, in Japan the 'Food Recycling Law' requires food waste producers to report the amount of food waste recycled, whereas in the US the 'Good Samaritan Law' supports the redistribution of food that would otherwise be wasted into other uses, such as for charities or food banks (Chong et al., 2016; Zurbrügg et al., 2014). In both cases, acceptance can be largely driven by the social and cultural perception around the positive impact of resource recovery processes and facilities in the community. If they are perceived to promote climate change mitigation, and address local deficiencies or inefficiencies, including employment, energy and fertiliser shortages, these are highly accepted by the community (especially in rural areas where jobs and energy supply are of greater concern) further enhancing socio-economic processes and improving the human, social and cultural capital (Achillas et al., 2011; Chong et al., 2016).

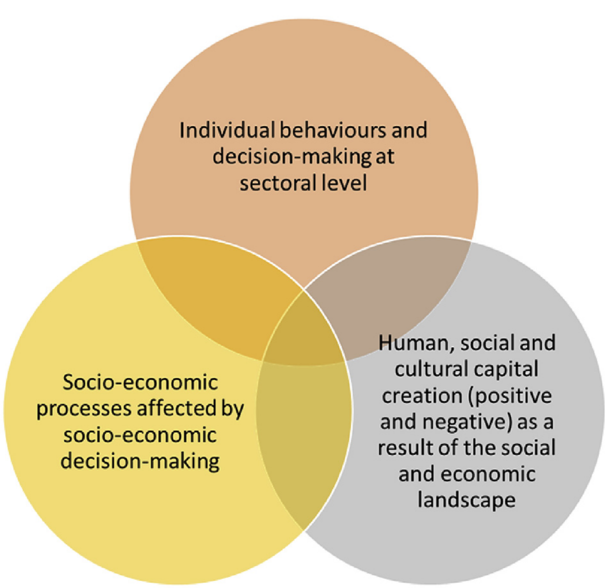

Fig. 2. Elements that affect social impacts associated with the lifecycle of MCPs. Adapted from: UNEP and SETAC (2009).

In fact, community-owned projects, for example run by community-based organisations (CBOs) or associations and cooperatives, have been stated as promoting deeper social involvement and increasing the success rate of resource recovery projects (Chong et al., 2016). In urban environments, where community cohesion may be lacking, educational and information dissemination programmes can aid the acceptance of resource recovery technologies and processes (Achillas et al., 2011). Taking reference from past studies, other types of social factors that affect the sustainability of RRfW systems is the interaction between private sector stakeholders and social capital (Zurbrügg et al., 2014). These interactions might influence the fair distribution of RRfW system benefits and impacts between citizens, affecting as such the acceptability of facilities planning and setup, and the collection and recovery systems used. In order to gain positive results, *participation rate* in the implemented schemes needs to be high, as well as participation in the community decision-making processes that govern the measures taken and imposed to the community. Collaboration, gender equity, employment, motivation, interest, and influence (power) are other important attributes that must be taken into account when assessing social impacts related to resource recovery systems (Zurbrügg et al., 2014).

The importance of socio-economic considerations becomes even more crucial in low-income countries. In these countries the RRfW systems are mainly operated by the informal recycling sector (IRS), typically by marginalised urban poor (waste pickers) (Velis et al., 2012; Wilson et al., 2015) where such activities constitute their main source of livelihood. Systems analysis tools that assess the waste and resource management at city level, such as the 'Wasteaware' benchmarking indicators (Wilson et al., 2015), and those that provide guidance on how to socially include and integrate the informal waste pickers into formalised legitimate operations ('InteRa') (Velis et al., 2012; Wilson et al., 2015), already explicitly include and try to quantify socio-economic dimensions.

The recently developed social LCA (sLCA) sets out key social phenomena relevant to the assessment of positive and negative social impacts of a MCP over its lifecycle (for example, on human rights, working conditions, and health and safety) (Hellweg and Milà i Canals, 2014; Klöpffer, 2003). Public health and safety are important factors within the society, with a close link to the economy and environment. In regards to the workforce necessary per each unit of MCP produced, collected, recovered and/or reprocessed there is a need to account for the *working hours*, the *wage*, as well as the employment conditions (job quality) (Table 5) (Gregson and Crang, 2015). The worker-hours needed in each process associated with the MCPs lifecycle is useful in understanding where data needs to be collected on-site, and where generic information might be sufficient to support the analysis of the added value offered by the workforce. However, *working hours* alone, cannot provide insightful views on the potential social impacts arising from worker-hours put into each process of the supply chain. The specific context in which the work is undertaken, as well as the country and condition of the facility where the work is carried out, might provide more input in relation to *working hourly wage* rates, unpaid labour, child labour, forced labour and informal

Table 5
Key social metrics used in RRfW assessment methods.

Criterion/Metric	Description	Unit	References
Acceptability	Acceptability of a RRfW policy, intervention or action to local/regional community and/or overall population; needs to be further researched.	Unspecified (potentially semi-quantified or qualitative)	(Chong et al., 2016; Balkema et al., 2002; UNEP and SETAC 2009)
Participation rate (in RRfW)	Level of involvement/participation in resource recovery, distribution, use, separation and collection; householders' and professionals willingness to separate waste at source.	% (on population)	(Chong et al., 2016; Balkema et al., 2002; UNEP and SETAC 2009)
Participation (in decision making)	Level of involvement of local (and regional) residents in the RRfW projects: from attending town hall meetings to community-led projects.	Various	(Chong et al., 2016)
Social function and equity	Equitable distribution of system's benefits and impacts within a community and social function provided by the RRfW system (including a range of social aspects such as time requirement, convenience, prestige, gender, vulnerable groups).	Unspecified (potentially semi-quantified or qualitative)	(Chong et al., 2016; UNEP and SETAC 2009; den Boer et al., 2007; Zurbrugg et al., 2014)
Child labour	Employment of children, esp. when in a dangerous or unsuitable environment for them in the RRfW system which depends on the child's age, the type and hours of work performed and the conditions under which it is performed.	Unspecified (potentially semi-quantified or qualitative)	(UN 2016; Wu et al., 2014)
Working hours	Measured based on the worker-hours required at each stage of the RRfW system.	hours	(UNEP and SETAC 2011; Wu et al., 2014)
Working hourly wage	Measured based on the number of working hours by taking into account, e.g. the living wage in the country, minimum wage in the country, and average wage in the sector.	% (paid wage/minimum or liveable wage)	(UNEP and SETAC 2009)
Health and safety (of workers)	*Safety:* Measured based on the number of reportable injuries by the average number of people employed per year. *Health:* Measured by the days sick leave by the average number of people employed per year (also in comparison with national/regional average across sectors).	Number of accidents at work per 100,000 workers Days sick leave per worker	(HSE 2016; Cameron et al., 2008)
System safety	As perceived by the neighbours as regards e.g. risk of accidents or malfunctions with increased release of (toxic) emissions and subsequent impacts on health.	Unspecificed (potentially semi-quantified or qualitative)	(Chong et al., 2016)
NIMBY syndrome	Opposition by residents in regards to their proximity to a RRfW facility, measured based on the fraction of citizens in support and against the siting of a new facility nearby (while in principle accepting the technology or need for RRfW).	% (on local residents)	(Kikuchi and Gerardo, 2009)
Job creation	Number of jobs created from the RRfW system.	No. of jobs or FTE	(UNEP, 2010)
Employment or job quality	Working and employment conditions.	Unspecified (potentially semi-quantified or qualitative)	(den Boer et al., 2007; Gregson and Crang 2015)
Local deficiencies	Addressing local requirements for resources (e.g. fertiliser) or energy (e.g. district heating).	Various	(Chong et al., 2016)
Noise pollution	Noise generated during each stage of the RRfW system.	Number of people exposed to unhealthy noise levels (day/night)	(den Boer et al., 2007)
Odour	Unpleasant odours caused by the degradation of materials (e.g. food waste, biomass) during resource recovery and management processes.	Number of people exposed to odours	(den Boer et al., 2007)

work conditions (UNEP and SETAC, 2009). Incidents at work that are used to reflect the *health and safety* metrics can be accounted based on the number of people in the working environment usually using 100,000 people as a reference (Table 5) (Cameron et al., 2008; HSE, 2016).

Child labour in several parts of the supply chain (e.g. in agriculture, mining, manufacturing, construction, waste picking (IRS), and processing) constitutes an important metric in RRfW systems assessment, because it can radically change the value of the recovered resources. There are many forms of *child labour* worldwide including forced labour and debt bondage (to pay off debts incurred by parents and grandparents) (UN, 2016; Wu et al., 2014). *Child labour*, which tends to be more pronounced in the informal sector can be dangerous and harmful, morally reprehensible, and can violate the child's freedom and human rights (UN, 2016; UNICEF, 2005). For resource recovery to be appropriately valued it must be decoupled from unethical processes of recovery based on *child labour.*

When public decisions are made on the location of a waste management facility the so-called *NIMBY (not in my backyard)* syndrome can be of concern (Table 5). *NIMBY* reflects the propensity of local citizens and officials to obstruct action regarding the siting of necessary but unwanted waste facilities and other locally unwanted land uses (LULUs) in their own community (Kikuchi and Gerardo, 2009). This syndrome is largely an effect of community perceptions on resource recovery and disposal facilities; specifically on the perception of the visual impact, noise, hygiene, safety, handling, transport and land space requirements of such facilities, on which most people are opposed to (den Boer et al., 2007; Kikuchi and Gerardo, 2009; Morrissey and Browne, 2004). Education and awareness raising campaigns could alter this opposition, and provide a better community cohesion when it comes to implementing strategies for resource recovery improving as such the human and cultural capital (Chong et al., 2016).

Odour and insects problems created by inappropriate resource extraction (e.g. mine tailing polluting ecosystems), disposal and collection (e.g. collection regime followed and types of collection) as well as *noise* pollution are additional metrics that can be taken into account in assessing RRfW systems (den Boer et al., 2007). The disposal method used, as well as the proximity of a treatment facility to a community, can be an important factor contributing to the augmentation of *odour* and *noise pollution* impacts (Table 5) (CECED, 2003). Hygiene aspects associated with resource recovery options are decisive to their successful implementation (Defra, 2007), as well as the annoyance caused by unpleasant odours or noise during the implementation thereof (Bottero et al., 2011).

The literature signifies the infancy of assessing the social impact of RRfW systems also by listing social and societal dynamics, phenomena or criteria to be taken into account in social impact assessment and decision support rather than proposing concrete and tested metrics and methods of measurement. Table 5 follows this state-of-the-art and presents either a proposed metric or a social criterion for the assessment. The unit of measurement is sometimes 'unspecified'. Generally, quantified metrics are desirable, but the social reality is often not as 'countable' as the environmental, technical and economic aspects, and therefore also semi-quantitative and qualitative metrics are to be considered for the assessment of social aspects.

A further hint that social aspects of RRfW are to be further researched and expanded is that the emerging concerns and metrics of subjective well-being and quality of life and their more objective drivers have not been taken up (Bache, 2015; Stiglitz et al., 2009).

6. Technical metrics

Materials, components and products (MCPs) are used because of the technical properties they possess: steel is strong; polypropylene films can be food-safe, lightweight and airtight; copper is conductive and malleable; light bulbs produce light, etc. The maintenance and retention of these properties extends the lifecycle of the components/products; whereas the degradation of these properties by aging, use, contamination, or changes in the service for which MCPs are purposed for, may determine whether they have reached their EoU or EoL status. Even so, materials contained within components/products that reached their EoL, can still independently maintain their full or considerable part of their potential value as technical engineered materials (metals, paper,

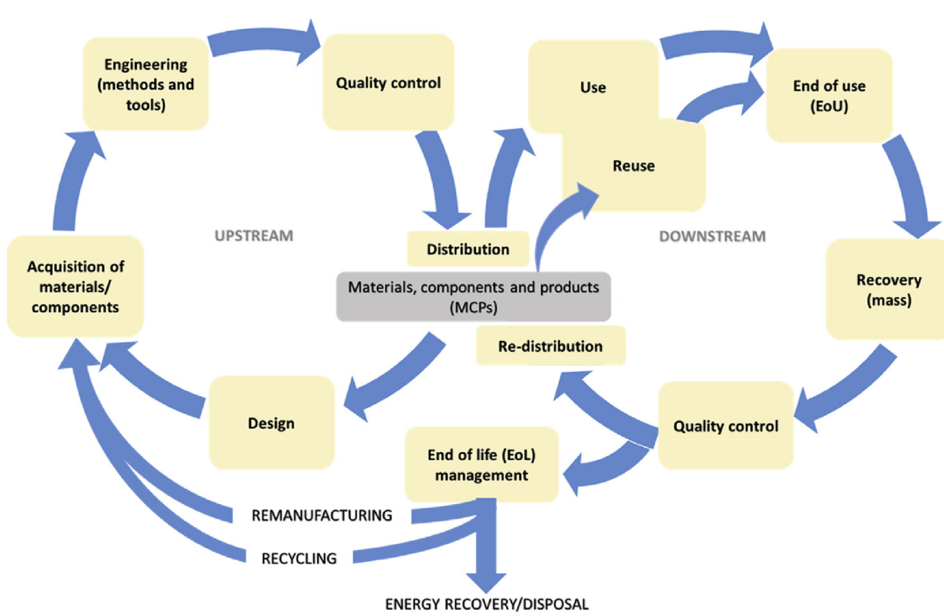

Fig. 3. The interconnection between upstream and downstream MCPs cycles in promoting their recovery after their EoU and EoL stages. Adapted from: Iacovidou et al. (2017b).

plastics). As such, technical metrics related to the MCPs lifecycle are prerequisites in conveying valuable information about how the physical and functional properties – and, as a consequence – the value of MCPs changes across the RRfW system (Tanzil and Beloff, 2006).

There is a variety of metrics associated with MCPs that range from those used in the manufacturing/engineering sectors (e.g. product yield, productivity, throughput; but also size, Young's modulus, melting point, load capacity, fracture resistance, density, toughness, modular design, corrosion resistance properties) (Wernick and Ausubel, 1995), to those used in green chemistry (e.g. atom economy, E-factor, effective mass yield) (Constable et al., 2002; Henderson et al., 2010) of which detailed description can be found elsewhere (Askeland and Phulé, 2003; Constable et al., 2002; Henderson et al., 2010). Even more interestingly, there can be multiple pathways to recover technical value starting from the same (engineered) used material. For example, paper can be re-pulped to produce new paper (if of suitable quality) and remain in the technosphere; it can incorporated to a degree in composting, returning some organic matter back to the biosphere; or it can be combusted in EfW to produce electricity and heat. In each of these resource recovery operations, a different set of metrics is relevant and applicable.

When components and products reach their EoU stage, they may still contain significant technical value that can be captured; hence, decisions made at this stage are important and may be affected by many factors (i.e. engineering, business, environmental, and societal factors) (Ziout et al., 2014). Accordingly, component/product recovery has come into increasing prominence in the industry and estimating the residual technical value of MCPs at their EoU and EoL stage is an essential prerequisite of promoting their reusability (Robotis et al., 2012), remanufacturability (Fang et al., 2015a; Hatcher et al., 2011, 2013), recyclability and recoverability potential. For clarification, EoU stage is usually followed by a quality control stage, at which reusability leads to a second or *n-th* cycle of a MCP's service life (Iacovidou et al., 2017b), whereas EoL stage means that the component/product can no longer be used and thus, remanufacturing, recycling and energy recovery are the optimal routes, as shown in Fig. 3. Component/product features and characteristics may govern the selection and feasibility of the above practices, and therefore have an effect upon the profitability of these strategies, and the environmental and social aspects associated with them.

The *reusability* or *reuse potential* refers to the ability of a component or product to retain its functionality after the end of its primary life (i.e. EoU stage) (Iacovidou and Purnell, 2016). To 'measure' the *reusability* of a component/product, static information on the physical and engineered characteristics (e.g. designed lifetime, usage period, etc.) and the way components and products are made and/or fixed together, needs to be available (Table 6). In addition to that, dynamic information that governs the transformation of their functional/physical characteristics during their use, must also be made available in order to generate the knowledge required for determining the durability of a component/product over a specific service and ability to continue providing this service after their primary EoU stage (Iacovidou et al., 2017b). These characteristics may vary depending on cultural, historical and organisational aspects (Iacovidou and Purnell, 2016), and therefore, this metric should be used with care when assessing the *reusability* of components/products in a specific RRfW systems. Moreover, the selection of technology and/or process available for the application and/or recovery of the component/product are also important in promoting and retaining their *reusability* (Park and Chertow, 2014). For example, the *reusability* for bricks can be somewhere between 50% and 95% contingent on the time allowed for dismantling, the care taken and the materials used for binding (e.g. cement based mortar vs. lime based mortar) (Leal et al., 2006; WRAP, 2008a, b).

Remanufacturability complements reuse in that it promotes the restoration of durable used products back into the manufacturing process with low investment costs (Gutowski et al., 2011; Hatcher et al., 2011; Ijomah et al., 2007; Ramoni and Zhang, 2012). This process involves the complete disassembly of a product, during which each component is cleaned, inspected for damage, sorted, reconditioned and/or reprocessed to its original equipment

Table 6
Key technical metrics used in RRfW management assessment methods.

Metric	Description	Unit	References
Reusability	Amount of MCPs that retain their functionality and physical attributes after the end of their primary life, on a weight or item basis.	t (or item)/t (or item) input into the system	(Ardente and Mathieux, 2014)
Remanufacturability	Potential to restore a component/product to like-new condition through measuring, disassembly, cleaning, inspection and sorting, part repair/refurbishment/replacement, reassembly and final testing, on a weight or item basis.	% wt. or item	(Bras and Hammond 1996a, b, Amezquita et al. 1995)
Mass recyclability	Amount of MCPs collected and/or sorted for recycling, on a weight basis.	t/t input into the system	(Ardente and Mathieux, 2014)
Technical recyclability	Proportion of the material, or component made of only one material collected for recycling that will be recycled for producing high quality recycled MCPs.	% wt.	(Ardente and Mathieux, 2014; Cerdan et al., 2009)
Mass recoverability	Amount of MCPs, and/or proportion of the material in the component/product that is captured after the EoU stage.	t or % wt.	Adapted term based on Council Directive 2005/64/EC
Energy recoverability	Energy embodied in the materials, components (made of only one material), and component's and product's parts recovered by the EfW plants in the form of electricity and heat.	kWh/t input	(Ardente and Mathieux, 2014)
Lower heating value (LHV) or net calorific value	Amount of heat released by combusting a specified quantity of MCPs (initially at 25 °C) and returning the temperature of the combustion products to 150 °C, which assumes the latent heat of vaporization of water in the reaction products is not recovered.	J/kg or kWh/t	(Allacker et al., 2014)
Technical recoverability of components and products	Relates to components (made of more than one material) and products only. Assessed based on the component's and product's weight share that can be extracted for reuse, recycling, energy recovery and disposal, using the eco-design principles.	% wt.	(Mathieux et al., 2008; Cerdan et al., 2009)
Technological advancement	*Process based:* Advances in technology that improve the efficiency of technologies used in the processing steps of the RRfW system. *MCP based:* advances in the designed characteristics of existing and/or new MCPs.	Unspecified (potentially quantified or semi-quantified)	Ijomah et al. 2007

manufacturer specifications (if feasible). It is then reassembled – often together with new parts – into a new product that can offer functionality as good as, or better than, that of a brand new product (Asif et al., 2012; Hatcher et al., 2011; Ismail et al., 2014). This series of industrial processes, i.e. disassembly, cleaning, inspection and sorting, part repair/refurbishment/replacement, reassembly and final testing represents the key elements of the *remanufacturability* metric (Table 6) (Amezquita et al., 1995; Bras and Hammond, 1996a,b); methods proposed for measuring each one of these elements can be found elsewhere (Amezquita et al., 1995; Bras and Hammond, 1996a,b). This list of elements is in accordance with those suggested by Fang et al. (2015a), which include fastener accessibility, disassembly complexity, disassemblability, and recoverability (Fang et al., 2015a). As recoverability constitutes a metric with a wider meaning herein, it is not considered further in the context of remanufacturability alone. Nonetheless, it must be emphasised that for remanufacturing to be meaningful the product lifetime, rate of technical innovation, and failure rate of components have to also be taken into account as they influence the return rate of products from their EoU to their EoL stage (Östlin et al., 2009). At the same time the efficiency and effectiveness of the remanufacturing process depends largely on decisions made during the design process (Ijomah et al., 2007); hence, indicating the importance of linking upstream and downstream cycles of the production-consumption-disposal system, as shown in Fig. 3.

It is widely speculated that remanufacturing, due to the avoided resource use, energy, and emissions associated with new component and product production, can offer greater economic and environmental savings than recycling (Gutowski et al., 2011; Hatcher et al., 2011; Lund and Mundial, 1984; Ramoni and Zhang, 2012). This is because recycling may require additional labour, energy, and machinery, resulting in additional costs, energy consumption and carbon emissions (Ramoni and Zhang, 2012). Nevertheless, recycling is the most widely known resource recovery process practiced worldwide, both in the formal and informal recycling sectors, reducing energy consumption, GHG emissions and pollution when recycled materials are substituting primary feedstock and suitable pollution control is applied. As such 'recyclability' metrics are of great importance in RRfW systems.

Hitherto, *recyclability* is also affected by the features and characteristics of MCPs, as well as on the existence of viable recycling processes and suitable technologies thereof. Factors such as, the number of different materials used in components and products (e.g. use of plastics of the same or compatible types increase recyclability), the way materials are fixed together (the degree of liberation: e.g. welding metallic parts together with plastic parts might minimise its recyclability), the use of substances, laminated materials or compounds (e.g. paint, pigments and marks) that may lead to contamination (Cerdan et al., 2009), can largely affect the *recyclability* of components and products. In that regards, attaining the right information from the upstream cycle of MCPs supply chain system (as shown in Fig. 3) can prove beneficial in addressing these issues. For this purpose various metrics have been proposed for recyclability - indicatively: i) *mass recyclability* (e.g. amount of MCPs sorted for recycling on a weight basis); and ii) *technical recyclability* (e.g. maximum capture of value from recyclates via producing high quality recycled products) (Table 6) (Ardente and Mathieux, 2014).

To elaborate, concrete's *mass recyclability* can be affected by its composition (mix design), strength, purity (e.g. based on the presence of pollutants found in paint and plaster), and form (e.g. cast-in-situ, pre-cast, or in unit materials such as blocks, tiles and stair units) (Iacovidou and Purnell, 2016). The *technical recyclability* of concrete however, can be affected by the use of paint, whereas paint is regarded as a remedial solution in rust problems associated with steel used in bridges (Horvath and Hendrickson, 1998), prolonging the lifetime of steel components without affecting as much their *technical recyclability*.

In regard to the *technical recyclability*, the allocation of secondary materials in new MCPs production represents an important step in closing the material loops (Allacker et al., 2014). This allocation is largely affected by the quality of secondary materials, as well as their mass flow. The designed features of the primary material produced, and the waste management and recycling processes followed in a given RRfW system, may greatly influence the conservation or degradation of the quality of a material. Degradation of a material's quality will determine whether the material is suitable for 'closed' or 'open-loop' recycling. In the strict, narrow definition of closed-loop recycling, the recycled materials can replace their primary counterparts for the original use within one (or *n-th,* depending on the type of material) lifecycle, given that their quality has remained high (i.e. no significant degradation has occurred). On the contrary, in open-loop recycling the recycled material cannot replace the primary material for the original use due to quality degradation during or after primary use. Notably, these terms may be used in a much wider way, with for example the mixing (via compatibilisers) of different packaging plastic polymers to a new polymer being also perceived as closed loop recycling, regardless of the fact that it is a form of 'cascading'. Where quality degradation occurs then cascading of the recycled material to a different (lower) quality product may be the optimal option for recovering its value. This is often called a cascade recycling system, and is based on the sequential use of materials for the production of new MCPs (different from the original use), of which quality continues to be degraded over successive lifecycles, until it becomes too low for the MCP to be used any further as a raw material in another lifecycle (Bartl, 2015; Kim et al., 1997). Detailed information around the cascading of materials/components can be found elsewhere (Bartl, 2015; Kim et al., 1997). Besides quality, the quantity of the recycled material is another factor that may affect its *technical recyclability*. If there is no economic value, the recyclable material is no longer used and is diverted into other waste management options. Therefore, in a cascade recycling system, the quality, the quantity, and the economic aspect of the recyclable material should be considered (Kim et al., 1997).

In all cases, 'quality' would be defined relatively, deepening on fit for purpose for the intended new use, and critically affected by the fundamental feature of waste, that of heterogeneity at different scales (spatial, material, temporal), as for example is defined in the theory of sampling (TOS) (Esbensen and Wagner, 2014a, 2014b). Regarding spatial variability, the degree of concentration of resources (or reversely dispersion) over geographies and components/products, should also be a critical factor of any theoretical and/or operational definition of *recyclability*. However, such aspects are seldom considered by the current definitions, at least directly: there is still need to resort to fundamental concepts and invent new, strictly defined metrics capturing critical aspects of what *recyclability* could mean. And the same applies to *recoverability*.

Recoverability can refer to physical (mass) recoverability, called here *mass recoverability* (e.g. the potential for recovering (capturing) components, parts of products or materials at their EoU stage; as adapted from guidance on the type-approval of motor vehicles with regard to their reusability, recyclability and recoverability based on European Directive 2005/64/EC (European Union, 2005)) (Table 6); and the recoverability that is based on energy recovery, thus named here as *energy recoverability* (e.g. the energy embodied in the component/product's parts that is recovered though EfW plants in the form of electricity and heat) (Table 6) (Ardente and Mathieux, 2014). Mathieux et al. (2001, 2008) in their studies have also defined *recoverability* as the ability of the component, product (or its components) and the constitutive materials to be captured, and which may evolve during the component/product's lifecycle, as the available technologies and materials change with time (Mathieux et al., 2001, 2008). They assessed recoverability based on the use of the *technical recovery indicator (TRI)*, which for consistency was named here as *technical recoverability of components and products*, that expresses the component and/or product weight share that is captured for reuse, remanufacture, recycling or energy recovery (Fang et al., 2015a; Mathieux et al., 2001). *Technical recoverability of components and products* can be assessed based on the 11 eco-design principles suggested in the study of Cerdan et al. (2009) which include, reusable parts, recyclable materials, reversible joints, same material joints, parts with label, tools for disassembling, time for disassembly, intelligent material, time for battery changing, laminated or compound materials and painted, stained or pigmented surfaces (Table 6) (Cerdan et al., 2009).

In general, technical metrics are highly controlled by a specific MCP and its features, whereas the availability of appropriate technologies for its recovery can be a prevailing factor of the implementation of best practice. As such, *technology advancement* can be a metric that captures both the technological advances in MCPs design (MCP based), and of the technologies used at industrial level (process based) (Table 6). The process based *technological advancement* metric refers to the capacity of existing technologies used in a facility to provide maximum efficiency when it comes to the recovery and processing of resources (Ijomah et al., 2007). For example, a less environmentally sound technology in manufacturing facilities can have a negative impact on the *remanufacturability* of its output, as well as on the environmental and logistic aspects associated with it; whereas the efficiency of screen sorting and eddy current in removing paper and cardboard, and aluminium, respectively, in combination with the efficiency of NIR technologies to sort plastics, may affect their *mass recyclability* in the material recovery facilities (MRFs). However, at the MCP level, *technology advancement* should in theory progress alongside MCPs redesign and development, so that when new MCPs become available, their reusability, remanufacturability, recyclability and recoverability potential remains plausible (Ijomah et al., 2007). For example, because of rapid software and hardware innovation, mobile phones are often so outdated by their EoU stage, that their *reusability* (if possible) could be meaningless in certain socio-economic contexts. Table 6 lists key generic technical metrics in assessing the value of recovery of resources, as reported in the literature.

7. Discussion

The study highlights that there are many frameworks, methods and tools suitable for assessing RRfW systems. Notwithstanding the potential for most of them to support decision-making processes in RRfW systems, a number of shortcomings (e.g. lack of transparency, large number of assumptions, consideration of only one process or sub-system of the supply chain system) limit their potential for adopting a whole systems approach. This is because most of these frameworks, methods and tools concentrate on a single domain of value, usually environmental or economic, neglecting to account for value (impacts or benefits) in other domains. Moreover, they often also collapse multiple dimensions of value onto a few or even one metric, losing detail and clarity. Nonetheless, a holistic valuation of the RRfW systems is necessary in capturing all benefits and impacts relevant and important to all stakeholders, as well as the trade-offs and synergies associated with MCPs. For example technical aspects referring to the physical properties of MCPs will ultimately determine their reusability and/or recyclability, while social aspects will determine the viability and acceptability of an RRfW processes; hence, must be included in any analysis.

As illustrated in Fig. 4, the concurrent consideration of dimensions of value from each of the four domains (i.e. environmental, economic, social and technical), has been largely overlooked. However, a holistic resource recovery system's evaluation for complex value optimisation requires input from all four domains of value.

Measuring concurrently dimensions of value from the four value domains as they evolve along with the flows and the transformations of materials within a system is critical, not only because of the need to grasp the complexity of RRfW systems, but also of the need to demonstrate their viability, sustainability and compatibility with upstream design, manufacture and distribution systems. A holistic description (with metrics), evaluation and assessment of RRfW systems enables sound decision-making that increases the potential for MCPs to be properly designed, produced, used, and managed when they reach their EoU and EoL status (as presented in Fig. 3, Section 6).

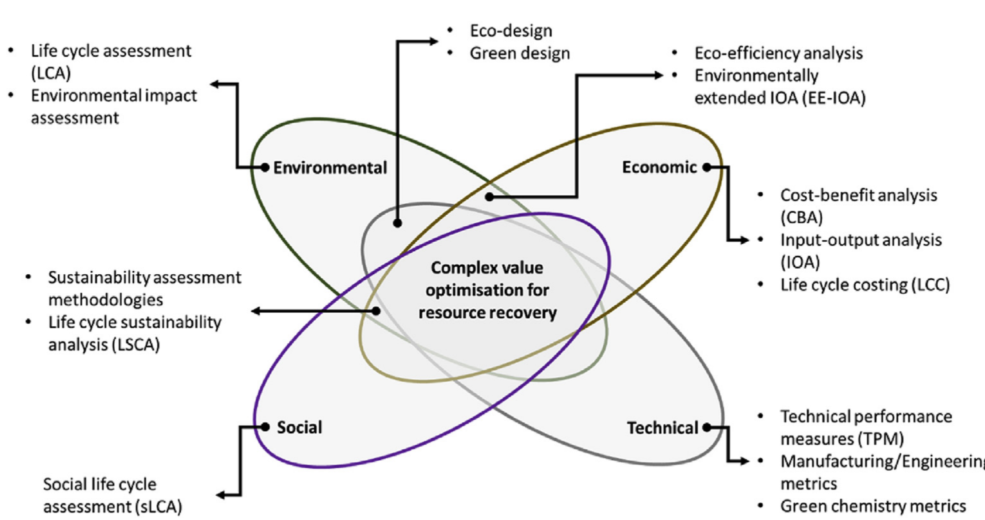

Fig. 4. Frameworks, methods and tools used for the appraisal of aspects from the domains of resource value. Methods and tools which combine different domains are linked to the points of overlap.

In this paper, we critically examined the multitude of metrics embedded within various frameworks, methods and tools and classified them into four domains: environmental, economic, technical and social. We found that:

- Environmental metrics are more extensively used than others; to the point where some dimensions of environmental value (e.g. carbon footprint) have multiple conflicting metrics associated therewith. This reflects where the development focus of decision-making frameworks, and of assessment methods and tools has been placed over recent decades. This focus is not unjustified considering that environmental metrics provide an important measure of the ability to meet sustainability principles associated with MCPs lifecycle, aligned with the planet's ecological capacity.
- Economic metrics are focussed on creation/loss of monetary value and often neglect wider economic issues. CBA is essentially a ‘partial equilibrium’ tool ill-suited to system-wide, long run (i.e. holistic) assessment. Furthermore, the need to decouple materials consumption from economic development means moving yet further away from conventional economic thinking. There is widespread appetite for improved economic assessment frameworks, not least given more general recognition of the system-wide and long-run impacts of infrastructure, including waste treatment infrastructure.
- Social metrics are the least used of the four, and yet the benefits and impacts of RRfW on our society are ultimately the most important. These metrics depend on many interlinked relationships that make their targeted selection and assessment within the RRfW system a challenging task. Nonetheless a sufficient number of social metrics exist across multiple assessment techniques that, if collated and selected correctly, could provide a basis for useful advances in tackling the problem of social value accounting. The low number and low level of operationalisation of social metrics also mirrors the usually quite low level involvement of social scientists in assessments of RRfW systems, pointing to the need to establish a truly interdisciplinary practice of such assessments.
- Technical metrics are generally specific to a given MCP or process, but a general subset of technical dimensions relevant to RRfW can be identified; but technical metrics are often insufficiently defined in the literature. Key aspects such as, variability and dispersion are not extensively considered. Such technical metrics are set to reflect and capture the functionality of MCPs; a product, component or material is technically functional if its relevant technical properties, e.g. strength, purity, plasticity and operability, have not deteriorated to the point where it no longer fulfils any valuable function. Technical value also depends on the intended pathway through which value is to be recovered, and the available know-how and skills to recover this value in the most beneficial way, invoking interaction between upstream and downstream cycles of supply chains as well as the interdependence between technical and social metrics. Gaining an understanding of the MCPs physical and technical characteristics and properties and their relevance to the function provided is an essential step towards determining the optimal way of recovering this function at their EoU and EoL stage.

Wider political aspects are not neglected herein; they are reflected in and analysed via the interconnections between the environmental, economic, social and technical domains of value. Yet these have an important role to play; given the geopolitical landscape, leadership changes and regulatory constraints the benefits and impacts of RRfW and the decisions derived from a holistic assessment and evaluation, will differ widely from one region to another.

Whereas there are metrics that are well known and developed (e.g. *direct carbon emissions*, *costs of raw materials*, *net profit*), there are others that are less developed (e.g. *reusability*, *mass recoverability*, *technical recoverability*) or less flexible for wider use by many stakeholders (e.g. *embodied carbon*, *child labour*, *NIMBY syndrome*). This raises concerns around the practicality of using metrics that some stakeholders either may not be in a position to understand, or are unable to gather the data required for the incorporation of such metrics in their evaluation process. Where the associated data are (made) available, metrics that are simple, transparent, and easy to understand and measure will be most useful for evaluating the environmental, economic, social and technological dimensions of resource value, and ideally the foundations of sound decision and policymaking in the area of resource management (Krajnc and Glavic, 2003; Singh et al., 2012; Tanzil and Beloff, 2006).

It must be highlighted that the metrics listed in this study are only those that have been regularly observed and used in waste management and resource recovery systems' assessment studies. It is likely that some metrics have been overlooked, and others are yet to be developed. Not all metrics are relevant to all stakeholders involved in the RRfW system; selection of metrics must be specific to the system under investigation (i.e. the type of material, component or product, and their pathways to value recovery), and the issues that the assessment aims to address. Our future research work aims to improve the lists of metrics and provide guidance for the selection of metrics for evaluating a RRfW system under a specific context. Development of this guidance will be based on multi-criteria decision analysis (MCDA) and the ‘systems of provision’ approach (Fine and Leopold, 1993), which can explicitly and straightforwardly bring together the various domains of value (Brooks, 2015; van Kempen, 2003). Each system of provision involves a particular valuable economic good or service, integrally connected to specific agents and groups (e.g. private business, consumers, workers, state actors) in processes of production, exchange, distribution and consumption, usually for profit (Brown and Robertson, 2014).

Our approach will first aim to establish, for each resource recovery system, what are the relevant actors, groups, processes, and structures involved in the provision of the relevant goods and services. This will facilitate the identification of the multiple values flowing through the system, and the various interests (goals) of different groups and agents within the system. Such an approach nurtures a realistic *narrative framework* within which to contextualise the RRfW system under study aiding the modelling process and the interpretation of results. For example, the identification of appropriate system boundaries and metrics will be greatly aided by this framework, in a way that is not possible were the system is conceptualised in terms of the fiction of *homo economicus* found in standard CBA approaches.

8. Conclusions

The transition to a circular economy requires materials, components and products (MCPs) to be retained in the economy for longer. This presumption requires the assessment and evaluation of resource recovery from waste (RRfW) systems, upstream and downstream of the point where wastes are generated, in order to enable sound policy and decision-making processes. To make a significant step towards achieving this, a holistic evaluation of RRfW systems based on MCPs multi-dimensional ‘complex value’, i.e., the holistic sum of their environmental, economic, social and technical benefits and impacts and how these are distributed across the system over time, is increasingly required. This evaluation would provide insights into the sustainability of retaining the functionality and value of MCPs, and how to enable this to become realised. This study reveals that existing frameworks, methods and tools currently used for assessing waste management and resource recovery systems do not adequately account for the complex value of MCPs across supply chain systems; this is because of the lack of accounting for all domains of value simultaneously. It highlights that there is a diverse range of useful and informative metrics extant in the literature that could be combined for assessing the complex value of MCPs, but the need for them to be simple, transparent and easy to measure must be an important precondition in selecting the suite of metrics that are best able to assess the RRfW system under investigation. The gap between the need for multi-dimensional evaluation of RRfW systems and the lack of coherence in the pool of metrics that currently exists in this field (as well as those that have perhaps been overlooked) needs to be urgently addressed. Future developments should focus on closing this gap to provide the foundations for a new approach that can address complex value and properly evaluate the trade-offs, synergies and problem shifting that interventions in resource recovery processes or systems intended to promote a circular economy may cause.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2017.07.100.

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