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**AVALIAÇÃO DA SUSTENTABILIDADE DO CICLO DE VIDA DE
RESÍDUOS SÓLIDOS NO BRASIL**

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**AVALIAÇÃO DA SUSTENTABILIDADE DO CICLO DE VIDA DE RESÍDUOS
SÓLIDOS NO BRASIL**

Tese apresentada ao Instituto de Ciência e Tecnologia, Universidade Estadual Paulista (Unesp), Campus de São José dos Campos, como parte dos requisitos para obtenção do título de LIVRE-DOCENTE, nas disciplinas de “GESTÃO E GERENCIAMENTO DE RESÍDUOS SÓLIDOS”, “MATERIAIS E RECICLAGEM” e “TRATAMENTO DE RESÍDUOS SÓLIDOS”, junto ao Departamento de ENGENHARIA AMBIENTAL.

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DEDICATÓRIA

Dedico esta Tese, e a conquista a ela associada, à alma e Luz de meu Pai,
Rui Pereira de Souza.

" A universidade deve ser um lugar onde as máscaras caem, onde a verdade é buscada, não importa quão dolorosa possa ser". F. Nietzsche

RESUMO

SOUZA, R.G. Avaliação da Sustentabilidade do Ciclo de Vida de Resíduos Sólidos no Brasil. São José dos Campos (SP): Universidade Estadual Paulista (Unesp), Instituto de Ciência e Tecnologia; 2023.

Resíduos Sólidos Urbanos (RSU) e Resíduos de Equipamentos Eletroeletrônicos (REEE) são resíduos sólidos amplamente gerados no mundo, provocando significativos impactos ambientais, sociais e econômicos. Os impactos de sustentabilidade associados aos RSU e REEE não se restringem às etapas após o fim da vida útil dos respectivos bens; em uma visão de ciclo de vida, os impactos compreendem também aqueles provocados pelas etapas da cadeia produtiva, logística direta e reversa dos bens que lhes deram origem – ou daqueles que podem ser beneficiados pela sua reciclagem. Avaliar os impactos ao longo do ciclo de vida de produtos e resíduos é essencial para identificar os pontos críticos que requerem maior controle, e assim avaliar soluções alternativas, dando suporte à tomada de decisões. A Avaliação do Ciclo de Vida (ACV) é atualmente a metodologia mais robusta para avaliação de impactos ambientais, pois mapeia os processos do ciclo de vida delimitados em um sistema de produto, seus respectivos fluxos de entrada e saída de materiais, substâncias, e energia, e associa a esses fluxos diversas categorias de impactos ambientais. A robustez da ACV levou à necessidade de adaptá-la para também contemplar impactos sociais e econômicos, e de forma integrada, avaliar a sustentabilidade de produtos e sistemas, dando origem à Avaliação da Sustentabilidade do Ciclo de Vida (LCSA). No desenvolvimento e aplicação da LCSA, tem-se dado ênfase à avaliação de sistemas de resíduos sólidos, pois apresentam problemas complexos de sustentabilidade, especialmente em países em desenvolvimento, como o Brasil. O histórico de publicações do candidato demonstra uma atuação científica consistente, desenvolvendo estudos relacionados a este assunto, com temas que contemplam: a modelagem de ACV (ambiental) para avaliar sistemas de resíduos sólidos no Brasil; a importância da ACV Consequencial (CLCA), para identificar e avaliar impactos em processos marginais afetados por mudanças no gerenciamento de resíduos sólidos; adoção de Métodos de Estruturação de Problemas (PSM) combinada com revisão sistemática da literatura e consulta a *stakeholders*, para determinar categorias de impacto social e econômico do ciclo de vida; aplicação da ACV social (SLCA) e da CLCA a problemas de resíduos sólidos, apresentando o estado-da-arte e diretrizes metodológicas para futuros estudos; e geração de dados primários e regionalizados, que são um dos principais gargalos para estudos de ACV e LCSA. Compreende-se que o próximo ciclo de estudos deste candidato deve não somente aprofundar-se na modelagem de CLCA e SLCA para resíduos, mas contemplar temas associados como: LCSA consequencial, *ecodesign*, função das cadeias de resíduos para a economia circular e Objetivos de Desenvolvimento Sustentável (ODS), e análises químicas para geração de dados primários relevantes.

Palavras-chave: Resíduos Sólidos. Avaliação do Ciclo de Vida. Gestão do Ciclo de Vida. Avaliação da Sustentabilidade do Ciclo de Vida.

ABSTRACT

SOUZA, R.G. Solid Waste Life Cycle Sustainability Assessment in Brazil. São José dos Campos (SP): São Paulo State University (Unesp), Institute of Science and Technology; 2023.

Municipal Solid Waste (MSW) and Waste Electrical and Electronic Equipment (WEEE) are waste streams widely produced worldwide, provoking significant environmental, social and economic impacts. The sustainability impacts related to MSW and WEEE are not restricted to the end-of-life stages of respective goods; in a life cycle perspective, the impacts also comprehend those provoked by production stages, direct and reverse logistics of the original goods – or of those that can be affected by their recycling. Assessing life cycle impacts of products and wastes is essential to identifying hotspots that require more control, and thus assess alternative solutions, supporting decision-making. Life Cycle Assessment (LCA) is currently the most robust methodology for environmental impacts assessment, as it maps the life cycle processes within a product system, their respective input-output flows of materials, substances and energy, and associates to these flows several environmental impact categories. LCA robustness led to the need of adapting it to also comprehend social and economic impacts, and integrated, to assess the sustainability of products and systems, giving rise to Life Cycle Sustainability Assessment (LCSA). In developing and applying LCSA, there is much emphasis to assessing waste management systems, as they present complex sustainability issues, especially in developing countries like Brazil. The publications history of the candidate shows a consistent scientific performance, developing studies related to this subject, in themes that involve: LCA modelling to assess waste management in Brazil; the relevance of Consequential LCA (CLCA) to identify and evaluate impacts to marginal processes affected by changes in waste management; adoption of Problem Structuring Methods (PSM) combined with systematic literature review and stakeholders consultation, to determine social and economic life cycle impact categories; application of Social LCA (SLCA) and CLCA to waste management problems, presenting the state-of-art and methodological guidelines for further studies; and generation of primary and regionalised data, one of the key bottlenecks for LCA and LCSA studies. The candidate understands that his next cycle of studies must not only delve into CLCA and SLCA modelling for waste management, but comprehend related themes like: Consequential LCSA, ecodesign, role of waste streams to circular economy and Sustainable Development Goals (SDG), and chemical analyses to produce relevant primary data.

Keywords: Solid Waste. Life Cycle Assessment. Life Cycle Management. Life Cycle Sustainability Assessment.

LISTA DE ABREVIATURAS E SIGLAS

ACV	Avaliação do Ciclo de Vida
BSC	Balanced Scorecard
CCV	Custeio do Ciclo de Vida
CLCA	Consequential Life Cycle Assessment
CPP	Composição Probabilística de Preferências
E-waste	Electronic Waste
LCA	Life Cycle Assessment
LCD	Liquid Crystal Display
LCSA	Life Cycle Sustainability Assessment
MCDA	Multicriteria Decision Aid/Analysis
MSW	Municipal Solid Waste
PNRS	Política Nacional de Resíduos Sólidos
PSM	Problem Structuring Methods
REEE	Resíduos de Equipamentos Eletroeletrônicos
RSU	Resíduos Sólidos Urbanos
SLCA	Social Life Cycle Assessment
TCC	Trabalho de Conclusão de Curso
WEEE	Waste Electrical and Electronic Equipment

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1 INTRODUÇÃO

Resíduos Sólidos Urbanos (RSU) e Resíduos de Equipamentos Eletroeletrônicos (REEE) são tipos de resíduos sólidos amplamente gerados pela população mundial, o que provoca significativos impactos ambientais, sociais e econômicos. Ao fim da vida útil, quando os bens de consumo se tornam resíduos, seguem-se etapas de coleta e destinação final que podem provocar impactos diversos, como: contaminação da água, ar e solo, emissão de gases de efeito estufa, acidentes de trabalho, desperdício de materiais com valor econômico, dentre muitos outros (Beula & Sureshkumar, 2021; Gómez-Sanabria et al, 2022).

Os impactos de sustentabilidade associados aos RSU e REEE não se restringem apenas às etapas após o fim da vida útil dos respectivos bens; em uma perspectiva de ciclo de vida, os impactos destes resíduos compreendem também aqueles provocados pelas etapas da cadeia produtiva, logística direta e reversa dos bens que lhes deram origem – ou daqueles que podem ser afetados pela sua reciclagem. Por exemplo, os impactos ambientais de um resíduo de bateria de íons de lítio não se limitam aos provocados pela sua destinação final, em aterros, lixões, incineradores ou mesmo sua reciclagem. A produção deste produto para consumo também gera intensos impactos ambientais, desde a extração das matérias-primas, gerando depleção de recursos naturais, emissão de materiais particulados e efluentes tóxicos; a produção industrial das baterias, que também gera emissões e resíduos; o transporte de insumos, subcomponentes e produtos finalizados, muito associado à emissão de material particulado e gases de efeito estufa; e até mesmo a fase de uso, que pode gerar acidentes e vazamentos (Herrerias-Martínez et al., 2021; Mrozik et al., 2021). Assim, quando os resíduos não são reciclados, os insumos necessários para a produção de novos produtos devem necessariamente ser supridos por fontes primárias; inversamente, quando há o crescimento da reciclagem e recuperação de recursos a partir de resíduos, há menor dependência de insumos primários (virgens), evitando-se ao menos parcialmente os impactos associados à sua obtenção (Ducoli et al., 2022; Fahimi et al., 2022).

Avaliar os impactos ao longo do ciclo de vida de produtos e resíduos é

essencial para identificar os pontos críticos que requerem maior controle ambiental, e desta forma buscar e comparar soluções alternativas, dando suporte à tomada de decisões e políticas públicas. A Avaliação do Ciclo de Vida (ACV; LCA em inglês) é atualmente a metodologia mais robusta para esta avaliação de impactos ambientais, pois mapeia todos os processos do ciclo de vida delimitados em um sistema de produto, seus respectivos fluxos de entrada e saída de materiais, substâncias, insumos e energia, e associa a esses fluxos diversas categorias de impactos ambientais, como: aquecimento global, acidificação, eutrofização, depleção de recursos naturais, toxicidade humana e ecotoxicidade, dentre outras. Ao quantificar, classificar e caracterizar os impactos associados a cada um dos fluxos de entrada e saída de cada processo (ex. emissões atmosféricas, recursos naturais, energia), a ACV calcula um indicador para cada categoria de impacto, e permite assim mensurar os impactos totais de um sistema, e identificar os processos e fluxos críticos para cada impacto (ABNT, 2009). Em uma abordagem mais abrangente, a ACV Consequencial (CLCA em inglês), é possível avaliar os impactos de uma decisão tomada em um sistema de produto (ex. nova tecnologia de reciclagem) propagados ao longo de toda a cadeia de sistemas interligados a este, até a fronteira de processos marginais – aqueles que são afetados em no mínimo 1% de impacto pela decisão. Desta forma, a CLCA tem o potencial de modelar e avaliar problemas complexos de sustentabilidade, como as cadeias produtivas e ambientais associadas a sistemas de gerenciamento de resíduos sólidos (Bernstad Saraiva et al, 2018).

A robustez metodológica e extensiva aplicação da ACV para avaliar impactos ambientais ao longo do ciclo de vida levou à necessidade de adaptá-la para também contemplar impactos sociais e econômicos, e de forma integrada, poder avaliar a sustentabilidade de produtos e sistemas. Nos últimos anos, especialistas têm se esforçado em desenvolver bases metodológicas para a Avaliação Social do Ciclo de Vida (S-LCA em inglês), a Avaliação Econômica do Ciclo de Vida (que tem sido geralmente resumida ao Custeio do Ciclo de Vida – CCV), e a integração das três dimensões na Avaliação da Sustentabilidade do Ciclo de Vida (LCSA em inglês) (UNEP/SETAC, 2011, Souza et al, 2016).

No âmbito do desenvolvimento e aplicação da LCSA e seus componentes metodológicos, tem-se dado ênfase na avaliação de sistemas de resíduos sólidos.

Estes sistemas apresentam problemas complexos de sustentabilidade, especialmente em países em desenvolvimento, como o Brasil. Diante de cenários de expressiva disposição final inadequada em lixões e aterros controlados, intensa atividade do setor informal na coleta e processamento de resíduos, amplo mercado formal e informal de reciclagem, complexa cadeia logística nacional e internacional, e carência de dados sobre grande parte das etapas do ciclo de vida, o desenvolvimento de abordagens de LCSA que sejam capazes de avaliar a sustentabilidade destes sistemas torna-se essencial (Sala et al, 2013; Souza et al, 2016; Berardocco et al, 2022). Visando expressar a complexidade destes sistemas e seus aspectos de sustentabilidade, tem-se tornado necessário desenvolver uma abordagem Consequencial da LCSA (ex. Voss et al, 2022). Ainda, devido à necessidade de uma robusta avaliação e integração entre os aspectos sociais, ambientais e econômicos, muitos dos quais não podem ser avaliados de forma quantitativa ou não possuem dados disponíveis (especialmente em países em desenvolvimento), o desenvolvimento da LCSA vem progressivamente integrando métodos de apoio à tomada de decisão, como métodos de estruturação de problemas sociais complexos (PSM) e métodos de análise multicritério (MCDA) (e.g. Souza et al, 2015; Souza et al, 2016; Ren et al, 2018; Angelo & Marujo, 2019; Oliveira & Souza, 2023).

A importância da LCSA para sistemas de resíduos também abrange a crescente necessidade de desenvolver e selecionar produtos mais sustentáveis, de expandir a reciclagem e reduzir a dependência de matérias primas virgens. Em um país como o Brasil, é essencial compreender os impactos de sustentabilidade das práticas comumente adotadas, como por exemplo, a disposição final irregular de resíduos eletroeletrônicos, seu processamento inadequado por atores informais (ex. lixiviação ácida para obtenção de ouro), e num pensamento de ciclo de vida, a própria obtenção de matérias primas (haja vista a tragédia ambiental e humanitária associada à extração de ouro na Amazônia, e os problemas socioambientais das minas de Cobalto no Congo e de Lítio na América Latina), que poderia ser evitada com a ampla reciclagem (Souza et al, 2016; Amnesty International, 2017; Schüller et al, 2018; Harper et al, 2019).

Dentre os principais desafios para a realização de estudos robustos de LCSA aplicados ao gerenciamento de resíduos, particularmente em países em

desenvolvimento, está a necessidade de coleta e geração de dados primários e regionalizados – aqueles que expressam com maior fidelidade o contexto regional sendo estudado. Os dados coletados de fontes primárias e regionais permitem uma percepção mais transparente da realidade, reduzindo incertezas e facilitando a tomada de decisão (Ripa et al, 2017). A forma de obtenção destes dados envolve a consulta a partes interessadas e respectiva documentação (indústrias, empresas de gerenciamento de resíduos, prefeituras, órgãos ambientais etc.), questionários para avaliações qualitativas, amostragem de resíduos, emissões e efluentes, e análises em escala de laboratório (e.g. Souza et al, 2016; Jiang et al, 2020; Erakca et al, 2022).

Esta Tese contempla 12 artigos. O texto foi preparado de acordo com o Artigo 6o da Resolução UNESP número 27 de 14 de abril de 2009, que prevê a defesa de uma tese ou texto que sistematize criticamente parte da obra do candidato, como cumprimento dos requisitos para obtenção do título de Livre-docente.

2 PROPOSIÇÃO

A proposta desta Tese é sistematizar e discutir estudos realizados sobre a gestão da sustentabilidade do ciclo de vida de resíduos sólidos no Brasil, com foco em Resíduos Sólidos Urbanos (RSU), e Resíduos de Equipamentos Eletroeletrônicos (REEE). Estes estudos compreendem diferentes aspectos do ciclo de vida destes resíduos, desde as estimativas de geração, Avaliação do Ciclo de Vida (ACV) e suas derivações – Avaliação Social do Ciclo de Vida (S-LCA) e Avaliação da Sustentabilidade do Ciclo de Vida (LCSA), recuperação de recursos, economia circular, gestão estratégica e compras sustentáveis. Além de 9 estudos publicados em revistas indexadas em bases científicas internacionais, esta Tese também inclui 3 artigos que estão em processo de avaliação por periódicos indexados de alto impacto.

Os trabalhos foram ordenados em uma sequência cronológica, de forma a sintetizar a produção científica do candidato durante o Mestrado, Doutorado, Pós-Doutorado e Docência na UNESP. As publicações compreendidas nesta Tese são:

- Artigo 1: Souza, R. G., Cordeiro, J. S. Mapeamento cognitivo e Balanced Scorecard na gestão estratégica de resíduos sólidos urbanos. *Gest. Prod.* 2010; 17(3): 483–496.
- Artigo 2: Souza, R. G., Rosenhead, J., Salhofer, S. P., Valle, R. A. B., Lins, M. P. E. Definition of sustainability impact categories based on stakeholder perspectives. *J. Cleaner Prod.* 2015; 105: 41–51. doi: n°. 10.1016/j.jclepro.2014.09.051
- Artigo 3: de Souza, R. G., Clímaco, J. C. N., Sant’Anna, A. P., Rocha, T. B., do Valle, R. de A. B., Quelhas, O. L. G. Sustainability assessment and prioritisation of e-waste management options in Brazil. *Waste Manag.* 2016; 57: 46–56. doi: n°. 10.1016/j.wasman.2016.01.034
- Artigo 4: Bernstad Saraiva, A., Souza, R. G., Valle, R. A. B. Comparative lifecycle assessment of alternatives for waste management in Rio de Janeiro – Investigating the influence of an attributional or consequential approach. *Waste Manag.* 2017; 68: 701–710. doi: n°. 10.1016/j.wasman.2017.07.002
- Artigo 5: Bernstad Saraiva, A., Souza, R. G., Mahler, C. F., Valle, R. A.

- B. Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences. *J. Cleaner Prod.* 2018; 202: 488–496. doi: n°. 10.1016/j.jclepro.2018.08.038
- Artigo 6: Abbondanza, M. N. M., Souza, R. G. Estimating the generation of household e-waste in municipalities using primary data from surveys: A case study of Sao Jose dos Campos, Brazil. *Waste Manag.* 2019; 85: 374–384. doi: n°. 10.1016/j.wasman.2018.12.040
 - Artigo 7: Mancini, S. D., de Medeiros, G. A., Paes, M. X., de Oliveira, B. O. S., Antunes, M. L. P., Souza, R. G., Ferraz, J. L., Bortoleto, A. P., de Oliveira, J. A. P. Circular Economy and Solid Waste Management: Challenges and Opportunities in Brazil. *Circ Econ Sustain.* 2021; 1(1): 261–282. doi: n°. 10.1007/s43615-021-00031-2
 - Artigo 8: Costa, A. M., Mancini, S. D., Paes, M. X., Ugaya, C. M. L., de Medeiros, G. A., Souza, R. G. Social evaluation of municipal solid waste management systems from a life cycle perspective: a systematic literature review. *Int J Life Cycle Assess.* 2022; 27(5): 719–739. doi: n°. 10.1007/s11367-022-02057-6
 - Artigo 9: Oliveira, D. F., Souza, R. G. Life cycle sustainability impact categories for sustainable procurement. *J Cleaner Prod.* 2023; 383. doi: n°. 10.1016/j.jclepro.2022.135448
 - Artigo 10 (em avaliação): Domingues, A., Franzoni, M., Padoan, F., Souza, R.G., Silva, J.R. Life Cycle Sustainability Assessment (LCSA) framework for Lithium-Ion batteries (LIBs) based on the Sustainable Development Goals (SDGs). Submetido ao *J Ind Ecol*, em 12/11/2022
 - Artigo 11 (em avaliação): Nascimento, G.A., Souza, R.G., Leachate analysis of Waste electrical and electronic equipment disposal in landfills and open dumps – context of a Brazilian city. Submetido ao *J Environ Pollut*, em dezembro/2022
 - Artigo 12 (em avaliação): Castro, J.P., Garcia, J.A., Souza, R.G., Pereira-Filho, E. Indium recovery from end-of-life e-waste: chemical determination and recycling viability. Submetido ao *J Waste Manag*, em 15/12/2022.

3 ARTIGOS

3.1 Artigo 1 – Mapeamento cognitivo e Balanced Scorecard na gestão estratégica de resíduos sólidos urbanos *

Souza, R. G., Cordeiro, J. S. Mapeamento cognitivo e Balanced Scorecard na gestão estratégica de resíduos sólidos urbanos. Gest. Prod. 2010; 17(3): 483–496. doi n°. 10.1590/S0104-530X2010000300004

ABSTRACT

The public sector services are inserted in a very dynamic environment forcing their administrators to focus on internal and external factors that determine the systems' reality. Particularly for the solid waste management in Brazil, this universe is defined by political pressures and limitations, influences by population's consuming patterns, and environmental inadequacies. For a sustainable development in these systems, it is necessary an alignment between the several aspects and the sustainability principles developing action strategies. Currently, Balanced Scorecard is a widely used strategic management tool, which main advantages are: the analysis, beyond the strictly financial and operational focus, of organizations' positioning under many perspectives, and the design of a strategy map, which visually communicates the hierarchy of organizations' intended goals. It is also structured. A corporate score card containing the objectives and their related measures, goals, and initiatives is also designed, both at organizational and departmental levels. This research analyses the applicability of this method for the solid waste management in a simulated trust between the cities of Araraquara and São Carlos in the state of São Paulo. This allowed working with more than one strategy, including the system's most probable management function workers. The results indicate a great applicability of the tool, and despite the few adaptations needed, the hierarchal strategy between the aspects of learning, growing, and internal processes, and those of the organization and society were kept.

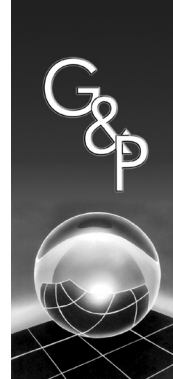
Keywords: Solid waste management; Sustainable development; Strategy; Balanced Scorecard; Cognitive mapping.

Mapeamento cognitivo e *Balanced Scorecard* na gestão estratégica de resíduos sólidos urbanos

Strategic solid waste management using Balanced Scorecard

Ricardo Gabbay de Souza¹

João Sérgio Cordeiro²



Resumo: Os serviços públicos estão situados num ambiente bastante dinâmico, exigindo de seus gestores uma boa percepção dos fatores internos e externos que determinam a realidade dos sistemas. Particularmente, na gestão de resíduos sólidos urbanos no Brasil, este universo se caracteriza por pressões e limitações políticas, influência dos padrões de produção e consumo da sociedade e inadequações ambientais. Para um desenvolvimento sustentável nestes sistemas, é necessário que os diversos aspectos se alinhem aos princípios da sustentabilidade, definindo uma estratégia de ação. Atualmente, uma ferramenta bastante utilizada para a gestão da estratégia é o *Balanced Scorecard*, cujas principais vantagens funcionais são a análise do posicionamento organizacional por múltiplas perspectivas, extrapolando o âmbito estritamente financeiro e operacional e a construção de um mapa estratégico que comunica visualmente a hierarquia das consecuições pretendidas para a organização. Este trabalho procurou analisar o comportamento deste método quando aplicado à gestão de resíduos sólidos urbanos e utilizou como objeto de estudo um Consórcio fictício entre Araraquara e São Carlos (SP). Os resultados apontam para uma boa aplicabilidade desta ferramenta, e, apesar das poucas adaptações, se manteve a hierarquia estratégica entre os aspectos de aprendizado e crescimento, processos internos, da sociedade e institucional.

Palavras-chave: Gestão de resíduos sólidos urbanos. Desenvolvimento sustentável. Estratégia. *Balanced Scorecard*. Mapeamento cognitivo.

Abstract: *The public sector services are inserted in a very dynamic environment forcing their administrators to focus on internal and external factors that determine the systems' reality. Particularly for the solid waste management in Brazil, this universe is defined by political pressures and limitations, influences by population's consuming patterns, and environmental inadequacies. For a sustainable development in these systems, it is necessary an alignment between the several aspects and the sustainability principles developing action strategies. Currently, Balanced Scorecard is a widely used strategic management tool, which main advantages are: the analysis, beyond the strictly financial and operational focus, of organizations' positioning under many perspectives, and the design of a strategy map, which visually communicates the hierarchy of organizations' intended goals. It is also structured. A corporate score card containing the objectives and their related measures, goals, and initiatives is also designed, both at organizational and departmental levels. This research analyses the applicability of this method for the solid waste management in a simulated trust between the cities of Araraquara and São Carlos in the state of São Paulo. This allowed working with more than one strategy, including the system's most probable management function workers. The results indicate a great applicability of the tool, and despite the few adaptations needed, the hierarchal strategy between the aspects of learning, growing, and internal processes, and those of the organization and society were kept*

Keywords: *Solid waste management. Sustainable development. Strategy. Balanced Scorecard. Cognitive mapping.*

1 Introdução

A gestão pública de resíduos é um sistema que, tradicionalmente, tem recebido uma baixa prioridade estratégica dos Governos nas diversas esferas – especialmente as prefeituras, as principais responsáveis

por este serviço público, de acordo com a Constituição Federal de 1988. Como consequência, estas entidades não desenvolvem uma percepção ampla da cadeia de relações causais relativa a este sistema, o que se reflete

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no cenário atualmente padrão no Brasil: inadequação ambiental da disposição final dos rejeitos, riscos à salubridade e qualidade de vida da população e catadores, com baixo direcionamento e capacitação de recursos, ineficiência na sua aplicação e pouco envolvimento direto da sociedade.

Como resposta a estes fatores, algumas experiências no Brasil apontam a tendência em transferir a gestão do saneamento ambiental para autarquias e consórcios públicos (estes últimos regidos pela Lei Federal Nº 11.107/05), tendência esta que será cada vez mais estimulada, a partir do sancionamento da Política Nacional de Resíduos Sólidos (previsto para 2010, após 21 anos de tramitação do Projeto de Lei). Estas formas de entidades possuem autonomia administrativa e orçamentária, pois constituem entes jurídicos distintos. Os consórcios possuem ainda a vantagem da integração e compartilhamento de experiências, visões e recursos, por se tratarem de uma associação entre governos com criação de uma empresa sustentada por legislação municipal específica. Por outro lado, aumenta sobre as gerências destas formas de entidades as pressões externas, o que exige uma maior capacidade de sistematizar as relações e ações diversas, por parte dos gestores.

O presente trabalho tem como objetivo analisar na gestão de resíduos sólidos urbanos o processo de estruturação da estratégia baseado no *Balanced Scorecard*, bastante utilizado atualmente principalmente no âmbito das empresas privadas, sendo subsidiado pelo mapeamento cognitivo e os conceitos gerais sobre sustentabilidade. Para tanto, considerou-se um consórcio intermunicipal recentemente estabelecido entre os municípios de Araraquara e São Carlos (SP), com o intuito de verificar o comportamento do processo de concepção de um modelo estratégico de sustentabilidade em uma empresa pública, cuja alta gerência é composta por mais de uma pessoa.

2 Breve cenário da gestão de resíduos sólidos no Brasil

A gestão ou gerenciamento de resíduos sólidos compreende a priori todos os processos de administrar a operacionalização das atividades de coleta, tratamento e disposição final de resíduos. No entanto, como em qualquer sistema produtivo, existem outras funções além da operacional que fornecem o suporte e definem o posicionamento deste sistema em seu ambiente.

Como afirma Slack (2002), toda organização possui três funções centrais e outras de apoio (Desenvolvimento, Produção e *Marketing*), o que não quer dizer que haja necessariamente unidades específicas para cada uma, pois, nas diferentes naturezas das organizações, estes papéis podem ser desempenhados até mesmo por um único indivíduo. As designações destas funções também variam, pois *Marketing* não é um termo adequado aos serviços públicos, embora sua essência seja a de manter o veículo de comunicação e garantir a satisfação dos clientes.

Desta forma, a alta gerência da gestão de resíduos sólidos está envolvida com o sistema produtivo e igualmente com as áreas de contato e suporte à população, relações com os fornecedores e planejamento e desenvolvimento. As fronteiras do serviço são delimitadas pelas conexões totais de que dispõe com seu ambiente (Quadro 1).

A Associação Nacional dos Sistemas Municipais de Saneamento fez um levantamento de experiências brasileiras nestes serviços públicos (ASSOCIAÇÃO NACIONAL DOS SERVIÇOS MUNICIPAIS DE SANEAMENTO, 2006), as quais foram analisadas em função de seus alinhamentos aos princípios do desenvolvimento sustentável. De acordo com as experiências que mais se destacaram, é possível esboçar uma provável estrutura organizacional a partir de padrões apresentados:

- Centro de Educação Socioambiental: parceria com universidades, democratização do

Quadro 1. Processos e conexões externas da função operacional da gestão de resíduos.

Processos	Atividades	Conexões
Acondicionamento	Instalação de condicionadores, separação, acondicionamento, degradação	População, trabalhadores informais, intempéries, animais e processos naturais
Coleta e transporte	Manuseio, compactação, cumprimento do trajeto, transferência dos resíduos	Fornecedores (equipamento, combustível, etc.), trabalhadores formais, meio ambiente, população, infraestrutura urbana
Transferência	Receptação, condução, estocagem, triagem, carregamento de veículos	Trabalhadores, mercado de recicláveis, fornecedores
Tratamento	Reciclagem, compostagem, tratamento físico, químico, biológico	Mercado de recicláveis, trabalhadores, população, meio ambiente
Disposição final	Dimensionamento e preparação, receptação, confinamento, degradação, drenagem, tratamento de efluentes	Meio ambiente, população, fornecedores, trabalhadores

Fonte: Monteiro, et al. (2001) e Tchobanoglous (1993).

- conhecimento técnico, programas de visitação e cursos e oficinas para alunos e professores;
- Desenvolvimento de Recursos humanos: assistência social, saúde e segurança do trabalho, assessoria aos trabalhadores e suas famílias; recrutamento e treinamento;
 - Equipe e Laboratório de Qualidade: aferição dos equipamentos, medição dos serviços, definição de sistema de indicadores, *benchmarking*, capacitação, Certificação ISO 9001;
 - Desenvolvimento: modernização dos sistemas, utilização de SIG, integração de políticas e programas com outras ações da administração pública;
 - Financeiro: desenvolvimento de banco de dados de preços públicos e de projetos técnicos, realização de pregão e licitações, adequação do sistema tarifário, concessão de isenções e concepção de mecanismos de compensação; busca de financiamentos;
 - Jurídico: adequação às leis ambientais e ao Plano Diretor, agilização dos trâmites na Câmara de Vereadores; defesa dos interesses da organização e da sociedade;
 - Conselho Consultivo: participação e controle da população; realização de Conferências; elaboração de diagnósticos e planejamento estratégico; e
 - Departamento de Comunicação Social e Relações Comunitárias: atendimento ao público, democratização de informações, resposta imediata a solicitações; realização de parcerias e articulação de eventos com a sociedade

(ASSOCIAÇÃO NACIONAL DOS SERVIÇOS MUNICIPAIS DE SANEAMENTO, 2006).

2.1 Alguns problemas na gestão de resíduos sólidos brasileira

Analisando no nível mais operacional e técnico, o tratamento dado no Brasil para seus resíduos sólidos ainda está muito aquém de uma situação adequada. A grande maioria dos materiais ainda é disposta inadequadamente, e as experiências de coleta seletiva ainda são muito pontuais. No entanto, existem ainda outros fatores que também são dignos de preocupação por parte da gestão destes sistemas. O Quadro 2 apresenta alguns exemplos de problemas que estão diretamente relacionados aos resíduos sólidos no país.

A Pesquisa feita pela ASSEMAE (ASSOCIAÇÃO NACIONAL DOS SERVIÇOS MUNICIPAIS DE SANEAMENTO, 2006) avaliou uma grande amostragem das experiências de saneamento no Brasil para definir aquelas que estavam alinhadas aos princípios da sustentabilidade. O Quadro 3 apresenta alguns exemplos deste alinhamento, que apontam para uma inovação na percepção estratégica dos gestores com relação aos padrões atuais. Estas decisões podem ter sido deliberadas em um Plano ou estar implícitas na visão de seus responsáveis, mas todas elas indicam uma busca de melhor compreensão do ambiente externo e otimização das relações estabelecidas com este.

3 Aspectos estratégicos do desenvolvimento sustentável

Como se observa no Quadro 3, as ações que (proposital ou implicitamente) se encaminham para a sustentabilidade são guiadas por princípios, ou seja,

Quadro 2. Alguns problemas relacionados aos resíduos sólidos no Brasil.

Aspectos técnicos e operacionais	Adequação das áreas de disposição final: 42% das cidades Existência de coleta seletiva: 7% das cidades Custos da coleta seletiva: – Santos: R\$ 587/t – Recife: R\$ 82/t
Baixa disponibilidade de recursos	Gastos: ≤ 5% do orçamento municipal Gastos com pessoal específico: ≤ 5% do orçamento municipal 54% dos municípios não cobram taxa de lixo Terceirização = 1,24 empresas/município
Padrões de produção e consumo	Geração de resíduos domiciliares: 228.413 t/dia (1,3 kg/hab.dia) Aquisição de veículos: 6% do rendimento mensal Demanda/oferta de computadores no País: 96%
Dinamismo do mercado de recicláveis	Reciclagem de Alumínio = 87%; Latas de aço = 49,5%; Papel = 43,9% do total Coleta seletiva formal: Alumínio = 1%; metais = 9% do total Cooperativas na coleta seletiva: 43% dos municípios com coleta seletiva Catadores na área de disposição final: 28% dos municípios
Participação da população na gestão	Existência de conselhos ambientais: 22,3% dos municípios Grau de identificação da população com as entidades: 35%

Fonte: Instituto Brasileiro de Geografia e Estatística (2000, 2002, 2003, 2004a, 2004b) e Compromisso Empresarial para a Reciclagem (2008).

conceitos fundamentais que descrevem de forma ampla o comportamento ideal desejado. As definições de desenvolvimento sustentável são bastante polêmicas até hoje, mas pode-se afirmar que são constituídas fundamentalmente por dois momentos: um de percepção e monitoramento da realidade, e outro de direcionamento das ações. Para o primeiro, dada a amplitude do conceito em questão, costuma-se fazer a análise por dimensões e para o segundo, princípios genéricos ou específicos para as situações ou sistemas em foco.

A ONU (ORGANIZAÇÃO DAS NAÇÕES UNIDAS, 2001) atualmente sugere quatro grandes dimensões para a análise e definição de indicadores de sustentabilidade: social, ambiental, econômica e

institucional. É conveniente descompô-las em temas em subtemas. Por exemplo, para a dimensão social, os temas: equidade, saúde, educação, etc; para a ambiental: atmosfera, águas, biodiversidade, etc. A dimensão ambiental está relacionada ao uso dos recursos naturais e à degradação do meio ambiente, bem como à sua preservação e conservação em benefício das gerações atuais e futuras. A social compreende os fatores para satisfação das necessidades humanas, a justiça social e a melhoria da qualidade de vida. Na esfera econômica, inclui-se não somente o desempenho macroeconômico e financeiro, como também os impactos quando do consumo de recursos naturais e energia primária, e a reprodução econômica sustentável a longo prazo

Quadro 3. Exemplos de ações positivas no saneamento ambiental brasileiro.

Princípio	Objetivos	Decisões
Universalidade	Ampliar os serviços institucionais Conceder isenções de taxa Universalizar a coleta seletiva	Modernização Tarifas mais baixas Condições para organização de catadores e suas famílias
Equidade	Conceder isenções Racionalizar despesas	Conferência Municipal de Saúde Realizar pregão
Integralidade	Racionalizar esforços Aplicar o conceito de saneamento ambiental	Criar Consórcio Assumir a gestão de resíduos e educação ambiental
Titularidade municipal	Defender solidez da empresa pública	Impedir abertura de capital
Gestão pública	Gestão responsável e integrada Desvincular contabilidade das contas da prefeitura Obter continuidade na administração dos serviços	Compartilhar decisões com população e especialistas Criar Autarquia Usar técnicos do próprio quadro
Participação e controle social	Deliberação de políticas e ações da gestão ambiental pela população Melhorar canal de comunicação com consumidor Desenvolver e disseminar conhecimento Impedir controle da empresa pública pela iniciativa privada	Conferência e Política Municipal de Saneamento Ambiental Parceria com universidades e ONGs Criar Centro de Educação Ambiental Controle social instituído por Lei Criar Departamento de Comunicação
Intersetorialidade	Democratizar saúde Ressocializar detentos Oferecer condições dignas e perspectivas aos catadores Adequar-se aos novos desafios como a ocupação desordenada Potencializar ações conjuntas	Parceria para trabalho conjunto Assistência Social, Saúde, Educação Revisar Lei de Uso e Ocupação Política de Saneamento integrada a outras políticas públicas Ações preventivas
Qualidade dos serviços	Minimizar perdas Certificação ISO 9001 Eficiência na tomada de decisões Desenvolver os recursos humanos	Combater desperdício Medição dos serviços Treinamento, Assistência e Segurança do Trabalho
Acesso	Fim do clientelismo na avaliação de usuários inadimplentes	Criar Fundo Social Maior consumo subsidia menor consumo Lei de responsabilização pelos resíduos

Fonte: Associação Nacional dos Serviços Municipais de Saneamento (2006).

(eficiência da produção versus alteração nas estruturas de consumo). A importante dimensão institucional refere-se à orientação política, capacidade institucional e esforços despendidos para a implementação efetiva do desenvolvimento sustentável.

Quanto aos princípios para norteamento das ações, existe certo consenso sobre as diretrizes fundamentais, dentro das referências científicas e documentos relevantes como a Agenda 21, marco das discussões globais sobre desenvolvimento sustentável. Milanez (2002) compilou da bibliografia um conjunto de quinze princípios genéricos para a sustentabilidade: elementar (direito à vida saudável e produtiva), geração de renda, gestão cooperativa e participativa, equidade, eficiência econômica responsável, poluidor-pagador, paz, soberania e relações internacionais (ou entre esferas do Governo), respeito às condições locais, responsabilidade intergeracional, avaliação de impactos sociais e ambientais, precautório, preventivo, uso racional dos recursos naturais e compensação dos impactos e danos.

4 Estruturação da estratégia utilizando o *Balanced Scorecard*

Mintzberg (1987) explica que a estratégia pode ser definida por cinco formas (“5 *Pês*”), mais ou menos determinantes no comportamento geral das organizações:

- Plano: um conjunto de diretrizes, com a deliberação de um curso de ação específico;
- Pretexto: uma manobra específica para superar os concorrentes;
- Padrão: uma consistência que pode ser inferida dos comportamentos dentro da empresa;
- Posição: situação da empresa em seu meio (ambiente interno versus externo); e
- Perspectiva: visão ampla que representa a cultura e a “personalidade” organizacionais (MINTZBERG, 1987).

Estes aspectos configuram-se em um modelo de decisões representado por objetivos estratégicos, aos quais são alinhadas as políticas e sequências de ação – programas – da organização (ANDREWS, 1980; QUINN, 1980). Tal é a hierarquia normalmente apresentada pelos mapas cognitivos. O mapeamento cognitivo é uma técnica para estruturação de problemas complexos, que faz uma representação visual mais próxima da realidade, ao identificar os conceitos essenciais (‘constructos’, no mapa), e as relações causais entre eles (‘setas’).

O processo de mapeamento consiste em, a partir de depoimentos originados por entrevistas ou material teórico, separar as frases em conceitos principais, conceitos estes que devem prioritariamente estar direcionados à ação (com verbos ou substantivos

que indiquem ação). Os objetivos (conquistas) são normalmente posicionados no topo do mapa, com as direções estratégicas baixo, e as opções potenciais (normalmente atividades) na base. Ou seja, de forma que as relações causais apresentem um fluxo de baixo para cima. As setas podem ser direcionadas em um ou nos dois sentidos, e podem representar relações de causalidade direta ou inversa (em que ‘mais’ de uma causa implica em ‘menos’ de uma consequência). Os benefícios em utilizar esta técnica são: estruturar, organizar e analisar as ideias sobre um problema; identificar objetivos, pontos críticos, *loops* e conflitos; esclarecimento e referência para entrevistas posteriores, e busca do consenso (ACKERMAN; EDEN; CROPPER, 1992).

O *Balanced Scorecard* (BSC) também atua na sistematização desta hierarquia estratégica, comunicando-a visualmente por meio dos *scorecards* (‘quadros de indicadores’) e o mapa estratégico, uma forma de mapa cognitivo responsável pelo ‘balanceamento’ dos objetivos segundo perspectivas (grupos de temas com a mesma natureza). Os *scorecards* são quadros, também categorizados por perspectivas, que representam, para cada objetivo estratégico, as respectivas metas, indicadores de desempenho e ações. Eles permitem esclarecer e controlar os aspectos relativos a cada objetivo estratégico definido.

O mapa estratégico é um diagrama que conecta os objetivos em uma sequência lógica e verticalmente causal, classificados por perspectivas, as quais normalmente apresentam a seguinte hierarquia causal:

Aprendizado e Crescimento → Processos Internos
Clientes → Finanças (KAPLAN; NORTON, 1996).

Embora esta configuração seja bastante comum e eficiente na representação das estratégias de empresas privadas, sempre há a possibilidade de redefinição das perspectivas, de acordo com a natureza dos objetivos estratégicos. Esta adaptação é particularmente interessante nas estratégias para o setor público, no qual a lucratividade não é o objetivo principal. Normalmente, o que ocorre é a redefinição da perspectiva ‘Clientes’ como ‘População’ ou ‘Sociedade’, e seu posicionamento no topo do mapa, acima ou ao lado da perspectiva financeira (KAPLAN; NORTON, 2004). Um exemplo de aplicação do BSC no setor público brasileiro é a Prefeitura de Porto Alegre (RS).

O processo de concepção do BSC obedece à seguinte sequência:

Visão (para o futuro) → Perspectivas → Objetivos
→ Pontos Críticos → Indicadores → Ações (OLVE; ROY; WETTER, 1999)

Inicialmente é definida qual a Missão da organização (‘por que existimos?’), seguida da Visão (‘onde queremos chegar no futuro?’). Esta visão deve nortear a definição das perspectivas estratégicas, e dos objetivos, processo este que pode ser auxiliado pela utilização

do mapeamento cognitivo. Uma vez definidos os objetivos, são identificados os pontos críticos que serão mensurados pelos indicadores, e que irão nortear as ações para conquistar os objetivos. Desta forma, a visão estratégica pode ser eficazmente comunicada para toda a organização, e são construídos *scorecards* individuais para os principais departamentos, ou seja, objetivos, metas e ações específicas que permitirão o alinhamento de cada departamento aos objetivos organizacionais.

5 Estudo de caso: concepção de um *Balanced Scorecard* em um consórcio intermunicipal para gestão de resíduos sólidos entre os municípios de Araraquara e São Carlos – SP

Para este trabalho, foram realizadas entrevistas abertas e semiestruturadas, feitas individualmente com pessoas que ocupam cargo de gerência nos sistemas de resíduos sólidos dos dois municípios em estudo. Considerou-se, como membros da alta direção do consórcio, o Superintendente do DAAE-Araraquara, Autarquia responsável pelos serviços de saneamento ambiental neste município, e o Diretor de Política Ambiental de São Carlos, vinculado à Secretaria Municipal de Desenvolvimento Sustentável, Ciência e Tecnologia, nos anos de 2006 e 2007.

As entrevistas com a alta direção respeitaram dois momentos:

- Entrevistas temáticas: foram feitas perguntas buscando extrair a visão dos gestores sobre temas associados às dimensões da sustentabilidade. Ex.: *Quais os fatores que atualmente influenciam a implantação de políticas de longo prazo na gestão de resíduos do município?* (Tema: Continuidade Administrativa, Dimensão Institucional). *Quais devem ser as condições ideais para o tratamento dos resíduos sólidos locais e regionais?* (Tema: Conservação do Meio Ambiente, Dimensão Ambiental);
- Entrevistas baseadas nos princípios da sustentabilidade: os princípios para o desenvolvimento sustentável compilados por Milanez (2002) e mencionados acima foram apresentados aos altos gestores, os quais foram inquiridos a interpretar estes conceitos sob a realidade local e especificamente a gestão de resíduos sólidos, de forma a adequá-los e reescrevê-los de forma mais específica; e
- Entrevistas para construção do *Balanced Scorecard* do Consórcio: de posse dos objetivos estratégicos obtidos, foram derivados os indicadores e possíveis ações estratégicas, além de construído o mapa estratégico.

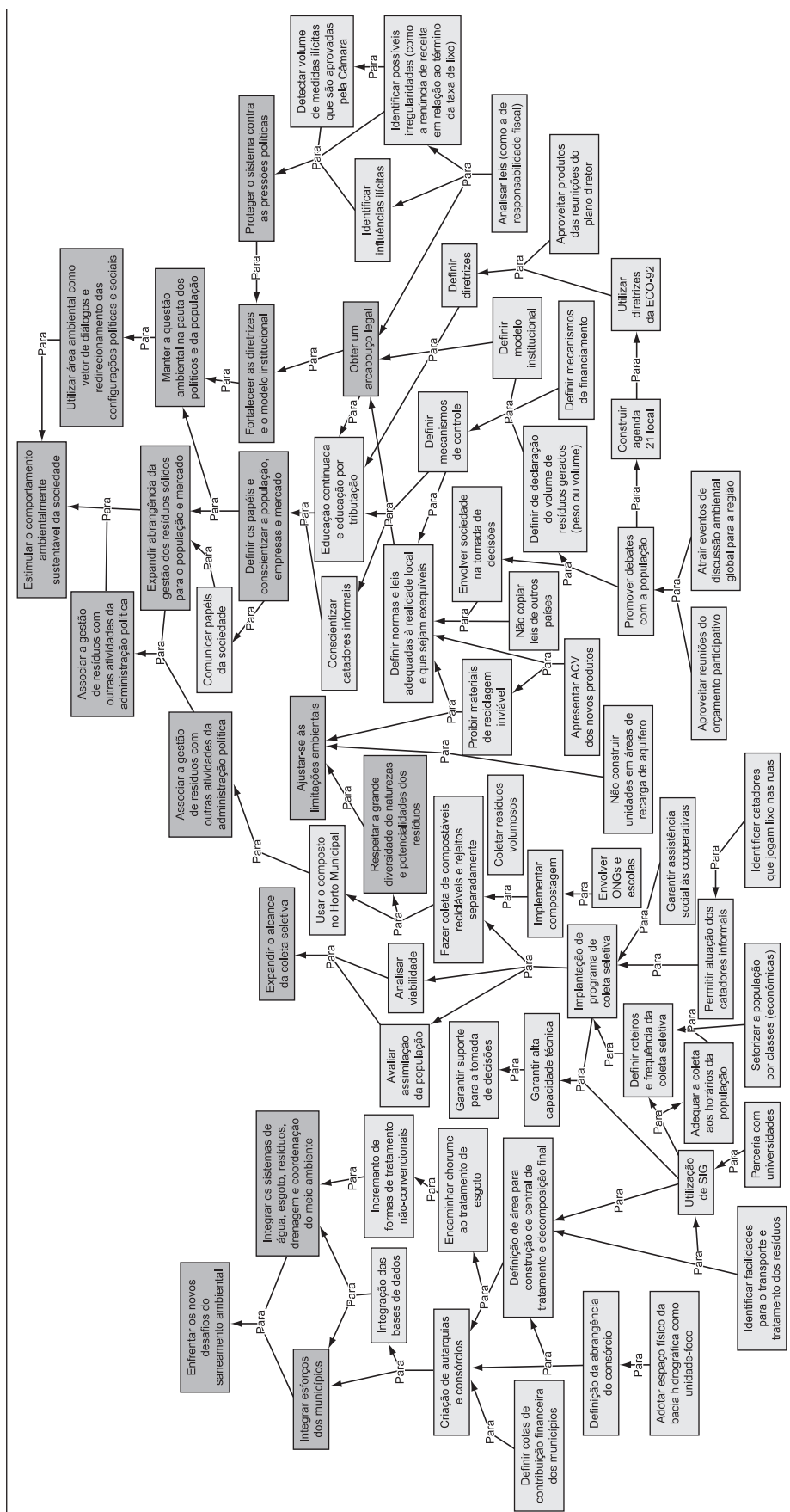
Também foram feitas entrevistas com ocupantes de cargos de chefia de departamentos nestes sistemas em estudo, tendo sido apresentados os objetivos gerais do Consórcio, para que posteriormente fossem interpretados sob as responsabilidades daquelas funções específicas, derivando objetivos restritos, bem como indicadores e possíveis ações.

Com relação às entrevistas temáticas aplicadas à alta gerência, as respostas foram gravadas, transcritas, e então foram identificados os conceitos e termos nucleares que permearam os discursos. Neste processo, utilizou-se o mapeamento cognitivo para verificação da hierarquia entre os fatores mencionados e para verificar aqueles que podem ser objetivos estratégicos e aqueles que somente se referem a ações, instrumentos ou alternativas. O mapa cognitivo final pode ser visto na Figura 2, onde os potenciais objetivos estão destacados em amarelo. Para melhor compreensão do processo de mapeamento, considere-se o exemplo abaixo, referente a uma citação do Gestor 2 (Figura 1):

[...] a diretriz tem que ser seguida, o arranjo institucional também, e daí comunicar todos os papéis. Qual o papel do Governo, o do cidadão, o da empresa, o do comércio, todo mundo tem que ter um papel definido, cada um com seu papel, você tem que mexer no arcabouço legal, normas, leis... [...] Pra população, por exemplo, tem que definir meta. Meta legal quer dizer: leis, normas que sejam exequíveis. Não adianta fazer uma lei sem pé nem cabeça que ninguém vai cumprir. Tem que fazer coisa que dê pra fazer, que seja exequível, que se tenha instrumento e meio pra fazer a lei ser cumprida. Tratar com eficiência, essas coisas. Não adianta você querer copiar uma lei que deu certo na França e querer aplicar aqui, que não vai dar certo.



Figura 1. Exemplo de hierarquia de conceitos segundo a visão dos gestores. Fonte: Souza (2007).



A Figura 2 traduz os discursos dos dois gestores entrevistados, com respeito às questões relacionadas à adequação de seus sistemas segundo os temas de sustentabilidade definidos. Percebe-se claramente que a hierarquia obtida elucida que constructos são relativos a opções potenciais, ou ações – aqueles dos quais somente saem setas, e quais são aqueles que mais se assemelham a pontos críticos ou objetivos estratégicos potenciais – aqueles que concentram grande quantidade de setas, e aqueles que somente recebem setas. Além disto, foram observadas as características dos conceitos, ou seja, se correspondem a ações diretas, ou se se assemelham a objetivos, ou seja, pontos a serem conquistados.

Quanto às entrevistas baseadas nos princípios da sustentabilidade, pela própria natureza destes conceitos – o direcionamento das ações –, apresentaram fatores mais próximos daquilo que caracteriza um objetivo estratégico potencial, como os identificados no exemplo a seguir:

- Princípio da gestão cooperativa e participativa:
 - reconhecer e respeitar as diversidades da sociedade, oferecendo uma estrutura de apoio (Gestor 2);
 - integrar os níveis de Governo para garantir a continuidade de políticas (Gestor 2); e
 - integrar horizontalmente as partes para a gestão ambiental adequada, pela conscientização, união das consciências e respeito ao meio ambiente e sua diversidade (Gestor 1 – SOUZA, 2007).

Os conceitos principais ou potenciais objetivos estratégicos obtidos por estas duas formas de entrevistas foram reunidos e, então, agrupados de acordo com a semelhança de suas naturezas (*clusters* de conceitos). Dos onze grupos que surgiram, foi feita uma síntese que resultou nos onze objetivos estratégicos do Consórcio. Por exemplo, para o Objetivo O5:

- Grupo de objetivos potenciais N°. 5:
 - integrar esforços dos municípios para enfrentar os desafios do saneamento ambiental (Gestor 1, Gestor 2, Entrevistas Temáticas);
 - integrar os sistemas de saneamento ambiental (Gestor 1, Gestor 2, Entrevistas Temáticas);
 - integrar os níveis de governo para garantir continuidade de políticas (Gestor 2 – Princípios – Gestão Cooperativa e Participativa);
 - associar G.R.S.U. a outras atividades da administração pública (Gestor 1 – Entrevistas Temáticas);

- expandir a gestão de resíduos sólidos para a população e o mercado (Gestor 2 – Entrevistas Temáticas);
- associar a gestão de resíduos às atividades diversas da sociedade (Gestor 1 – Entrevistas Temáticas);
- integrar horizontalmente as partes para a gestão ambiental adequada, pela conscientização, união das consciências e respeito ao meio ambiente e sua diversidade (Gestor 1 – Princípios – Gestão Cooperativa e Participativa); e
- manter a questão ambiental na pauta dos políticos e da população (Gestor 1 – Entrevistas Temáticas).

Objetivo O5: Integrar e associar a gestão de resíduos com os outros sistemas de saneamento, setores da administração pública, outros municípios, níveis de governo e atividades da sociedade civil, promovendo a união de esforços e de consciências (Integração dos Esforços).

Os onze objetivos finais obtidos, que estão apresentados no Quadro 4, foram submetidos a uma avaliação pelos altos gestores, verificando o seu grau de atendimento a critérios obtidos na literatura sobre as características ideais dos objetivos estratégicos corporativos – ter estabilidade; dar uma orientação clara para o futuro; justificar o comprometimento; ter alcance corporativo; referir-se a pontos críticos atuais; e traduzir a essência da vitória (HAMEL; PRAHALAD, 1989; KAPLAN; NORTON, 1997). Foram atribuídas pelos gestores notas de 0 a 10 para esta verificação de objetivo versus critério. Notou-se uma boa eficiência no resultado final deste processo, conforme se observa na Tabela 1 (as notas dadas pelos gestores estão entre parênteses, e as médias em negrito).

Nota-se que os objetivos com maior média são: O2, O5, O8, O11. Isto indica uma priorização, por parte dos gestores, para os aspectos relacionados à mobilização da sociedade e o envolvimento direto com a gestão. Portanto, isto poderia caracterizar um perfil da gestão deste Consórcio. O objetivo com a menor média (7,33) foi o O7 – *Melhor Capacitação Técnica*, o qual foi considerado ainda assim essencial pelos gestores e assim mantido na estratégia final.

Também foi verificada a maior influência dos princípios em relação aos temas na consecução dos objetivos estratégicos. Esta medição se deu pelo número de conceitos-chave (objetivos potenciais) contidos nos grupos que foram compilados, em relação a qual momento de entrevistas – temáticas ou sobre os princípios – surgiram os conceitos. Esta relação está apresentada na Figura 3, em termos de porcentagem de conceitos surgidos por cada tipo de

entrevista (temas ou princípios), em relação ao total de objetivos potenciais de cada grupo (*cluster*). O resultado somente comprova que as discussões sobre os princípios os posiciona no topo da hierarquia estratégica, como elementos da Visão para o futuro, derivando diretamente deles os objetivos estratégicos, ao passo que os temas assemelham-se às perspectivas do BSC, que são detalhadas em termos de objetivos, pontos críticos e ações (como no mapa da Figura 2).

Para a construção do mapa estratégico, foram apresentados aos gestores os objetivos finais, e foi solicitado que conectassem estes conceitos por meio de setas:

- Setas em um único sentido: relação causa-efeito direta de um objetivo em relação ao outro; e
- Setas em duplo sentido: relação de causalidade mútua entre dois objetivos.

As setas também variavam em função da forma de causalidade entre os objetivos: *setas com linha cheia*, para relações em que mais de um implica em mais do outro, e *setas com linha tracejada*, em que mais de um provoca menos do outro, ou vice-versa. Embora um tanto confuso, o mapa cognitivo auxiliar da Figura 4 permitiu uma melhor compreensão sobre como os gestores visualizavam as relações causais existentes entre os objetivos; também permitiu identificar suas características hierárquicas, ou seja, se potencialmente

são objetivos de topo ('entram' mais setas do que 'saem'), ou de base (mais setas 'saindo').

Ao final, as conexões foram quantificadas (Tabela 2) segundo a entrada ou saída em cada constructo de objetivo estratégico. Analisando os dados da Tabela 2, foi possível compreender a posição hierárquica aplicável a cada objetivo, embora este processo não tenha prescindido de certa subjetividade pelo pesquisador, o que, no entanto, é comum nos processos de mapeamento. Notou-se que a distribuição dos objetivos caracterizava as perspectivas desta estratégia, e assim foi definido o mapa estratégico final para o Consórcio (Figura 5).

Observou-se que os fatores mais fundamentais da estratégia (aqueles dos quais saem mais setas do que entram, como a *Sensibilização e Conscientização*), estavam associados à perspectiva inicial do BSC de *Aprendizado e Crescimento*, ao passo que os objetivos finais (recebem mais que originam setas, como o *Fortalecimento Institucional*), se relacionavam à Dimensão *Institucional* dos conceitos de sustentabilidade. Os aspectos intermediários puderam ser classificados dentro das perspectivas de *Processos Internos (Operacionais)* e da *Sociedade*. Estabeleceu-se, assim, uma hierarquia estratégica que respeita o modelo original do *Balanced*

Tabela 1. Avaliação dos objetivos em função dos critérios estratégicos.

		Essência da vitória	Pontos críticos atuais	Alcance corporativo	Justificar comprometimento	Orientação clara para o futuro	Estabilidade e consistência
O1	Otimizar ciclo de vida dos resíduos	(10, 10) 10	(7, 8) 7,5	(8, 8) 8	(7, 8) 7,5	(10, 10) 10	(10, 10) 10
O2	Sensibilização e conscientização	(10, 10) 10	(10, 10) 10	(10, 10) 10	(10, 10) 10	(5, 10) 7,5	(7, 10) 8,5
O3	Respeitar diversidades	(7, 10) 8,5	(10, 10) 10	(5, 10) 7,5	(8, 10) 9	(8, 10) 9	(5, 10) 7,5
O4	Eficiência da coleta seletiva	(7, 10) 8,5	(7, 10) 8,5	(10, 10) 10	(7, 10) 8,5	(7, 10) 8,5	(10, 10) 10
O5	Integração da gestão	(10, 10) 10	(8, 10) 9	(10, 10) 10	(10, 10) 10	(5, 10) 7,5	(10, 10) 10
O6	Fortalecimento institucional	(7, 10) 8,5	(10, 10) 10	(7, 10) 8,5	(7, 10) 8,5	(5, 10) 7,5	(8,10) 9
O7	Melhor capacitação técnica	(8,8) 8	(7, 8) 7,5	(9, 8) 8,5	(8, 8) 8	(6, 5) 5,5	(8, 5) 6,5
O8	Internalizar custos	(8,10) 9	(10, 10) 10	(5, 10) 7,5	(10, 10) 10	(10, 10) 10	(10, 10) 10
O9	Precaução nos projetos	(8,10) 9	(10, 10) 10	(7, 10) 8,5	(10, 10) 10	(10, 10) 10	(10, 10) 10
O10	Compensar danos	(8,10) 9	(6, 10) 8	(8,10) 9	(8,10) 9	(5, 10) 7,5	(8,10) 9
O11	Gestão das informações	(9,10) 9,5	(8,10) 9	(10, 10) 10	(10, 10) 10	(10, 10) 10	(10, 10) 10

Fonte: Souza, 2007.

Quadro 4. Objetivos estratégicos finais definidos para o Consórcio.

Objetivos	Descrição
O1: otimização do ciclo de vida dos resíduos	Reduzir, controlar e retornar ao máximo os resíduos à cadeia produtiva, respeitando as limitações da natureza e a diversidade dos materiais, pela ACV inclusive dentro da gestão de resíduos, atenção aos novos produtos e recursos naturais não renováveis e a promoção de políticas de minimização de rejeitos.
O2: sensibilização e conscientização	Promover a conscientização da população, empresas, cooperativas, e catadores informais para assumirem um papel ambientalmente correto.
O3: respeito às diversidades	Garantir que a gestão e as políticas respeitem as diversidades existentes na sociedade, favorecendo a boa qualidade de vida, inclusive dos catadores.
O4: eficiência da coleta seletiva	Fazer uma coleta seletiva eficiente, expansiva e adequada aos horários da população.
O5: integração dos esforços	Integrar e associar a gestão de resíduos com outros sistemas de saneamento, setores da administração pública, outros municípios, níveis de governo e atividades da sociedade civil, promovendo a união de esforços e consciências.
O6: fortalecimento institucional	Fortalecer a instituição e suas diretrizes contra as pressões externas, sejam políticas ou nas relações de interdependência e estratégicas, por meio da soberania e de um consistente arcabouço legal.
O7: melhor capacitação técnica	Buscar a alta capacitação técnica para elaboração e análise de projetos e suporte à tomada de decisões no planejamento, adequando-se à realidade do sistema e evitando a imitação de outras experiências.
O8: internalização dos custos	Considerar indissociáveis a eficiência econômica e ambiental, ao internalizar os custos ambientais e valorar os recursos naturais e os impactos na medida do possível.
O9: precaução nos projetos	Prever, na elaboração dos projetos, os impactos e sua reversibilidade e os custos da futura remediação e medidas mitigadoras. A escolha da área de disposição final deve buscar as áreas com maior impermeabilidade natural, embora haja tecnologia para tal proteção, mantendo os padrões de qualidade dos recursos naturais inclusive para os municípios vizinhos.
O10: compensação dos impactos	Fornecer compensação social e ambiental às vítimas e áreas de impacto de projetos, utilizando a moral e a idoneidade de forma a superar o dano causado.
O11: gestão das informações	Promover a boa gestão das informações para facilitação das avaliações e comunicação dos riscos e instruções à população, valendo-se para tal da potencialidade das universidades.

Fonte: Souza (2007).

Tabela 2. Número e tipos de conexões dos objetivos estratégicos.

O1 – Otimização do ciclo de vida	4 (Entrada = 3; Saída = 3; Duplo sentido = 2)
O2 – Sensibilização e conscientização	7 (Entrada = 4; Saída = 7; Duplo sentido = 4)
O3 – Respeito às diversidades da sociedade	5 (Entrada = 2; Saída = 5; Duplo sentido = 2)
O4 – Eficiência da coleta seletiva	5 (Entrada = 3; Saída = 3; Duplo sentido = 1)
O5 – Integração dos esforços	5 (Entrada = 5; Saída = 5; Duplo sentido = 5)
O6 – Fortalecimento institucional	6 (Entrada = 6; Saída = 3; Duplo sentido = 3)
O7 – Melhor capacitação técnica	7 (Entrada = 4; Saída = 7; Duplo sentido = 4)
O8 – Internalização dos custos	5 (Entrada = 3; Saída = 4; Duplo sentido = 2)
O9 – Precaução nos projetos	4 (Entrada = 3; Saída = 1; Duplo sentido = 0)
O10 – Compensação dos impactos	6 (Entrada = 6; Saída = 0; Duplo sentido = 0)
O11 – Gestão das informações	6 (Entrada = 5; Saída = 6; Duplo sentido = 5)

Fonte: Souza (2007).

Scorecard, e introduz com relevância os aspectos do desenvolvimento sustentável, de forma integrada.

Por meio do mapa estratégico final, verifica-se que a estratégia inicia-se por um processo de capacitação interna, gestão das informações, e mobilização da população. Isto é muito coerente, especialmente pela semelhança com a perspectiva de base tradicional do BSC – Aprendizado e Crescimento, as primeiras conquistas a serem feitas antes de promoverem-se as transformações no processo operacional. Em seguida, a dimensão operacional, concentrada numa coleta seletiva mais eficiente, na elaboração de projetos bem

fundamentada tecnicamente, e a otimização do ciclo de vida dos resíduos, o que implica no monitoramento do ciclo e busca da minimização de rejeitos. Todos estes aspectos permitem, por um lado, a integração dos esforços entre governo e sociedade e, por outro, o respeito às diversidades, este principalmente permitido pela precaução nos projetos. Havendo tais conquistas na perspectiva da Sociedade, é possível buscar o maior fortalecimento do Consórcio e a internalização dos custos, pois é adquirida confiabilidade no sistema e, com as conquistas anteriores, é possível distribuir as responsabilidades financeiras também pela sociedade. Todos estes aspectos permitem ao sistema oferecer compensação aos impactos causados, o que demanda recursos financeiros e envolvimento dos diversos atores do governo e sociedade. Este objetivo de topo, no entanto, refere-se a algo indesejável (os impactos) e que tende a ser minimizado de acordo com a evolução dos objetivos anteriores (observando a predominância de setas tracejadas que o alcançam).

Esta estratégia busca, inicialmente, construir uma maior solidez para o Consórcio (dimensão institucional no topo). Em outras palavras, por meio da busca pela sustentabilidade, visa-se conquistar a autossustentação do Consórcio, em seu primeiro momento. Possivelmente, uma estratégia posterior poderia dar maior ênfase, por exemplo, aos aspectos da sociedade e meio ambiente, uma vez que o Consórcio já se encontraria fortalecido institucional e financeiramente. Ainda assim, verifica-se que a

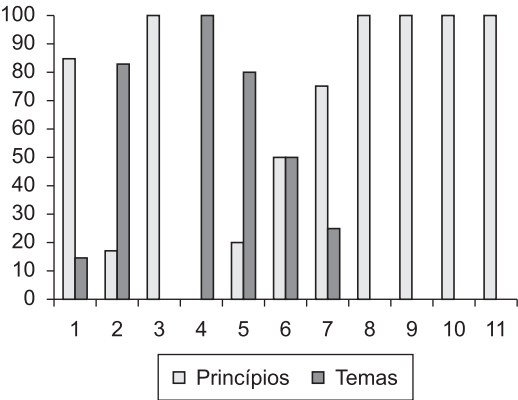


Figura 3. Influência dos princípios e dos temas nos objetivos finais. Fonte: Souza, 2007.

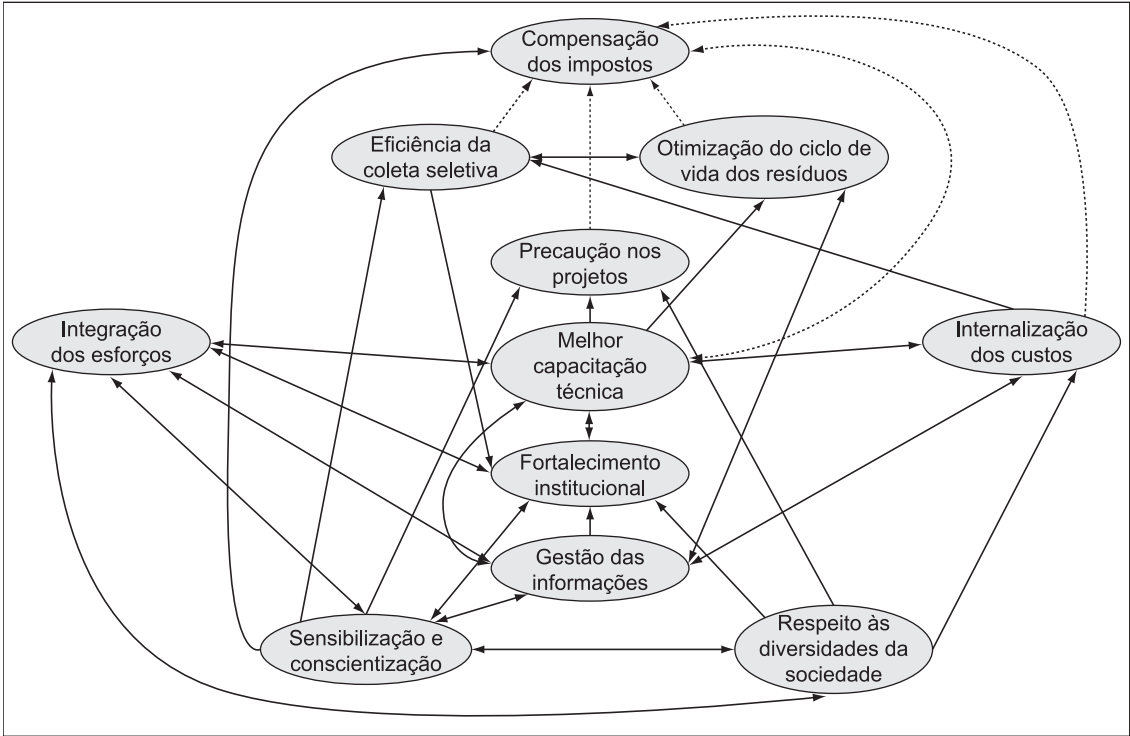


Figura 4. Mapa cognitivo inicial para os objetivos. Fonte: Souza (2007).

Quadro 5. Scorecard da perspectiva processos operacionais.

Objetivo	Indicadores	Iniciativas
O4 – Eficiência da coleta seletiva	– Quantidade de resíduos/mês coletada pela coleta seletiva/população/área (ou bairros)	– Fazer educação ambiental – Tornar obrigatória a separação dos resíduos
O9 – Precaução nos projetos	– N°. de impactos produzidos/mês, ano – Quantidade de pessoas e seres vivos afetados pelos impactos	– Elaborar projetos com alta capacidade técnica, transparência e publicidade para o conhecimento e apreciação da sociedade
O1 – Otimização do ciclo de vida dos produtos	– Quantidade e caracterização dos resíduos que chegam ao aterro – Taxa de crescimento da população/taxa de crescimento da produção de resíduos	– Programas de coleta seletiva e de compostáveis maximizados – Gerir resíduos sólidos perigosos, resíduos tecnológicos, de serviços de saúde e da construção civil

Fonte: Souza (2007).

Quadro 6. Scorecard do departamento jurídico.

Objetivos	Iniciativas
Monitorar leis reguladoras buscando reduzir os impactos dos novos produtos	Manter a atenção sobre as condições de inconstitucionalidade das leis aprovadas na Câmara permitindo a inserção de novos produtos
Buscar a inserção dos catadores num especial sistema previdenciário	– Criar Fundo de Aposentadoria Social no Consórcio
Proteção legal da instituição	– Estatuto do Consórcio de acordo com a Constituição – Estimular a formação de equipes multidisciplinares no planejamento
Estimular a aceitação favorável dos projetos pela população	Concessão de isenções de impostos e taxas em função da desvalorização do imóvel causada pelos projetos

Fonte: Souza (2007).

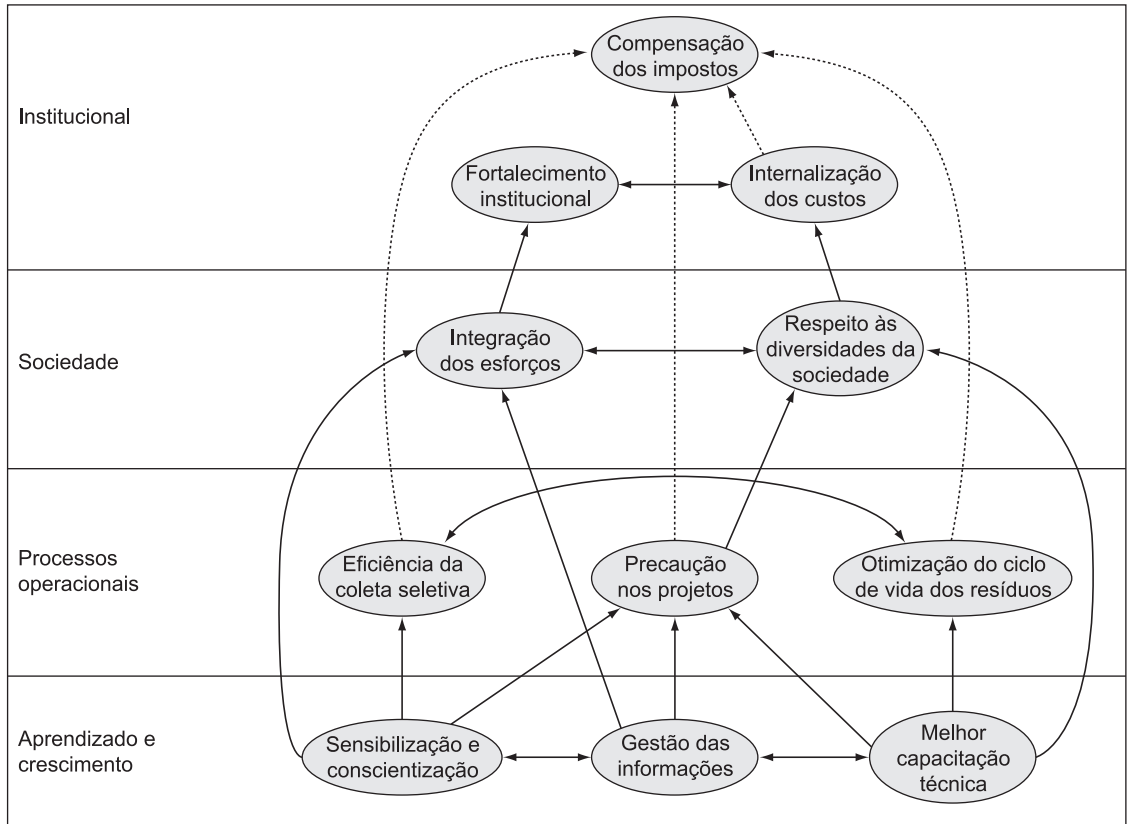


Figura 5. Mapa estratégico do Consórcio proposto para estudo. Fonte: Souza (2007).

sustentabilidade pode ser garantida pelos processos operacionais, que visam o menor impacto ambiental e social, e com maior eficiência dos recursos, e os objetivos da sociedade, que garantem o atendimento aos critérios de qualidade de vida, nos pontos em que se implica a gestão de resíduos. Fazendo um paralelo com os problemas apresentados no Quadro 2, observa-se que todos eles são combatidos pela estratégia, em todas as perspectivas, mas especialmente nos processos operacionais. Também há grande semelhança com os princípios e objetivos apresentados no Quadro 3, relativos às experiências de maior êxito no saneamento ambiental brasileiro.

O Quadro 5 apresenta o *scorecard* obtido junto à alta gerência para a Perspectiva dos *Processos Operacionais*. Este *scorecard* elucida como os gestores pretendem monitorar o desempenho dos objetivos O1, O4 e O9, e que ações são consideradas fundamentais para sua consecução. O Quadro 6 exemplifica um *scorecard* construído para o Departamento Jurídico, construído com o auxílio do Chefe da Procuradoria do DAAE-Araraquara. Neste, observa-se como tal Departamento pretende alinhar-se à estratégia do Consórcio, especialmente aos objetivos O1, O3, O6, O9 e O10. Em Souza (2007), também podem ser observados os demais *scorecards*, inclusive para os departamentos de Gerência Operacional, Recursos Humanos e Educação Ambiental.

6 Conclusões

De modo geral, considerou-se o *Balanced Scorecard* bastante aplicável e útil para os sistemas públicos, a exemplo da gestão de resíduos sólidos, analisada neste estudo. Também há boa aplicabilidade para modelos em que a alta gerência é constituída por mais de um gestor, todos com poderes de deliberação. Esta aplicabilidade é garantida, principalmente, pela utilização do mapeamento cognitivo, na estruturação do problema e busca do consenso entre as diferentes visões. Também é assegurada pela utilização dos temas e princípios da sustentabilidade como referências de topo, e do BSC como ferramenta para estruturação da estratégia. Desta forma, torna-se bastante viável e prático elucidar quais devem ser os objetivos estratégicos, a sua hierarquia, e quais as perspectivas estratégicas que os caracterizam. Também é importante observar que os princípios da sustentabilidade, por suas naturezas, já induzem um direcionamento tanto para as dimensões da sustentabilidade como para as perspectivas originais do BSC, o que permite uma visão estratégica integrada e bastante lógica e tangível, quando esclarecida em um mapa estratégico.

Os resultados principais obtidos neste estudo foram os objetivos estratégicos do Consórcio, e, conseqüentemente, o mapa estratégico e os *scorecards* gerais e dos departamentos. Demonstraram-se bastante coerentes com a problemática, e atenderam ao que se

esperava, com relação à integração da sustentabilidade e o BSC. Apesar de traduzirem visões relativas a um caso específico, podem ser referências importantes para a elaboração de estratégias com este mesmo intuito, seja para efeito de comparação, como para serem integradas a elas, desde que correspondam a aspectos relevantes das visões dos gestores. A estratégia obtida neste estudo também pode ser mais amplamente detalhada, aproveitando os objetivos atuais e integrando-os a estes novos objetivos, de forma a descrever com maior riqueza outras possibilidades de direcionamento de uma estratégia para a gestão pública de resíduos.

Por fim, considera-se que este trabalho configura-se numa referência interessante para o meio acadêmico, na elaboração de pesquisas relacionadas ao tema, e para os sistemas públicos de resíduos brasileiros. Aos primeiros, recomenda-se o aprofundamento da pesquisa sobre a metodologia adotada, e sua aplicação em outros casos, como os outros sistemas públicos de saneamento, ou outros setores da administração pública. Aos gestores dos sistemas públicos de resíduos no Brasil, aconselha-se que tomem este material como referência para a tomada de decisões, para uma compreensão mais crítica de seus próprios sistemas e como estímulo para a elaboração de suas próprias estratégias para sustentabilidade dos serviços prestados e dos processos de gestão.

Referências

- ACKERMAN, F.; EDEN, C.; CROPPER, S. **Getting started with cognitive mapping**. 7th ed. University of Warwick: Young OR Conference, p. 1992. p. 65-82.
- ANDREWS, K. R. **The concept of corporate strategy**. Homewood, IL: Richard D. Irwin, 1980.
- ASSOCIAÇÃO NACIONAL DOS SERVIÇOS MUNICIPAIS DE SANEAMENTO. **Experiências de êxito em serviços municipais de saneamento**. Brasília: ASSEMAE, 2006.
- COMPROMISSO EMPRESARIAL PARA A RECICLAGEM. **Pesquisa ciclosoft**. CEMPRE, 2006. Disponível em: <www.cempre.org.br>. Acesso em: 30 jan. 2007.
- HAMEL G.; PRAHALAD, C. K. **Objetivo estratégico**. Londres: Harvard Business Review, 1989.
- INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA. **Economia informal urbana**. Rio de Janeiro: IBGE, 2003.
- INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA. **Indicadores de desenvolvimento sustentável**. Rio de Janeiro: IBGE, 2004b.
- INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA. **Pesquisa da produção industrial**. Rio de Janeiro: IBGE, 2004a.
- INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA. **Pesquisa nacional de saneamento básico**. Rio de Janeiro: IBGE, 2000.

- INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA. **Pesquisa nacional de orçamentos familiares**. Rio de Janeiro: IBGE, 2002.
- KAPLAN, R.; NORTON, D. P. **The balanced scorecard**. Boston: Harvard Business School Press, 1996.
- KAPLAN, R; NORTON, D. P. **A estratégia em ação: balanced scorecard**. Boston: Harvard Business School Press, 1997.
- KAPLAN, R; NORTON, D. P. **Strategy maps: converting intangible assets into tangible outcomes**. Boston: Harvard Business School Press, 2004.
- MILANEZ, B. **Resíduos sólidos e sustentabilidade: princípios, indicadores e instrumentos de ação**. 2002. 207 f. Dissertação (Mestrado em Engenharia Urbana)-Universidade Federal de São Carlos, São Carlos, 2002.
- MINTZBERG, H. **Five P's for strategy**. California Management Review, 1987.
- MONTEIRO, J. H. P. et al. **Manual de gerenciamento integrado de resíduos sólidos**. Rio de Janeiro: IBAM, 2001.
- OLVE, N. G.; ROY, J.; WETTER, M. **Performance Drivers: a practical guide to using the Balanced Scorecard**. Chichester: John Wiley & Sons, 1999.
- ORGANIZAÇÃO DAS NAÇÕES UNIDAS. **Indicators of sustainable development: guidelines and methodologies**. Nova York: ONU, 2001.
- QUINN, J. B. **Strategies for change: logical incrementalism**. Londres: Richard D. Irwin, 1980.
- SLACK, N. **Administração da produção**. São Paulo: Atlas, 2002.
- SOUZA, R. G. **Gestão de resíduos sólidos: utilização do Balanced Scorecard**. 2007. 159 f. Dissertação (Mestrado em Engenharia Urbana)-Universidade Federal de São Carlos, São Carlos, 2007.
- TCHOBANOGLIOUS, G. **Integrated solid waste management: engineering principles and management issues**. Nova York: McGraw-Hill, 1993.

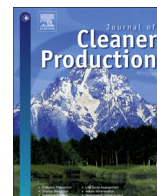
3.2 Artigo 2 – Definition of sustainability impact categories based on stakeholder perspectives *

Souza, R. G., Rosenhead, J., Salhofer, S. P., Valle, R. A. B., Lins, M. P. E. Definition of sustainability impact categories based on stakeholder perspectives. J. Cleaner Prod. 2015; 105: 41–51. doi: n°. 10.1016/j.jclepro.2014.09.051

ABSTRACT

Sustainability assessment is an important approach for decision-support in waste management systems. Life Cycle Sustainability Assessment (LCSA) is the most promising tool for this purpose, but it still lacks a formal methodological framework. One requirement for establishing an effective LCSA methodology is the necessary involvement of stakeholders in decisions on LCSA models, which still constitutes a gap in current assessments. In this paper we develop and apply a methodology for stakeholder consultation regarding the selection of LCSA impact categories (focussing, in this case, only on social and economic issues). The methodology is based on decision science concepts and tools with an emphasis on the elicitation of stakeholders' perspectives depicted in cognitive causal maps. Results from a case study illustrate a useful contribution from the methodology, revealing both well-established and innovative impact categories with very positive feedback from stakeholders. A methodological issue is the sensitiveness in defining endpoint and midpoint levels of impact categories. The method revealed straightforward and effective, with no necessity for quantitative ratings and minor influence of the analysts' perspectives.

Keywords: Life cycle sustainability assessment; Impact categories; Stakeholder perspectives; Cognitive maps; Problem structuring methods.



Definition of sustainability impact categories based on stakeholder perspectives



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1. Introduction

Sustainability assessment has developed to be an important approach to support decision-making in waste management and other decision contexts (Kaufman et al., 2010; Wagner, 2011; Menikpura et al., 2012; Aparcana and Salhofer, 2013). However, taking the perspectives of involved stakeholders into consideration is still a major gap if such applications are to be successful for real decision problems.

The most promising methodology for sustainability assessment is Life Cycle Sustainability Assessment (LCSA). It consists in evaluating and often comparing potential social, economic and

environmental impacts that can result from alternative *product systems*² that could be considered for implementation in the real world.

LCSA, which can be expressed by: $LCSA = LCA + LCC + SLCA$ (Jørgensen et al., 2013), aggregates potential environmental, economic and social impacts using three complementary methodological *modules*:

- LCA (sometimes eLCA – *environmental Life Cycle Assessment*), the environmental module whose methodological framework is well established, being standardized in ISO 14040 to 14044 and richly developed in the ILCD Handbook (EC-JRC, 2010);
- LCC (*Life Cycle Costing*), which usually focuses on costs for different actors along the chain; this is not yet standardized, but some suggested methodological guidelines do exist (Swarr et al., 2011); and

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² Each product system describes a chain of processes within a product's life cycle (for example from manufacturing to waste disposal – gate to grave). The currently operating system is usually regarded as one of the options, in order to assess impacts of the system 'as is'.

- SLCA (*Social Life Cycle Assessment*), still under constant development due to its higher levels of subjectivity but also provided with some suggested guidelines (UNEP-SETAC, 2009).

The social, economic and environmental impacts are characterized by a set of *impact categories*³ and their respective performance indicators. The selection of impact categories is a crucial step in performing a sustainability assessment of available alternatives that can enhance real world systems.

The process for selecting LCSA impact categories may differ between the methodological modules, but both SLCA and LCC approaches need to be anchored to the “root” LCA standard (Swarr et al., 2011; UNEP-SETAC, 2009). In LCA, “the selection of impact categories must be comprehensive in the sense that they cover all relevant environmental issues related to the analysed system” (EC-JRC, 2010). Therefore in LCSA this must also be true for social (SLCA) and economic (LCC) issues.

In LCC, impact categories relate strictly to direct costs experienced by one or more actors in the product life cycle (e.g. supplier, producer, user or consumer, end-of-life agent), narrowly referring just to real money flows (Swarr et al., 2011). Thus the costs associated with waste disposal fees can be attributed to consumers; with take back programmes to producers; and landfill development and closure to society. One limitation of LCC is that it only refers to costs, and therefore does not cover other economic aspects, such as: profits; competitive advantage; impacts on the product market and on the regional economy; etc.

Regarding SLCA, Finkbeiner et al. (2010) argue that the “selection of social criteria and their quantification is still one of the major challenges”, as “there is currently no uniform usage of a standardized set of indicators”. According to the same authors, “there are still research needs and consensus needs of the involved stakeholders”. The UNEP Guidelines for Social Life Cycle Assessment (UNEP-SETAC, 2009) highlights the importance of developing impact categories based on “social issues of interest to stakeholders and decision makers”.

The involvement of stakeholders is indeed one major gap in current LCSA implementations – as it is for each of its components, when applied independently. Concerning LCA, Sala et al. (2012) state that “the interested parties should be involved in order to better define the decision context and the purpose of the study, but in practice an LCA is carried out for one actor only”.

But how can LCSA applications successfully cover “all relevant issues”, if they are based on a “one actor only” perspective (usually of the client or that of the analysts themselves)? In the light of the methodological recommendations cited above, it would appear all too likely that disregarding relevant stakeholders perspectives will lead to the incompleteness or failure of LCSA models and their results.

As identified by Sala et al. (2012), there is still no structured framework where “stakeholders will not only serve as audience but as active, informed and responsible parties in the decision making process”. The objective of this paper is to suggest and demonstrate a methodological approach for the selection of LCSA impact categories based on consultation of real stakeholders, and elicitation and structuring of their perspectives.

To do so we will apply some concepts and tools from the decision sciences to the task of selecting impact categories in LCSA. The field of decision sciences has developed tools and skills to manage multiple stakeholder perspectives, including the selection of

decision criteria. It would appear likely that such tools can make a significant contribution to the development of an overall LCSA framework.

To demonstrate the applicability of such tools to the selection of LCSA impact categories, we investigate here a real decision problem, which involves sustainability assessment in waste management: the definition of a model for the Brazilian Waste Electric and Electronic Equipment (WEEE) reverse logistics system, which must be implemented in accordance with the recent National Solid Waste Policy (PNRS, 2010).

2. Background theory

2.1. Impact categories as decision criteria

Our methodology is based on a combination of theories drawn from LCA and the decision sciences. To do this we need to demonstrate that LCSA impact categories can be interpreted as *decision criteria*, by which alternative product systems can be evaluated in order to define the best solution for implementation.

In decision sciences, a *decision criterion* or *attribute* is defined as “a performance indicator employed to measure the impact of adopting each decision alternative on the organizational objective being pursued” (Franco and Montibeller, 2009). We can assume that the fundamental objectives for any product system assessed in LCSA are: to avoid negative impacts and maximize positive impacts on some well-defined sustainability (social, economic, environmental) *areas of protection* (AoP). In the case of eLCA studies, the scope needs to cover the potential impacts on the three environmental AoP: human health, natural environment and natural resources (EC-JRC, 2010). In the case of LCSA studies, the purpose, according to Jørgensen et al. (2013), is to assess the impacts of product systems on: poverty in the present generation; and to maintain the stock of capital for people living in the near and long-term future. Certainly, many other strategic objectives could also be considered, like culture and education, life quality, damage to ecosystem etc. Thus, the impact categories can be seen as the criteria by which we can measure the impact of each decision alternative on those objectives.

2.2. Definition of impact categories in current LCSA applications and in WEEE LCA studies

To the present, most LCSA studies (Bachmann, 2012; Ostermeyer et al., 2013; Martínez-Blanco et al., 2014) have adopted impact categories specified in established frameworks like the UNEP SLCA Guidelines (Norris, 2013) and an LCC core reference (Swarr et al., 2011). These frameworks organize pre-defined impact categories and subcategories related to each stakeholder group. In those LCSA applications, choice of impact categories is usually made by the analysts themselves, rather than deriving those from the perspectives of stakeholders involved in each problem situation. An interesting approach is to derive categories from a Sustainability SWOT of a product (Pesonen and Horn, 2012), drafted with stakeholders and experts.

Regarding WEEE systems, many LCA studies have been carried out by LCA and WEEE specialists worldwide (Hischier et al., 2005; Wäger et al., 2011; Bigum et al., 2012; Traverso et al., 2012; Rocchetti et al., 2013). Those studies present a standard set of LCA impact categories:

- Damage to human health; damage to ecosystem diversity; resource scarcity (*endpoints*);
- Global warming; acidification; human toxicity; eutrophication; resource depletion (*midpoints*).

³ In ISO (2006a), an impact category is defined as a “class representing environmental issues of concern to which life cycle inventory analysis results may be assigned”. For LCSA, this definition is extended to social and economic issues.

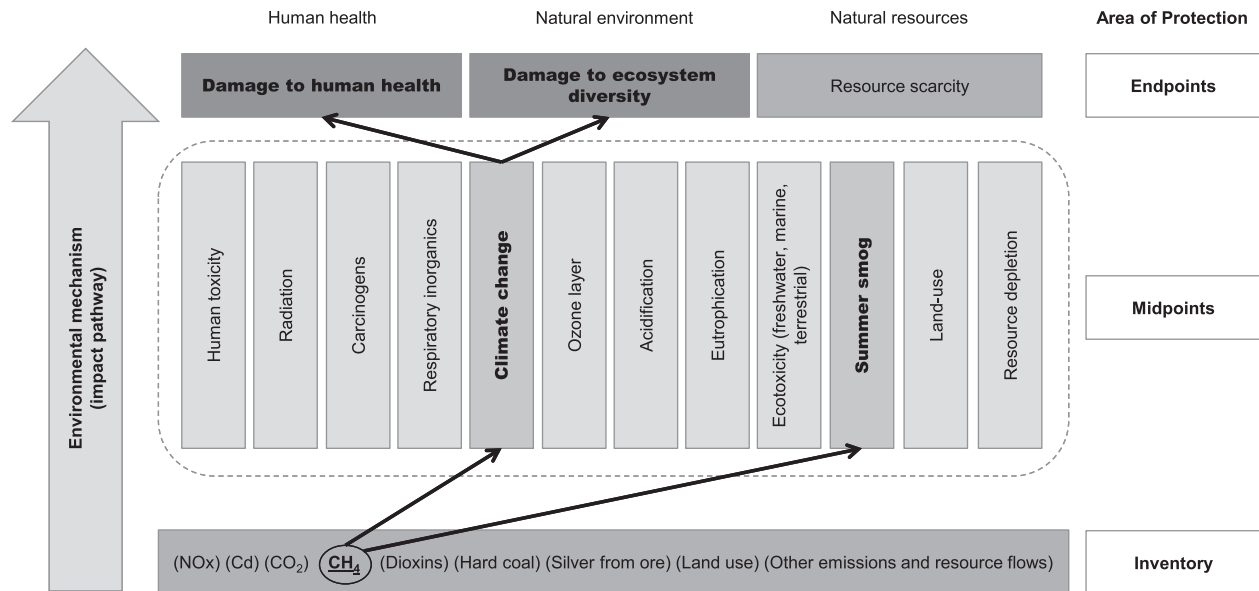


Fig. 1. Illustrative impact pathway for Environmental LCA (adapted from EC-JRC, 2010).

2.3. Cause-effect chains for impact categories

In LCA most of the environmental effects of a considerable range of resource flows and emissions are already mapped and known, as they depend essentially on physical and chemical mechanisms and parameters. This establishes a clear cause-effect hierarchy based on data from LCA inventories (compiled and quantified inputs and outputs for a system), which is called the *impact pathway* (Fig. 1). For example, it is well established that an increase in CH₄ emissions contributes to climate change, which has negative effects on both human health and the natural environment (see Fig. 1).

Areas of protection depend on the decision context, and so are defined by the scope of the study. Based on the impact pathway it is possible to select an adequate set of impact categories that are likely to affect each AoP. Impact categories can be defined by the ultimate impacts on the AoP, i.e., at the *endpoints* of the impact pathway. But they can also be defined at *midpoint* levels, intermediate consequences of inventory (system processes) emissions and resource flows that converge and aggregate to configure the endpoint impacts. Individually or combined, those midpoint impacts will provoke the effects described by the endpoints. This is why an LCA study should not combine endpoint and midpoint categories together, as they would produce redundancy in the impacts assessment. In practice midpoints are usually preferred in eLCA, as “on midpoint level a higher number of impact categories is differentiated and the results are more accurate and precise compared to the three Areas of Protection at endpoint level that are commonly used for endpoint assessments” (EC-JRC, 2010).

In LCA, an important stage fully based on analysis of the impact pathway is *characterization*. A *characterization model* reflects the environmental mechanism that connects inventory results, category indicators and sometimes category endpoints. As explained in ISO (2006b), the *characterization model* is used to derive the *characterization factors*, which are applied to convert inventory data to the common unit of the category indicator. For example, the “global warming” midpoint impact category (Fig. 1), measured by the indicator “tons of CO₂ equivalent”, is characterized by an IPCC model (IPCC, 2013) which defines the conversion factors (Global Warming Potentials – GWP) for emissions of greenhouse gases such as CO₂ (factor value = 1), methane (factor value = 28) or chloroform

(factor value = 16). As analysis of the mechanism moves towards endpoints, more subjectivity enters the characterization model. Global warming (midpoint) can provoke damage to human health (endpoint category, see Fig. 1) such as blindness or heart attack. In this endpoint category, however, these health impacts are combined through a calculation of years of life lost.⁴ This is an inevitably subjective aggregation of a complexity of health problems related to global warming, and is thus less informative for environmental assessment than the midpoint indicators.

The mechanism illustrated in Fig. 1 leads us to the notion that there is a cause-effect hierarchy connecting the emissions and resource flows produced by each product system to midpoint and endpoint impacts. This causal chain is well established for environmental impacts. In LCC and SLCA, there is also an implicit cause-effect hierarchy of midpoint, endpoint and areas of protection (Jørgensen et al., 2008). UNEP-SETAC (2009) argued that there are indeed causal chains connecting social impacts themselves, as well as others connecting social, environmental and economic ones, but these chains are “at present usually disregarded” and need to be investigated.

One possible reason for these social and economic causal chains being implicit, disregarded and in need of further investigation is that they do not depend solely on physical mechanisms, but also on human behaviours and values. This brings us back to the importance of considering stakeholder perspectives when selecting sustainability impact categories. Appropriate tools should elicit the perspectives of involved stakeholders, in order to structure such socio-economic causal chains.

2.4. Causal maps as a support for selecting decision criteria

Eliciting and structuring stakeholder perspectives as a support for decision-making are features of the family of approaches known as *Problem Structuring Methods* (PSMs). PSMs are designed to

⁴ The category indicator for “damage to human health” in the Eco-Indicator 99 method is Disability Adjusted Life Years: DALY = YLL + YLD, where YLL = years of life loss (reduced time expectancy) and YLD = Year equivalent Lost Due to morbidity of related diseases.

support decisions in problems characterised by such features as: complex systems of changing and interacting problems; the need for these problem situations to be ‘managed’ (rather than ‘solved’); the existence of alternative types and levels of explanation and phenomena of concern; solutions which are judged ‘good’ or ‘bad’ by interested parties themselves (instead of ‘true or false’ judgements by analysts); involvement with areas of greatest human concern, rather than with tactical/operational questions; and the salience of participants’ perceptions, values and interests (Rosenhead and Mingers, 2001).

In PSMs, an important methodological step is usually to draw diagrams that can visually represent stakeholders’ perceptions about the decision context. The use of maps or other graphical representations to support reasoning is a feature in several areas of knowledge. Methods which are quite widely used are Concept Maps (Novak and Cañas, 2008), Thinking Maps (Hylerle, 2008) and Mind Maps (Buzan, 1994). Mostly they are used to enhance meaningful learning and critical thinking, and are helpful in uncovering “unknown unknowns”.

One commonly used diagrammatic method in PSMs is the *causal map*. Causal maps are an integral component of a widely used PSM known as Strategic Options Development and Analysis (SODA). SODA is a methodology composed of a number of steps or stages. The first of these is to interview a number of individuals who between them represent all the key stakeholders, and to build individual causal maps (*cognitive maps*) for each of them. These individual maps are then merged into a single, group map that is discussed and enriched by the stakeholders, usually meeting in a workshop format. This process facilitates the establishment of workable and feasible agreements on the problematic situation that the group is confronting (Eden and Ackermann in Rosenhead and Mingers, 2001).

When organized in causal maps, stakeholders’ decision-making discourses can be used to generate a means-ends structure (Fig. 2, based on Keeney, 1996; Ackermann et al., 1992; Montibeller and Belton, 2006). In this topology, decision-makers’ ends/goals are positioned at the top of the map, with means/options at the bottom (Montibeller and Belton, 2006), as in Fig. 2 (The causal chain shown by the arrows in Fig. 2 represents a segment of the merged map built in our case study, see Section 4).

One of the main contributions of causal maps to the decision-making process is the support they provide for the selection of

decision criteria. In order to assess decision alternatives, broader strategic objectives at the very top of the maps must be decomposed into more “operational” ones (Franco and Montibeller, 2009), or broken into their “logical parts” (Keeney, 1996). Such operational objectives are defined as *criteria* to measure the degree of attainment of the concern expressed by broader objectives. As such, they can be translated into performance indicators, and adopted as *decision criteria* to assess alternatives (Franco and Montibeller, 2009).

Both the impact categories in the impact pathway (Fig. 1), and fundamental and operational objectives in causal maps (Fig. 2), measure the levels of good or bad *effects* provoked by lower-level decision alternatives, or, in LCSA, by alternative systems with their respective inventory flows and emissions. The basic value of causal maps for LCSA is then to identify desirable and undesirable effects reflecting implicit or explicit issues of social and economic concern for the stakeholders.

Care needs to be taken in establishing a set of decision criteria, as there are pitfalls for the unwary. Such a set of criteria needs to satisfy some desired properties, such as those presented in Table 1. This is also true for a workable set of LCA impact categories. Despite differences of terminology, there is a high degree of convergence between decision science and LCA main references (ISO, 2006a; ISO, 2006b; EC-JLC, 2010) about the expected characteristics of a coherent set of decision criteria – or, in our case, the set of LCSA impact categories.

The use of PSMs to support decisions towards sustainability is an increasing trend. Leith et al. (2014) discussed the importance of multi-stakeholder consultation, the role of analysts as facilitators, and the contribution of problem structuring to sustainability. Some researches applied PSMs to the problem of managing forests (Wolfslehner and Vacik, 2011; Marcot et al., 2012), as for developing indicators for further multicriteria decision analysis (MCDA). Nevens et al. (2013) included participatory problem structuring as an essential step of a framework to support transition towards sustainable cities. At present, no study has been published in the use of PSMs to support definition of impact categories in LCSA.

2.5. Multiple stakeholders in waste management

Waste management systems are complex and involve a diversity of stakeholders, often with conflicting interests. Let us take as an example the case which we will describe more fully in Section 4:

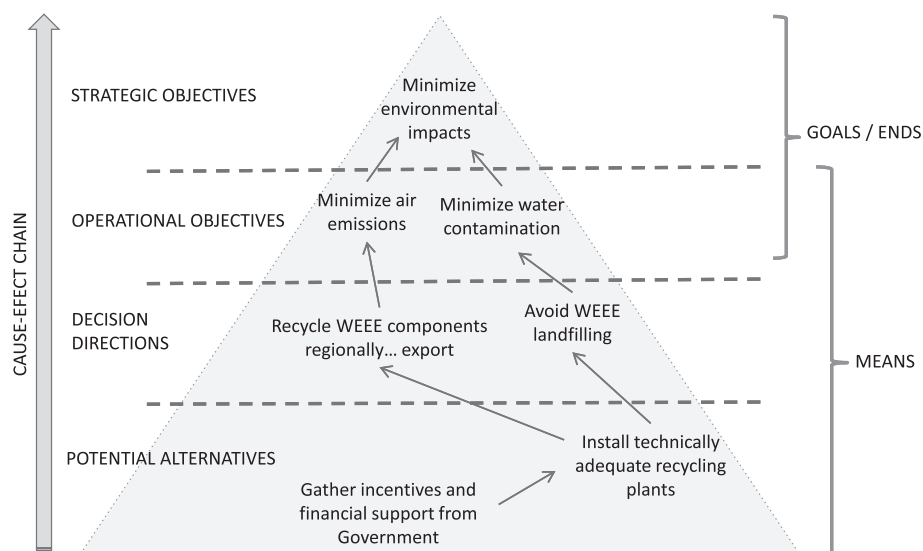


Fig. 2. Structure of a causal map, case study Brazilian WEEE reverse logistics.

Table 1

Comparison of the desirable properties of decision objectives (criteria) and of LCA impact categories.

Properties (references)	Decision objectives (Franco and Montibeller, 2009)	LCA impact categories (ISO, 2006a; ISO, 2006b; EC-JRC 2010)
Essential	Consider all essential objectives in the decision	Consider the essential objectives/goals
Understandable	Clear meaning for all the members of the decision group	Comprehensive, internationally accepted
Operational	It should be possible to measure the performance of decision alternatives against each of the fundamental objectives	Impact categories should represent characterized and aggregate midpoint and endpoint impacts of inputs and outputs (inventory) of the product system through category indicators
Nonredundant	They should not measure the same concern twice	Avoid double counting
Concise	The smallest number of objectives required for the analysis	Environmentally relevant
Preferentially independent	Performance measurement of decision alternatives on one objective disregarding all other objectives, allowing for the use of an aggregation function	Aggregation of impacts

the Brazilian Waste Electric and Electronic Equipment (WEEE) reverse logistics systems. In PNRS (2010), *reverse logistics* is defined as an “instrument for social and economic development characterized by a set of actions, procedures and means focused on enabling the collection and delivery of solid waste to the business sector, for recycling, in their own or other production cycles, or for environmentally sound final disposal”. PNRS determines that responsibility for planning and implementing such systems is shared by manufacturers, importers, distributors and retailers. Representatives of these stakeholders groups are required to submit proposed models for the WEEE reverse logistics systems, to be approved by the Ministry of Environment and selected by a Commission. After approval, those stakeholders and the Federal Government must sign a *sectoral agreement*, a contract that specifies procedures and roles for Brazilian WEEE management.

PNRS also adopts the principles of *polluter-pays* and *shared responsibility for the product life cycle*. This makes other actors such as consumers, waste pickers and their cooperatives, private companies and municipalities legally responsible for their specific roles within WEEE systems. Their organised representatives can also take part in decision-making about the sectoral agreement.

This complexity of actors necessarily implies a complexity of interests, expectations and concerns, which in turn will be reflected in a variety of expected goals and impacts (and possibly conflicting interests). WEEE reverse logistics, then, can expect to require a range of accommodations between social, economic and environmental criteria.

3. Methodological approach

The methodology which we developed to take account of actual stakeholder perspectives in order to identify and select impact categories is illustrated in Fig. 3. The methodology combines elements of both SODA and multi-criteria methods.

The first step is to identify and consult stakeholders. Questions should be formulated preferably using “why?” to identify effects, and “how?” to identify the causes of determined issues of concern (Keeney, 1996). Stakeholder consultation involves working both individually and in a workshop environment.

In our study only individual interviews were carried out, for all steps involving stakeholder consultation, during the years of 2012 and 2013, through personal visits to stakeholders and specialists in Rio de Janeiro and Sao Paulo, Southampton and Vienna, as well as via internet and telephone calls. All interviews were recorded and transcribed as source material for further steps of the methodology. Logistical issues made the holding of workshops impractical: participants lived in different towns and worked for different organisations. The researchers had no participation in sectoral agreement meetings, nor played any role as formal group decision facilitators, but rather carried out an independent analysis with stakeholder consultation.

In the standard SODA process individual interviews are needed in order to build individual maps (step 2). These maps are used to capture and structure the concepts elicited in interviews. They are then merged into a final group map (step 3), by identifying overlaps and connections between the elicited concepts. At the workshops stakeholders are encouraged to iteratively interpret and enrich the merged map (Eden and Ackermann, 2001), and thereby contribute to the necessarily subjective evaluation of impact categories. In our case, workshops were substituted by additional individual interviews, sequenced cyclically in order to update results iteratively until no further modifications were considered necessary by consulted stakeholders.

Based on the cause-effect hierarchy of the final merged map, potential impact categories can be identified (step 4), according to their topological position as midpoint or endpoint effects. The latter are usually found at the end of causal chains (destinations of arrows), at the upper areas of the map, while midpoints are positioned immediately below them (intermediate destinations of arrows). There need to be two initial sets of impact categories, one relating to endpoints and the other to midpoints. In LCSA studies we suggest the adoption of impact categories based on midpoints rather than endpoints, as was discussed in Section 2.2.

The initial set of potential midpoint impact categories is then assessed according to the properties described in Table 1. The

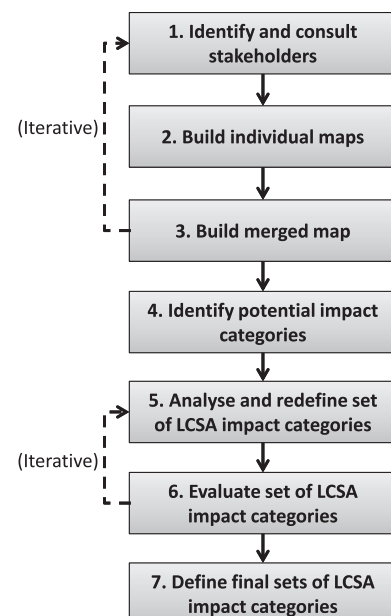


Fig. 3. Methodological approach for the selection of LCSA impact categories.

analysts suggest a redefinition of initial categories (step 5), adapting them to better satisfy the properties of *nonredundancy*, *preferential independence* and being *operational*. Some clear *dependency* relations between potential categories can be identified in the merged map. Such dependencies can give rise to *redundancy* through double counting. Another way to detect potential redundancy is by comparing the suggested indicators for each impact category to check for repetition. Finally, analysts need to check the *operationality* of suggested indicators, in order to confirm their practical measurability. Redefinitions suggested to the initial set of categories may involve the elimination and merging of impact categories (combining two potential categories into a single one, and/or sharing their performance indicators).

Stakeholders and analysts must discuss this suggested set of categories, and may redefine it as much as needed (step 6), until it satisfies all the properties in Table 1. The merged map is used as a major supporting reference in this step. Should a new category be included by stakeholders, the cause-effect chain must be discussed and reviewed in order to understand the new causal connections and hierarchy resulting from this change. Such redefinition can also involve merging duplicate or dependent impact categories into a single one, possibly sharing their performance indicators.

4. Case study: modelling WEEE reverse logistics systems in Brazil

The context for this case study was the development and implementation of a formal WEEE reverse logistics system in Brazil. Development of this system involves decisions from manufacturers, importers, distributors and retailers of electric and electronic equipment (EEE), along with government at all levels and preferably with consumers, waste pickers and other societal representatives.

WEEE reverse logistics incorporates a number of stages, and each of these can be implemented through alternative configurations. These system configurations include a variety of means for: the implementation of WEEE delivery stations; WEEE collection and delivery; transport operations and procedures; and WEEE sorting, reuse, recycling and treatment, as well as final disposal of residues. Also the roles of each actor can vary between models, as each alternative allocates different responsibilities along the chain (PNRS, 2010).

Decision criteria are needed in order to evaluate and compare possible models for the overall reverse logistics system. As argued previously in this paper, such criteria should most appropriately be based on potential sustainability impacts defined in accordance with stakeholders' perspectives.

For this research, we interviewed representatives of stakeholder groups (EEE manufacturers; commerce; State Government; *university business incubators*⁵ of waste pickers' cooperatives; waste pickers' cooperatives for WEEE dismantling and commercialisation; a social program for computer refurbishment; an environmental regulation agency) and built their respective cognitive maps. Specialists were also interviewed (on environmental legislation; on waste management; on WEEE; on reverse logistics; on environmental journalism), to provide a background to the analysis of stakeholder perspectives.

Certain important actors, such as consumers, importers and distributors, are clearly missing from our coverage of stakeholders.

But we were able to include a range of specialists and societal representatives whose inputs can go some way towards providing an overall perspective of the relevant issues. We believe the range of current interviewees is sufficient to provide a rich understanding of the context, and so a safe base for this application of the methodology, and for an assessment of its potential contribution to LCSA.

4.1. Causal maps

After recording and transcribing each stakeholder interview, we applied the causal mapping technique to the text in order to build individual maps. These individual maps were then combined into a merged map, by identifying and then merging similar concepts. For example, the manufacturer's individual map contained an operational objective of "add value in WEEE recycling". The retailer has also included the concept "add value", indicating a point at which their respective maps could be merged, enabling the cause-effect networks of the two individual maps to be linked. This procedure was extended to all the maps and their respective causal links.

The merged map was further enriched by 'interrogating' it to i) identify additional potential causes contributing to any particular concept, or further outcomes potentially affected by that concept; and ii) search for more detailed explanations of specific issues found in the merged map, seeking to clarify their possible connections to other existing or emerging issues in the map. This enrichment of the merged map was achieved through additional questions such as: "Why is X important?"; "how does W relate to Y?"; "how can U contribute to Z?"; etc.

Here is an example of this enrichment process. An early version of the merged map included the economic aim of "making the WEEE system operational and economically feasible". A direct consequence, identified from the interviews, was to "reduce currently high costs to manufacturers and retailers", which in turn enables the system to "offer good EEE prices to consumer", with onward effects on the wealth of the national economy. In follow up interviews using a first version of the merged map we asked stakeholders: "why else is it important to make the WEEE system feasible?". We were told that it can also "assure the maintenance of processes and efficiency of the new reverse chain", which in turn can "maximize materials recovery and recycling", with consequential environmental and economic benefits (Fig. 4). In this way the map was made more complete.

4.2. Environmental impact categories

As argued before, LCA impact categories have most commonly been defined technically, rather than based on stakeholders perspectives. A list of applicable WEEE LCA categories is presented in Section 2.2.

While expressed in technical terms, this list of LCA categories is clearly aligned to the overall environmental concerns revealed by the stakeholders of our case study (water and soil contamination, air pollution, risks to human health). This suggests that the stakeholder-based methodology proposed here is not contradictory to established methods of selecting environmental impact categories, but complementary to them.

4.3. Social and economic impact pathways

Unlike the environmental, the social and the economic impact categories are not dependent on physical or chemical attributes, but must rely strongly on stakeholders viewpoints. It is here that the methodology we propose finds its rationale. Examination of the merged map arising from our interviews enabled us to identify

⁵ Business incubators are instruments for local economic and employment development that aim to assist entrepreneurs with enterprise start-ups (OECD, 1999). In Brazil there are many university business incubators supporting the development of waste pickers' cooperatives.

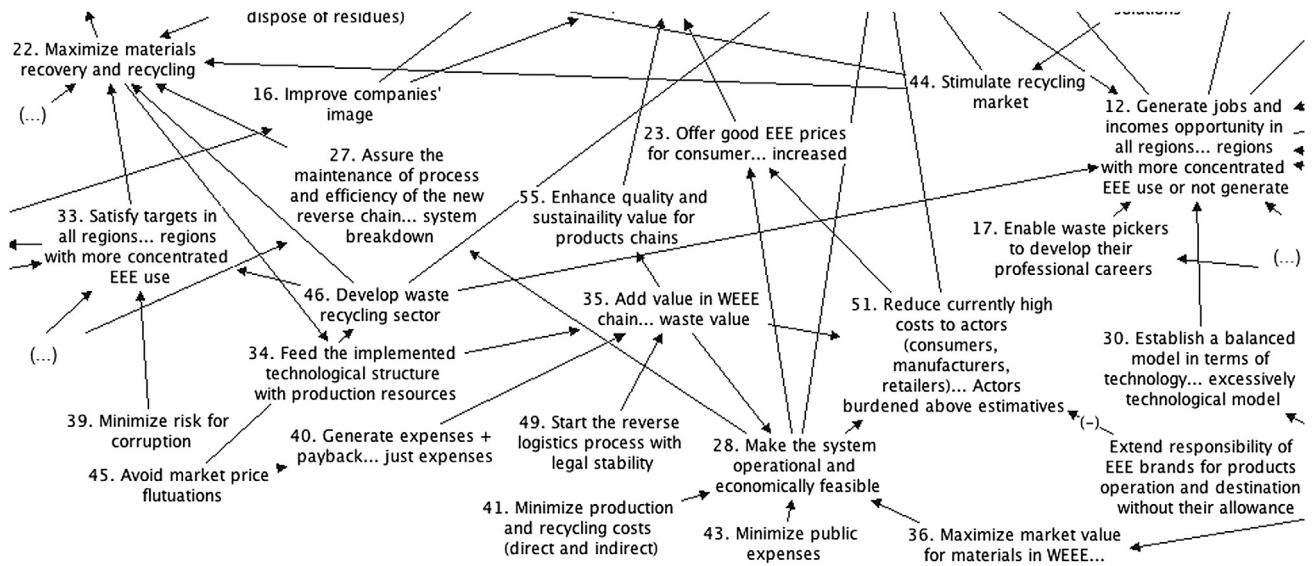


Fig. 4. Part of the merged map with enriched details from second round of interviews.

midpoint/endpoint hierarchies for both social and economic issues of concern, as shown respectively in Figs. 5 and 6.

One difficulty in the interpretation of the final causal map was to detect the implicit borders that distinguish endpoint and midpoint zones. Endpoints are the more straightforward to identify. They are the ultimate goals and so will be the final aims of the causal arrows, which will be found at the top of the merged map. (They also indicate the implicit Areas of Protection.). Below the endpoints, we proceeded by selecting potential midpoint categories that resemble operational objectives, while clearly distinguishable from critical decisions or potential options, as described in Fig. 2.

It should be noted that even among midpoint issues there is an internal causal hierarchy defining different midpoint levels, rather than the “flat” structure presented by midpoints in the environmental impact pathway (Fig. 1). This implies that there are various possible combinations of midpoint categories that could be used for an LCSA study. This is why the proposed methodology must include a way of selecting the most adequate set of categories.

Some of the potential impact categories identified are consistent with reference frameworks for LCC and SLCA. This is true for “working conditions” and “generation of jobs and incomes” (Fig. 5), mentioned in some SLCA approaches (UNEP-SETAC, 2009; Aparcana and Salhofer, 2013). The same holds for “direct and indirect costs” in Fig. 6, which can be considered the core for LCC assessment in other established frameworks (Swarr et al., 2011).

The interpretation of stakeholders perspectives using causal maps also gave rise to some interesting novel impact categories. One such case is “digital inclusion” (Fig. 5), which can be a positive effect of good WEEE systems on the social Areas of Protection, especially “human development in all Brazilian regions”. In the Brazilian context, according to stakeholders' maps, digital inclusion can be facilitated both via the donation of refurbished or used computers to NGOs and communities, and by the capacity development of waste pickers' cooperatives, providing them with a powerful resource for managing their business. Other innovative categories were: “self-esteem” (social) and “competitiveness of

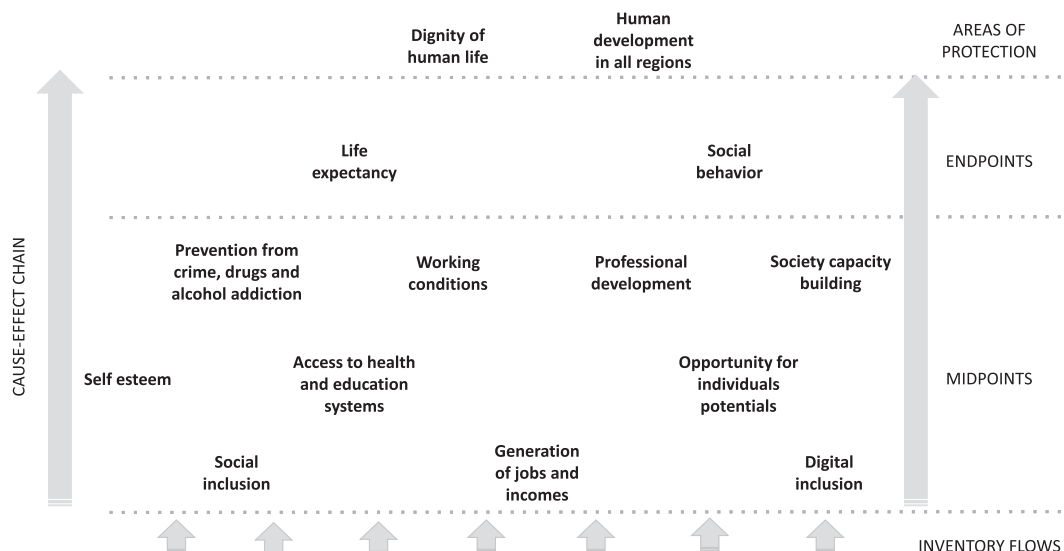


Fig. 5. Potential midpoint and endpoint social impact categories for the case study.

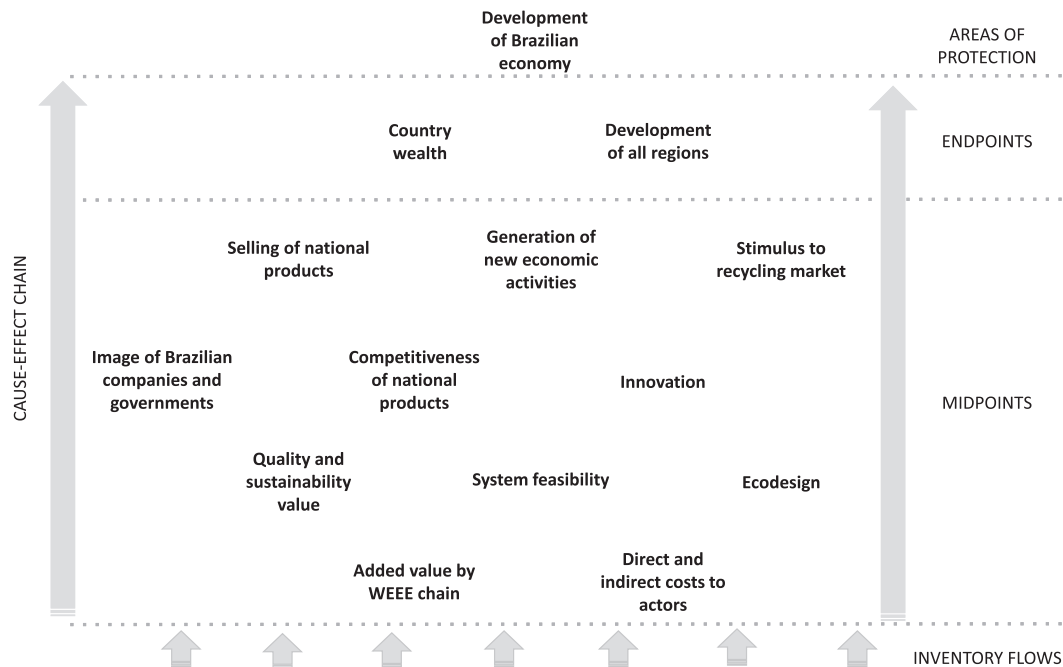


Fig. 6. Potential midpoint and endpoint economic impact categories for the case study.

formal EEE products” (economic). Without the systematic mapping of all aspects mentioned by the various stakeholders, such innovative issues might be neglected.

Another consequence of this enhanced perspective is that issues that are usually regarded as endpoints in literature references can now be viewed instead as midpoints for broader concerns. For example: rather than *working conditions* being seen as an SLCA endpoint category (UNEP-SETAC, 2009), we can now see it, in a wider context, as a means for achieving improved *life expectancy* and *human development in Brazilian regions*. Similarly, in the economic dimension, *direct and indirect costs* can be viewed as bottom-level midpoints that affect other midpoints such as *system feasibility*, with consequences for the endpoints *country wealth* and *development of all regions*.

4.4. Selection of the most adequate set of LCSA impact categories

Figs. 5 and 6 summarize the initial sets of identified social and economic impact categories. The next step was to evaluate and refine the potential set of social and economic midpoint impact categories towards a final set, which could be considered adequate when, according to the judgement of stakeholders, it satisfies the properties in Table 1. There are two levels, one for the endpoint categories and the other for all midpoint issues. It is the latter which are the focus of our research.

The analysts undertook a first analysis of the initial midpoint set, by identifying and eliminating *dependent*, *redundant* or *non-operational* impact categories. For example, we considered that product “competitiveness” and “selling” are strongly causally related, and so are not preferentially independent. We also considered that both “ecodesign” and “quality and sustainability value” could be regarded as embraced within “innovation”, so these three categories are redundant. “Self-esteem” can be considered to be a non-operational category. In this way the analysts developed a modified set of impact categories which overcame the identified inconsistencies.

The modified set suggested by analysts needed to be iteratively redefined by stakeholders until they agree that it satisfied all the properties of Table 1. This should preferably be achieved through discussion in a facilitated workshop format, but in this research practical considerations ruled out the convening of a workshop; however we were able to carry out sequential individual interviews with a subset of the stakeholders. This subset included a representative of the State Government; of the EEE manufacturers; and of a WEEE dismantling waste pickers’ cooperative. After all reviews, the final set of LCSA impact categories that satisfied all desired properties, as defined with stakeholders, is shown in Tables 2 and 3.

The midpoint categories in Tables 2 and 3 include some which are well-established in the literature, some which have been applied in other LCSA studies on waste management, but also some unforeseen categories. The equivalents of social impact categories such as “working conditions”, “formal employment” or “access to healthcare and education” (Table 2) are to be found in overall S-LCA handbooks (UNEP-SETAC, 2009; Norris, 2013) and in accounts of practical studies (e.g. Aparcana and Salhofer, 2013). Other categories obtained in our study, however, such as “social inclusion” (socialisation of workers and social/psychological assistance, as in Table 2), are not found in such sources. The closest category in the UNEP reference (Norris, 2013) is “social benefit/social security” which incorporates such aspects as health insurances, but makes no reference to psychological aspects or to support to workers in vulnerable social categories. Aparcana and Salhofer’s (2013) category “psychological working conditions” refers to satisfaction within the company rather than to mental disorders or psychological support to workers. On the other hand, some categories usually found in S-LCA on waste management, such as “child labour” or “freedom of association” (UNEP-SETAC, 2009; Aparcana and Salhofer, 2013) were not mentioned by consulted stakeholders, therefore not being concerned issues in the context of our case study.

Table 2
Social midpoint impact categories for LCSA of Brazilian WEEE systems.

Social impact categories	Suggested indicators
Social inclusion (socialising and social assistance for WEEE workers and their families)	- Number of WEEE workers and relatives provided with social and psychological assistance - Number of new WEEE workers that come from such groups: women; informality; prison; slums; alcoholism; drugs addiction; crime; physical and mental disabilities^a
(Formal and informal) employment and generation of income with opportunity to professional development	- Number of WEEE workers (formal and informal) per occupation; - Average income of WEEE workers (formal and informal) per occupation; - Number of workers that undertook professional training and refresher courses^b
Risks and working conditions	- Number of WEEE workers (formal and informal) working in adequate conditions (equipment, protection, training); - Occurrence of job accidents and diseases directly related to risks of the WEEE chain
Access to healthcare, education, environmental education and digital inclusion	- Number of WEEE workers and their relatives provided with health insurance; - Number of WEEE workers and their relatives per level of education; - Number of individuals (workers and relatives, community) benefited by the WEEE chain with digital inclusion and environmental education

^a This indicator was modified to merge two potential impact categories that had inter-dependency.

^b This indicator was included by stakeholders.

4.5. Some issues regarding the application of the methodology

Based on the results of our study we can make some comments on the operation of the methodology in practice. First, in this application it proved to be simply infeasible to hold workshop

Table 3
Economic midpoint impact categories for LCSA of Brazilian WEEE systems.

Economic impact categories	Suggested indicators
System feasibility and efficiency	- Direct and indirect costs per actor and stage of the system (US\$); - Profit and avoided costs per actor and stage of the system (US\$); - WEEE per destination/Total WEEE collected; - Demand/capacity of the WEEE system
Awareness and adhesion to WEEE reverse logistics ^a	Number of citizens and companies delivering WEEE to formal collection points
Innovation and generation of new economic activities	- Number of recently created companies within the EEE and WEEE chain; - Number of companies within the EEE and WEEE chains with innovation recognized by the Brazilian Ministry of Science and Technology
Competitiveness of the formal EEE and WEEE products ^b	- Increase rate of prices of formal products due to increased costs by reverse logistics; - Amount of informal products collected by the WEEE system; - Evolution of the EEE and WEEE informal market

^a This impact category and respective indicator were included in substitution to the former category “Image of Brazilian companies and governments”, suggested by the analysts in the first redefined set.

^b This category with respective indicators was included by stakeholders in substitution to “Selling of national products”, suggested by the analysts in the first redefined set.

meetings with the stakeholders. There is no doubt that a more thorough exploration and discussion with them would have been advantageous. However this issue was substantially overcome by the iterative interviews held with individual stakeholders as the methodology progressed, which enabled missing information to be incorporated in the map.

Second, no work of this kind can be entirely free of subjectivity. So although analysts can suggest redefinitions of impact categories, it is important that the consultation with stakeholders (step 6 of the methodology) incorporates not only the suggested redefined set, but also the impact pathway or the merged map containing all the original potential categories identified. In our study some potential categories that were excluded by the analysts in the first analysis (step 5) were brought back by stakeholders at the next step. This possibility of iterative redefinition confers a degree of robustness on our methodology.

Third, we would like to highlight that the suggested process did not require any strong interference by the analysts. As a result, feedback from stakeholders was highly positive: they felt satisfied with the completeness of the defined set of impact categories, and indeed identified with it. Such a sense of ownership of the results by the stakeholders is a significant benefit of Problem Structuring Methods which extends also to the other steps of our methodology.

While offering significant support to LCSA studies, especially in facilitating stakeholder consultation, our methodology still presents some practical issues that need to be addressed if it is to deliver its full potential contribution to sustainability assessment:

- *Consulting all relevant stakeholders:* Identification and consultation of all relevant stakeholders is fundamental to many steps of our methodology. There is a risk of missing important stakeholders in the consultation. The best way to overcome this issue is by analysing carefully the statements of consulted stakeholders' in order to identify new stakeholders. Another approach is to ask interviewed stakeholders who they think are respected people in the field and need to be consulted.
- *Consulting consumers:* In particular, consumers are a very heterogeneous group, and their consultation requires a rather complex approach which may either follow or not the steps described in our methodology. Representatives of pressure groups can be incorporated in the process, if they represent all classes within the “universe” of consumers (different geographic areas, different age groups, differing affluence, different occupations etc.). One possibility is to carry out a survey that allows consumers from a population sample to input subjective explanations regarding their perceived WEEE issues;
- *Determining the hierarchic level of concepts within the cause-effect chain:* the richer the merged map is, the more complex are its elements and causal connections. The task of determining the hierarchical level of issues (inventory, midpoints or endpoints) is a demanding one for analysts and decision-makers. If necessary, tangles at this stage (step 4) can be refined during steps 5 and 6 of the methodology;
- *Determining when it is time to stop:* the interactive process of interviewing stakeholders is time intensive. As more stakeholders are interviewed new ideas and issues emerge and the process of preparing and analysing maps becomes more complex. There is a need to decide when to stop this process. One pragmatic resolution of the dilemma is to observe closely when new inputs start predominantly to repeat issues and connections already raised, albeit in slightly different formulations;
- *Evaluating potential sets of impact categories:* the evaluation of potential impact categories (step 6) is irreducibly subjective, and relative values for categories can vary between stakeholders. Workshop discussion processes can help to facilitate consensus. This evaluation of particular categories is also highly dependent

on the product system being studied (defined by the LCSA scope), which makes it impossible to provide a standard impact pathway for LCC and SLCA studies. For example, an LCSA study on a Brazilian agricultural company could reveal such social impact categories as “forced labour” or “respect of indigenous rights” (Norris, 2013), which are commonly major concerns in this context; in WEEE management, by contrast, these categories will not be relevant, but others such as “informal workers” are likely to be major issues;

- *Applying the methodology comparatively*: while the methodology is in principle context-free, the data it elicits from stakeholders is not. Therefore it will not provide a standard LCC and SLCA impact pathway or set of impact categories for WEEE management, as is possible in LCA.

5. Summary and conclusions

This paper proposes a methodology that accesses the perspectives of relevant stakeholders for the identification and selection of sustainability impact categories. We applied the methodology to the case of WEEE reverse logistics in Brazil.

Based on this study we can draw the following conclusions:

- Causal maps can make a useful contribution to structuring impact pathways and to the selection of social and economic impact categories based on real stakeholders' perspectives;
- In our case study causal mapping and the involvement of stakeholders has revealed potential impact categories that have already been established in other SLCA and LCC methodologies, but also other quite innovative and unique ones;
- Endpoint impact categories reflect best the essential sustainability concerns for systems studied in LCSA, but their measurement is problematic. Midpoint impact categories, in contrast, are more operational, but they need to be assembled with care, and refined to ensure that their properties render them logically and practically coherent. Analysis of causal dependency among issues is a good starting point for such refinement;
- The methodology described in this paper has the potential to facilitate consultation with stakeholders in SLCA modelling. Results (maps and the defined impact categories) showed to be rich and concise. Feedback from stakeholders was highly positive, revealing that they identified with and felt ownership of the results obtained.

Some remaining methodological issues remain regarding the identification and involvement of all relevant stakeholders, the definition of endpoint and midpoint levels, and the qualitative evaluation of potential sets of impact categories. Progress here will be most effectively achieved through practical applications of the approach. One possibility for future work could be to compare both the selected impact categories and the results of LCSA studies on two or more cases involving similar decision situations but in different cultural contexts (for example, WEEE reverse logistics modelling in Brazil, India, China and other developing countries). There is also a clear potential for applying this methodology beyond waste management for other types of complex sustainability issues involving multiple stakeholders, perspectives and interests.

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References

- Ackermann, B.F., Eden, C., Cropper, S., 1992. Getting started with cognitive mapping. In: 7th Young OR Conference. University of Warwick, pp. 65–82.
- Aparcana, S., Salhofer, S., 2013. Development of a social impact assessment methodology for recycling systems in low-income countries. *Int. J. Life Cycle Assess.* 18 (5), 1106–1115. <http://dx.doi.org/10.1007/s11367-013-0546-8>.
- Bachmann, T.M., 2012. Towards life cycle sustainability assessment: drawing on the needs project's total cost and multi-criteria decision analysis ranking methods. *Int. J. Life Cycle Assess.* 18 (9), 1698–1709. <http://dx.doi.org/10.1007/s11367-012-0535-3>.
- Bigum, M., Brogaard, L., Christensen, T.H., 2012. Metal recovery from high-grade WEEE: a life cycle assessment. *J. Hazard. Mater.* 207–208, 8–14. <http://dx.doi.org/10.1016/j.jhazmat.2011.10.001>.
- Buzan, 1994. *The Mind Map Book*. Penguin Books.
- Eden, C., Ackermann, F., 2001. SODA: the principles. In: Rosenhead, J., Mingers, J. (Eds.), *Rational Analysis for a Problematic World Revisited*. John Wiley and Sons, Chichester.
- European Commission, Joint Research Centre, Institute for Environment and Sustainability, 2010. *International Reference Life Cycle Data System (ILCD) Handbook – General Guide for Life Cycle Assessment – Detailed Guidance*. JRC, IES, First ed. Publications Office of the European Union, Luxembourg.
- Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. *Sustain* 2 (10), 3309–3322.
- Franco, L.A., Montibeller, G., 2009. Problem Structuring for Multicriteria Decision Analysis Interventions. *Wiley Encyclopedia of Operations Research*.
- Hischier, R., Wäger, P., Gaughhofer, J., 2005. Does WEEE recycling make sense from an environmental perspective? *Environ. Impact Assess. Rev.* 25 (5), 525–539.
- Hyerle, D., 2008. *Visual Tools for Transforming Information into Knowledge*. Corwin Press, Thousand Oaks, CA.
- IPCC, 2013. *Climate change 2013: the physical science basis*. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, B.M. (Eds.), *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- ISO, 2006a. *International Standard ISO 14040: Environmental Management — Life Cycle Assessment — Principles and Framework*.
- ISO, 2006b. *International Standard ISO 14044: Environmental Management — Life Cycle Assessment — Requirements and Guidelines*.
- Jørgensen, A., Le Bocq, A., Nazarkina, L., Hauschild, M., 2008. Methodologies for social life cycle assessment. *Int. J. Life Cycle Assess.* 13 (2), 96–103.
- Jørgensen, A., Herrmann, I.T., Bjørn, A., 2013. Analysis of the link between a definition of sustainability and the life cycle methodologies. *Int. J. Life Cycle Assess.* 18 (8), 1440–1449.
- Kaufman, S.M., Krishnan, N., Themelis, N.J., 2010. A screening life cycle metric to benchmark the environmental sustainability of waste management systems. *Environ. Sci. Technol.* 44 (15), 5949–5955.
- Keeney, R., 1996. Value-focused thinking: identifying decision opportunities and creating alternatives. *Eur. J. Oper. Res.* 92, 537–549.
- Leith, P., O'Toole, K., Haward, M., Coffey, B., Rees, C., Ogier, E., 2014. Analysis of operating environments: a diagnostic model for linking science, society and policy for sustainability. *Environ. Sci. Policy* 39, 162–171. <http://dx.doi.org/10.1016/j.envsci.2014.01.001>.
- Marcot, B.G., Thompson, M.P., Runge, M.C., Thompson, F.R., McNulty, S., Cleaves, D., et al., 2012. Recent advances in applying decision science to managing national forests. *For. Ecol. Manag.* 285, 123–132. <http://dx.doi.org/10.1016/j.foreco.2012.08.024>.
- Martínez-Blanco, J., Lehmann, A., Muñoz, P., Antón, A., Traverso, M., Rieradevall, J., Finkbeiner, M., 2014. Application challenges for the social life cycle assessment of fertilizers within life cycle sustainability assessment. *J. Clean. Prod.* 69, 34–48. <http://dx.doi.org/10.1016/j.jclepro.2014.01.044>.
- Menikpura, S.N.M., Gheewala, S.H., Bonnet, S., 2012. Framework for life cycle sustainability assessment of municipal solid waste management systems with an application to a case study in Thailand. *Waste Manag. Res.* 30 (7), 708–719.
- Montibeller, G., Belton, V., 2006. Causal maps and the evaluation of decision options—a review. *J. Oper. Res. Soc.* 57 (7), 779–791.
- Nevens, F., Frantzeskaki, N., Gorissen, L., Loorbach, D., 2013. Urban transition labs: co-creating transformative action for sustainable cities. *J. Clean. Prod.* 50, 111–122. <http://dx.doi.org/10.1016/j.jclepro.2012.12.001>.
- Norris, C.B. (Ed.), 2013. *The Methodological Sheets for Sub-categories in Social Life Cycle Assessment (S-LCA)*. United Nations Environment Programme and SETAC.
- Novak, J.D., Cañas, A.J., 2008. *The Theory Underlying Concept Maps and How to Construct and Use Them*. Institute for Human and Machine Cognition. <http://cmmap.ihmc.us/publications/ResearchPapers/TheoryCmaps/TheoryUnderlyingConceptMaps.htm>.
- OECD, 1999. *Business Incubation: International Case Studies*. OECD Publishing.
- Ostermeyer, Y., Wallbaum, H., Reuter, F., 2013. Multidimensional Pareto optimization as an approach for site-specific building refurbishment solutions applicable for life cycle sustainability assessment. *Int. J. Life Cycle Assess.* 18 (9), 1762–1779. <http://dx.doi.org/10.1007/s11367-013-0548-6>.
- Pesonen, H.-L., Horn, S., 2012. Evaluating the sustainability SWOT as a streamlined tool for life cycle sustainability assessment. *Int. J. Life Cycle Assess.* 18 (9), 1780–1792. <http://dx.doi.org/10.1007/s11367-012-0456-1>.
- PNRS, 2010. *Brazilian Law No. 12305/2010*.

- Rocchetti, L., Vegliò, F., Kopacek, B., Beolchini, F., 2013. Environmental impact assessment of hydrometallurgical processes for metal recovery from WEEE residues using a portable prototype plant. *Environ. Sci. Technol.* 47 (3), 1581–1588.
- Rosenhead, J., Mingers, J. (Eds.), 2001. *Rational Analysis for a Problematic World Revisited*. John Wiley and Sons, Chichester.
- Sala, S., Farioli, F., Zamagni, A., 2012. Life cycle sustainability assessment in the context of sustainability science progress (part 2). *Int. J. Life Cycle Assess.* 18 (9), 1686–1697.
- Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.-L., Ciroth, A., Brent, A.C., Pagan, R., 2011. *Environmental life cycle costing: a code of practice*. Society of Environmental Chemistry and Toxicology (SETAC), Pensacola.
- Traverso, M., Asdrubali, F., Francia, A., Finkbeiner, M., 2012. Towards life cycle sustainability assessment: an implementation to photovoltaic modules. *Int. J. Life Cycle Assess.* 17 (8), 1068–1079.
- UNEP-SETAC, 2009. *Guidelines for Social Life Cycle Assessment of Products*. UNEP.
- Wäger, P., Hischer, R., Eugster, M., 2011. Environmental impacts of the Swiss collection and recovery systems for Waste Electrical and Electronic Equipment (WEEE): a follow-up. *Sci. Total Environ.* 409 (10), 1746–1756.
- Wagner, J., 2011. Incentivizing sustainable waste management. *Ecol. Econ.* 70 (4), 585–594.
- Wolfslehner, B., Vacik, H., 2011. Mapping indicator models: from intuitive problem structuring to quantified decision-making in sustainable forest management. *Ecol. Indic.* 11 (2), 274–283. <http://dx.doi.org/10.1016/j.ecolind.2010.05.004>.

3.3 Artigo 3 – Sustainability assessment and prioritisation of e-waste management options in Brazil *

* de Souza, R. G., Clímaco, J. C. N., Sant'Anna, A. P., Rocha, T. B., do Valle, R. de A. B., Quelhas, O. L. G. Sustainability assessment and prioritisation of e-waste management options in Brazil. *Waste Manag.* 2016; 57: 46–56. doi: n°. 10.1016/j.wasman.2016.01.034

ABSTRACT

Brazil has an increasing rate of e-waste generation, but there are currently few adequate management systems in operation, with the largest share of Waste Electrical and Electronic Equipment (WEEE) going to landfill sites or entering informal chains. The National Solid Waste Policy (2010) enforces the implementation of reverse logistics systems under the shared responsibility of consumers, companies and governments. The objective of this paper is to assess sustainability and prioritise system alternatives for potential implementation in the metropolitan region of Rio de Janeiro. Sustainability criteria and decision alternatives were defined by elicitation of stakeholders. The adopted multicriteria approach combines Life Cycle Assessment with qualitative evaluations by a small sample of regional experts with knowledge of the problem. The recommended system consists of a hybrid WEEE collection scheme with delivery points at shops, metro stations and neighbourhood centres; a pre-treatment phase with the involvement of private companies, cooperatives and social enterprises; and full recycling of all components in the country.

Keywords: Waste electric and electronic equipment; Reverse logistics; Multicriteria decision analysis; Composition of probabilistic preferences; Life cycle assessment; Sustainability assessment.



Sustainability assessment and prioritisation of e-waste management options in Brazil



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1. Introduction

Adequate e-waste management is still a challenge in most parts of the world, especially in developing countries. It is estimated that Brazil generated 3.8 kg of Waste Electrical and Electronic Equipment (WEEE) per capita in 2008 (Araújo et al., 2012), and 7 kg/capita in 2014 (StEP, 2015). This may be less than Mexico's 2014 generation rate (8.2), but it is more than the other BRICS countries, with the exception of Russia (China 4.4 kg/capita, India 1.3, South Africa 6.6, Russia 8.7) (StEP, 2015). Despite such a rapidly increasing generation rate, only a few adequate WEEE management systems are currently operating in the country. A large share of the e-waste produced is still disposed mixed with household waste and is destined for landfill sites, or informal chains operated by waste pickers, cooperatives and scrap dealers. The estimated recycling rate for the country is 2% (Bandini, 2009 *apud* Araújo, 2012).

In Rio de Janeiro city, the composition of collected household waste in 2012 indicated that 3.7 k tonnes of WEEE were sent to landfill sites (COMLURB, 2013), wasting valuable and non-renewable resources with considerable environmental risks. It is known that rough recycling techniques like burning cables and acid leaching are commonly applied by the informal sector in the country (Souza, 2014; Lundgren, 2012). This is an insalubrious and inefficient practice to recover materials. In addition, the country seems to be an illegal receiver of e-waste from developed countries in North America (Lundgren, 2012). Illegal and informal activities are also responsible for a large amount of Electrical and Electronic Equipment (EEE) consumed in the country; in 2014 non-official markets accounted for 1.5 million purchased computers, corresponding to 15% of the total for the year, and 35% of desktops (ABINEE, 2015).

In order to tackle those issues and to implement adequate e-waste management, the Brazilian Solid Waste National Policy – PNRS (Brazil, 2010) enforced the implementation of WEEE reverse logistics under the shared responsibility of EEE producers, importers, distributors and retailers (direct chain), with broader responsibilities of governments and other actors. In order to implement reverse logistics, those EEE direct chain actors must analyse

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different WEEE reverse logistics options, assess technical and economic feasibility, and propose a model satisfying the objectives and principles defined in the Policy as far as possible, which include the protection of public health and environmental quality; incentivising the recycling industry and resource recovery; integrating waste management; making the articulation of the different sectors; promoting operational and financial sustainability; giving stimulus to Life Cycle Assessment, and to the integration of waste pickers' cooperatives (Brazil, 2010). Proposals of WEEE reverse logistics systems have been submitted by those actors to a council of related Federal Ministries, headed by the Ministry of the Environment (MMA), which is entrusted with analysing, suggesting alterations and selecting the model to be implemented in the country. By August 2013 four proposals had been received, but due to the complexity of these multiple interests as well as the complexity of the decision problem, namely the tasks of building a coherent set of criteria and to evaluate and compare system alternatives, two years later there was still no approved final model. Proponents pointed out some issues that still needed to be addressed, such as: the implementation of a clearly communicated recycling fee; the control of imported electronic products and the simplification of WEEE transportation and WEEE ownership (Brazil, 2015).

PNRS stimulates this decision process to be reproduced at lower government levels. For instance, a São Paulo state resolution calls for industry to introduce WEEE reverse logistics proposals. Commitments must be signed by the end of 2015 (Sao Paulo, 2015). Local government resolutions are essential because municipalities are legally responsible for Municipal Solid Waste Management (MSWM) and because reverse logistics systems must be aligned with the mandatory Municipal Waste Management Plans. The Brazilian National Solid Waste Plan (PLANARES) targets indicate that the implementation of PNRS waste management strategies must start first with the largest cities, with progressive expansion to the smallest ones.

Adequate WEEE management implementation should consider a set of sustainability criteria, aligned with both the PNRS objectives and context-specific stakeholder values. Souza et al. (2015) elicited the perspectives of stakeholders involved in the Brazilian WEEE context and specifically in Rio de Janeiro, and using decision science techniques derived a set of relevant social and economic criteria to support this particular decision. These criteria were: social inclusion; employment and generation of income; professional development; health risks and working conditions; workers access to education and healthcare; system feasibility and efficiency; population awareness and adhesion to reverse logistics; innovation and stimulus of new economic activities; and competitiveness of formal EEE products in regard to the informal ones.

Assessment of environmental performances needs a systematic approach which calculates impacts based on system modelling and resources flows along the different EEE/WEEE life cycle stages. Because of its capacity to analyse complex systems and a large amount of data, Life Cycle Assessment (LCA) has been widely applied in the context of waste management and particularly WEEE management. Besides this fact, there are still few LCA applications in Brazilian waste management.

Despite the need for relevant information to assess potential impacts of system alternatives and to make decisions on Brazilian e-waste management, there is a lack of an adequate database. Collection of primary data is often obstructive, especially in regional, local and organisational scales. A practical solution to facilitate such decision could be to promote the integration of Multicriteria Decision Analysis (MCDA) with LCA, qualitative evaluations of social and economic indicators. A robust approach to sustainability assessment and prioritisation of alternatives should be a multicriteria method that, among other features, allows for the adoption of

a life cycle perspective and for a non-compensatory integration of both quantitative and qualitative indicators (Cinelli et al., 2014).

Regarding qualitative assessment of waste management sustainability indicators. In some contexts the available sample of evaluators with proper knowledge of the problem may not be sufficient to enable statistical analyses of the qualitative and quantitative measures, but the intervention of human expertise must still be considered in the decision. This can be the case, for example, of technical councils that may be organised to evaluate local WEEE management systems in Brazilian cities.

The objective of this paper is to assess sustainability and prioritise system alternatives for e-waste management in the city of Rio de Janeiro, Brazil. Specific objectives are:

- to develop an approach and consult experts for qualitative evaluation of social and economic relative performances of the system alternatives;
- to build a multicriteria analysis model, adequate for cases with small samples of evaluators; and
- to integrate MCDA with Life Cycle Assessment and with qualitative social and economic assessment.

This paper builds upon previous studies of some of the authors, namely Souza et al. (2013), where e-waste management system alternatives for Brazil/Rio de Janeiro were identified based on stakeholder elicitation; and Souza et al. (2015), where a set of sustainability criteria for Brazilian/Rio de Janeiro e-waste management was derived from stakeholder perspectives. The present study targets e-waste management specialists and decision-makers in Brazil, and seeks to recommend a solution to the decision problem, leading to the implementation of an e-waste management system in Rio de Janeiro.

2. Background knowledge

2.1. E-waste management in the Brazilian National Solid Waste Policy

According to the National Solid Waste Policy (PNRS), consumers (both population and institutions) are responsible for disposing of their e-waste separately at adequate delivery points defined in Municipal Solid Waste Plans. Retailers and distributors are responsible for returning the products to producers and importers, who in turn have to provide adequate treatment and final disposal of refuse. PNRS (Brazil, 2010) permits that these actors formalise one or more management entities, which can outsource reverse logistics operations to private waste management companies, MSWM schemes, skilled cooperatives or social enterprises. PNRS stimulates involvement of cooperatives if they have adequate training, working conditions and environmental licences to perform the required activities.

Proposals of a WEEE reverse logistics system for the country should be presented to the Ministry of the Environment by producers, importers, distributors and retailers. The selected model must be formalised into a Sectorial Agreement (SA), a contract signed by all aforementioned parties. It can also be specific to State SA and to Municipal SA. The SAs have to detail, among other information:

- descriptions of the set of integrated activities by each participant in the reverse logistics system, in the processes of collection, storage, transport, recycling and final disposal, indicating:
 - technical recommendations at each stage;
 - criteria to install and operate delivery points;
 - adopted collection schemes;
 - procedures and responsibilities for sorting, reuse, recycling, treatment, and final disposal activities;

- possible hiring of waste pickers' cooperatives and associations;
- assessment of social and economic impacts of the reverse logistics system (Brazil, 2010).

2.2. E-waste management situation and feasibility assessment in Brazil

In order to structure the Sectorial Agreement, the Brazilian government previously requested a feasibility study (FS) of the WEEE reverse logistics implementation in Brazil (ABDI, 2012). This FS pointed out that Brazil produces around 1 million tonnes of WEEE per year and that around 4 thousand collection points in 2016 would be necessary in order to achieve around 70% of the collection rate. The FS also pointed out that Brazil has 94 WEEE recycling facilities, most of them in the state of São Paulo (Southeast of Brazil, the same region as the state of Rio de Janeiro). Due to the large geographical distances within Brazil, the installation of sorting units that store WEEE components before their transportation to recycling facilities was suggested. In order to optimise cost, the FS recommend that cities with more than 150 thousand inhabitants must have a sorting unit. In addition, the FS proposed a WEEE reverse logistics model coordinated by one or more management entity and divided in two stages:

- *Primary stage*: involves WEEE collection and transportation to sorting units. At this stage, the consumer should dispose of small devices in collection points available at EEE shops. The industry, importer or management entity should provide door collection of large devices. The commerce or management entity is responsible for storing WEEE collected and transporting it to sorting units.
- *Secondary stage*: involves WEEE storage and transportation from sorting units to recycling facilities and final disposal. The management entity should coordinate the sorting units and the transportation to recycling facilities.

The FS discussed alternatives and made recommendations in eight key modelling variables: sources of resources for implementation; responsibility for orphan products (from illegal or informal markets); targets for recovery and recycling; level of responsibility of the public administration; WEEE hazardousness classification; reuse in the reverse logistics system; WEEE sorting by brands; proportional responsibility for WEEE; and competition model. Their recommendations were, respectively: shared costs by the actors in the EEE chain (from industry to consumer); no recommendation for orphan products; recycling target is 100% of products declared by the SA signed companies; government will be responsible for providing resources for research, infrastructure and campaigns; WEEE should be treated as non-hazardous but only be dismantled by recyclers; delivery points and consumer service should be made available to organise the reuse chain; WEEE should be monitored by sampling at the sorting units for identification of orphans, data checking and reporting to authorities; cost share for each producer is defined in proportion with their relative sales in the previous year; and stimulating competition among several management entities created by partnerships of producers and importers.

The FS was the base for a call for SA proposals to be presented by producers, importers, distributors and retailers. The Ministry of Environment received 11 proposals in 2013, but only 4 were accepted for evaluation (Veloso, 2015).

2.3. Existing studies in Brazilian e-waste management

There is increasing interest in developing studies on Brazilian WEEE management, but there are currently few published studies on this topic. Araújo et al. (2012) proposed a model to estimate WEEE generation and applied it at national-level using primary

data from the EEE market. Franco and Lange (2011), using a survey of householders' behaviour, estimated the WEEE generation in the city of Belo Horizonte and tracked the current flows, identifying that most is either donated, kept or sold.

Trying to describe the current situation, Saavedra and Ometto (2012) identified some existing initiatives referring to state legislation, social and digital inclusion and recycling companies. Oliveira et al. (2012) provided an overview on current e-waste management practices in the country, highlighting the need for a well-defined model, the existence of a cascade reuse market, the lack of companies for complete e-waste recycling (PCBs are exported), the need for an efficient collection scheme, and the need to include waste pickers. Ongondo et al. (2011) added the evidence of WEEE being dumped. Quariguasi Frota Neto and Van Wassenhove (2013) found that existing WEEE take-back initiatives in the country are led by large multinational manufacturers, but are less advanced than those performed.

Focusing the development of solutions for the e-waste problem in the country, Pimentel et al. (2013) presented a research project, called Ambientronics, which aims at technological development to support the Brazilian recycling industry. Chatterjee and Kumar (2009) proposed an outsourcing model which integrates non-formal operators in collection, disassembly and segregation of e-waste, whereas formal actors process PCBs for resource recovery. Campos et al. (2014) highlighted the relevance of creating a 5R network – Reduce, Redesign, Recycle, Reuse and Repurpose. Hirayama and Saron (2015) analysed the composition of Brazilian waste computer equipment (WCE) and concluded that implementation of stricter regulations for identification of thermoplastic polymers in WCE in Brazil is an important step for successful mechanical recycling of these materials.

Other studies discussed Brazilian legal framework in comparison to other countries. These highlighted the need for national and global standardisation (Sant'Anna et al., 2014), the limitation of the PNRS in just requiring implementation of reverse logistics without an efficient management and control framework (Barboza et al., 2014), and the need for decisions from competent authorities observing diverging interpretations of legal stipulations and administrative procedures (Brandmann and Altvater, 2012). None of these identified studies focused on modelling and selection of an e-waste management model for the country or a city by sustainability assessment and prioritisation of alternatives.

2.4. LCA studies in WEEE management

Life Cycle Assessment (LCA) has been established as a technique to quantify the potential environmental impact using a systematic approach. LCA can provide objective indicators to compare processes or products, being an important tool in environmental management and pollution prevention (ISO, 2006). It has been widely applied in the context of waste management (Laurent et al., 2014a) and particularly WEEE management (Wäger et al., 2011; Bigum et al., 2012; Hong et al., 2015; among others). There are few LCA studies in Brazilian waste management (e.g. Mendes et al., 2003; Leme et al., 2014; Reichert and Mendes, 2014), and none specifically for Brazilian household WEEE (although Rubin et al., 2014 analyse a generic recycling system that could be applied in Brazil, as suggested by the authors).

Several studies have used LCA to analyse and compare different scenarios for e-waste treatment, aiming to support decisions, identify key factors and improve opportunities associated with all stages within a determined system boundary. Using LCA, Huisman et al. (2008) concluded that the implementation of the European WEEE Directive contributed to reducing environmental impacts in all categories evaluated. Emphasis is set on the reduction of 36 million tons of CO₂ and 34 million tons of CFCs, which

are no longer discharged into the environment due to WEEE recycling.

Rubin et al. (2014) used global data to apply LCA and compare two processes for recovering copper from PCBs, both adopting mechanical and electrochemical processing. They concluded that the process which employs aqua regia (a combination of nitric and chloridric acid) had the best environmental performance. Because of the narrow scope of their LCA study, with little necessity for regional data like distances or local WEEE composition, this recommended solution can be employed in Brazil or elsewhere.

2.5. Qualitative sustainability assessment of waste management

There are just a few studies adopting qualitative evaluations in waste management, most of which are relatively recent. They are essentially based on the application of survey questionnaires to a sample of specialists or stakeholders (Glew et al., 2013; Khalili and Ehrlich, 2013; Nichols et al., 2013), or judgment by the analysts based on observation of the context (Troschinetz and Mihelcic, 2009; Manhart, 2011; Al Sabbagh et al., 2012).

Specifically in the context of e-waste management, Manhart (2011) assessed a reference (baseline) and two alternative scenarios for an international recycling cooperation between developing and developed countries: in the baseline, e-waste generated in low-income countries undergoes crude recycling and uncontrolled disposal; in the first scenario, e-waste components containing valuable metals are exported from developing to developed countries for resource recovery; and in the second scenario, low-income countries import e-waste from developed ones for pre-treatment, and then export it back for recovery of precious metals. Some of these scenarios are rather similar to the alternative systems analysed in the present study (Section 3.1). Qualitatively evaluating the scenarios by own judgment, based on the study of context information, the author concluded that Scenario 1 allows for “better management of hazardous substances, higher recycling rate of scarce and valuable metals, reduced pressure on mining, lower GHG emissions, income generation for the urban poor, and investments into social and environmental standards”.

2.6. MCDA sustainability assessment in waste management

There is a remarkable number of publications on the use of MCDA in waste management decision problems. Soltani et al. (2015) carried out a literature review of MCDA applications in MSWM, identifying 68 references published up to 2013. Additionally, we could identify another 27 articles from 2014 and 2015 (searched at Scopus in 4/Sep/2015). Probably the most common MSWM decision problem analysed with MCDA is the location of facilities and landfills (as studied by Liu et al., 2013; Eiselt and Marianov, 2015; Hamzeh et al., 2015).

Among the 68 studies identified by Soltani et al., only 26 have acknowledged multiple stakeholders using MCDA, mainly for assigning weights to criteria. Most of these stakeholders were experts (69% of the 26 papers), governments or municipalities (62%) and public or residents (50%). Most of the papers adopted a popular MCDA method called AHP (34 of total studies and 15 with stakeholders), which is an aggregative and compensatory method, whilst other MCDA methods were used in 15 papers with stakeholder consultation. In this case, the methods PROMETHEE and ELECTRE, which are not compensatory, were used respectively in 1 and 3 papers. Although less commonly used, non-compensatory methods (in which the criteria are not aggregated in a single synthesizing criterion) are preferable when assessing sustainability, because they enforce a strong sustainability approach (for instance, a good economic performance cannot com-

pensate a bad environmental or social performance) and because they can handle uncertainty (Cinelli et al., 2014).

There are just a few studies of MCDA applications in WEEE management. Rousis et al. (2008) used PROMETHEE to examine 12 alternative WEEE systems for Cyprus, and concluded that the best option is “partial disassembly and forwarding of recyclable materials to the native existing market and disposal of the residues at landfill sites”. In this study, the scores were attributed by the analysts and validated by experts. Gamberini et al. (2010) analysed alternative WEEE transport routes in a region of Italy, evaluated with LCA and technical parameters, and ranked using fuzzy optimisation. Wibowo and Deng (2015) also adopted a fuzzy approach, aiming to evaluate e-waste recycling programs in Sri Lanka by consulting a group of three decision makers. Each one evaluated the alternatives considering four criteria, representing the three sustainability dimensions and a technical category. The evaluations were measured on a qualitative 5-point scale from Very Poor to Very Good. The best ranked program was “Recovery of precious metals and other recyclable materials such as metals, plastic from e-waste”. A problem with this approach is that the decision criteria are too broad and may lead to high subjectivity, whilst an advantage is the ability to handle subjectiveness and imprecision of qualitative evaluations.

Specifically regarding Brazilian WEEE, Guarnieri et al. (2014) proposed an MCDA framework to the problem of selecting third-party reverse logistics providers. The proposed model comprehended the following criteria categories: logistics, financial, capacity/infrastructure, value added services to customers, alliances with suppliers, and environmental issues. It is not proposed as either a method for performance evaluations, nor as an MCDA approach to prioritise the alternatives.

Among the identified studies in WEEE MCDA, the most similar to the present research was proposed by Wibowo and Deng (2015), because it worked with a small sample of evaluators (decision makers) and with qualitative indicator performances estimated by the decision makers using predefined scales. The study of Gamberini et al. (2010) is similar to the present in the aspect of integrating LCA evaluations with other performance indicators in an MCDA approach. The principal differences are associated with the decision problem analysed, the sustainability criteria and the approaches adopted for MCDA qualitative evaluation (ours are described in Section 3).

2.7. The relevance to waste management of combining MCDA and LCA

LCA, qualitative assessment and MCDA are innovative, powerful tools to improve waste management practices. The Brazilian National Solid Waste Policy mentions explicitly the objective of stimulating LCA implementations and implicitly the need for qualitative assessment by social actors. MCDA is not mentioned, but could be an appropriated tool for obeying its principle that, when managing solid waste systems, “a systemic view that considers environmental, social, cultural, economic, technological, and Public Health variables” should be adopted.

The three previous sections showed that there is vast literature underlining the specific role of each of these tools in waste management. Kiddee et al. (2013) has highlighted the usefulness of LCA to ameliorate most e-waste problems to human health and the environment, admitting nevertheless, that it must be complemented by other tools in order to enhance the decision robustness. Troschinetz and Mihelcic (2009) explored the benefits of using qualitative data when there is a lack of numeric indicators to evaluate performances of waste management systems. Wibowo and Deng (2015) concluded that MCDA is an effective and efficient approach to evaluate the performances of alternative e-waste recycling programs in a specific situation. It is a complex and challeng-

ing problem, as it involves “several decision makers and multiple evaluation criteria with the presence of subjective and imprecise assessments”.

The present paper looks for an integrated approach combining those three tools, which represents an innovation in WEEE theory and practice. Such an integrated approach offers an interesting potential solution to the current challenge of incorporating all dimensions of sustainability in waste management decisions. Some authors have already anticipated the importance of this integration. Munda (2008) argues that every sustainability assessment approach is based on some form of multicriterial aggregation, explicit or not. For Cinelli et al. (2014), sustainability assessment must happen in a structured, transparent and reliable way and MCDA can largely contribute to this; moreover, they stress that an adequate MCDA method for this purpose should be able to integrate LCA and qualitative indicators. When adopting the strong sustainability paradigm, such integration should not involve a compensatory aggregation of the criteria (Valle and Clímaco, 2015).

3. Materials and methods

3.1. Alternative scenarios under evaluation

The alternative WEEE systems considered for implementation in Rio de Janeiro are based on the previous mapping by Souza et al. (2013). The baseline scenario A (Fig. 1) corresponds to the existing collection and to the treatment activities for WEEE generated in Rio de Janeiro, with emphasis on informality and landfilling. As there is no available data that indicates the total amounts of WEEE collected and processed by the existing chains (besides the landfill waste composition), the WEEE flows represented in the baseline scenario were assumed but may not correspond to reality. Rather, this scenario is relevant for assessment because it allows for perceiving the relative importance of informality and landfilling to sustainability performances.

All the other four scenarios analysed (B1, B2, C1 and C2) consider the same hybrid solution for WEEE pre-treatment, consisting of 20% of WEEE being sorted and dismantled by social enterprises and skilled cooperatives, and 80% by formal WEEE recycling com-

panies. No informality takes place in these scenarios; cooperatives involved are adequately trained and licensed. These scenarios also consider that most WEEE components are adequately recycled regionally, assuming that recycling plants are located around São Paulo (ABDI, 2012). Some functional appliances and components are refurbished by a social program in a model similar to the *Fábrica Verde* (Green Factory), a Rio de Janeiro governmental social project (suspended early 2015) that offered training to youngsters from favelas in computer dismantling, refurbishment and maintenance. Refurbished computers were donated to NGOs and favela communities.

Alternatives B1 and B2 (Fig. 2) have a collection scheme based on WEEE delivery only at EEE shops. In B1 PCBs are exported for recycling in Europe, whilst in B2 all components are recycled regionally.

C1 and C2 (Fig. 3) consider a hybrid collection scheme with delivery points at EEE shops, metro stations and neighbourhood stations. C1 and C2 differ in the same way as B1 and B2.

3.2. LCA modelling

The LCA model used in this research was developed and is described, seeking to satisfy the recommendations by Laurent et al. (2014b) for LCA applications in waste management. It also satisfies the guidance of ISO 14040 (ISO, 2006).

LCA was applied to evaluate and compare the environmental performances of the alternative systems proposed in Section 3.1 (goal of the study). The functional unit was 1 ton of WEEE (Swiss mix as reported in SWICO, 2013).

Because there is limited information about current e-waste flows and processes as currently occurring in Brazil, this study used international data mostly from the Ecoinvent database (Hischier et al., 2007) and others (SWICO, 2013; Hong et al., 2015). The modelling also adopted assumptions in the representation of Brazilian e-waste treatment processes. Table 1 summarises the data used, model assumptions and sources of data.

The distance of 150 km adopted as a pattern in all transport processes is a standard used in the Ecoinvent database when there is no detailed local data. A sensitivity analysis was applied to all variables marked with “Own Assumption” Table 1, with a variation

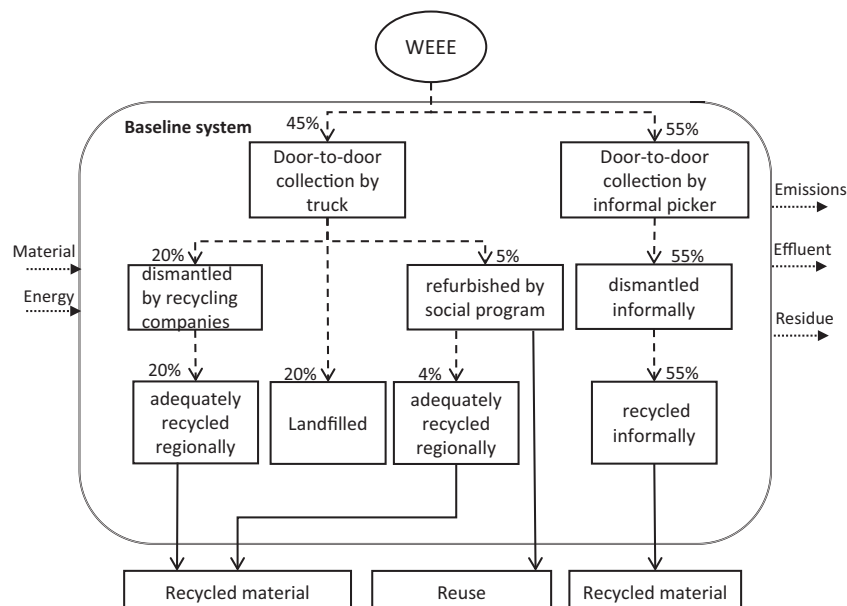


Fig. 1. Baseline system with respective WEEE flows.

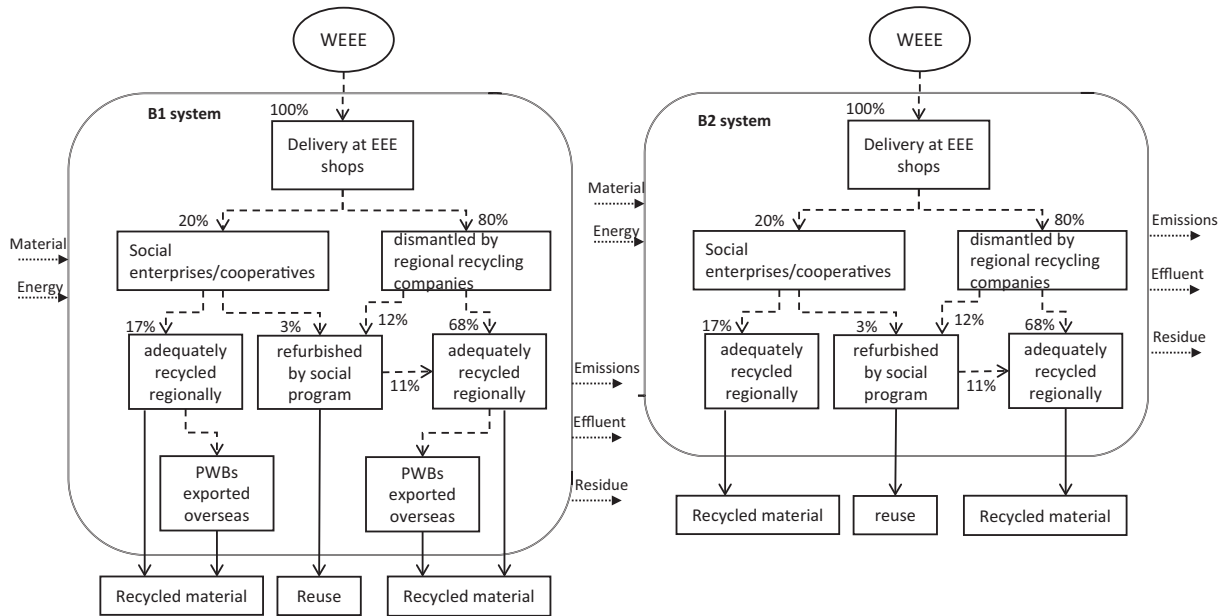


Fig. 2. B1 and B2 alternative systems with respective WEEE flows.

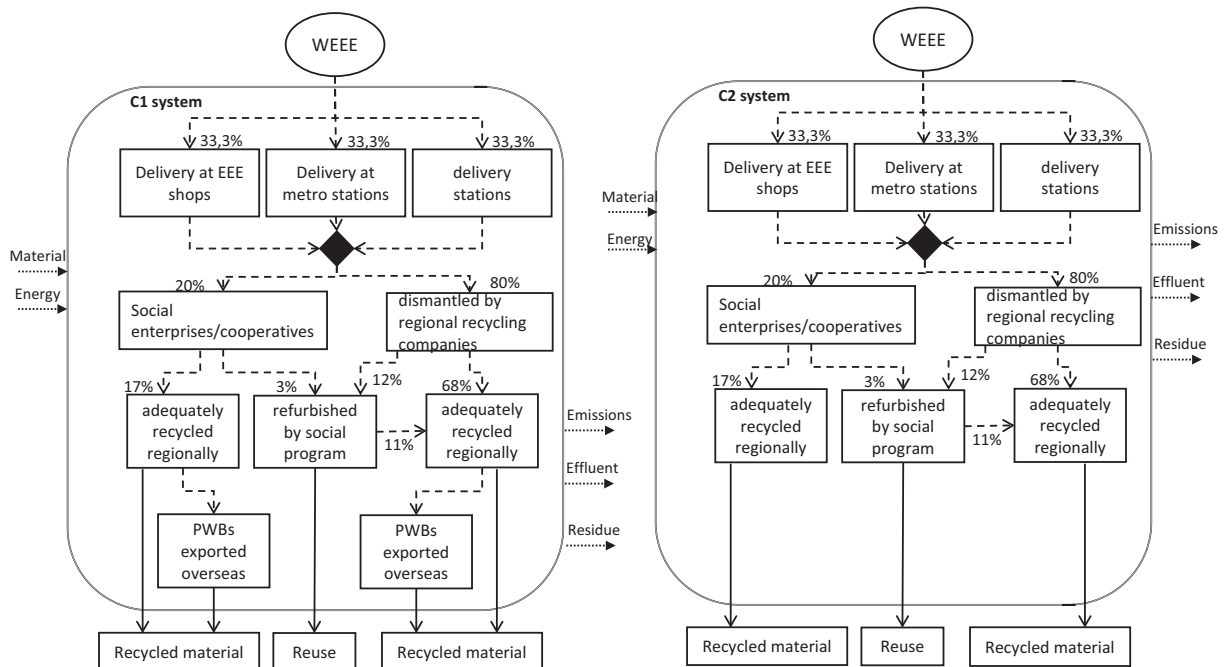


Fig. 3. C1 and C2 alternative systems with respective WEEE flows.

of 50% in the values. There was no allocation of impacts, but Ecolnvent usually adopts the economic allocation.

In order to support LCA data analysis and interpretation, this study used SimaPro® 8.1, a piece of software that supports various Life Cycle Inventory (LCI) databases. Among the Life Cycle Impact Assessment (LCIA) methods provided by the software, CML (Center of Environmental Science of Leiden University) has been chosen to classify and characterise inputs and outputs from the e-waste treatment process (e.g. energy, transportation, emissions, etc.) into potential environmental impacts. The impact categories adopted are in line with previous WEEE LCA applications found in literature

(Section 2.3). These are: Abiotic Depletion (and Fuels); Global Warming; Human Toxicity; Ecotoxicity (Fresh Water, Marine, Terrestrial); and Acidification.

3.3. Qualitative assessment of social and economic indicators

3.3.1. Social and economic impact categories

The social and economic impact categories assessed in this study are those previously defined in Souza et al. (2015). These categories were defined specifically for the decision problem under study, based on the elicitation of Rio de Janeiro and Brazilian stake-

Table 1
LCA model input data, assumptions and sources.

Process	Data used and assumptions	Source
Door-to-door collection by truck	Assumed 150 km between home and recycling companies with small truck	Own assumption
Door-to-door collection by pickers	Assumed work labour has no environmental impact	Own assumption
Delivery at EEE shops	Assumed 150 km between delivery points and recycling companies with medium truck	Own assumption
Delivery at metro stations		
Delivery at delivery points		
Dismantled by recycling company	Assumed manual disassembly of e-waste in parts and material as reported by the SWICO organisation	SWICO (2013)
Refurbishment by social program	Data collected at <i>Fabrica Verde</i> social program in Brazil which refurbished computers for donation (15% of e-waste received)	Collected by the authors
Informal dismantling	Assumed Chinese data with no treatment at end-life disposal	Hong et al. (2015)
Informal recycling		
Adequately recycled regionally	Assumed European data from Ecoinvent database for dismantling companies. In case of regional PWB recycling, Chinese data was used with treatment of end-life disposal	Hischier et al. (2007) and Hong et al. (2015)
Social Enterprises/cooperatives		
PWB recycling exported overseas	Assumed European data for recycling with use of pyrometallurgical process to recycle PWB and recovery metals	Hischier et al. (2007) and Classen et al. (2009)
Landfill sites	Assumed European data for landfilling with inert material	Doka (2009)

holders directly involved with the WEEE context. Table 2 summarises the social and economic impact categories together with respective indicators under evaluation.

3.3.2. Specialists sample, process of inquiry and data compilation

The set of social and economic indicators was evaluated individually by a group of five experts. All of them are from universities or research institutes in Rio de Janeiro and São Paulo, and all of them have published in the areas of Brazilian waste management, WEEE and/or waste Life Cycle Assessment, in indexed international journals. For the purposes of this research, this group works as a consultancy team for the decision-maker – the WEEE management entity or the municipality. This way, the framework for decision-support, except for the LCA model, is completely based on the perceptions and judgments of regional stakeholders and experts.

The number of experts selected is aimed at the minimum possible evaluations to make a mathematical analysis acceptable. The mathematical approach used is explained in Section 3.4. The present study adopted a decision context where few experts with knowledge of the problem are available to make qualitative evaluations, in order that the approach can be replicated in other critical decision situations.

The inquiry process consisted of either face-to-face or online interviews. In the first case, evaluations were made in writing, whilst in the second case a Word file is used. First, the system alternatives and the sustainability categories were presented to the evaluators. Furthermore, they also have had the opportunity to clarify possible doubts and to validate these decision parameters.

Table 2
Sustainability impact categories and indicators evaluated in this study. Source: Adapted from Souza et al. (2015)

Categories	Indicators
Economic	System feasibility Direct and indirect costs – total Profit and avoided costs – total System efficiency Amount of e-waste with adequate treatment/total collected Balanced demand/capacity of the system Population awareness and adhesion to reverse logistics Number of citizens and companies delivering E-WASTE to formal collection points Innovation and generation of new economic activities Number of recently created companies within the e-product and e-waste chains Number of companies within the EEE and E-WASTE chains with innovation recognised by the Brazilian Ministry of Science and Technology Competitiveness of formal products in regard to informal ones Increase in prices of formal e-products due to increased costs by formal e-waste reverse logistics Growth of the informal market (e-products and e-waste)
Social	Social inclusion Number of e-waste workers and relatives provided with social and psychological assistance Number of new e-waste workers that come from such groups: women; informality; prison; slums; alcoholism; drug addiction; crime; physical and mental disabilities Formal employment Number of formal e-waste workers Generation of income Average income of e-waste workers – formal and informal Opportunity for professional development Number of workers that undertook professional training and refresher courses Health risks and working conditions Number of e-waste workers (formal and informal) working in adequate conditions (equipment, protection, training) Occurrence of accidents at work and diseases directly related to the risks of the e-waste chain Access to healthcare Number of e-waste workers and their relatives provided with health insurance Access to education Number of e-waste workers and relatives with high level of education

Secondly, they were asked to evaluate the relative performances of alternatives for each criterion, according to their perceptions. For this evaluation, a continuous interval scale (Fig. 1a) was presented blank, indicating a direction of preference from the lower (–) to the higher (+) impact in each criterion. Evaluators had to mark the relative position of each alternative in that scale, taking into consideration the distances between alternatives themselves and the worst/best possible scores. An example of such evaluation by one of the consulted experts is presented in Fig. 4b.

The reason for using such a scale instead of the Likert scale – the five-point scale (Very Poor – . . . – Very Good) is that it can be more intuitive to evaluators (alternatives are placed on the line as if they are in a “race”). It is also possible to translate the data into a Likert scale by defining thresholds.

After obtaining evaluations from the consulted experts, the relative distances of alternatives in the scales were measured with a rule, and normalised in the form percentage of the total length of the line. Normalisation also involved the correction of preference

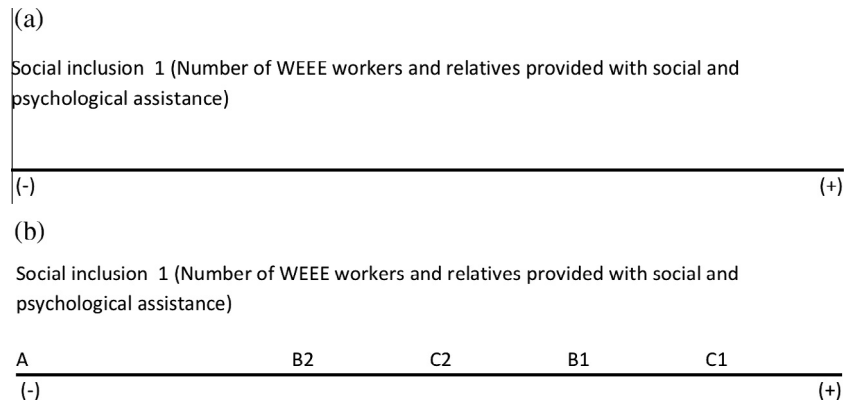


Fig. 4. Example of the evaluation scale used for qualitative assessment. (a) Blank scale. (b) Filled evaluation by an expert.

directions in the form of 0% to the worst and 100% to the best possible performances. Data provided by all experts were compiled in Excel tables.

3.4. Multicriteria analysis: composition of probabilistic preferences

The present study required a MCDA approach refusing the possibility of bad performances being compensated by good performances (i.e., non-compensatory), allowing the integration of qualitative and quantitative indicators, and permitting a reasonably trustworthy analysis of qualitative evaluations by a small sample of specialists. A good option is the recently developed Composition of Probabilistic Preferences (CPP). CPP is a MCDA approach which takes into account imprecise measurements. Evaluation of an alternative is given by the probability of such alternative being the best or the worst in comparison to all other alternatives.

The main steps in CPP are: (a) defining the criteria and the alternatives; (b) transforming the initial evaluations into probabilistic distributions; (c) calculating probabilities of preference for each alternative; and (d) obtaining the global preferences. This method is fully described in Sant'Anna (2015).

In CPP, the required input data are the preference evaluations, in our case established by five experts, as outlined above. When scores of alternatives in each criterion are obtained by those evaluations, the sets of measurements for a same alternative can be treated as samples of that distribution, and can be used to estimate all of its parameters. For the purposes of this study, the performance measurements by evaluators in the continuous scale were divided into nine preference classes, defined as profiles with stretched extremes and shrunk centrals, in order to avoid empty or too crowded classes, and thus making a better distinction of the alternatives.

A variation called CPP Beta was designed to analyse evaluations made by multiple experts. The beta distribution is applied to cases where asymmetry is assumed, thus permitting more flexibility in modelling the sample distribution. It also adopts an evaluation metric from 0 to 1, which is suitable to the scale adopted in the present study (Section 3.3). The density of the beta distribution is $f(x) = [(x-L)/(U-L)]^{\alpha-1} [(U-x)/(U-L)]^{\beta-1} / \text{Beta}(\alpha, \beta)$, for x varying between L and U and Beta denoting the beta function with positive shape parameters α and β . The standard beta distribution, with extremes 0 and 1, is obtained by substituting the observed x by $(x-L)/(U-L)$. In the beta model, it is possible to model the gain in precision with the increase in the information, by fixing $\alpha + \beta = N$, denoting N as the number of experts. In this distribution the variance decreases with the sum of the parameters α and β (where α is $f(x)$ times N , and β is $(1-f(x))$ times N).

The beta distribution is unimodal (has a single local maximum), which occurs for $\alpha > 1$ or $\beta > 1$. Assuming that the x_{ijk} for alternative

i according to criterion j are allowed to vary on bounded intervals (a_{ij}, b_{ij}) which are sufficiently far from L and U (0 and 1), the initial vector of evaluations $(x_{ij1}, \dots, x_{ijN})$ gives rise to a unimodal standard beta distribution with shape parameters α_{ij} and β_{ij} . Theoretically, unimodality is desirable in CPP because the values obtained from experts' evaluations are a location around which the variable is distributed. Because the evaluation scale is a vector of observations, some experts will necessarily be attracted to extreme evaluations. If this is observed, distorting the expected unimodal shape of the beta distributions for many values for i and j , this attraction can be compensated by slightly dislocating the measures to the interval between $1/(N-1)$ and $(N-2)/(N-1)$. For instance, unimodality is granted replacing x_{ijl} by $[1 + (N-3)(x_{ijl} - a_{ij}))(b_{ij} - a_{ij})]/(N-1)$. When $N = 1, 2$ or 3 , it is not possible to have $1/(N-1) < (N-2)/(N-1)$. Because of this, CPP beta requires a number of experts $N > 3$, in order to have a completely acceptable interval.

CPP calculates the following parameters: probabilities of preference of alternatives in each criterion, which are further combined in a final probability of preference; and the importance parameter of each criterion (capacity). Capacity is calculated to each possible subset of criteria. It is different from criteria weighting because there is not a fixed weight but combined values taking into account the relative importance of each criterion in a specific evaluation. The importance of each criterion is calculated by the Shapley index. Considering the possibility of interaction between sustainability categories, the preferences in each dimension can be combined into a final probability of preference with the use of the Choquet integral, which is a non-compensatory model (Grabisch et al., 2008).

In the present study, CPP Beta was modelled as following: (1) each dimension was assessed independently, assuming interaction between criteria, and using the Choquet integral to aggregate their respective probabilities of preferences; (2) the best alternative in each dimension was selected to the next stage; (3) the criteria which did not significantly distinguish between the selected alternatives were discarded; only the criteria where there was a minimum gap of 0.25 between at least two alternatives were selected for the next step; (4) the probabilities of preferences of selected alternatives (step 2) in the distinguishing criteria of the three dimensions (step 3) were aggregated, again with the use of the Choquet integral to admit interactions, in a final composition of probabilistic preferences.

4. Results and discussion

The environmental assessment of scenarios using LCA is summarised in Fig. 5. Scenarios B1 and C1, which performed best in most of the impact categories, are very similar to each other. B2

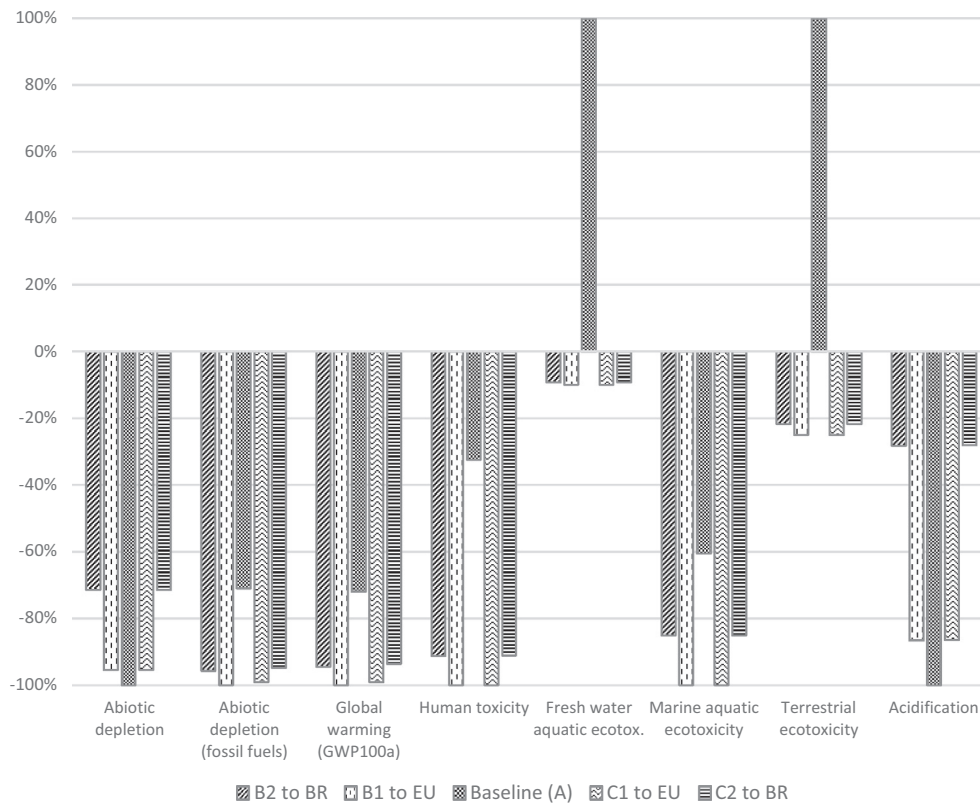


Fig. 5. Results of the LCA for the alternative WEEE system scenarios.

and C2 are also very similar. Scenario A (baseline) is the worst solution in almost all categories.

Observing the LCA results and sensitivity analyses, it is possible to conclude that WEEE collection and transport activities had little contribution to environmental impacts, in comparison to the recycling and treatment processes. Based on the life cycle perspective and restricted by the model assumptions (Table 1), all system alternatives present benefits in almost all environmental impact categories, except for system A in categories related to fresh water aquatic ecotoxicity and terrestrial ecotoxicity, which is related to contamination by inadequate disposal of hazardous material. Differences in the three ecotoxicity categories are explained by the large variations of characterisation factors of determined substances in the Life Cycle Inventory (e.g. Hydrogen fluoride has a $1\text{E}+7$ highest impact potential for marine toxicity than for fresh water toxicity).

On the other hand, due to the large uncertainties, no available method is recommended to address marine and terrestrial ecotoxicity (JRC, 2011) and no conclusion should be taken about those categories. The large amount of copper, zinc and nickel released into soil in the informal recycling contributes to the major impacts on the fresh water toxicity in baseline system.

Results indicate that recycling PCBs in Europe is more positive to the environment than recycling regionally, considering the assumption that Brazil would adopt a technology similar to the Chinese (Table 1). We have to consider that Brazil is still developing a national technology to recycle PCBs, and when this technology and national data is made available, those results may be changed.

These results also point out that the collection scheme with WEEE delivery only at EEE shops is environmentally better than the hybrid scheme with metro and neighbourhood stations. The sensitivity related to assumed distances was analysed for 50% variation, and there was no significant difference (<5%) to the final LCA results.

The qualitative assessment of social and economic categories, as it could be expected, presented significant variations in the evaluations of the experts. This is mainly due to subjectivity in the interpretation of system alternatives and impact categories, though the inquiry process searches to minimise this effect with clear explanations.

CPP was applied to integrate the criteria within each dimension, using the Choquet integral, and the minimum and maximum probabilities of preference for each alternative in each criterion were calculated. Table 3 presents the results of this round. In the economic dimension, C1 and C2 were the best alternatives (class 7 of 9); in the social dimension, B2 is the most recommended system.

Applying CPP to the environmental dimension, B1 is the best-ranked alternative. In the social dimension, B2 is the best alternative, and in the economic assessment, C1 and C2 (economic) are the highest ranked alternatives (Table 4). These four alternatives were selected to the next assessment stage, which is the integration of the three dimensions, adopting only the criteria that provoked a considerable distinction of the alternatives in each dimension (at least 0.25 between two alternatives). Results of this final stage are summarised in Table 4.

Table 3

Minimum probability of preference and classification of alternatives (in a 9-point scale) in the social and economic dimensions.

System	Economic		Social	
	Min.	Rank	Min.	Rank
A	0.04766	2	0.01798	5
B1	0.01580	6	0.07203	6
B2	0.05232	6	0.02254	7
C1	0.00770	7	0.01625	5
C2	0.06817	7	0.06190	5

Table 4
Final ranking of alternatives.

System	Maximum probabilities of preferences (shapley values of criteria)						Final rank (%)
	0.1621 Acidification	0.1546 Resource depletion	0.1885 System efficiency 2	0.1819 Innovation 2	0.1950 Social inclusion 2	0.1179 Health risks 2	
B1	18.60%	18.74%	13.82%	13.24%	12.88%	29.56%	21.73%
B2	36.04%	36.70%	12.84%	34.32%	24.89%	19.66%	33.41%
C1	16.10%	19.55%	33.80%	15.59%	21.27%	34.94%	29.04%
C2	29.27%	25.01%	39.54%	36.85%	40.95%	15.84%	38.36%

Adopting the CPP approach described in Section 3, C2 was considered the best WEEE management system to be implemented in Rio de Janeiro. This system consists of a hybrid WEEE collection system with delivery points at EEE shops, metro stations and neighbourhood centres, and comprehends recycling of all components (PCBs included) regionally in Brazil.

This solution is different than that proposed by Manhart (2011) for developing countries, which consisted of exporting PCBs to developed countries (more similar to B1 or C1). This can be explained by the different approaches adopted to assess the alternatives. Whilst the method by Manhart (2011) was based on the author's judgements in six decision criteria (Section 2.5), the current research adopted a framework fully based on stakeholder perspectives and experts evaluations, fully designed for the specific Brazilian context. On the other hand, B1 or C1 could be an intermediate step for WEEE management implementation in the country, before the installation of adequate technology to recycle PCBs.

The adopted methodology in this study can be considered adequate to the kind of problem that is under consideration – lack of data and few evaluators available. Positive points are the integration of LCA with qualitative evaluations, the assumption of interactions between criteria, the non-compensatory aggregation, and the presentation of results as probabilities of preference. A limitation of the approach is that $N > 3$ is a mathematical requirement for the equations, but does not guarantee as precise results as could be obtained with statistical inference and a larger sample of experts. This approach can be a feasible solution to similar decision-support problems with both quantitative data and qualitative assessment by a small sample of evaluators.

5. Conclusions

The present paper developed and applied an approach to prioritise alternative WEEE management systems in Brazil according to their sustainability performances, comprehending the integration of LCA with qualitative evaluations, and considering a small sample of evaluators. This multicriteria approach, which integrated LCA, CPP and qualitative measures, can be useful in cases where there is a lack of data and a small number of evaluators with knowledge of the problem. However, precision of results could only be ensured with the use of statistical inference, which requires a larger sample of respondents. So, it adds scientific value to waste management literature and practice by proposing a much needed method that integrates LCA, qualitative evaluation and MCDA, and that does not require a large number of evaluating experts.

Another contribution of this paper is suggesting the most adequate WEEE take-back system for the case of Rio de Janeiro, with basis on a sustainability assessment of available alternatives. The recommended solution is a hybrid collection system with delivery points at EEE shops, metro stations and neighbourhoods, integration of social enterprises and cooperatives in the pre-treatment processes, and adequate recycling of all components in the country. Achieving this scenario, however, may require a progressive implementation, in which a starting point could be organising

collection and pre-treatment phases and avoiding informality and landfilling, but temporarily exporting PCBs to developed countries with adequate technologies.

Further steps in this research are: to make qualitative evaluations with a large sample of both national and international experts, in order to allow for statistical inference; to apply different MCDA methods using the same collected data, in order to test their performances in such a decision problem; to collect primary data for more precise evaluations, both for the environmental assessment (LCA) and social and economic indicators. Another possibility is to integrate GIS with the methodology in search for logistics optimisation (routing and siting).

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References

- ABDI, 2012. *Logística Reversa de Equipamentos Eletroeletrônicos Análise de Viabilidade Técnica e Econômica*. ABDI, Brasília.
- ABINEE, 2015. Economic Panorama and Sectorial Performance – 2015. Brazilian Association of Electrical and Electronic Industry–ABINEE. Available in Portuguese. <<http://www.abinee.org.br/programas/50anos/public/panorama/#20/z>>.
- Al Sabbagh, M.K., Velis, C.A., Wilson, D.C., Cheeseman, C.R., 2012. Resource management performance in Bahrain: a systematic analysis of municipal waste management, secondary material flows and organizational aspects. *Waste Manage. Res.* 30 (8), 813–824. <http://dx.doi.org/10.1177/0734242X12441962>.
- Araújo, M.G., Magrini, A., Mahler, C.F., Bilitewski, B., 2012. A model for estimation of potential generation of waste electrical and electronic equipment in Brazil. *Waste Manage.* 32 (2), 335–342. <http://dx.doi.org/10.1016/j.wasman.2011.09.020>.
- Barboza, M.R., Gonçalves, R.F., D'Onofrio, E., Costa, I., 2014. Reverse logistics of information and communication technology equipment: a comparative assessment of laws and programs. In: IFIP WG 5.7 International Conference on Advances in Production Management Systems, APMS 2014, 439, 2, pp. 114–121. http://dx.doi.org/10.1007/978-3-662-44736-9_14.
- Bigum, M., Brogaard, L., Christensen, T.H., 2012. Metal recovery from high-grade WEEE: a life cycle assessment. *J. Hazard. Mater.* 207–208, 8–14. <http://dx.doi.org/10.1016/j.jhazmat.2011.10.001>.
- Brandmann, C., Altwater, M., 2012. Growing regulatory complexity as challenge – strategies for managing compliance with international waste regulations. Paper presented at the Electronics Goes Green 2012+, ECG 2012 – Joint International Conference and Exhibition, Proceedings.
- Brazil, 2015. Ministério do Meio Ambiente. Desafios e oportunidades da logística reversa como instrumento da política nacional de resíduos sólidos. In: 6o Forum Internacional de Resíduos Sólidos, São José dos Campos.
- Brazil, Presidência da República, 2010. National Solid Waste Policy (Política Nacional de Resíduos Sólidos – Lei No. 12.305/2010) (in Portuguese).
- Campos, T.R.T., Fonseca, M.V.A., Morais, R.M.N., 2014. Reverse logistics: a route that only makes sense when adopting a systemic vision. *WIT Trans. Ecol. Environ.* 180, 41–52. <http://dx.doi.org/10.2495/WIT140041>.
- Chatterjee, S., Kumar, K., 2009. Self sustained model of electronic waste management and recycling. *Electron. Inform. Plann.* 37 (3–4), 57–69. <www.scopus.com>.
- Cinelli, M., Coles, S.R., Kirwan, K., 2014. Analysis of the potentials of multi criteria decision analysis methods to conduct sustainability assessment. *Ecol. Ind.* 46, 138–148. <http://dx.doi.org/10.1016/j.ecolind.2014.06.011>.
- Classen, M., Althaus, H.-J., Blaser, S., Tuchscheid, M., Jungbluth, N., Doka, G., Faust Emmenegger, M., Sharnhorst, W., 2009. Life cycle inventory of metals. Final Report Data v2.1, No 10. EMPA dubendorf, Swiss Centre for Life Cycle Inventories, Dubendorf, CH.

- COMLURB, 2013. Caracterizações Gravimétrica e Bacteriológica de Resíduos Sólidos Domiciliares da Cidade do Rio de Janeiro – 2013. Rio de Janeiro.
- Doka, G., 2009. Life cycle inventories of waste treatment services. Ecoinvent Report No. 13. Swiss Centre for Life Cycle Inventories, Dübendorf.
- Eisel, H.A., Marianov, V., 2015. Location modeling for municipal solid waste facilities. *Comput. Oper. Res.* 62, 305–315. <http://dx.doi.org/10.1016/j.cor.2014.05.003>.
- Franco, R.G.F., Lange, L.C., 2011. Flow of e-waste at the city of belo horizonte, minas gerais, brazil. [Estimativa do fluxo dos resíduos de equipamentos eletrônicos: tricos e eletrônicos no município de Belo Horizonte, Minas Gerais, Brasil]. *Engenharia Sanitaria e Ambiental* 16 (1), 73–82. <http://dx.doi.org/10.1590/S1413-41522011000100011>.
- Gamberini, R., Gebennini, E., Manzini, R., Ziveri, A., 2010. On the integration of planning and environmental impact assessment for a WEEE transportation network—a case study. *Resour. Conserv. Recycl.* 54 (11), 937–951. <http://dx.doi.org/10.1016/j.resconrec.2010.02.001>.
- Glew, D., Stringer, L.C., McQueen-Mason, S., 2013. Achieving sustainable biomaterials by maximising waste recovery. *Waste Manage.* 33 (6), 1499–1508. <http://dx.doi.org/10.1016/j.wasman.2013.03.005>.
- Grabisch, M., Kojadinovic, I., Meyer, P., 2008. A review of methods for capacity identification in choquet integral based multi-attribute utility theory. Applications of the kappalab R package. *Eur. J. Oper. Res.* 186 (2), 766–785. <http://dx.doi.org/10.1016/j.ejor.2007.02.025>.
- Guarnieri, P., Sobreiro, V.A., Nagano, M.S., Marques Serrano, A.L., 2014. The challenge of selecting and evaluating third-party reverse logistics providers in a multicriteria perspective: a Brazilian case. *J. Cleaner Prod.* 96, 209–219. <http://dx.doi.org/10.1016/j.jclepro.2014.05.040>.
- Hamzeh, M., Ali, Abbaspour R., Davalou, R., 2015. Raster-based outranking method: a new approach for municipal solid waste landfill (MSW) siting. *Environ. Sci. Pollut. Res.* 22 (16), 12511–12524. <http://dx.doi.org/10.1007/s11356-015-4485-8>.
- Hirayama, D., Saron, C., 2015. Characterisation of recycled acrylonitrile-butadiene-styrene and high-impact polystyrene from waste computer equipment in Brazil. *Waste Manage. Res.* 33 (6), 543–549. <http://dx.doi.org/10.1177/0734242X15584845>.
- Hischier, R., Classen, M., Lehmann, M., Sharnhorst, W., 2007. Life Cycle inventories of electric and electronic equipment: production, use and disposal. Ecoinvent Report No. 18. EMPA/Technology & Society Lab, Swiss Centre for Life Cycle Inventories, Dübendorf, 2007.
- Hong, J., Shi, W., Wang, Y., Chen, W., Li, X., 2015. Life cycle assessment of electronic waste treatment. *Waste Manage.* 38, 357–365. <http://dx.doi.org/10.1016/j.wasman.2014.12.022>.
- Huisman, J., Delgado, C., Magalini, F., Kuehr, R., 2008. 2008 Review of Directive 2002/96 on Waste Electrical and Electronic Equipment (WEEE). United Nations University, Bonn.
- ISO, 2006. Environmental Management—Life Cycle Assessment – General Principles and Framework. ISO 14040. ISO, Geneva.
- Joint Research Centre, 2011. International Reference Life Cycle Data System (ILCD) Handbook – Recommendations for Life Cycle Impact Assessment in the European Context. Publications Office of the European Union, Luxembourg.
- Khalili, N.R., Ehrlich, D., 2013. A qualitative multi-criteria, multi stakeholder decision making tool for sustainable waste management. *Prog. Ind. Ecol.* 8 (1/2), 114.
- Kiddee, P., Naidu, R., Wong, M.H., 2013. Electronic waste management approaches: an overview. *Waste Manage.* 33 (5), 1237–1250. <http://dx.doi.org/10.1016/j.wasman.2013.01.006>.
- Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschil, M., Christensen, T.H., 2014a. Review of LCA studies of solid waste management systems – part I: lessons learned and perspectives. *Waste Manage.* 34 (3), 573–588. <http://dx.doi.org/10.1016/j.wasman.2013.10.045>.
- Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., et al., 2014b. Review of LCA studies of solid waste management systems – part II: methodological guidance for a better practice. *Waste Manage.* 34 (3), 589–606. <http://dx.doi.org/10.1016/j.wasman.2013.12.004>.
- Leme, M.M.V., Rocha, M.H., Lora, E.E.S., Venturini, O.J., Lopes, B.M., Ferreira, C.H., 2014. Techno-economic analysis and environmental impact assessment of energy recovery from municipal solid waste (MSW) in Brazil. *Resour. Conserv. Recycl.* 87, 8–20. <http://dx.doi.org/10.1016/j.resconrec.2014.03.003>.
- Liu, H.-C., Wu, J., Li, P., 2013. Assessment of health-care waste disposal methods using a VIKOR-based fuzzy multi-criteria decision making method. *Waste Manage.* 33 (12), 2744–2751. <http://dx.doi.org/10.1016/j.wasman.2013.08.006>.
- Lundgren, K., 2012. The global impact of e-waste addressing the challenge. International Labour Office, Programme on Safety and Health at Work and the Environment (SafeWork), Sectoral Activities Department (SECTOR), Geneva.
- Manhart, A., 2011. International cooperation for metal recycling from waste electrical and electronic equipment: an assessment of the “best-of-two-worlds” approach. *J. Ind. Ecol.* 15 (1), 13–30. <http://dx.doi.org/10.1111/j.1530-9290.2010.00307.x>.
- Mendes, M.R., Aramaki, T., Hanaki, K., 2003. Assessment of the environmental impact of management measures for the biodegradable fraction of municipal solid waste in São Paulo city. *Waste Manage.* 23 (5), 403–409. [http://dx.doi.org/10.1016/S0956-053X\(03\)00058-8](http://dx.doi.org/10.1016/S0956-053X(03)00058-8).
- Munda, G., 2008. Social Multi-Criteria Evaluation for a Sustainable Economy. Springer, Heidelberg.
- Nichols, A., Grose, J., Bennallick, M., Richardson, J., 2013. Sustainable healthcare waste management: a qualitative investigation of its feasibility within a county in the south west of England. *J. Infect. Prev.* 14 (2), 60–64. <http://dx.doi.org/10.1177/1757177412471411>.
- Oliveira, C.R.D., Bernardes, A.M., Gerbase, A.E., 2012. Collection and recycling of electronic scrap: a worldwide overview and comparison with the Brazilian situation. *Waste Manage.* 32 (8), 1592–1610. <http://dx.doi.org/10.1016/j.wasman.2012.04.003>.
- Ongondo, F.O., Williams, I.D., Cherrett, T.J., 2011. How are WEEE doing? A global review of the management of electrical and electronic wastes. *Waste Manage.* 31 (4), 714–730. <http://dx.doi.org/10.1016/j.wasman.2010.10.023>.
- Pimentel, M., Rocha, J., Filho, S.E., Rocha, T.B., Moraes, D., 2013. Ambientronic: a Brazilian program to support the development of innovative projects in e-waste reverse logistics. In: Paper Presented at the Proceedings of the 5th International Conference on Management of Emergent Digital EcoSystems, MEDES 2013, pp. 285–288. <http://dx.doi.org/10.1145/2536146.2536188>.
- Quariguasi Frota Neto, J., Van Wassenhove, L.N., 2013. Original equipment manufacturers' participation in take-back initiatives in Brazil: an analysis of engagement levels and obstacles quariguasi and van wassenhove OEM participation in take-back initiatives in Brazil. *J. Ind. Ecol.* 17 (2), 238–248. <http://dx.doi.org/10.1111/jiec.12019>.
- Reichert, G.A., Mendes, C.A.B., 2014. Life cycle assessment and decision making support in integrated and sustainable municipal solid waste management. [Avaliação do ciclo de vida e apoio à decisão em gerenciamento integrado e sustentável de resíduos sólidos urbanos]. *Engenharia Sanitaria e Ambiental* 19 (3), 301–313. <http://dx.doi.org/10.1590/S1413-41522014019000001145>.
- Rousis, K., Moustakas, K., Malamis, S., Papadopoulos, A., Loizidou, M., 2008. Multi-criteria analysis for the determination of the best WEEE management scenario in Cyprus. *Waste Manage.* 28 (10), 1941–1954. <http://dx.doi.org/10.1016/j.wasman.2007.12.001>.
- Rubin, R.S., Castro, M.A.S.D., Brandão, D., Schalch, V., Ometto, A.R., 2014. Utilization of life cycle assessment methodology to compare two strategies for recovery of copper from printed circuit board scrap. *J. Cleaner Prod.* 64, 297–305. <http://dx.doi.org/10.1016/j.jclepro.2013.07.051>.
- Saavedra, Y.M.B., & Ometto, A.R., 2012. E-waste in the Brazilian context. Paper presented at the Electronics Goes Green 2012+, ECG 2012 – Joint International Conference and Exhibition, Proceedings.
- Sant'Anna, L.T., Machado, R.T.M., De Brito, M.J., 2014. E-waste in Brazil and abroad: legal differences and the urgent need of a global standardization. [Os resíduos eletroeletrônicos no Brasil e no exterior: Diferenças legais e a premência de uma normatização mundial]. *Revista De Gestao Social e Ambiental* 8 (1), 37–53.
- Sant'Anna, A.P., 2015. Probabilistic Composition of Preferences, Theory and Applications. Springer, Heidelberg.
- Sao Paulo, 2015. RESOLUÇÃO SMA No 45, DE 23 DE JUNHO DE 2015: define as diretrizes para implementação e operacionalização da responsabilidade pósconsumo no Estado de São Paulo, e dá providências correlatas. Secretary of Environment, pp. 1–3.
- Soltani, A., Hewage, K., Reza, B., Sadiq, R., 2015. Multiple stakeholders in multi-criteria decision-making in the context of Municipal Solid Waste Management: a review. *Waste Manage.* 35, 318–328. <http://dx.doi.org/10.1016/j.wasman.2014.09.010>.
- Souza, R.G., Rosenhead, J., Lins, M.P.E., Valle, R.A.B., 2013. Multiple perspectives on waste management systems: towards more effective models for decision support. In: Proceedings Sardinia 2013, Fourteenth International Waste Management and Landfill Symposium.
- Souza, R.G., 2014. Modelling and Sustainability Assessment of Waste Management Systems Based on Multiple Stakeholders' Perspectives. PhD Thesis, Program of Production Engineering, Federal University of Rio de Janeiro.
- Souza, R.G., Rosenhead, J., Salhofer, S.P., Valle, R.A.B., Lins, M.P.E., 2015. Definition of sustainability impact categories based on stakeholder perspectives. *J. Cleaner Prod.* 105, 41–51. <http://dx.doi.org/10.1016/j.jclepro.2014.09.051>.
- StEP, 2015. E-Waste World Map. <www.step-initiative.org/step-e-waste-world-map.html> (accessed September 12th, 2015).
- SWICO, 2013. Technical Report 2013. <<http://www.swicorecycling.ch/downloads/dokumente/technical-report-swico-sens-slrs-2013.pdf/1400>>.
- Troschinetz, A.M., Mihelcic, J.R., 2009. Sustainable recycling of municipal solid waste in developing countries. *Waste Manage.* 29 (2), 915–923. <http://dx.doi.org/10.1016/j.wasman.2008.04.016>.
- Valle, R.A.B., Clímaco, J.C.N., 2015. A new tool to facilitate quantitative assessment of green activities – A trial application for Rio de Janeiro. *Technol. Forecast. Soc. Change* 98, 336–344. <http://dx.doi.org/10.1016/j.techfore.2015.03.000>.
- Velo, Z.M.F., 2015. Desafios e oportunidades da logística reversa como instrumento da política nacional de resíduos sólidos. In: 6° Forum Internacional de Resíduos Sólidos. São José dos Campos, 2015. <<http://www.6firs.institutoventuri.org.br/imagens/apresentacoes/Zilda.pdf>>.
- Wäger, P.A., Hischier, R., Eugster, M., 2011. Environmental impacts of the swiss collection and recovery systems for waste electrical and electronic equipment (WEEE): a follow-up. *Sci. Total Environ.* 409 (10), 1746–1756. <http://dx.doi.org/10.1016/j.scitotenv.2011.01.050>.
- Wibowo, S., Deng, H., 2015. Multi-criteria group decision making for evaluating the performance of e-waste recycling programs under uncertainty. *Waste Manage.* 40, 127–135. <http://dx.doi.org/10.1016/j.wasman.2015.02.035>.

3.4 Artigo 4 – Comparative lifecycle assessment of alternatives for waste management in Rio de Janeiro – Investigating the influence of an attributional or consequential approach *

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ABSTRACT

The environmental impacts from three management alternatives for organic fraction of municipal solid waste were compared using lifecycle assessment methodology. The alternatives (sanitary landfill, selective collection of organic waste for anaerobic digestion and anaerobic digestion after post-separation of organic waste) were modelled applying an attributional as well as consequential approach, in parallel with the aim of identifying if and how these approaches can affect results and conclusions. The marginal processes identified in the consequential modelling were in general associated with higher environmental impacts than average processes modelled with an attributional approach. As all investigated waste management alternatives result in net-substitution of energy and in some cases also materials, the consequential modelling resulted in lower absolute environmental impacts in five of the seven environmental impact categories assessed in the study. In three of these, the chosen modelling approach can alter the hierarchy between compared waste management alternatives. This indicates a risk of underestimating potential benefits from efficient energy recovery from waste when applying attributional modelling in contexts in which electricity provision historically has been dominated by technologies presenting rather low environmental impacts, but where projections point at increasing impacts from electricity provision in coming years. Thus, in the present case study, the chosen approach affects both absolute and relative results from the comparison. However, results were largely related to the processes identified as affected by investigated changes, and not merely the chosen modelling approach. The processes actually affected by future choices between different waste management alternatives are intrinsically uncertain. The study demonstrates the benefits of applying different assumptions regarding the processes affected by investigated choices – both for provision of energy and materials substituted by waste management processes in consequential LCA modelling, in order to present outcomes that are relevant as decision support within the waste management sector.

Keywords: Lifecycle assessment; Solid waste management; Municipal solid waste; CLCA; Consequential.



Comparative lifecycle assessment of alternatives for waste management in Rio de Janeiro – Investigating the influence of an attributional or consequential approach



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1. Introduction

Environmentally sound municipal solid waste (MSW) is a worldwide concern, but of particular priority in transition countries with large urban areas, dense populations and challenging geographical conditions. The per capita waste generation in countries like Brazil still is low (1.041 kg/capita and day in 2013) (Abrelpe, 2014), compared to Sweden and the US (1.263 and 1.995 kg/capita and day in 2013 respectively) (Waste Sweden, 2014; EPA, 2015). However, the total generation of MSW in Brazil in 2013 was estimated to almost 210,000 tons per day, an increase

by 4.1% compared to the year before, while population increased by 3.7% (Abrelpe, 2014). Although waste management in Brazil is focused primarily on eliminating waste disposal in open dumps, the newly implemented National Solid Waste Policy, (PNRS, 2010) opens the possibilities for technological leapfrogging and introduction of more advanced waste-to-energy and material recovery technologies, not necessary leading to disposal in sanitary landfills. The environmental benefits of such strategies could be investigated through use of lifecycle assessment (LCA) methodology.

According to Laurent et al. (2013), there is an evident confusion of concepts and terminology surrounding different types of Lifecycle Inventory (LCI) modelling frameworks in previous LCAs of solid waste systems. One of the reasons identified behind this is a lack of

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adequate goal definition, and subsequently, lacking stringency in the choice of LCI modelling framework. The selected type of LCI modelling framework has a significant influence on the definition of system boundaries (EC, 2010). Attributional lifecycle assessment (ALCA) employs a system-modelling approach in which the inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule (Sonnemann and Vigon, 2011). Consequential LCA (CLCA), on the other hand, is change-oriented and quantifies the effects associated with changes in the life cycle of a system brought about by a decision (Curran et al., 2005). In this way, the consequential approach seeks to take the environmental assessment a step further, in order to analyse how environmental burdens may vary in response to changes when processes are linked via market mechanisms beyond the foreground system (Vázquez-Rowe et al., 2013). Such linkages occur for example when waste is used for energy or material recovery, as waste-based goods are released on the market, with the effect of substituting other product-systems.

Several studies have been performed in recent years, using a consequential framework for LCI modelling of waste management systems. Boesch et al. (2014) compared an attributional and a consequential modelling of a Swiss MSW management scenario, considering substitution of both fossil and renewable energy to improve consequential LCI (CLCI) modelling, referring to recommendations by Mathiesen et al. (2009). No considerations were taken to affected suppliers in the modelling of material recovery, and average data was used for these processes in the study. Seigné-Itoiz et al. (2015) present a CLCA of different alternatives for end-of-life treatment of plastic waste. The identification of marginal electricity was based on projections from national governments and the IEA. Assumptions of substituted heat were not justified and average data was used for modelling of production of virgin plastic substituted through material recycling. Comparing ALCA and CLCA modelling of end-of-life treatment of construction materials, Sandin et al. (2014) refer to literature data in identifying marginal heat provision processes, while data representing current average technologies is used for modelling of substituted materials. The influence of technology development was evaluated through 'explorative' scenarios, investigating the possible range of future outcomes by assuming a single technology to be representative for average future technology. Tonini and Astrup (2012) chose to investigate both short- and long-term changes related to the introduction of a waste refinery system, using CLCI modelling. Average processes were used in the modelling of virgin production of metals, plastics and glass as well as inorganic fertilizers substituted by recycled waste materials and organic fertilizers respectively. Cimpan et al. (2015) consider substitution of both energy and virgin material in a CLCA of a Danish waste management system. While the affected suppliers of virgin metals, plastics and glass not could be identified, marginal nitrogen, phosphorus and potassium fertilizers were identified as calcium ammonium nitrate, diammonium phosphate and potassium chloride, based on Hamelin (2013). In a review of Danish and Swedish consequential LCAs of waste management systems, Mathiesen et al. (2009) demonstrated the wide application of the methodology suggested by Ekvall and Weidema (2004) for identification of affected electricity provision. However, the same authors also stated that reviewed studies did not apply the suggested methodology consistently, and that the current practice should be improved by 1) using combined affected technologies, i.e. a complex set of marginal technologies; 2) using long-term perspectives by identifying affected technologies in several possible future scenarios, and 3) identifying the affected technologies based on energy system analysis with realistic geographical distributions and distribution in time.

The aim of the present paper was to investigate different waste management alternatives under two different LCI-modelling approaches. The objective to discussed if and why different treatment alternatives come across as more beneficial than others, depending on the chosen approach, and how this can affect input given to decision makers. In the present study, the environmental performances of three different scenarios for future development of the municipal solid waste management (MSWM) system in the city of Rio de Janeiro are assessed with an attributional as well as a consequential LCI-modelling approach. The functional unit is set to the collection and treatment of 1 ton of domestic waste, as collected by the municipal waste collection agent.

Energy provision is the background system of principal relevance to the system studied in the present study. MSW management is also increasingly integrated with material provision systems through material recovery and recycling, where environmental benefits can arise through avoidance of virgin material provision. Once a process avoided by waste-to-energy and material recycling or recovery is identified, the affected technology, also referred as marginal (Weidema et al., 2009), should be identified. This is commonly a key issue in consequential LCA modelling. A specific objective is therefore also to compare two different approaches for identification of the processes and technologies affected by the changes investigated in the consequential assessment of environmental impacts.

2. Methodology

Two different approaches were applied for identification of processes and technologies affected by the changes investigated in the present study. Firstly, the methodology suggested by Weidema et al. (2009) is used in a consistent manner for identification of single marginal technologies for production of electricity. This analysis is compared with the official national energy balance plan. Affected technologies and processes related to the handling of biofertilizers from waste management were identified through the approach presented by Weidema et al. (2009), using two different approaches for market delimitation. In the attributional modelling, average data was used for modelling of electricity and fertilizer production. As system enlargement was applied also in the attributional modelling, and factors such as rebound effects were not addressed in the consequential modelling, the largest difference between the two approaches lies within the identification of the unit processes linked to each system.

2.1. Scenario description

The selected functional unit is 1000 kg of mixed municipal solid waste, with the composition presented by the local government (COMLURB, 2014). The fraction of organics in domestic waste in the city of Rio de Janeiro has decreased from 60.7 to 52.8% over the period 2005–2013. However, over the same period, the amount of domestic waste increased by 25%, resulting in an overall increase of domestic organic waste by 8.5% (COMLURB, 2014). Thus, handling of this fraction is of large importance, and was also selected to be the focus of the present study. The scenarios investigated in the study are presented in Table 1.

2.2. LCA study methodology

Exclusively covering the MSW end-of-life phase, this LCA study was conducted following the international standards ISO 14040 and 14044 (ISO, 2006a and b). All scenarios were modelled in the EASETECH software, developed by DTU. This software is based on material flow modelling and enables generation of input- and

Table 1
Scenarios compared in the present study.

Scenario	Description
S1	0% separate collection of organics. Landfilling of all MSW with collection of landfill gas and combustion of the same for generation of electricity (current situation)
S2	Separate collection of 50% organics for anaerobic digestion. Combustion of generated biogas in engine generating electricity. Landfilling of residual waste (including non-separated organics). Collection and combustion of landfill gas for generation of electricity. Use of digestate as fertilizer on farmland
S3	Common collection of all MSW and post-separation of organics (75% of total) in material recover facility (MRF). Anaerobic digestion of mechanically separated organic waste. Combustion of generated biogas in engine generating electricity. Landfilling of residual waste and of digestate produced in AD. Collection and combustion of landfill gas for generation of electricity

process-specific results. Lifecycle impact assessment (LCIA) categories and methods recommended by ILCD (2011) were used. These categories were: Climate change (as Global Warming Potential, GWP), Ozone Depletion Potential (ODP), Particulate matter (PM), Photochemical Oxidant Formation (POF), Eutrophication marine (EU-marine), Acidification, Human Toxicity cancer effects (HT-carc and non-carc). Due to problems of importing some ecoinvent v3 datasets to the EASETECH software, modelling of production of fertilizers was made in Simapro. LCIA methods used in

these two softwares were compatible in all cases, exempt for ecotoxicity. This impact category was therefore excluded from the assessment.

2.3. Lifecycle inventory data

Current and potential future systems for MSW management in Rio de Janeiro city were modelled, including collection, pre-treatment (when relevant), final treatment and further handling of by-products/secondary waste. The composition of domestic waste was based on official statistics, presented by COMLURB (2014). According to this data, the fraction of biowaste, plastic, paper and metal waste reached 52.8, 19.0, 16.8 and 1.6% respectively in 2013. The fraction of other combustible waste (leather, wood, textiles, etc.) and inserts reached 4.9% and 4.5% respectively the same year (COMLURB, 2014). The relation between animal and vegetable food waste was set to 12 and 88% respectively based on Bernstad and Andersson (2015). Characteristics of waste fractions were based on Riber et al. (2009). Diesel use in collection and transport to the treatment site was assumed to 1.30 L/t wet waste (ww) (Angelo et al., 2017). The same amount of diesel per ton ww was assumed in all scenarios, as two-compartment trucks, simultaneously collecting organic waste and residual waste, assumedly would perform source-separation. Data used in the modelling of waste handling is presented in Tables 2 and 3.

Table 2
Parameters used in modelling of landfill and anaerobic digestion. LFG = Landfill gas.

Parameter	Value	Unit	Reference
<i>Landfill</i>			
Years of utility	18	Years	Abrelpe (2012)
Leachate collection (year 1–30)	90	%	Assumed
Leachate collection (year 31–100)	50	%	Assumed
Methane oxidation in top cover	10	%	Assumed
Density	1	t/m ³	Assumed
Height	50	M	Abrelpe (2012)
Diesel use (operation)	0.003	kg/t waste	da Silva (2011)
Diesel use (construction)	3.4E-06	kg/t waste	da Silva (2011)
Concrete (construction)	0.00011	kg/t waste	da Silva (2011)
High density polyethylene (HDPE) (construction)	0.003	kg/t waste	da Silva (2011)
Gas collection (year 1–50)	50	%	Abrelpe (2012)
Gas collection (year 51–100)	0	%	Abrelpe (2012)
Flaring of collected LFG	40	%	Abrelpe (2012)
Use of electricity in leachate treatment	0.443	kW h/t waste	Yoshida et al. (2014)
Electricity recovery (LFG)	36	%	Genrup and Jonshagen (2011)
<i>Anaerobic digestion</i>			
Electricity use in pre-treatment	12.6	kW h/t ww	Waste Management Sweden (2013)
Electricity use in dry AD	14	kW h/t ww	DWMT (2014)
Methane emissions in dry AD	0.5%	% of CH ₄	DWMT (2014)
DS in digestate (prior to dewatering)	35	%	DWMT (2014)
Electricity use in dewatering (digestate)	35	kW h/t ww	Balmér (2012)
DS in digestate (post dewatering)	60	%	Balmér (2012)
Transport of digestate	0.007	L diesel/km, t	Angelo et al. (2017)
Electricity recovery (biogas)	36	%	Genrup and Jonshagen (2011)

Table 3
Nutrient content in generated digestate and parameters used in modelling of on-land application.

Parameter	Value	Unit	Reference
N ₂ O-N emissions (direct) ^a	1.25	% of N-tot	IPCC (2006)
NH ₃ -N emissions ^a	15	% of N-tot	OSB/WSTB (2000)
NO ₃ -N emissions ^a	20	% of N-tot	OSB/WSTB (2000)
Incoming N content ^b	4.85	kg N-tot/ton ww	Based on modelling
Incoming P content ^b	0.65	kg P-tot/ton ww	Based on modelling
Incoming K content ^b	1.48	kg K-tot/ton ww	Based on modelling
Application of digestate	20	MJ/t digestate	Borgshed et al. (2003)
Application of mineral fertilizers	0.36	MJ/kg N-tot	Finnveden et al. (2000)

^a Digestate and mineral fertilizers on farmland.

^b In food waste only, as kg per ton mixed incoming waste.

Carbon storage in landfill as well as in soil after application of digestate on farmland (in scenario 3), was assumed to zero in the based case of the present study. According to [Smith et al. \(2001\)](#), the sequestration potential is strongly dependent on the type of organic material (i.e. the amount of resistant organic matter such as lignin) and climate (i.e. increases in cold climates, and decreases in warm and moist climates). The fraction of biodegradable carbon in food waste which is not mineralized over the investigated time-frame (in this case limited to 100-years) was estimated to 11% by [Smith et al. \(2001\)](#), representative for northern European conditions. However, as these estimations are made for conditions very different to the ones in tropical moist areas, carbon sequestration was excluded in the base case of the study. The potential influence of considering carbon sequestration from landfills as well as from digestate spread on farm land was however investigated in a sensitivity analysis.

In the attributional approach, national and global averages, based on historical data were used. Electricity provision was modelled as the average Brazilian electricity mix (based the process for medium voltage electricity from grid in ecoinvent 3.0. For details, see [Table 1 SI](#)). However, emissions of GHG were adjusted to represent official statistics of the GHG-emissions from the national electricity system over the period 2010–2014, based on national statistics, resulting in 0.07547t CO₂-eq/MW h ([MME, 2015](#)). This process is referred to as “average mix” in the following. Global average data for fertilizer provision (Nitrogen fertilizer (as N), Phosphate fertilizer (as P₂O₅) and Potassium (as K₂O), from the ecoinvent 3.0), were also applied.

For the consequential modelling of the system, identification of marginal processes was made based on two different approaches, explained in detail in the below.

2.4. Identification and modelling of affected processes

[Laurent et al. \(2013\)](#) presented a guidance to facilitate identification of the most adequate LCI-modelling framework in a specific situation. According to the guidance, LCAs generated as basis for decision-making should be modelled as a mix of long-term marginal processes when large-scale consequences on the background system can be expected. In a Brazilian perspective, waste-to-energy is still rare. However, recent changes in the national waste management legislation firmly point at the necessity of increasing the disposal of solid waste in sanitary landfills with energy recovery, as well as source-segregating organic waste in order to achieve recovery of nutrients ([IPEA, 2014](#)). In Brazil, with a generation of almost 210,000 tons of MSW per day ([Abrelpe, 2014](#)), changes in the current management could imply large-scale consequences on background systems in the local/regional context. Due to the lack of transfer-capacity.

2.4.1. Affected technologies for Brazilian electricity provision – identification based on method presented by [Weidema et al. \(2009\)](#)

According to the methodology suggested by [Weidema et al. \(2009\)](#), the procedure for identification of which unit processes to link, has four steps:

- Identifying the scale and time horizon of the potential change studied
- Identifying the limits of a market
- Identifying trends in the volume of a market
- Identifying changes in supply and demand

Detailed information about how these steps were handled is presented in the [SI](#), while a summary is presented in the below.

The changes investigated in the present study are considered small (marginal) in a national context, implying that it does not

affect the determining parameters of the overall market situation, and studied changes are assumed to be long-term, with potential installation of new capacity. According to national statistics ([ONS, 2015](#)), the electricity demand in Brazil has increased by more than 20% over the period 2009–2015. National projections point at a continuing increase in electricity demand ([PNE, 2014](#)). As the majority of the Brazilian biomass fueled power plants are dependent on waste material from the sugarcane industry, the expansion of bioenergy is constrained by developments in sugar and ethanol demand. Lack of investments in transmission capacity from north to south could constrain both hydro and wind power installations. There are plans of new transmission capacity, to be taken into use from the second part of 2021 ([ONS, 2015](#)). However, even if these plans are realized, they are insufficient in relation to the needs. The government projects a necessary increase in generation capacity placed in the south/southeastern parts of the country ([PDE, 2014](#)). This capacity must be ready to be started and generate electricity within a few hours after notice ([PDE, 2014](#)).

Results from the latest national electricity auctions show that large hydropower followed by wind power are the most competitive suppliers, delivering at 155 and 179 BRL/MW h respectively ([EPE, 2015](#)). However, due to constraints presented above, it was assumed that the single marginal technology must be flexible in terms of location and operation. Several potentially relevant fuels (coal, oil derivatives and natural gas) can be found within the Brazilian territory. However, the majority of the Brazilian coal reserve has low quality, with high content of ash and sulfur, and low energy content ([ANEEL, 2009](#)). Vast increase in diesel use is limited by the lack of national refinery capacity. Important reserves of natural gas have been confirmed on Brazilian territory. In December 2014, the national production of natural gas reached 98.15 Mm³/day ([ANP, 2015](#)), an increase of the internal production by 7–10% from 2012 to 2014, while the national consumption increased by 22% in 2014 ([ANP, 2015](#)). The import of natural gas from Bolivia, with a capacity of 30 Mm³/day is therefore vital for the Brazilian market and prolonged and potentially increased transfer capacities are discussed ([Portal Brasil, 2014](#)). The last years have demonstrated a political will to decrease the dependency of the import of natural gas from Bolivia, resulting in investments in import of liquid natural gas (LNG) to the southeastern parts of the country. This was first seen as a temporary measure in the light of the draught in 2012, resulting in reduced hydropower generation, but renewed import licenses has increased the possibilities of this import becoming a stabile input to the Brazilian electricity matrix. The major part (over 75%) of imported LNG in 2012 originated from Qatar, Nigeria and Trinidad and Tobago and the end use is primarily generation of electricity ([ANP, 2015](#)). The cost for this electricity is thereby high, making it likely that new construction of generation capacity would decrease the demand, and at the current, natural gas appears to be the most likely fuel for marginal electricity production, seen in a mid-term perspective.

2.4.2. Affected technologies for Brazilian electricity provision – identification based on national plans and projections

In 2014, the Brazilian government presented a ten-year plan for national energy expansion ([EPE/MME, 2014](#)). The plan is built on expectations of a yearly GDP growth of 3.2% and a population increase of 0.7% per year. The structure of the national economy is expected to be the same as in the year 2014, i.e. dominated by services (67.1%), followed by industry (27.3%) and animal agriculture (5.5%). An increased demand of electricity is assumed to increase the installed electricity production capacity by 56% between the years 2015 and 2024, equal to 74,600 MW over the same period. The largest increase is assumed to take place in the northern part of the country. Based on given permits and existing plans for installation of new generation capacity, a profile of the

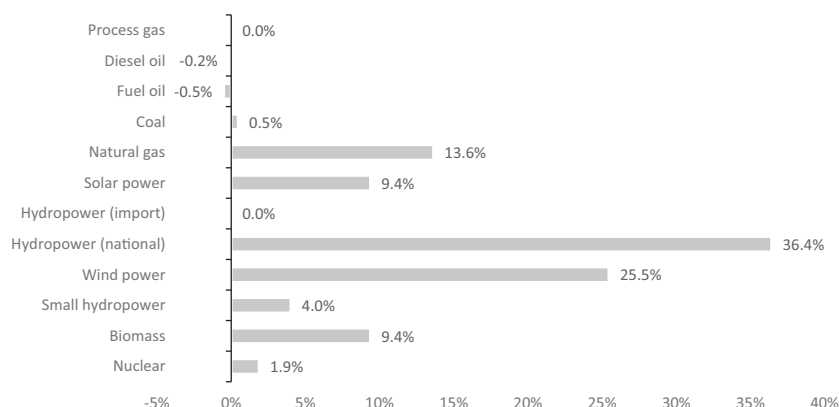


Fig. 1. Changes in Brazilian installed electricity generation capacity over the coming ten years (PDE, 2014). Values add up to 100%, representing the total increase in installed capacity (74,600 MW) over the period 2015–2024.

changes in electricity generation capacity over the coming ten years could be established (Fig. 1).

2.4.3. Affected technologies for production of fertilizers

Brazil consumes around 32.2 Mt of mineral fertilizers each year (ANDA, 2015). In 2014, the use of nitrogen, phosphorus and potassium fertilizers increased by 4.7%, 2.4%, and 5.9% respectively (ANDA, 2015). Only around 30% of the demand is produced internally, and several national strategies have been developed to decrease the country's dependency on imports. An increasing demand has been followed by increasing prices. The national costs for import of fertilizers to Brazil hit a new record in 2015 (ANDA, 2015).

Due to the rather low concentration of nutrients in digestate from organic MSW, the transport distance must be kept to a minimum. It would thereby not be relevant to assume application of digestate on crops not cultivated in the absolute proximity of the anaerobic digestion plant. Thus, in order to identify the types of fertilizers affected by the generation of digestate in the case-study area, the dominating crops in the region were identified as tomato, mandioc and banana (IBGE, 2013).

An assessment of the types of fertilizers commonly used for these crops shows that urea is the most relevant choice of marginal mineral nitrogen fertilizer. Amongst potassium fertilizers, previous studies have shown that potassium chloride is the dominant product, covering 98% of the Brazilian demand (FAO, 2002). In the case of phosphate fertilizers, simple superphosphate was the only phosphate fertilizer used in both mandioc and banana production, and thus, assumed to be the marginal one. In the modelling of marginal fertilizers, adjustment of average global datasets were made to represent Brazilian marginal conditions in terms of electricity use in the fertilizer production chain, using the two approaches for identification of marginal electricity production presented above.

Substitution of composted green waste by produced digestate was assessed in parallel to the substitution of marginal mineral fertilizers. Emissions from low-tech composting of organic waste and application of the same on farmland would in this case be avoided. If no alternative use is found for the waste previously composted, it is likely that this will degrade naturally, partly anaerobically, or be subject to open-air burning. Emissions from open-air burning of vegetation were gathered from Yokelson et al. (2008), and emissions from composting were based on default values presented by IPCC (2006) in the case of CH₄ and N₂O, and Shen et al. (2011) in the case of NH₃. Emissions from on-land application of digestate and compost were assumed to be the same as in the case of digestate, based on Nordic Council of Minister (2007). Substitution of compost was made on nutrient content basis, assuming nitrogen

as the determining macronutrient, using average data presented by Andersen et al. (2011) for nitrogen content in compost. This resulted in a substitution ratio of 1.73 tons of green waste per ton organic waste treated by anaerobic digestion.

3. Results

The two methods used for identification of marginal electricity production generated results with largely different environmental impacts (Fig. 2). While emissions related to GWP, OD, POF, A and E (marine) are higher from electricity based on combustion of natural gas in power plants, the complex marginal mix assessed through official projections generate higher impacts in terms of HT (canc. and non-canc.) and PM. Impacts to HT and PM are primarily related to combustion of biomass.

The emission profiles from electricity production presented above were applied in modelling of the three waste treatment scenarios presented in Table 1, resulting in the impacts presented in Fig. 3 (assuming that production of mineral fertilizers are affected by production of digestate) and 4 (assuming that composting is affected by production of digestate). Results are presented in relation to each of the three waste treatment scenarios (S1 (current situation), S2 (source-segregation of 50% of organics for biogas) and S3 (post-sorting of organics for biogas)) under three different assumptions for electricity provision (current average – used in attributional modelling, combustion of natural gas – used in consequential modelling (C1), and marginal mix based on the national energy plan – used in consequential modelling (C2)).

Results show that scenario 3 (anaerobic digestion of organic waste after post-separation) is the preferable alternative in relation to most of the included impact categories when using the attributional approach as well as the consequential approach, assuming that digestate will substitute mineral fertilizers. However, when assuming that digestate will substitute green compost and that natural gas is the affected technology for electricity provision, scenario 2 is preferable in relation to the largest number of impact categories. The performances in each impact category can be explained as following:

GWP – Unrecovered methane from landfills dominate emissions, and thus, emissions increase with increasing amount of landfilled organic waste, while substitution of electricity and fertilizers reduce emissions, making S2 the preferred scenario when applying a consequential modelling, as the benefits from substitution of marginal fertilizers are higher in case, while S3 is the preferred alternative when attributional modelling is applied (Fig. 3). Assuming that composting is substituted by digested produced in S2 result in lower avoidance of GWP compared to

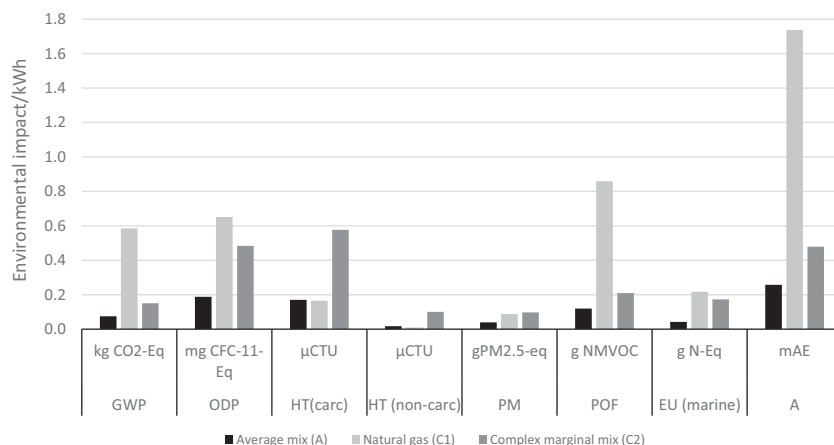


Fig. 2. Comparison between the average mix (A), based on currently available LCI data combined with most recent average data on GHG-emissions from national electricity generation, and the modelling of the two marginal electricity generation capacities identified in the study (C1 and C2).

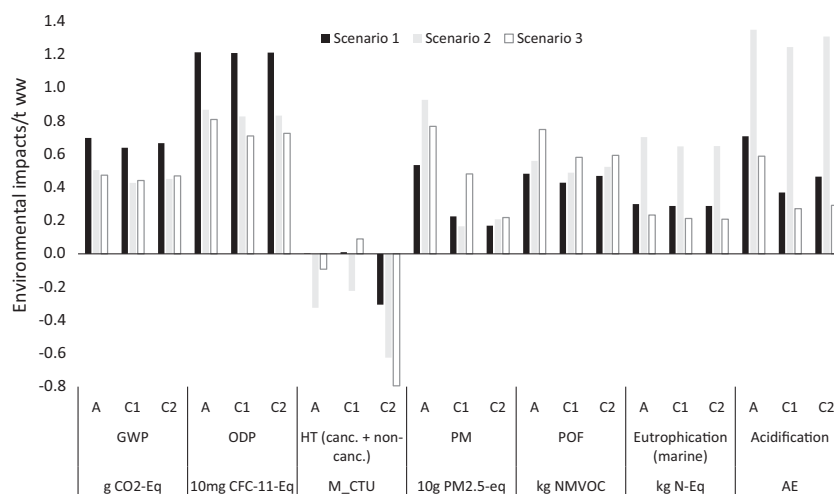


Fig. 3. Results from modelling of three different treatment alternatives for domestic waste (S1-S3) with attributional (A) and consequential modelling (marginal electricity provided through natural gas (C1) or complex marginal mix (C2), in all cases assuming that digestate substitutes mineral fertilizers. Negative emissions signify that net-emissions from scenarios are negative.

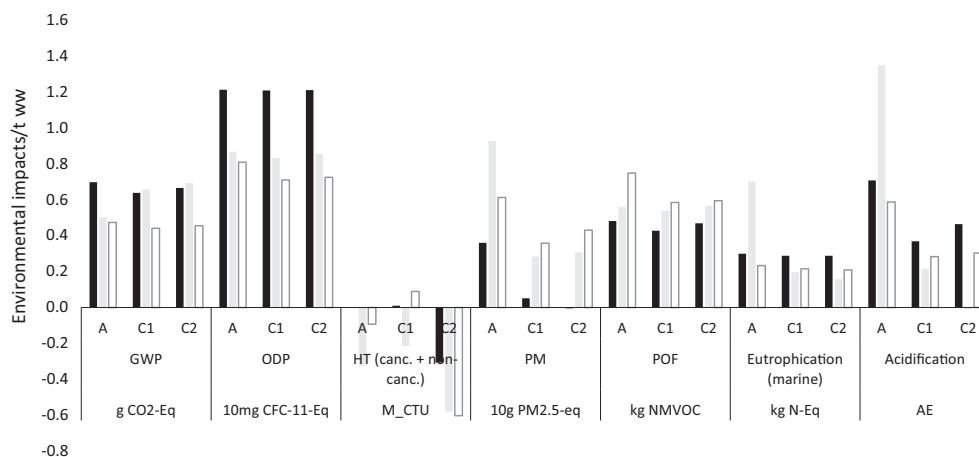


Fig. 4. Results from modelling of three different treatment alternatives for domestic waste (S1-S3) with attributional (A) and consequential modelling (marginal electricity provided through natural gas (C1) or complex marginal mix (C2), in all cases assuming that digestate substitutes compost. Negative emissions signify that net-emissions from scenarios are negative.

the substitution of mineral fertilizers (Fig. 4). This is an effect of the relatively low net emission from the composting process as well as from the burning of waste in open air when compost is substituted.

ODP – Also here, landfill processes dominate emissions. Emissions are slightly higher in the attributional modelling, mainly due to a lower reduction of ODP per recovered unit of electricity

generated in the different scenarios. Thus, reduction of ODP is higher in S2 and S3 where more electricity is recovered per ton waste.

HT – Substitution of the complex marginal energy results in large avoidance of toxic emissions, primarily as a result of the need for some of the elements used in construction/operation of solar, wind and nuclear power. Thus, with the C2 approach, S3 is more beneficial as this scenario results in the highest electricity recovery. Due to a much lower toxicity impact per kWh produced electricity with the C1 and attributional approach, substitution of fertilizers plays a more important role, making S3 the most beneficial alternative. The result is a change in the hierarchy between compared scenarios when comparing attributional and consequential modelling approaches.

PM – Substitution of marginal electricity result in larger reduction of PM compared to substitution of average electricity. As this is reflected also in provision of fertilizers, reduction of particulate matter is largest from S2. The result is a change in the hierarchy between compared scenarios when comparing the attributional and the two consequential modelling approaches.

POF – Emissions are primarily influenced by combustion of produced biogas and transports. Substitution of marginal electricity reduces impacts, but other processes are more relevant for overall results.

E – Eutrophication is to a large extent related to emissions from use of organic fertilizers (digestate) on farmland, and contribution is therefore higher from S2 (Fig. 3). However, when assuming that digestate substitutes composting, emissions resulting in eutrophication are avoided (Fig. 4). This is largely related to the avoided emissions of ammonia from the composting process. These are rather high, as an effect of assumedly high temperatures reached during the composting process, based on previously established correlation between temperature and ammonia emissions (Beck-Friis et al., 2001). Thus, the modelling approach (attributional or consequential) can have a large impact on overall results in relation to this impact category, depending on the identification of affected processes.

A – Also acidification is strongly related to emissions from farmland as well as from composting. Thus, similarly to the case of eutrophication, the identification of affected processes in consequential modelling will determine to what extent the modelling approach is decisive to results within this impact category.

3.1. Sensitivity analyses

In order to investigate the impact of some of the many assumptions made in the consequential modelling of the processes substituted by goods provided by compared waste management alternatives, a number of sensitivity analyses were performed.

The differences in environmental impacts from production and use of urea and nitrate based fertilizers respectively can be vast (Harrison and Webb, 2001). Thus, in a second sensitivity analysis, calcium ammonia nitrate (CAN), was assumed to be the marginal nitrogen fertilizer. Global average data for provision of CAN was used in the modelling (Ecoinvent, 2013), containing, amongst others data for production of nitric acid. Production of nitric acid can lead to large emissions of greenhouse gases due to production of nitrous oxide as a waste product. Catalytic cleaning of nitrous oxide can however reduce nitrous oxide emissions by 70–90% (Jenssen and Kongshaug, 2003). This technology is currently introduced in production plants primarily in Northern Europe (Yara, 2015). Catalytic cleaning has also been introduced on Brazilian production plants (UNFCCC, 2006). The effects of such technology developments, resulting in a reduction of emissions of N_2O in ammonia nitrate production by 80%, were assessed. The choice of nitrogen fertilizer also affects emissions from on-land application of nitrogen fertilizers. Emission factors considered for estimating NH_3 from urea and ammonium nitrates, as well as CO_2 releases from urea application on farmland, were based on Hamelin et al. (2012). These adjustments would have an effect principally on the environmental impact categories climate change and eutrophication. As presented in Fig. 5, substitution of CAN would reduce contribution to GWP by 3–5% and contribution to eutrophication by 10% in S2, when compared to the assumption that urea is the marginal fertilizer. On the other hand, technology development of nitric acid production, and thus reduced emissions of N_2O from provision of CAN, would increase contribution to GWP by 2–4% compared to the base case, causing a change in the hierarchy between compared scenarios in relation to the impact category climate change.

In countries where domestic household waste is segregated at source for anaerobic digestion, nutrient recycling (with focus on phosphorus) is one of the main motivations (SEPA, 2016). As discussed above, the Brazilian consumption of phosphorus has increased vastly in later years, and further increase is expected (DNPM, 2014). According to the Brazilian National Department for Mineral Production (DNPM), the national reserves of P_2O_5 were estimated to 315 million tons in 2015, and several projects are currently under development to increase the Brazilian phosphorus production (DNPM, 2014). The cadmium content of phosphate rock varies vastly, between lower levels in igneous rocks (up to 0.25 mg Cd/kg rock) to higher levels in sedimentary rocks (up to 2.60 mg Cd/kg rock) (Smolder and Six, 2014). In later years, Brazil has reduced its import of highly cadmium containing sedimentary phosphorus fertilizers from countries such as Marrocos (DNPM, 2014). It could thereby be argued that recycled phosphorus in digestate in a Brazilian context will substitute fertilizers with a lower cadmium content compared to current global average. How-

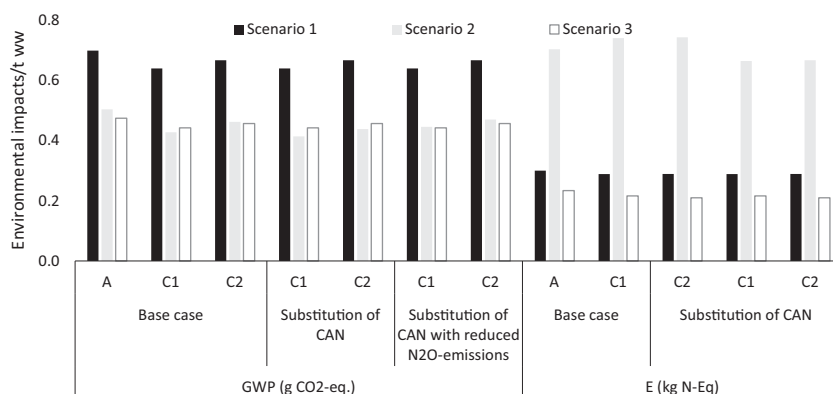


Fig. 5. Results from sensitivity analyses related to fertilizer substitution, change of assumed marginal mineral nitrogen fertilizer.

ever, the global trend in phosphate fertilizers is rather an increase in cadmium content (Cichy et al., 2014). Thus, assuming that the mineral fertilizer market is global, it is more likely that digestate will substitute phosphorous fertilizers with higher cadmium content than current global average. The impact of a lower (a reduction of 90% was assumed, based on average difference in cadmium content between sedimentary and igneous phosphorus) as well as higher cadmium content (an increase by 20% was assumed) compared to the average was assessed in a sensitivity analyses. A reduced amount of cadmium in fertilizers could reduce avoidance of toxic emissions seen in S2 by 5–25% (C2 and C1 respectively), while an increase by 20% would increase the avoidance of toxic emissions by 1–13% (C2 and C1 respectively) (Fig. 6). Neither the levels of increase nor decrease in the content of cadmium in fertilizers investigated in the sensitivity analyses would result in any changes of the hierarchy between investigated alternatives.

When assuming that the affected electricity is represented by the national energy plan (C2), variations of cadmium in fertilizers will have a low influence. This is explained by the use of cadmium in solar panels. As solar power will correspond to as much as 9.4% of overall changes in electricity production capacity until 2024, the commonly applied cadmium telluride (CdTe) technology, using water soluble and highly toxic cadmium chloride (CdCl_2), for production of solar panels will be dominant for the emissions related to human toxicity. However, use of magnesium chloride (MgCl_2) might substitute cadmium and thus vastly decrease toxicological impacts related to solar power (Major et al., 2014). The effect from such technological development was investigated, assuming that toxicological impacts would be reduced by 44% when MgCl_2 substitutes CdTe (Werner et al., 2011). Results state that this would decrease avoidance of human toxicity impacts by between 3% and 6% from C2 scenarios (see SI for details).

No consideration was taken to potential sequestration of biogenic carbon in landfills and soils where digestate was used as fertilizer. In a sensitivity analysis, the influence of assuming the levels of carbon sequestration assumed by Smith et al. (2001) for European condition (presented in Section 2.3) was investigated. Results show that inclusion of carbon sequestration would reduce greenhouse gas emissions by between 2% and 5% (see SI for details). The influence from carbon sequestration is relatively seen small when compared to results presented by Smith et al. (2001) and Manfredi and Christensen (2009). This is mainly a result of emissions from landfills dominating the overall greenhouse gas emissions in the present study, due to a low landfill gas recovery ratio. The effects of a prolonged lifetime of landfills when food waste is segregated were not assessed in the base case of the present study. The landfill used as model in the present study has an

expected lifetime of 18 years (Abrelpe, 2012). Scenarios S2 and S3 would increase the lifetime by 4.2 and 5.4 years respectively. As the next landfill constructed in the Rio de Janeiro City area will be placed even further away from the city centre, one of the effects of the diversion organic waste is avoided emissions from transports of waste over a longer distance for several years. In order to quantify the environmental impacts of this, the future location of the next landfill to which waste would be transported is necessary. However, in a sensitivity analysis, it was assumed that the transport distance would increase by 50%. The largest effect of such a change is related to emissions of particulate matter, increasing by 85 and 102% in C1 and C2 modelling respectively in case of no diversion of food waste (S1), while emissions would increase by less than 20% in S3 with the largest diversion of food waste from landfills (see SI for details).

4. Discussion

Results from the case study show that a change from the current waste management strategy, focused on sanitary landfills, to post-separation of organic fraction food waste could decrease GHG-emissions from waste management by more than 30% when applying an attributional modelling. However, with a consequential modelling, assuming substitution of electricity generated from natural gas and marginal mineral fertilizers through source-segregation of organic waste, this strategy appears more beneficial in relation to GHG-emissions. Thus, results suggest that the chosen methodological approach could have a large effect on the relative performance of compared treatment alternatives. This differs from previous conclusions presented by Sandin et al. (2014), stating that the choice of methodological approach does not seem to influence the relative performance of different end-of-life treatment alternatives. However, large differences in absolute emissions from the two modelling approaches were identified by the same authors, as well as in the present study.

The two approaches used for identification of marginal electricity would in both cases increase emissions related to all assessed environmental impact categories, except human toxicity, when compared to average data used in attributional modelling. As more electricity is produced than consumed in all compared scenarios, consequential modelling will in most cases results in higher avoidance of environmental impacts, compared to attributional. This is to some extent counter-acted by electricity used in the waste treatment process. Thus, the differences between the compared approaches might appear smaller than expected.

Findings in the present paper partly follows the ones presented by Lund et al. (2011), showing that a complex marginal electricity mix renders lower environmental impacts compared to the commonly used assumption of natural gas being the marginal fuel for electricity provision. However, as the present study is not limited merely to an assessment of GHG-emissions, it shows that a complex marginal electricity mix can generate higher impacts than the single marginal technology in relation to other impact categories. The chosen approach used for identification of marginal electricity would, however, in the present study, change the hierarchy between compared waste management scenarios in relation to only one or two of the assessed impact categories, depending on the processes assumedly substituted by produced digestate.

Similar to the approach chosen by Boesch et al. (2014) and Seigné-Itoiz et al. (2015), the present paper uses governmental long-term planning documents as basis for the identified marginal electricity mix. The many uncertainties connected to long-term energy plans will surely reduce the predictability in this type of documents. As shown by Mathiesen et al. (2009), such plans are seldom fully realized, for several different reasons. Thus, as pro-

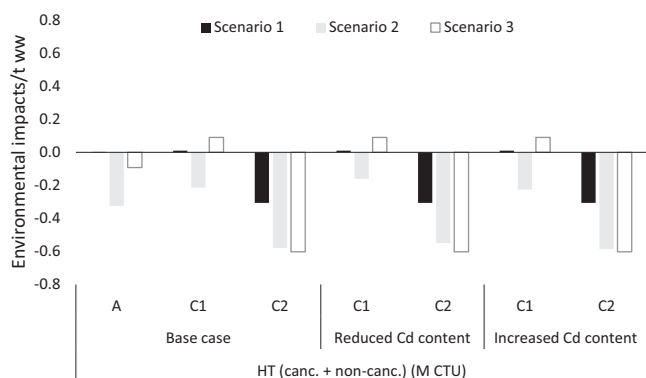


Fig. 6. Results from sensitivity analyses related to changes in cadmium (Cd) content in substituted phosphorus fertilizer.

posed by Mathiesen et al. (2009), it could be interesting to create alternative projections, adjusted for some key assumptions, such as high or low oil prices, and political will for investments in nuclear energy.

Different from several other LCAs of waste management alternatives, the present study provides an attempt to identify marginal technologies not only for energy provision, but also for provision of materials substituted by goods from the waste management process. The assumption of the product substituted by digestate being compost or mineral fertilizers will have a large influence on the impact categories climate change and eutrophication. This shows the importance of identifying affected suppliers also of products recovered from waste management systems, in order to generate relevant outcomes for decision support, although this not always is seen in previous consequential LCAs of waste management systems (Tonini and Astrup, 2012; Boesch et al., 2014; Seigné-Itoiz et al., 2015). Different from studies presented by Cimpan et al. (2015), the present study does not account for cascading effects, i.e. release of capacity in landfills when organic waste is treated through anaerobic digestion. Villanueva and Wenzel (2007) stated that alternative use of treatment capacity should be included in LCA of material recycling, but that the consequence in the long term is not avoided treatment through capacity release, but avoided construction of new capacity, as the market will adjust for a trend towards an increased material recovery. This was in the present study investigated in a sensitivity analysis, demonstrating that prolonged lifetime could result in vast benefits, primarily in relation to emissions of particulate matter through reduced risks of increasing transport distance. Including other cascading effects (for example assuming that released capacity in landfills would avoid use of uncontrolled dumpsites), can certainly increase even further the difference between results and conclusions between attributional and consequential modelling of different waste management alternatives.

As stated by Ekvall and Weidema (2004), it is not reasonable to assume that the environmental properties of the technologies in the future are accurately described with data used in attributional modelling, as these represent current or even historical technologies. However, it can also be risky to consider future technological advances in environmental assessments of future systems. Scenario modelling, applying dynamic modelling, describing expected and surprise-free future scenarios has been suggested for investigation of probable technological developments (Ekvall and Weidema, 2004). This was in the present paper represented by exploring the effects of a change from cadmium telluride to magnesium chloride in solar power cells as well as by large scale use of catalytic reduction of nitrous oxide emissions from production of nitrous fertilizers. Result from these sensitivity analyses state that the assumed technology level can be of vast importance to overall environmental impacts from investigated treatment alternatives.

In summary, the present study shows that not only the chosen modelling approach (attributional or consequential), but also the identification of marginal processes can affect both absolute impacts and ranking of compared waste management scenarios. As pointed out by Ekvall and Weidema (2004), the future is inherently uncertain. The actual future processes substituted by the goods generated in the different waste management treatment alternatives are defined by factors that we cannot predict at present, such as future prices of chemical fertilizers and fuels, as well as political decisions on taxes etc. This could be interpreted as if LCA results generated with consequential modelling are associated with larger uncertainties than results generated with attributional modelling (Schmidt and Weidema, 2009). It should, however, be remembered that although there may be large uncertainties concerning what and how unit processes are affected by a change in

the studied system, the answer cannot be to opt for an alternative that certainly is not accurate, reflecting the past rather than the future. In countries like Brazil, electricity provision has historically been dominated by technologies that present rather low environmental impacts. However, as projections point at an increased use of electricity generation with larger environmental impacts, attributional modelling could result in a risk of underestimating potential benefits from efficient energy and material recovery of waste.

5. Conclusions

The present case study has shown that the chosen approach certainly can have an effect on conclusions drawn from LCAs of solid waste management systems. In cases where waste management results in substitution of electricity and materials through energy recovery and recycling, environmental benefits increase with an increased environmental burden from the products assumedly substituted. Thus, if the processes identified as affected by changes in the waste management system have a higher environmental impact than the average, it is likely that waste treatment alternatives including efficient energy and material recovery will come across as more beneficial when consequential modelling is applied – and the other way around. Thus, when compared to attributional modelling, application of a consequential modelling could both increase as well as decrease the environmental benefits from the different waste management alternatives compared in the present study, depending on the processes included in the consequential modelling. The many uncertainties related to identification of processes affected by the changes investigated in the specific LCA should not limit the use of a consequential modelling. Rather, systematic investigation of alternative processes can present the range in which results are most likely to be found, and the most relevant factors for different outcomes can be identified. The present study clearly shows that different approaches in identifying affected processes can affect overall results vastly. In order to facilitate the interpretation of gained results, a multi-criteria analysis could be performed, using relevant methods for weighting of different environmental impact categories. This will be developed in a second phase of the present project.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.wasman.2017.07.002>.

References

- ANDA, 2015. ASSOCIAÇÃO NACIONAL PARA DIFUSÃO DE ADUBOS - ANDA. Mercado de fertilizantes: janeiro-maio/2015, São Paulo. <http://www.anda.org.br/estatistica/comentarios.pdf>.
- Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2011. Mass balances and life cycle inventory of home composting of organic waste. *Waste Manage.* 31 (9–10), 1934–1942.
- ANEEL, 2009. Agência Nacional de Energia Elétrica. Atlas de Energia Elétrica do Brasil, parte 3: Carvão. http://www2.aneel.gov.br/arquivos/pdf/atlas_par3_cap9.pdf.
- Angelo, A.C.M., Bernstad Saraiva, A., Clímaco, J.C.N., Infante, E., Valle, R., 2017. Life Assessment and Multi-criteria Decision Analysis: Selection of a strategy for domestic food waste management in Rio de Janeiro. *J. Cleaner Prod.* 143, 744–756.
- ANP, 2015. Agência Nacional Do Petróleo. Gás Natural e Biocombustíveis, Produção Nacional de Derivados de Petróleo (metros cúbicos). <http://www.anp.gov.br/?pg=18599&m=&t1=&t2=&t3=&t4=&r=&ps=1449492438759>.
- Abrelpe, 2012. Atlas Brasileiro de Emissões de GEE e Potencial Energético na Destinação de Resíduos Sólidos. http://www.abrelpe.org.br/arquivos/atlas_portugues_2013.pdf.
- Abrelpe, 2014. Panorama dos resíduos sólidos no Brasil 2013, Abrelpe, São Paulo, Brazil. <http://www.abrelpe.org.br/Panorama/panorama2013.pdf>.

- Balmér, P., 2012. Strategies for use of sludge (Strategier för slam användning, in Swedish). VA-strategi AB. Presentation, VA-mässan, September 2012.
- Beck-Friis, B., Smars, S., Jonsson, H., Kirchmann, H., 2001. Gaseous emissions of carbon dioxide, ammonia and nitrous oxide from organic household waste in a compost reactor under different temperature regimes. *J. Agric. Eng. Res.* 78, 423–430.
- Bernstad, S.S.A., Andersson, T., 2015. Food waste minimization from a life-cycle perspective. *J. Environ. Manage.* 147, 219–226.
- Boesch, M.E., Vadenbo, C., Saner, D., Huter, C., Hellweg, S., 2014. An LCA model for waste incineration enhanced with new technologies for metal recovery and application to the case of Switzerland. *Waste Management (New York, N.Y.)*, 34 (2), 378–389. doi: <http://dx.doi.org/10.1016/j.wasman.2013.10.019>.
- Borgshed, J., Leander, J., Rönquist, E.-M., Steinwall, P., 2003. Systems Analysis of Waste Treatment in the Kalmar Region. Carl Bro Energy Consultancy (in Swedish), Kalmar, Sweden.
- Cichy, B., Jaroszek, H., Paszek, A., Tarnowska, A., 2014. Cadmium in phosphate fertilizers; ecological and economical aspects. In: Proceedings from XV Conference Environmental, vol. 10, pp. 837–842. <http://www.chemikinternational.com/wp-content/uploads/2014/10/3.pdf>.
- Cimpan, C., Rothmann, M., Hamelin, L., Wenzel, H., 2015. Towards increased recycling of household waste: Documenting cascading effects and material efficiency of commingled recyclables and biowaste collection. *J. Environ. Manage.* 157, 69–83.
- COMLURB, 2014. Caracterizações gravimétrica e bacteriológica de resíduos sólidos domiciliares. Cidade do Rio de Janeiro – 2013.
- Curran, M.A., Mann, M., Norris, G., 2005. The international workshop on electricity data for life cycle inventories. *J. Cleaner Prod.* 13 (8), 853–862.
- Da Silva, G.F., 2011. Aterro sanitário São João: estudo dos indicadores ambientais em emergência. Universidade Paulista – UNIP, Brazil. Master Thesis.
- DNPM, 2014. Brazilian National Department for Mineral Production (Departamento Nacional da Produção Mineral). Fosfato – Sumário Mineral 2014. <http://www.dnpm.gov.br/dnpm/sumarios/fosfato-sumario-mineral-2014/view>.
- DWMT, 2014. Database of Waste Management Technologies AD 2: Dry AD followed by covered windrows composting. <<http://www.epem.gr/waste-control/database/html/AD-01.htm>>.
- EC, 2010. International reference life cycle data system handbook. General guide for Life Cycle Assessment – Detailed Guidance. Joint Research Centre Institute for Environment and Sustainability, European Commission, Ispra, Italy.
- Ecoinvent 3.0. LCI Database, Ecoinvent Center, 2013. Swiss Center for Life Cycle Inventory. <http://www.ecoinvent.org/database/>.
- Ekvall, T., Weidema, B., 2004. System boundaries and input data in consequential life cycle inventory analysis. *Int. J. LCA* 9 (3), 161–171.
- EPA, 2015. Advancing Sustainable Materials Management: 2013 Fact Sheet, EPA, Washington, US. http://www.epa.gov/wastes/nonhaz/municipal/pubs/2013_advncng_smm_fs.pdf.
- EPE, 2015. Empresa de Pesquisa de Energia. Results from Auctions FA 2015, A-5 2015, A-1 Reserva 2015 and A-3 2015. <http://www.epe.gov.br/PDEE/Relat%C3%B3rio%20Final%20do%20PDE%202024.pdf>.
- EPE/MME, 2014. Plano Decenal de Expansão de Energia 2024. Empresa de Pesquisa Energética – EPE/Ministério de Minas e Energia, Brasília, Brazil.
- FAO, 2002. Fertilizer use by crop in Brazil. Chapter 4. The fertilizer sector. <http://www.fao.org/3/a-y5376e/y5376e08.htm>.
- Finnveden, G., Johansson, J., Lind, P., Moberg, Å., 2000. Life Cycle Assessments of Energy from Solid Waste. FMS Report 137.
- Genrup, M., Jonshagen, K., 2011. Integration of bio-fired gas turbines in combined heat and power generation. Värmeforsk Report 1164. Stockholm, Sweden.
- Hamelin, L., Jørgensen, U., Petersen, B.M., Olesen, J.E., Wenzel, H., 2012. Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark; a consequential life cycle inventory. *Global Change Biol. Bioenergy* 4, 889–907.
- Hamelin, L., 2013. Carbon management and environmental consequences of agricultural biomass in a Danish Renewable Energy strategy. PhD Thesis, Department of Chemical Engineering, Biotechnology and Environmental Technology. University of Southern Denmark (Faculty of Engineering).
- Harrison, R., Webb, J., 2001. A review of the effect of N fertilizer type on gaseous emissions. *Adv. Agron.* 71, 65–108.
- IBGE, 2013. IBGE, Diretoria de Pesquisas, Coordenação de Agropecuária, Produção Agrícola Municipal 2013. Rio de Janeiro.
- ILCD, 2011. Recommendations for Life Cycle Impact Assessment in the European Context. European Commission Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.
- IPCC, 2006. N2O emissions from managed soil as and CO2 emissions from lime and urea application. Guidelines for National Greenhouse Gas Inventories, vol. 4, Chapter 11, Agriculture, Forestry and Other Land Use.
- IPEA, 2014. PNRS Observatory: solid waste final destination. Institute for Applied Economic Research. Brasília, DF, Brazil. <<http://observatoriopnrs.org/destinacao-final/>> (Accessed 07.04.15).
- Jenssen, T.K., Kongshaug, G., 2003. Energy consumption and greenhouse gas emissions in fertiliser production. In: Proceedings No. 509. The International Fertiliser Society, London, UK.
- Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T.H., Hauschild, M.Z., 2013. Review of LCA studies of solid waste management systems – Part I: Key learnings and perspectives. *Waste Manage.* 34, 573–588.
- Lund, H., Mathiesen, N., Christensen, P., Schmidt, J.H., 2011. Energy system analysis of marginal electricity supply in consequential LCA. *Int. J. Life Cycle Assess.* 15, 260–271. <http://dx.doi.org/10.1007/s11367-010-0164-7>.
- Major, J.D., Treharne, R.E., Phillips, L.J., Durose, K., 2014. A low-cost non-toxic post-growth activation step for CdTe solar cells. *Nature*. <http://dx.doi.org/10.1038/nature13435>.
- Manfredi, S., Christensen, T.H., 2009. Environmental assessment of solid waste landfilling technologies by means of LCA-modeling. *Waste Manage.* 29, 32–43.
- Mathiesen, B.V., Münster, M., Fruergaard, T., 2009. Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. *J. Cleaner Prod.* 17 (15), 1331–1338. <http://dx.doi.org/10.1016/j.jclepro.2009.04.009>.
- MME, 2015. Ministério de Minas e Energia. Balanço Energético Nacional 2015. <http://www.mme.gov.br/web/guest/publicacoes-e-indicadores/balanco-energetico-nacional>.
- Nordic Council of Ministers, 2007. Biowaste – decision support tool for collection and treatment of source-sorted organic municipal solid waste. TemaNord 2007, 602 <http://norden.diva-portal.org/smash/get/diva2:701837/FULLTEXT01.pdf>.
- ONS, 2015. Operador Nacional do Sistema elétrico. PAR – Plano de Ampliação e Reforços.
- OSB/WSB, 2000. Ocean Studies Board and Water Science and Technology Board.
- PDE, 2014. Plano Decenal de Energia. EPE, Rio de Janeiro, Brasil.
- PNE, 2014. Plano Nacional De Energia (PNE) 2050. Empresa de Pesquisa Energética, Rio de Janeiro, Brazil.
- Portal Brasil, 2014. Petrobras firma acordo para importação de gás natural boliviano. Published in August 2014. (Accessed 20151020).
- Riber, C., Pedersen, C., Christiansen, T.H., 2009. Chemical composition of material fractions in Danish household waste. *Waste Manage.* 29 (4), 1251–1257.
- Sandin, G., Peters, G.M., Svanström, M., 2014. Life cycle assessment of construction materials: the influence of assumptions in end-of-life modelling. *Int. J. Life Cycle Assess.* 2014 (19), 723–731.
- Schmidt, J.H., Weidema, B., 2009. Response to the public consultation on a set of guidance documents of the International Reference Life Cycle Data System (ILCD) Handbook. 2-0 LCA Consultants.
- SEPA, 2016. Increased recycling of food waste. <http://www.naturvardsverket.se/Miljoarbete-i-samhället/Miljoarbete-i-Sverige/Uppdelat-efter-omrade/Avfall/Atervinning-av-matavfall/>.
- Sevigné-Itoiz, E., Gasol, C.M., Rieradevall, J., Gabarrell, 2015. Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. *Waste Manage.* 46, 557–567.
- Shen, Y., Ren, L., Li, G., Chen, T., Guo, R., 2011. Influence of aeration on CH4, N2O and NH3 emissions during aerobic composting of a chicken manure and high C/N waste mixture. *Waste Manage.* 31, 33–38.
- Six, L., Smolders, E., 2014. Future trends in soil cadmium concentration under current cadmium fluxes to European agricultural soils. *Sci. Total Environ.* 2014 Jul 1, 485–486, 319–28.
- Smith, A., Brown, K., Ogilvie, S., Rushton, K., Bates, J., 2001. Waste management options and climate change: final report. European Commission, DG Environment. http://ec.europa.eu/environment/waste/studies/pdf/climate_change.pdf.
- Sonnemann, G., Vigon, B., 2011. Global guidance principles for life cycle assessment databases – a basis for greener processes and products, UNEP/SETAC. <http://www.unep.fr/shared/publications/pdf/DT1410xPA-GlobalGuidancePrinciplesforLCA.pdf>.
- Tonini, D., Astrup, T., 2012. Life-cycle assessment of a waste refinery process for enzymatic treatment of municipal solid waste. *Waste Manage.* 32 (1), 165–176.
- UNFCCC, 2006. PROJECT DESIGN DOCUMENT FORM (CDM PDD). N2O Emission Reduction in nitric acid plant Paulínia, SP, Brazil. <https://cdm.unfccc.int/filestorage/8/A/5/8A5IO1X4L4L6W92OQOQZF0F8B0VBW/Brazil%20PDD.pdf?ta=Fp8bnlmOWtjDCL1CPmolyHTKQdCpDbqo>.
- Vázquez-Rowe, I., Rege, S., Marvuglia, A., Thénie, J., Haurie, A., Benetto, E., 2013. Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg. *Int. J. Life Cycle Assess.* 18 (8), 1593–1604. <http://dx.doi.org/10.1007/s11367-013-0604-210.1038/nature11811>.
- Waste Management Sweden, 2013. Pretreatment of food waste for biogas production (in Swedish: Förbehandling av matavfall för biogasproduktion). Report B2013:01. ISSN 1103–4092. Waste Management Sweden, Malmö, Sweden.
- Waste Sweden, 2014. Avfallsstatistik. Avfall Sverige, Malmö, Sweden. <http://www.avfallsverige.se/statistik-index/avfallsstatistik/>.
- Weidema, B.P., Ekvall, T., Heijungs, R., 2009. Guidelines for application of deepened and broadened LCA. Final report Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability – Calcas. Project no.037075.
- Werner, J.J., Zapf-Gottwick, R., Koch, M., Fischer, K., 2011. Toxic substances in photovoltaic modules. In: The 21st International Photovoltaic Science and Engineering Conference November 28th – December 2nd, 2011, Fukuoka, Japan.
- Villanueva, A., Wenzel, H., 2007. Paper waste – recycling, incineration or landfilling? A review of existing life cycle assessments. *Waste Manage. (Oxford)* 27, S29–S46.
- Yara, 2015. Garanti klimatavtryck. http://www.yara.se/images/YARA_GarantiKlimatavtryck_Sv_2014_tcm422-106645.pdf.
- Yokelson, R.J., Christian, T.J., Karl, T.G., Guenther, A., 2008. The tropical forest and fire emissions experiment: laboratory fire measurements and synthesis of campaign data. *Atmos. Chem. Phys.* 8, 3509–3527.
- Yoshida, H., Monster, J., Scheut, C., 2014. Plant-integrated measurement of greenhouse gas emissions from a municipal wastewater treatment plant. *Water Res.* 61, 108–118.

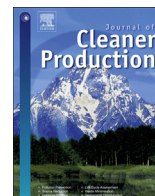
3.5 Artigo 5 – Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences *

* Bernstad Saraiva, A., Souza, R. G., Mahler, C. F., Valle, R. A. B. Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences. J. Cleaner Prod. 2018; 202: 488–496. doi: n°. 10.1016/j.jclepro.2018.08.038

ABSTRACT

Application of consequential lifecycle assessment modelling has gained increased interest in the area of solid waste management. In such assessments, identification of affected technologies and choices of system boundary setting are of key importance. With the aim of investigating how previous consequential lifecycle assessments of solid waste management systems have tackled these issues, a review was performed of 36 previously published studies. The intention is to contribute to improved understanding of the challenges of performing consequential lifecycle inventory modelling of solid waste management systems, which could facilitate future studies. Results demonstrate a strong relation between the selection of affected energy production technology and overall GHG-emissions. In general, assuming that energy provision from less polluting technologies is affected by studied changes will commonly discredit waste-to-energy technologies and promote material recycling. However, made choices were also in many cases not justified. Materials substituted by waste derived goods are frequently represented by average data. The detected inconsistency in how energy provision and material provision are modelled could result in biased results, and care should be taken to minimize this risk. Four aspects are identified where current practice in system boundary setting choices is diverse and where those choices could have significant influence on overall results; counterfactual waste management, fate of materials avoided through material recycling, cascading effects and rebound effects. This paper argues that there is a need for increased transparency and coherence in identification of affected processes/technologies, as well as for a broader approach in system boundary setting, if studies have the aim of serving as relevant input for decision makers.

Keywords: Lifecycle assessment; Solid waste management; Consequential modelling; Marginal processes.



Review

Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences

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1. Introduction

Life cycle assessment (LCA) has been presented as a decision-support tool, applying a holistic perspective in quantifying environmental impacts (European Commission, 2010). Its usefulness in providing valuable information for decision makers of solid waste management (SWM) systems has previously been demonstrated (EU, 2008). Nevertheless, in their review of a large set of LCAs of SWM-systems, Laurent et al. (2013) identified a certain confusion of concepts and terminology surrounding different types of lifecycle inventory (LCI) modelling frameworks. According to the authors, one of the main reasons could be a lack of adequate goal definition and insufficient reflection on the context situation in which the study is performed. Indeed, the context situation has previously been pointed out as being of key relevance for an adequate selection of LCI modelling framework, and the ILCD Handbook (European Commission, 2010) makes specific recommendations on this issue (later subjected to a critical review by Ekvall et al. (2016)). Regardless of the reasons given for selecting a particular LCI modelling approach, the choice made has undoubtedly a significant influence on the definition of system boundaries (European Commission, 2010). As discussed in Thomassen et al. (2008), ALCA and CLCA approaches are also likely to produce different results and offer different messages to end users. Attributional life-cycle assessment (ALCA) employs a system-modelling approach where the inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule (Sonnemann and Vigon, 2011). Consequential LCA (CLCA), on the other hand, is change-oriented and quantifies the effects associated with changes in the life cycle of a system brought about by a decision (Weidema et al., 2009; Curran et al., 2005). In this way, the consequential approach seeks to take the environmental assessment a step further, in order to analyse how environmental burdens may vary in response to changes with market implications, where processes are linked via market mechanisms beyond the foreground system (Vazquez-Rowe et al., 2013). Such linkages can occur for example when waste is used for energy or material recovery. Recovery implies that waste-based resources are released on the market, with the effect of substituting other product-systems. In CLCA, this is modelled through system expansion (Finnveden et al., 2009). Consequential LCI modelling should, according to Weidema et al. (2009), include the unit processes that change as a consequence of a decision, and identification of these processes has been presented as the key issue in consequential LCI modelling. Finnveden et al., 2009 and Mathiesen et al. (2009) have argued that the introduction of affected processes¹ in the LCI modelling involves

large uncertainties and significant time and effort are required to understand and reduce such uncertainties.

A number of reviews of SWM LCAs have been performed previously, with varying focus and objectives; some are limited to specific waste categories (Morris et al., 2013; Villanueva and Wenzel, 2007; Bernstad and la Cour Jansen, 2011; Lazarevic et al., 2010), and others to methodological conducts (Cleary, 2009; Gentil et al., 2010). Laurent et al. (2013) aimed at providing a comprehensive mapping and contextual analysis of LCA of SWM systems, but also at identifying potential misuses and misunderstandings and providing guidance to ensure robust application of the LCA methodology to SWM. The issue of consequential modelling was however not explored in details by the authors. Zamagani et al. (2012) argues that what distinguishes the two modes of LCA is the choice of the processes to be included in the system, but that the identification of those processes is often done inconsistently, using different arguments, which leads to different results. To our knowledge, no comprehensive review has until now been presented, focusing on identification and modelling of affected processes and technologies in CLCAs of SWM systems, in order to identify the augments most commonly used in this particular field, potential inconsistencies and, additionally, discuss these effects on results gained.

The objective of the present review is therefore not to add to debate on why and when attributional and consequential modelling should be used, nor the type of data most relevant to be used in different situations when applying consequential LCI modelling. These issues have been discussed thoroughly in several previous works (Suh and Yang, 2014), and is continuously an issue of debate (Ekvall et al., 2016). Rather, the objective of this paper is to describe how LCA practitioners working specifically with SWM.

- Justify the choice of consequential LCI modelling
- Select and justify the choice of data used in consequential LCI modelling

This was done by exploring choices made by LCA-practitioners in previously published LCAs of SWM systems performed with a consequential approach. The intention is to present a state of the art in the field and contribute to improved understanding of the challenges of performing consequential LCI modelling of SMW systems.

2. Method and material

The keywords *consequential*, *marginal*, *LCA*, *lifecycle assessment*, *life cycle assessment*, *CLCA*, *waste* and *residue* were applied in the databases Scopus, JSTOR and Science Direct, resulting in 177 hits. A screening of abstracts resulted in the selection of 36 scientific papers, presenting case studies where it is clearly stated that consequential modelling is applied, or where processes included in the assessment are described as marginal, i.e. affected by changes in the investigated SWM system. Only peer-reviewed original research papers published in English in scientific journals over the last ten years (2007–2017) were included. Papers studying treatment of solely residues/by-products from forest and agriculture were excluded. The main reason for this choice is the modelling of indirect land use changes (iLUC), as this is considered a topic on its own, and it would not be possible to include it in the present review

¹ The term “affected processes” will, in the present study, refer to any process (i.e. production of heat, electricity or materials) that may be affected by life cycle changes within the expanded system, while “technologies” in this context, will refer to specific technologies used for provision of these processes (i.e. wind power, production and combustion of natural gas etc.).

² The term “marginal” is commonly used in reviewed studies, referring to the processes and technologies affected by changes in the waste management system, with the underlying interpretation that changes are small enough to be approximated as infinitesimal, and that no changes are caused in the way the system is operated (Azapagic and Clift, 1999).

due to scope limitation. The review was performed with the intention of being as comprehensive as possible, including all studies that could be relevant for the aim of the study. However, relevant publications might have been excluded unintentionally. As the main objective of the study was to describe and discuss how researchers previously have justified the use of CLCA and the selection of data used in the modelling, conference papers were excluded as they commonly present a shortened version of a larger study, with reduced possibilities to justify made choices. The list of investigated studies and key information is presented in Table 1.

A matrix was established for the review, including i) justification of using consequential LCI modelling, ii) system boundary settings (included/excluded processes), iii) method(s) used for identification of affected processes/technologies, and iv) handling of uncertainties connected to the processes/technologies identified as affected by studied change. The outcomes of the review are presented and discussed below.

3. Results and discussion

3.1. Justification of using consequential LCI modelling

The choice between application of attributional or consequential modelling should be presented in the scope of the LCA (Laurent et al., 2013), and is of large importance since this has a strong influence on the LCA results (Ekvall et al., 2016). It could thereby be assumed that the chosen approach should have been thoroughly discussed and justified by authors of reviewed studies. However, the selected modelling approach is clearly justified only in half of the reviewed studies (SI).

In several studies, the application of CLCA is justified by a general focus on the consequences of a decision (Frøerger and Astrup, 2011; Tonini and Astrup, 2012; Seigné-Itoiz et al., 2014, 2015a,b). In other studies, the application of CLCA is justified more specifically by a wish to account for indirect effects and counterfactual waste management processes (Eriksson et al., 2007; Styles et al., 2016; Salemdéeb et al., 2017). Four of the reviewed studies have the outspoken aim to compare use of attributional and consequential modelling. In two of them, the two modelling approaches are applied with different objectives: the ALCA is used to attribute environmental impacts to different treatment alternatives, while the CLCA examines the potential environmental impacts from a shift between current management and a proposed alternative (Feraldi et al., 2013; Hums et al., 2016). In the other two, the comparison of results gained through ALCI and CLCI (attributional/consequential LCI) modelling is presented as an objective in itself (Boesch et al., 2014; Kua, 2015).

Justifying use of CLCI modelling by a will to focus on the consequences of a decision finds support in several references (Finnveden et al., 2009; Sonnemann and Vigon, 2011). This suggests that when LCA is used for decision-support, the LCI model should reflect the consequences of a decision in question, making consequential modelling more relevant. However, as seen in previous reviews of SWM systems, attributional modelling is by far more commonly applied on LCAs developed with the purpose of serving as decision support tool (Laurent et al., 2013). Thus, to many researchers in the field, CLCA still seems to be regarded as something new, in need of further investigation and interpretation, calling for an interest in comparing outcomes from ALCI and CLCI modelling and discussing the robustness of results gained through the use of these two approaches.

3.2. Methods used for identification of affected technologies²

In most reviewed studies, the substitution of conventional

goods by waste derived goods constitutes the main benefit related to waste-to-energy and material recovery alternatives. The identification of what is being substituted is therefore of large importance to assess overall environmental impacts from the investigated SWM system.

Methods used for identification of affected processes in reviewed studies could be grouped into seven categories (Table 2). In several cases, one method was used for identification of one type of process and another for identification of another. Methods and choices made in identification of affected processes and technologies are discussed below, divided in provision of energy and material respectively. More details are found in the SI.

3.2.1. Identification of affected energy technologies

Electricity and heat technologies affected by a change in the SWM system can be identified either as single or mixed technologies. Amongst reviewed studies, a clear majority identifies single technologies rather than a mix of affected technologies. For identification of single affected technologies, it is clear that the work of Weidema (2003, 2009) has had a strong influence in the field (Table 2). Several studies also refer to recommendations made by Mathiesen et al. (2009), namely the inclusion of two radically different technologies as affected options for production of electricity as two alternative base-scenarios (Eriksson et al., 2007; Merrild et al., 2008; Boesch et al., 2014).

In several cases, projections or plans from national or international energy entities are used. In these cases, the affected technologies are assumed to be the mix of projected new installation capacity according to international projections (Feraldi et al., 2013, regarded as long-term marginal based on EIA (2009)), national projections (Seigné-Itoiz et al., 2015b) or national objectives of phasing out specific fuels (Turconi et al., 2011; Martinez-Sanchez et al., 2016). Although use of such official sources could seem as a robust option, Mathiesen et al. (2009) showed that energy plans historically have rather limited relevance for projection of actual developments of national electricity markets.

Dynamic optimising models, such as energy systems analyses (ESA), can give a more complete description of the consequences of using or delivering electricity (Mathiesen et al., 2009). ESA can take into account effects on the utilisation of existing production facilities as well as effects on investments in new production facilities (Eriksson et al., 2007). Münster and Meibom (2010) argue that use of ESA is relevant for identification of affected energy production as it makes it possible to identify the combination of affected technologies in the short- as well as long-term. However, the ESAs used in reviewed LCAs were in most cases performed for other purposes than serving as LCI input. Thus, there is a risk that the ESA is outdated (Merrild et al., 2008) or that the context in which the ESA was performed not is entirely relevant for the system investigated in the LCA (Eriksson et al., 2007).

Differently from electricity production, substitution of thermal energy often depends on local conditions and production capacities connected to a specific district-heating network (Frøerger et al., 2010). It could therefore be argued that case specific existing production capacity should be assumed affected in the short-term. The review also shows that local or national average technologies are used as representation of affected technologies for substituted heat in several studies (Boldrin et al., 2011; Turconi et al., 2011; Tonini and Astrup, 2012; Carlsson et al., 2015). In the long-term perspective, Frøerger et al. (2010) argues that heat production from waste will contribute to phasing-out fossil fuels, a reasoning applied by Tonini and Astrup (2012), who identified different affected technologies for long- and short-term scenarios, exclusively substituting fossil fuels in the long-term horizon. However,

Table 1

Presentation of investigated studies and key data. MSW = Municipal Solid Waste, OFMSW = Organic Fraction Municipal Solid Waste, AD = Anaerobic Digestion, CHP = Combined Heat and Power.

Reference	Country	S	Treatment alternatives
Boesch et al., 2014	Switzerland	MSW	Incineration with metal recycling
Boldrin et al., 2011	General	OFMSW	Windrow composting ^a ; Tunnel composting ^a ; AD one stage wet ^a ; Combined technology ^a (AD + Composting)
Carlsson et al., 2015	General	Food waste	Wet AD ^a
Cimpan et al., 2015	Denmark	MSW	Biowaste pre-treatment and digestion + biogas CHP ^a ; WtE CHP
Eriksson et al., 2009	Sweden	Plastic waste	Incineration with CHP; Landfilling
Eriksson et al., 2007	Sweden	MSW	Incineration with CHP; Combustion of biomass with CHP; Recycling; Landfilling
Feraldi et al., 2013	US	Scrap tires	Mechanical recycling; Co-combustion at cement kiln
Fruergaard and Astrup, 2011	Denmark	MSW	Co-combustion; AD ^a ; mass burn incineration
Habert 2013	EU	Industrial waste (blast-furnace slags and coal combustion fly ashes)	Co-combustion in cement kiln
Hamelin et al., 2011	Denmark	Separated animal slurry	Field spreading as fertilizer ^a ; Decanter Centrifuge with Polyacrylamide ^a ; Screw Press ^a ; AD ^a
Hamelin et al., 2014	Denmark	Pig manure	Conversion to biofuel; Landfilling
Hums et al., 2016	US	Grease trap Waste	Remelting with primary aluminium
Johnson et al., 2013	US	Aluminium	Drying + milling; Biorefinery ^a
Karlsson et al., 2015	Sweden	Faba bean	AD ^a with CHP; Conversion to ethanol
Kimming et al., 2011	Sweden	Agricultural residues	Aggregate in concrete
Kua 2015	Singapore	Steel slag	Landfilling; Recycling; Incineration or Composting ^a
Manfredi et al., 2011	General	MSW fractions: organic, paper, plastic, aluminium, glass	
Millward-Hopkin et al., 2017	UK	Fly ash from coal combustion	Use in cement production; Landfilling; Prevention
Mü; nster and Meibom 2010	Denmark	Mixed waste; organic waste; RDF	Incineration; Co-combustion; AD ^a with CHP; Gasification
Martinez-Sanchez et al., 2016	Denmark	Food waste	Incineration; Co-digestion ^a ; Fodder treatment
Melamu and von Blottnitz 2011	South Africa	Sugar mill bagasse	Cellulosic ethanol biorefinery
Merrild et al., 2008	Denmark	Waste paper	Recycling; Incineration
Merrild et al., 2012	Denmark	MSW fractions: paper, cardboard, plastic, steel, aluminium, glass	Recycling; Incineration
Salemdeeb et al., 2017	UK	Food waste	Pig feed (wet/dry); AD ^a ; Composting ^a
Schmidt et al., 2007	Denmark	Waste paper	Recycling; Incineration; Landfill
Sevigné-Itoiz et al., 2014	Global	Aluminum old scrap	Recycling
Sevigné-Itoiz et al., 2015a	Spain	Waste paper	Recycling
Sevigné-Itoiz et al., 2015b	Spain	Plastic waste	Mechanical recycling; Recycling to RPL; Incineration with CHP; Landfill
Sørensen and Wenzel, 2014	Denmark	Bedpans	Recycling; Incineration
Styles et al., 2016	UK	Fodder and food waste	AD ^a with CHP; Composting ^a ; Incineration; Landfill
Tonini and Astrup 2012	Denmark	MSW	Enzymatic treatment in a biorefinery
Tonini et al., 2013	Denmark	OFMSW	AD ^a + Composting ^a (with CHP); Incineration with CHP; MBT + AD ^a or Composting ^a ; Bioreactor landfilling; Conventional landfilling
Turconi et al., 2011	Denmark and Italy	MSW	Incineration
Turk et al., 2015	Generic	Fly ash, steel slag	Aggregate in concrete
Vázquez-Rowe et al., 2013	Luxembourg	Maize	Conversion to biomethane
Zink et al., 2014	US	Smartphone	Traditional refurbishment, Repurposing using battery power, and Repurposing using portable solar power

^a Recycling of nutrients was assessed in the study.

the more fossil energy is substituted, the more likely is also the chance of non-fossils becoming affected by SWM systems in the future. This perspective was adopted by Fruergaard and Astrup (2011). As a result, combustion of biomass (wood and straw) was identified as the long-term affected technology.

The time horizon can be of large importance to the identification of affected energy technologies (Weidema et al., 2009). Even so, less than 40% of the reviewed studies present information of assumed time horizon for identification of affected energy technologies. Amongst these, a long-term perspective was applied in a majority of the studies. As stated by Weidema et al. (2009), even the effect of small, short-term changes are seldom isolated to the short-term

perspective, since each individual short-term purchase decision will contribute to the accumulated trend in the market volume, which is the basis for decisions on capital investment, and thus long-term changes, in free market situations. Thus, a long-term perspective could be of larger relevance in this type of studies. In five studies, short-as well as long-term horizons were applied and compared.³

³ It should be highlighted that different authors give different definitions to the terms short- and long-term. According to the European Commission (2010), short-term should be regarded as up to 5 years from present date. Tonini and Astrup (2012), use 15 years and Feraldi et al. (2013) use 10 years.

Table 2

Methods for identification of affected processes in reviewed studies. Sub-categories within each method are denoted with capital letters.

Method	Characteristics	Number of studies			Implication for CLCA modelling
		Power	Heat	Other ^a	
Reference to Weidema et al. (2009)	Identification of most and least competitive technology, when demand is increasing/decreasing respectively. Disregards constrained suppliers.	12	0	4	Reflecting consequences of studied decision, based on market logics.
Reference to Mathiesen et al., 2009	Fundamentally different affected energy technologies, including both fossil and renewable, should be considered (only relevant in relation to energy provision).	4	1	0	Reflecting the widest range of potential outcomes, ignoring market logics.
Statistics	A (Power and heat): Currently most common technologies (regional, national or local boundaries), based on recent statistics. B (Power and heat): Identifying the most expanding fuel in recent years. C (Power and heat): Identification of the fuel with historically highest/lowest cost. D (Power): Identification of principal fuel accommodating annual fluctuations over the last years.	4	9	9	A Ignoring consequences of studied decision on current situation. B, C and D Ignores that historic trends can be changed through political decisions, constrains etc.
Projections	A (Power): Projections of new installation capacity or national energy plans. B (Heat): Planned infrastructure projects (i.e. planned gas pipeline)	7	1	0	Projections commonly built on currently most competitive technology. Can include market logics or be based on historic trends. Plans can be less relevant in the long-term and have shown not to be representative for actual developments.
Energy system modelling	Identification of combinations of affected technologies in the short- as well as long-term, based on optimization of investments as well as production of electricity and heat, considering storage, transmission, prices and political decisions.	4	3	0	Enables modelling of short-term energy storage, price and policy dynamics over the timeframe of relevance. The context in which the ESA was performed might not being entirely relevant for the system or time horizon investigated in the LCA.
Other	- Experts opinion - Reasons of simplicity. - Creating optimized conditions for incineration vs recycling	3	2	2	Not related to the decision in question of the study.
Not stated	Choices were not motivated	10	9	1	Decreases the robustness of gained results.

^a Recycling techniques, virgin plastics, metals, glass and paper, wood, pulp, mineral fertilizers and fuels used in the transport sector.

Contrary to previous recommendations for consequential modelling (Weidema et al., 2009; Mathiesen et al., 2009), substituted electricity was in several studies modelled as current national or regional average grid mix, based on recent statistics (Manfredi et al., 2011; Zink et al., 2014; Salemdieb et al., 2017). Use of current grid-mix supposes that all technologies will respond in the same extent to changes in electricity demand, and includes also suppliers not affected by the change in focus of the study, why this approach should be avoided in CLCI (Hamelin, 2013).

3.2.2. Identification of affected materials

Material recycling of a product may lead to avoided production of primary material or avoided production of recycled material in another life cycle. According to Ekvall and Weidema (2004), the avoided production depends on the price elasticity of supply of collected material for recycling and on the price elasticity of demand for the recycled material in question. Assuming that the supply is completely price inelastic leads to the approximation that recycled material replaces 100% primary material. Assuming that the demand is completely price inelastic leads to the approximation that recycled material replaces 100% recycled material from other life cycles. It is obvious that this will have a large impact on results from LCAs of SWM systems including material recycling. At the same time, prices of recycled materials are commonly linked to the price of primary materials, which can fluctuate with economic cycles etc. (Frees, 2008). Thus, estimations of price elasticity may be uncertain and vary with the time horizon of a study.

The present review shows that the most common approach is assuming that the supply is completely price inelastic. This leads to the approximation that recycled material replaces 100% primary material of the same kind. According to Frees (2008), an argument for this approach is that it could be valid for materials for which there are no hindrances for recycling into the same material and where there is a growing demand. However, a thorough analysis of the material markets is seldom presented in reviewed studies. Thus,

this approach could be criticised for not taking into account that part of the recycled material may also replace recycled material from other systems, resulting in overestimations of environmental benefits from material recycling.

In general, it was assumed that the use of recyclables in manufacturing of new products would not influence the market situation for the product in question (Merrild et al., 2012). Production technologies for substituted products as well as recycling technologies where, with a few exceptions (Hamelin et al., 2014; Seign -Itoiz et al., 2014, 2015a,b), thereby commonly modelled as global or regional averages, using recent statistics. Differently from the case of district heating, materials are commonly sold on a global market, and the type of technology and electricity used in the production process by different suppliers can be of large relevance for overall environmental impacts from the material (Merrild et al., 2008; Seign -Itoiz et al., 2015b). Even so, different from in the case of energy, very few studies compared different choices of technologies affected by material recycling.

Only in cases where handling of one single type of waste was investigated (i.e. aluminium, paper, plastics or biowaste), did authors perform a systematic identification of the affected virgin materials substituted through material recycling. In all these cases, authors refer to the methodology presented by Weidema (2003 and 2009) for the identification of affected activities. However, also in these studies, the identified affected activities are at some point modelled as average processes. As an example, Seign -Itoiz et al. (2015a) identifies Brazilian bleached hardwood kraft pulp (BHKP) as the product affected by increased paper recycling in Spain. However, this process was modelled as an average process. However, according to Weidema et al. (2009) it could be assumed that the BHKP actually affected would be the one produced in the least efficient plants, reducing the demand for marginal rather than average electricity. Thus, even after an identification of affected processes, further effort is needed in order to provide relevant modelling of these processes and ensure overall consistency. In

many reviewed studies this is not done, potentially due to lack of data or possibilities to adjust datasets representing average production technologies.

As stated by Hamelin (2013), the main problem with use of average data is that it includes suppliers that are not affected by the decision under study. The historical data used in construction of average data can be of higher or lower relevance in CLCA. For materials where environmental performance in extraction and production is very similar globally, the use of electricity is low and technical advances are unexpected, the use of global average data might be relevant also in consequential modelling. However, this is not the case for many materials. As an example, Merrild et al. (2008) chose to present several different virgin pulp technologies as potentially affected by increased paper recycling, showing that the choice of virgin technologies influenced the result of the LCA, mainly due to large variations in energy use. Seigné-Itoiz et al. (2014) clearly show the large influence the choice of average and marginal data in modelling of substituted materials can have on overall results. According to their results, the avoided GHG-emissions related to aluminium recycling could be estimated to 6158 kg CO₂-eq./ton scrap collected when using average input data, while considering global markets and results marginal electricity in aluminium production result in avoidance of 18100 kg of CO₂-eq./ton of scrap collected. The difference is an effect of the identification of the actually most competitive global suppliers of virgin aluminium (including bauxite mining, alumina and smelting) and the marginal electricity mixes considered for respective supplier.

3.3. System boundary setting

The object of the consequential LCI is to include what is affected by a change in the use of a product or a service in the investigated life cycle (Ekvall and Weidema, 2004). Thus, unit processes are included in the system to the extent that they are expected to change as a consequence of a change in demand for the product/service in question (Weidema et al., 2009).

According to Ekvall and Weidema (2004), the boundaries of the system investigated should ideally be defined at the point where the consequences are so small, or the uncertainties so large, that further expansion of the boundaries will yield no information that is significant for any realistic decision. One problem with this recommendation is that it to some extent demands that the system initially must be expanded over this point, in order to identify to what extent the expansion renders significant information or not. The present review shows that previous recommendations on system boundary setting in CLCA commonly result in inclusion of processes with physical connection to the product/service in focus of the study, and energy and/or material substituted by goods generated through the investigated waste treatment (Fig. 1).

However, a different approach was chosen in some of the studies, including considerations of lost alternatives, fate of materials avoided through recycling, fate of surplus treatment capacity and lastly, rebound effects. The reasoning behind and implications of including these aspects in CLCA of SWM are discussed below.

3.3.1. Consideration of lost alternatives

The change-oriented nature of consequential modelling requires considerations of any variation in the current system resulting from investigated changes. Thus, the extent that the investigated use of waste material in our system results in reduced utilisation of other treatment alternatives is of relevance. As an example, using source-segregated household food waste for production of car fuel through anaerobic digestion in a system where this waste currently is incinerated with energy recovery, requires

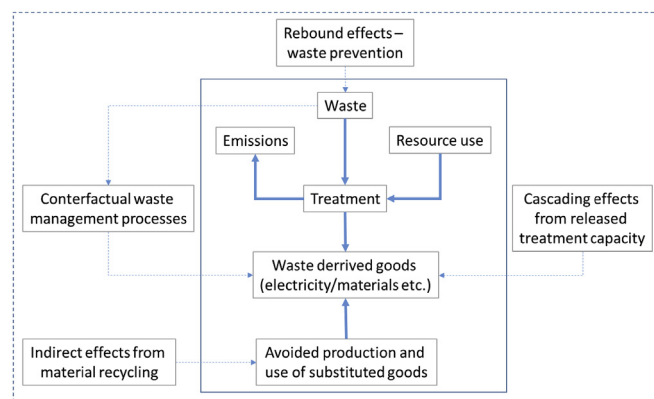


Fig. 1. Generic representation of the system boundaries most commonly applied in reviewed studies (within full line). Additional processes less commonly assessed within the studies are represented as boxes within the dotted line. Arrows represent a physical or economic connection between processes (boxes).

an assessment of the environmental loads avoided through reduced incineration. In addition, the affected processes needed for substitution of the electricity and heat no longer produced from food waste incineration could be considered. Hamelin et al. (2014) refer to goods no longer provided by the system after the investigated change as “lost alternatives”, while Salemddeeb et al. (2017) use the term “counterfactual waste management processes”, referring to the treatment alternatives from which waste is diverted. Styles et al. (2016) use the term “indirect effects”, acknowledging that, in some occasions, the alternative is not treatment, but rather use as i.e. fodder.

Counterfactual fates of waste handling are investigated by several authors (Eriksson et al., 2007; Tonini and Astrup, 2012; Feraldi et al., 2013; Hamelin et al., 2015; Styles et al., 2016; Salemddeeb et al., 2017). Styles et al. (2016) show that anaerobic digestion of brewery and bakery wastes increase GWP burdens from the system when these are diverted from current use as feed, if assumed that this increases the demand for feed produced from wheat. Tonini and Astrup (2012) argues that increased recycling of plastics and use of biomass for biogas-production in a biorefinery concept occurs at the expense of combustion of municipal waste with energy recovery. This is modelled as an increased need for electricity and heat provision from other sources, that will burden the investigated biorefinery system (Tonini and Astrup, 2012). However, the authors also consider benefits from reduced emissions from waste incineration as well as disposal of ashes and pollution control materials related to current handling (Tonini and Astrup, 2012). Counterfactual waste handling could also be relevant for secondary waste streams. Millward-Hopkins et al. (2017) investigated the effects from material recycling of low quality ashes from combustion of solid residual waste, compared to use of fly ash from coal incineration plants. According to the authors, an increased incineration of waste should be burdened by increased emissions of GHG, as it genders a need for import of high quality fly ashes from elsewhere, while low quality ashes are disposed of in the UK.

3.3.2. Indirect effects from material recycling

Increased recycling of waste materials can decrease the demand of virgin materials such as metals, plastics, paper and mineral fertilizers on the market. Recycled materials can also replace i) recycled material from other systems, ii) completely different types of material or iii) no material at all (Ekvall and Weidema, 2004). Amongst reviewed studies, it is commonly assumed that recycled

material substitutes virgin material to 100%. Although this approach might be attractive, it ignores other possible applications as well as any other consequences from material recycling. The issues lifted here could be applied to all studies in Table 1 where recycling of materials were assessed.

Substitution of virgin materials with recyclables will reduce also the supply of any by-products generated in the production of virgin materials. However, the demand of these by-products might not decrease, and it is relevant to discuss to what extent alternative provision of these materials should be included in the consequential modelling. As an example, it is well known that several metals are mined only as by-products. A recent investigation of 62 different metals and metalloids show that more than 60% have companionship (i.e. the production is dependent on the mining of host metals) greater than 50% (Nassar et al., 2015). The supply of gallium, indium, selenium and tellurium, rely largely on mining ores for the production of metals such as aluminium, copper and zinc. Based on this, it could be argued that any changes in the supply of by-products from for example copper production should be assessed in CLCA of systems where the demand of virgin copper is changed. Such aspects were however not included in any of the reviewed studies.

Paper recycling is a special case in waste CLCA, where the consequence of avoided use of wood is much debated. Reviewed studies assessing paper/cardboard waste assume substitution of virgin paper production through material recycling of waste paper, releasing wood and/or land for other uses. Whether this should be accounted for or not depends on the scarcity of forest area/wood and may thus depend on the time perspective applied in the study (Villanueva and Wenzel, 2007). In some cases, biomass saved through paper and cardboard recycling is assumed to remain in the forest (Eriksson et al., 2007; Seigné-Itoiz et al., 2015a,b). Others include the use of saved wood for energy purposes (Merrild et al., 2012; Schmidt et al., 2007). The latter is based on the assumption that wood in the future becomes a priority fuel/raw material of limited availability, due to energy policy decisions or fossil fuel scarcity (Villanueva and Wenzel, 2007). The alternative use of wood is a key issue when comparing recycling of paper and cardboard with incineration (Villanueva and Wenzel, 2007). The magnitude of the importance depends on the technology for production of electricity and heat assumedly replaced by wood, when this material no longer is used for paper production due to increased recycling. Merrild et al. (2012) assumes that coal is substituted by wood combustion, which results in vast benefits from paper recycling, while benefits according to Schmidt et al. (2007) are smaller, as natural gas assumedly was substituted by wood in their study. Fruergaard and Astrup (2011) describe biomass fuel (wood and straw) substituted by waste incineration as a non-constrained residual resource. If biomass substituted by waste-to-energy technologies is considered unconstrained, it implies that this material will not be used elsewhere to offset fossil fuels. The effect of biomass being constrained is the equivalent to assuming fossils as affected fuels. However, also residual biomass can provide several benefits to the systems they are recovered from, such as nutrient recovery and carbon storage in soils (Cherubini and Ulgiati, 2010), previously addressed in CLCA studies of bioenergy systems (Kimming et al., 2011; Ahlgren et al., 2013). Such aspects were not considered in any of the studies reviewed. If included, this could increase benefits from waste-to-energy treatment in scenarios where biomass was identified as affected technology.

3.3.3. Fate of surplus treatment capacity – cascading effects

Increased use of one treatment alternative can, in the short term, release capacity in other treatment alternative plants. As an

example, increased material recycling of waste previously combusted may imply that waste incineration plants instead take in other combustible waste. This introduces the question of what type of waste management is most likely to react to this increased demand for combustible waste. According to Cimpan et al. (2015), refused derived fuel (RDF) currently landfilled in the UK is the affected waste treatment in the EU. The reason is a current lack of incineration capacity and high domestic gate fees (Eunomia, 2013). Waste management systems releasing incineration capacity could thereby be credited emissions circumvented through avoided landfilling. According to Cimpan et al. (2015), inclusion of avoided landfilling of RDF in the UK increases climate benefits from investigated Danish source segregation systems by between 15 and 30%. Villanueva and Wenzel (2007) state that alternative use of treatment capacity should be included in LCA of material recycling. However, the consequence in the long term is not avoided treatment through capacity release, but avoided construction of new capacity, as the market over time will adjust for a trend towards an increased material recovery.

3.3.4. Rebound effects

Previous reviews have shown that although prevention according to the EU waste framework directive (EU, 2008) should be the preferred alternative for waste management, this alternative is seldom included in LCAs of SWM-systems (Laurent et al., 2013). Prevention of food waste was however investigated by Martinez-Sanchez et al. (2016). Authors also included potential rebound effects associated with the affected (marginal) consumption when there is a difference in costs to consumers between alternative scenarios providing the same service. In their study, this was exemplified by increased consumption of other goods through less spending on food, when food waste is prevented on end-consumer level. According to their results, food waste prevention generates high welfare gains as more services/goods could be consumed with the same income, but this consumption could result in negative environmental impacts if additional money is spent on goods that are more environmentally damaging compared to the (prevented) food. However, saved money could also be used in products that are less polluting, but costlier compared to the pre-prevention situation. This makes assumptions of the consumption affected by the change of main importance.

In comparative LCAs, it is of key importance to maintain the same functional output from all compared systems. Thus, it is necessary to compensate any savings in monetary spending by including the consequences of a corresponding increase in marginal spending, and vice versa for an increase in cost (Brandão and Weidema, 2014). In this context, it is relevant to highlight that different treatment alternatives for SWM will result in different costs for society. However, such rebound effects are commonly not lifted in previous research in this field.

3.4. Handling of uncertainties

As stated by Björklund et al. (2003), the reliability of LCA is affected by dependence on a variety of data from different sources and more or less subjective methodological choices. When compared to attributional modelling, the uncertainties involved in the identification of affected processes have been identified as yet another source to uncertainties in CLCAs (Finnveden et al., 2009). Other authors argue that the error is greater in attributional compared to consequential LCA (Weidema et al., 2009). This statement implies that the use of historical average data in ALCA is a worse estimation of the actual environmental impacts related to a specific product or service, than the approach taken when using

consequential modelling. As stated by Ekvall and Weidema (2004), describing the consequences of decisions means facing the general challenge of futures studies, as the future is inherently uncertain, and the actual future consequences of decisions are highly uncertain. Thus, such uncertainties can commonly not be reduced much further, but at least illustrated.

Several authors have suggested the use of different scenarios based on various assumptions, which together can illustrate how consequences might change under different market situations (Ekvall and Weidema, 2004; Zamagni et al., 2012). The present review shows that the approaches taken by researchers in order to investigate the influence from uncertainties commonly is rather simplified.

The most common approach amongst reviewed studies is to investigate the sensitivity of gained results in relation to the identified affected technologies for energy production. More than half (56%) of the reviewed studies performed such sensitivity analyses, and in several of these, more than one alternative technology were investigated. Mathiesen et al. (2009) recommended use of sensitivity analyses including fundamentally different technology alternatives for energy production, and this was the approach was used by most authors. Coal in power plants and coal or oil for heat production were commonly used as a worst-case scenario, compared to natural gas, wind power or use of biomass (Bosch et al., 2014; Fruergaard and Astrup, 2011; Carlsson et al., 2015; Tonini et al., 2013). In one case, the technologies identified as affected by increased need for electricity production through changes in the waste management system, were compared to current average data (Seigné-Itoiz et al., 2015b). When waste-derived goods are used in the transportation sector, production and use of either diesel or gasoline were identified as affected processes in all cases. Only in one study was this assumption investigated through a sensitivity analysis, assuming substitution of a renewable fuel (biodiesel) (Fruergaard and Astrup, 2011).

Outcomes from these sensitivity analyses usually show a high sensitivity to the selection of affected technology for energy provision on overall contribution to GHG-emissions from different waste treatment alternatives. In general, selection of less polluting technologies discredits waste-to-energy technologies and promotes recycling (Manfredi et al., 2011; Tonini et al., 2013; Carlsson et al., 2015).

4. Conclusions

Consequential lifecycle assessments (CLCA) of solid waste management (SWM) systems is a growing area of research. Previous studies commonly do not present relevant justifications of the choices made in identification of processes and technologies affected by studied changes. At the same time, the present review clearly demonstrates a strong relation between the selection of affected energy production technology and overall GHG-emissions. Thus, increased transparency is needed in the identification of the technologies for energy production affected by changes studied.

Materials substituted by waste derived goods are in reviewed studies commonly represented by global or regional/national average data. This can cause biased results in studies where material recycling is compared to waste-to-energy. Thus, increased attention is needed to guarantee coherent consequential modelling of all parts of the investigated system in order to avoid biased comparisons between different waste management alternatives. The review also shows that increased attention should be directed to the investigation of uncertainties related to identification of material flows affected by studied changes.

In terms of system boundary setting choices, four main areas

were identified where current practice is diverse and where choices could have significant influence on overall results; counterfactual waste management, fate of materials avoided through material recycling, cascading effects and rebound effects. These aspects can all be of relevance to address in CLCA of SWM systems, but were included very scarcely in reviewed studies. This indicates that CLCA currently is reduced to choices made on process modelling level, while there is a need to pay more attention to choices made on system modelling level.

The large influence from choices made in selection of substituted products/energy carriers calls for increased collaboration between the LCA-community and economists, with the aim of increasing the quality of economic modelling of substitution processes. With this said, it should be remembered that markets for different products rapidly can be altered by political measures, such as trade tariffs etc. creating inherent uncertainties that cannot be reduced. Increased transparency and justification of made choices, as well as of the effect of the same on overall results, is however a first step towards an improved interpretation and usefulness of CLCAs as decision support tool.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <https://doi.org/10.1016/j.jclepro.2018.08.038>.

References

- Ahlgren, S., Björklund, A., Ekman, A., Karlsson, H., Berlin, Börjesson, P., Ekvall, T., Finnveden, G., Janssen och Strid, I., 2013. LCA of Biorefineries – Identification of Key Issues and Methodological Recommendations. Report No 2013:25, f3 The Swedish Knowledge Centre for Renewable Transportation Fuels, Sweden. Available at: www.f3centre.se.
- Azapagic, A., Clift, R., 1999. Life cycle assessment as a tool for improving process performance: a case study on boron products. *Int. J. Life Cycle Assess.* 4 (3), 133–142. <https://doi.org/10.1007/BF02979447>.
- Bernstad, A., la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste – A Swedish full-scale case study. *Waste Manag.* 31 (8), 1879–1896. <https://doi.org/10.1016/j.wasman.2011.02.026>.
- Björklund, A., Johansson, J., Nilsson, M., Eldh, P., Finnveden, G., 2003. Environmental Assessment of a Waste Incineration Tax—case Study and Evaluation of a Framework for Strategic Environmental Assessment. FOI—Swedish Defence Research Agency, Department of Environmental Strategies Research—fms (FOI MEMO Dnr 01-1526:5, Fms-Report 184, ISSN 1404-6520). Available on: www.infra.kth.se/fms.
- Boesch, M.E., Vadenbo, C., Saner, D., Huter, C., Hellweg, S., 2014. An LCA model for waste incineration enhanced with new technologies for metal recovery and application to the case of Switzerland. *Waste Manag.* 34, 378–389.
- Boldrin, A., et al., 2011. Modelling of environmental impacts from biological treatment of organic municipal waste in EASEWASTE. *Waste Manag.* 31, 619–630.
- Brandão, M., Weidema, B.P., 2014. What can LCA learn from economics?. In: Proceedings from the LCA XIV International Conference, October 6–8, 2014, San Francisco, CA, United States, pp. 37–42.
- Carlsson, M., Naroznova, I., Möller, J., Scheutz, C., Lagerkvist, A., 2015. Importance of food waste pre-treatment efficiency for global warming potential in life cycle assessment of anaerobic digestion systems. *Resour. Conserv. Recycl.* 102, 58–66.
- Cherubini, F., Ulgiati, S., 2010. Crop residues as raw materials for biorefinery systems – a LCA case study. *Appl. Energy* 87 (1), 47–57.
- Cimpan, C., Rothmann, M., Hamelin, L., Wenzel, H., 2015. Towards increased recycling of household waste: documenting cascading effects and material efficiency of commingled recyclables and biowaste collection. *J. Environ. Manag.* 157, 69–83.

- Cleary, J., 2009. Life cycle assessments of municipal solid waste management systems: a comparative analysis of selected peer-reviewed literature. *Environ. Int.* 35, 1256–1266.
- Curran, M.A., Mann, M., Norris, G., 2005. The international workshop on electricity data for life cycle inventories. *J. Clean. Prod.* 13 (8), 853–862.
- EIA, 2009. Annual Energy Outlook 2009: with Projections to 2030. U.S. Department of Energy EIA Office of Integrated Analysis and Forecasting. DOE/EIA-0383(2009). Available at: <http://www.eia.doe.gov/oiaf/aeo/>.
- Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. *Int. J. LCA* 9 (3), 161–171.
- Ekvall, T., et al., 2016. Attributional and consequential LCA in the ILCD handbook. *Int. J. Life Cycle Assess.* 21, 293–296.
- Eriksson, O., et al., 2009. Plastic waste as fuel – CO₂ neutral or not? *Energy Environ. Sci.* 2, 907–914.
- Eriksson, O., Finnveden, G., Ekvall, T., Björklund, A., 2007. Life cycle assessment of fuels for district heating: a comparison of waste incineration, biomass- and natural gas combustion. *Energy Pol.* 35, 1346–1362.
- Eunomia, 2013. Residual Waste Infrastructure Review a High-level Analysis Issue 5. Eunomia Research & Consulting Ltd, Bristol, UK.
- European Commission, 2010. International Reference Life Cycle Data System (ILCD) Handbook – General Guide for Life Cycle Assessment - Detailed Guidance. March 2010. EUR 24708 EN, first ed. Publications Office of the European Union, Luxembourg, LU.
- EU, 2008. Directive 2008/98/EC of the European parliament and of the council of 19 november 2008 on waste and repealing certain directive (waste framework directive). <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098>.
- Feraldi, R., Cashman, S., Huff, M., Raahauge, L., 2013. Comparative LCA of treatment options for US scrap tires: material recycling and tire-derived fuel combustion. *Int. J. Life Cycle Assess.* 18, 613–625.
- Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. *J. Environ. Manag.* 91 (1), 1–21.
- Frees, N., 2008. Crediting aluminium recycling in LCA by demand or by disposal. *Int. J. Life Cycle Assess.* 13 (3), 212–218. <https://doi.org/10.1065/lca2007.06.348>.
- Fruergaard, T., Astrup, T., 2011. Optimal utilization of waste-to-energy in an LCA perspective. *Waste Manag.* 31, 572–582.
- Gentil, E.C., Damgaard, A., Hauschild, M., Finnveden, G., Eriksson, O., Thorneioe, S., Kaplan, P.O., Barlaz, M., Muller, O., Matsui, Y., Li, R., Christensen, T.H., 2010. Models for waste life cycle assessment: review of technical assumptions. *Waste Manag.* 30, 2636–2648.
- Habert, G., 2013. A method for allocation according to the economic behaviour in the EU-ETS for by-products used in cement industry. *Int. J. Life Cycle Assess.* 18, 113–126.
- Hamelin, L., Naroznova, I., Wenzel, H., 2014. Environmental consequences of different carbon alternatives for increased manure-based biogas. *Appl. Energy* 114, 774–782.
- Hamelin, L., Wesns, M., Wenzel, H., Petersen, B.M., Wesnæs, M., 2011. Environmental consequences of future biogas technologies based on separated slurry. *Environ. Sci. Technol.* 45, 5869–5877.
- Hums, M.E., Cairncross, R.A., Spataro, S., 2016. Life-cycle assessment of biodiesel produced from grease trap waste. *Environ. Sci. Technol.* 50, 2718–2726.
- Johnson, J.X., Mcmillan, C.A., Keoleian, G.A., 2013. Evaluation of life cycle assessment recycling allocation methods: the case study of aluminum Johnson et al. Evaluation of LCA allocation methods. *J. Ind. Ecol.* 17, 700–711.
- Karlsson, H., Ahlgren, S., Strid, I., Hansson, P.A., 2015. Faba beans for biorefinery feedstock or feed? Greenhouse gas and energy balances of different applications. *Agric. Syst.* 141, 138–148.
- Kimming, M., Sundberg, C., Nordberg, Baky, A., Bernesson, S., Norén, O., Hansson, P.A., 2011. Life cycle assessment of energy self-sufficiency systems based on agricultural residues for organic arable farms. *Bioresour. Technol.* 102 (2), 1425–1432.
- Kua, H.W., 2015. Integrated policies to promote sustainable use of steel slag for construction—a consequential life cycle embodied energy and greenhouse gas emission perspective. *Energy Build.* 101, 133–143.
- Lazarevic, D., Aoustin, E., Buclet, N., Brandt, N., 2010. Plastic waste management in the context of a European recycling society: comparing results and uncertainties in a life cycle perspective. *Resour. Conserv. Recycl.* 55, 246–259.
- Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T.H., Hauschild, M.Z., 2013. Review of LCA studies of solid waste management systems – part I: key learnings and perspectives. *Waste Manag.* 34, 573–588.
- Manfredi, S., Tonini, D., Christensen, T.H., 2011. Environmental assessment of different management options for individual waste fractions by means of life-cycle assessment modelling. *Resour. Conserv. Recycl.* 55, 995–1004.
- Martinez-Sanchez, V., Tonini, D., Muller, F., Astrup, T.F., 2016. Life-cycle costing of food waste management in Denmark: importance of indirect effects. *Environ. Sci. Technol.* 50, 4513–4523.
- Mathiesen, B.V., Münster, M., Fruergaard, T., 2009. Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments. *J. Clean. Prod.* 17 (15), 1331–1338. <https://doi.org/10.1016/j.jclepro.2009.04.009>.
- Melamu, R., von Blottnitz, H., 2011. 2nd Generation biofuels a sure bet? A life cycle assessment of how things could go wrong. *J. Clean. Prod.* 19, 138–144.
- Merrild, H., Damgaard, A., Christensen, T.H., 2008. Life cycle assessment of waste paper management: the importance of technology data and system boundaries in assessing recycling and incineration. *Resour. Conserv. Recycl.* 52, 1391–1398.
- Merrild, H., Larsen, A.W., Christensen, T.H., 2012. Assessing recycling versus incineration of key materials in municipal waste: the importance of efficient energy recovery and transport distances. *Waste Manag.* 32, 1009–1018.
- Millward-Hopkins, J., Busch, J., Purnell, P., Zwirner, O., Velis, C.A., Brown, A., Hahladakis, J., Iacovidou, E., 2017. Fully integrated modelling for sustainability assessment of resource recovery from waste. *Sci. Total Environ.* 612, 613–624.
- Morris, J., Scott Matthews, H., Morawski, C., 2013. Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. *Waste Manag. (Oxford)* 33, 545–551.
- Nassar, N.T., Graedel, T.E., Harper, E.M., 2015. By-product metals are technologically essential but have problematic supply. *Sci. Adv.* 1 (3), 1400180. <https://doi.org/10.1126/sciadv.1400180>.
- Salemdeeb, R., zu Ermgassen, E.K.H.J., Kim, M.H., Balmford, A., Al-Tabbaa, A., 2017. Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options. *J. Clean. Prod.* 140, 871–880.
- Schmidt, J.H., Holm, P., Merrild, A., Christensen, P., 2007. Life cycle assessment of the waste hierarchy—a Danish case study on waste paper. *Waste Manag.* 27, 1519–1530.
- Seigné-Itoiz, E., Gasol, C.M., Rieradevall, J., Gabarrell, X., 2014. Environmental consequences of recycling aluminum old scrap in a global market. *Resour. Conserv. Recycl.* 89 (0), 94–103. <https://doi.org/10.1016/j.resconrec.2014.05.002>.
- Seigné-Itoiz, E., Gasol, C.M., Rieradevall, J., Gabarrell, X., 2015a. Contribution of plastic waste recovery to greenhouse gas (GHG) savings in Spain. *Waste Manag.* 46, 557–567.
- Seigné-Itoiz, E., Gasol, C.M., Rieradevall, J., Gabarrell, X., 2015b. Methodology of supporting decision-making of waste management with material flow analysis (MFA) and consequential life cycle assessment (CLCA): case study of waste paper recycling. *J. Clean. Prod.* 105, 253–262.
- Sonnemann, G., Vigon, B., 2011. Global guidance principles for life cycle assessment databases – a basis for greener processes and products. UNEP/SETAC. <http://www.unep.fr/shared/publications/pdf/DT1410xPA-GuidancePrinciplesforLCA.pdf>.
- Styles, D., Dominguez, E.M., Chadwick, D., 2016. Environmental balance of the of the UK biogas sector: an evaluation by consequential life cycle assessment. *Sci. Total Environ.* 560–561, 241–253.
- Suh, S., Yang, Y., 2014. On the uncanny capabilities of consequential LCA. *Int. J. Life Cycle Assess.* 18 (6), 1179–1184. <https://doi.org/10.1007/s11367-014-0739-9>.
- Sørensen, B.L., Wenzel, H., 2014. Life cycle assessment of alternative bedpans: a case of comparing disposable and reusable devices. *J. Clean. Prod.* 83 (November), 70–79. <https://doi.org/10.1016/j.jclepro.2014.07.022>.
- Thomassen, M.A., Dalgaard, R., Heijungs, R., DeBoer, I., 2008. Attributional and consequential LCA of milk production. *Int. J. Life Cycle Assess.* 13, 339–349.
- Tonini, D., Astrup, T., 2012. Life-cycle assessment of a waste refinery process for enzymatic treatment of municipal solid waste. *Waste Manag.* 32 (1), 165–176.
- Tonini, D., Martinez-Sanchez, V., Astrup, T.F., 2013. Material resources, energy, and nutrient recovery from waste: are waste refineries the solution for the future? *Environ. Sci. Technol.* 47, 8962–8969.
- Turconi, R., et al., 2011. Life cycle assessment of waste incineration in Denmark and Italy using two LCA models. *Waste Manag. Res.* 29, S78–S90.
- Turk, J., Cotic, Z., Mladenovic, A., Sajna, A., 2015. Environmental evaluation of green concretes versus conventional concrete by means of LCA. *Waste Manag.* 45, 194–205.
- Vázquez-Rowe, I., Rege, S., Marvuglia, A., et al., 2013. *Int. J. Life Cycle Assess.* 18, 1593. <https://doi.org/10.1007/s11367-013-0604-2>.
- Villanueva, A., Wenzel, H., 2007. Paper waste – recycling, incineration or land-filling? A review of existing life cycle assessments. *Waste Manag. (Oxford)* 27, S29–S46.
- Weidema, B.P., 2003. Market Information in Life Cycle Assessment. Danish Environmental Protection Agency, Copenhagen (Environmental Project no. 863). <http://www2.mst.dk/Udgiv/publications/2003/87-7972-991-6/pdf/87-7972-992-4.pdf>.
- Weidema, B.P., Ekvall, T., Heijungs, R., 2009. Guidelines for Applications of Deepened and Broadened LCA. Deliverable D18 of Work Package 5 of the CALCAS Project. www.leidenuniv.nl/cml/ssp/publications/calcas_report_d18.pdf.
- Weidema, B.P., 2009. Avoiding or ignoring uncertainty. *J. Ind. Ecol.* 13 (3), 354–356.
- Zamagni, A., Guinée, J., Heijungs, R., Masoni, P., Raggi, A., 2012. Lights and shadows in consequential LCA. *Int. J. Life Cycle Assess.* 17, 904–918.
- Zink, T., Maker, F., Geyer, R., Amirtharajah, R., Akella, V., 2014. Comparative life cycle assessment of smartphone reuse: repurposing vs. refurbishment. *Int. J. Life Cycle Assess.* 19, 1099–1109.

3.6 Artigo 6 – Estimating the generation of household e-waste in municipalities using primary data from surveys: A case study of Sao Jose dos Campos, Brazil*

* Abbondanza, M. N. M., Souza, R. G. Estimating the generation of household e-waste in municipalities using primary data from surveys: A case study of Sao Jose dos Campos, Brazil. *Waste Manag.* 2019; 85: 374–384. doi: n°. 10.1016/j.wasman.2018.12.040

ABSTRACT

Estimating the generation of Waste from Electrical and Electronic Equipment (WEEE) in municipalities is relevant for the adequate planning of take-back systems. However, current WEEE estimation studies have focused primarily at the country level, and relied mostly on secondary data from literature, especially when adopting discrete average lifespans for each type of appliance. This paper aims at developing and applying a WEEE estimation method in a Brazilian city, by obtaining primary data that reflects the differences in WEEE generation among the various social and economic profiles in the city. A survey with a stratified sample was carried out to collect representative data from each city Zone. Results show that lifespan distributions are most desirable for WEEE estimation studies than discrete averages, and that the lifespan profiles in the city are considerably different than previous values adopted in other Brazilian studies. Results also show significant variations of lifespan profiles for different types of WEEE among the different Zones of the city, which reflect the social and economic differences.

Keywords: *E-waste; Waste from Electrical and Electronic Equipment (WEEE); E-waste generation; WEEE generation.*



Estimating the generation of household e-waste in municipalities using primary data from surveys: A case study of Sao Jose dos Campos, Brazil



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ABSTRACT

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1. Introduction

Estimating the generation of Waste from Electrical and Electronic Equipment (WEEE), also known as E-waste, is crucial for the planning of take-back schemes aiming at the further recycling and adequate treatment of hazardous waste. Planning for WEEE take-back is often integrated with the Municipal Solid Waste Management (MSWM) systems, because Municipal Solid Waste (MSW) usually includes household WEEE. Municipalities also play a key role in enabling and maximizing WEEE collection by avoiding informal WEEE schemes, developing specific policies as well as educating local consumers (Gregory et al., 2009; UNEP, 2012; Baldé et al., 2015). Even under legal frameworks, such as the Extended Producer Responsibility (EPR), where municipalities are not directly responsible for WEEE take-back, the municipal territory is still the primary geographical scope for planning efficient e-waste reverse logistics (UNEP, 2012).

Although consistent estimates for WEEE generation in municipalities or regions are required to design proper WEEE take-back systems and assess their respective feasibility, most of the current studies are focused on estimates of WEEE generation at the national level. For example, the E-waste World Map (StEP, 2017)

estimated the annual WEEE generation (total and per capita) for all countries based on a correlation with the Purchasing Power Parity (PPP) of the population. Many other studies developed WEEE generation estimates at the national level. Some of those studies are concerned with specific types of appliances (eg: LCD – Lee and Cooper (2008); mobile phones – Polak and Drapalova (2012); lead acid batteries – Cabral Neto et al. (2016)); whilst other studies (eg: Araujo et al., 2012; Ikhlayel, 2016) approach several types of WEEE, and some just estimate the total amount of WEEE produced in the countries (ex.: StEP, 2017).

Estimations of WEEE generation at the national level do not provide much useful information for the planning of take-back schemes in municipalities and regions, because they do not detail the effects of different social and economic contexts on local WEEE generation (e.g. PPP variations on a same city or region). Estimating only the total amounts of WEEE also do not provide enough information for take-back planning, because the heterogeneity of existing electrical and electronic equipment (EEE) results in different compositions of recyclable or hazardous materials (Huisman et al., 2008), thus demanding specific collection and treatment strategies. For example, in order to evaluate the feasibility of local WEEE dismantling units, it is necessary to know the expected amount of WEEE generated in one year which will be received by each unit, and the respective composition of appliances types in terms of percentages (Spitzbart et al., 2016). The implementation

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of WEEE dismantling units is principally recommended at the local or regional levels due to the more efficient logistics, whilst further recycling and treatment infrastructures can cover a wider geographical range (Wang et al., 2012; De Souza et al., 2016; Spitzbart et al., 2016). Moreover, at least to size WEEE collection, which is the first stage of take-back systems, it is necessary to have accurate WEEE generation estimates obtained at the city level (Gregory et al., 2009; UNEP, 2012; Baldé et al., 2015).

The objective of this study is to develop and apply a method to estimate WEEE generation in municipalities, using primary data collected by applying a survey research to a stratified population sample. This study intends to add to other WEEE generation estimates studies, by providing updated and detailed data from Brazil, as well as by showing the advantages of adopting lifespan distributions obtained by survey research, instead of relying on discrete average lifespans obtained from literature.

Specific objectives:

- To define a reliable estimation method that does not depend on secondary data obtained from literature, and which is able to integrate different social and economic contexts and the respective WEEE generation in the different areas of a city;
- To test the proposed method by carrying out a case study in São José dos Campos, State of São Paulo, Brazil;
- Based on this case study, to describe the EEE consumption and WEEE lifespan profiles in São José dos Campos, producing information to serve as a reference for further WEEE estimation studies in the country;
- To estimate future WEEE generation in São José dos Campos in order to support WEEE take-back planning.

1.1. WEEE estimation methods and data availability

One of the main limitations for estimating WEEE generation in municipalities is the lack of proper methods and necessary local data. Some WEEE estimation studies adopted correlation methods that associate the amount of e-waste generated with the GDP per capita or the PPP of population (e.g. Huisman et al., 2008; StEP, 2017). On one hand, these studies require the previous availability of WEEE generation data in order to fit a regression equation that allows for further projections. On the other hand, the goodness of fit may not be sound enough to allow for trustable estimates (e.g. in Huisman et al. (2008) the coefficient R for WEEE/head $\times$ GDP/head was around 0.55 with data from some EU Member States).

Most WEEE estimation studies apply mass balance methods, which are recommended by UNEP (2007), and rely on data such as Products on Market (POM) in each year (i.e. the number of purchased appliances), current stocks (equipment in use) in year t , and lifespan profiles of each type of appliance (Wang et al., 2013). There are several available mass balance methods for WEEE estimates (Wang et al. 2013; Ikhlayel, 2016), and the choice of the most adequate method for each case depends, to a large extent, on the available data. Some of them vary slightly depending on the adopted variable; for example, the Carnegie Mellon method adopts a discrete average lifespan value for each appliance, rather than a lifespan distribution as adopted by the Market Supply method (Wang et al. 2013). For detailed information on the available WEEE estimation methods, please refer to Wang et al. (2013) and Ikhlayel (2016). More explanation on the Market Supply method is given in Section 2.2.

In addition to data availability, data quality is also a critical issue in WEEE estimates using mass balance models. Particularly regarding the lifespan variable, especially in developing countries where there is a lack of data, hence discrete average lifespan values are obtained from previous literature, reflecting the contexts of dif-

ferent countries in previous decades (e.g. Araujo et al., 2012; ABDI, 2013; Rodrigues et al., 2015).

1.1.1. Improving data availability and quality using surveys

Considering the drawbacks of WEEE estimates using traditional methods as described previously, surveys with population samples can be useful to determine not only local lifespan profiles, but also stocks (i.e. EEE in use), composition and average weight of appliances (Wang et al., 2013). The use of surveys has been a methodological tool in recent studies, such as Kim et al. (2013), who applied a survey to 1000 persons to determine the lifespan of EEE, supported by the Weibull distribution. The same study has also collected data regarding the number of existing appliances at households (stocks), by age of the appliances. Other studies that propose using surveys combined with the Weibull distribution are: Steubing et al. (2010) and Chirapat et al. (2012). The study carried out by Steubing et al. (2010) obtained a rather small sample of responses to their survey questionnaire ($n = 47$), considering that they aimed at obtaining residence times of appliances in households at the national level in Chile. On the other hand, Chirapat et al. (2012) were able to obtain a sample of 800 households to assess WEEE generation at the national level in Thailand. In both cases, the aim was to assess WEEE generation in the country, rather than in a particular city.

1.2. WEEE take-back framework and situation in Brazil

The Brazilian National Solid Waste Policy (PNRS) (Brasil, 2010a) established as mandatory the implementation of WEEE take-back schemes under the shared responsibility of EEE producers, importers, distributors and retailers, as well as the Brazilian Ministry of Environment (MMA). Those EEE market stakeholders must arrange Sectorial Agreements (SA), which are contracts formalized between the representatives of those sectors detailing the take-back scheme to be implemented in the country in terms of: responsibilities and roles, operation, technologies, financial management, integration with MSWM, among others. Such SA must be submitted to MMA, who will analyse, discuss, validate and sign the document, in order to allow for the implementation of the take-back system in the country. By late 2018, SA proposals were submitted to MMA, but not yet approved and implemented due to the lack of agreement on some critical points such as: the definition of WEEE hazardousness, legislation on transboundary transport of hazardous materials, framework for environmental licensing of WEEE management units, among others. The SA can also be formalized at the state or municipal levels, with the approval and engagement of state or local governments.

Nevertheless, the implementation of formal WEEE take-back systems in Brazil are most likely to occur progressively, starting from the biggest cities and states to the smallest ones. Such progressive implementation of WEEE take-back is recommended by the decree n° 7404/2010 (Brasil, 2010b), which regulates the PNRS. It was also considered in a preliminary feasibility study for WEEE take-back implementation in Brazil (ABDI, 2013), which was commissioned by the MMA and EEE market representatives.

Meanwhile, the WEEE situation in the country is poorly controlled, and remains characterised by either a strong participation of the informal recycling sector, and the mixing of WEEE with MSW being destined to landfills or open dumpsites. Adequate WEEE recycling rate in Brazil is estimated at only 2% (Bandini, 2009 apud Araújo et al., 2012). WEEE accounts for less than 1% of the waste collected by recyclables collection schemes (CEMPRE, 2016). Another common route for Brazilian WEEE is to be landfilled mixed with MSW. In the city of Rio de Janeiro, for example, 3.7 k tonnes of WEEE were destined to the urban landfill in 2012 (COMLURB, 2013), increasing to 8.6 tonnes of WEEE in 2016 (COMLURB, 2016). There

is strong evidence of WEEE being improperly recycled by the informal sector, using rough techniques like burning or acid leaching, and also of WEEE being dumped in the open (Ongondo et al., 2011; de Oliveira et al., 2012; De Souza et al., 2016). Other critical factors regarding WEEE management in Brazil are:

- Low coverage of separate recyclables collection: only 15% of the Brazilian population met this criterion in 2016 (CEMPRE, 2016);
- Inadequate waste disposal: In 60% of the cities, which correspond to 42% of the MSW in Brazil, the waste was disposed of on inadequate sites in 2015 (ABRELPE, 2016);
- Large-scale activity of informal waste pickers: It was estimated that there were around 7500 waste pickers in urban areas in 2008, of which 18% are in the State of Sao Paulo, and 5500 are children under 14 years old. Furthermore, 27% of the cities acknowledged the presence of waste pickers at the final waste disposal areas (IPEA, 2012).

1.3. WEEE generation estimates in Brazil

Few studies have focused on WEEE generation estimates in Brazil. Araujo et al. (2012) estimated the nationwide generation of some WEEE categories such as: TVs, refrigerators, freezers, washing machines, audio systems (mature markets); as well as computers and mobile phones (non-mature markets). For mature markets, they adopted the “leaching” model, whilst for non-mature markets the Time-Step model was used (please refer to Wang et al. (2013) for detailed explanation on those methods). The lifespan data adopted was obtained from diverse literature, whilst stock data was gathered from the national census.

The study commissioned by the Federal Government and carried out by ABDI (2013) is a technical reference for the feasibility analysis of WEEE take-back in Brazil. They adopted the Carnegie Mellon method (Market Supply with discrete average lifespan). POM data was acquired from industry and commerce representatives, however the lifespan averages do not correspond to actual data from Brazil, but rather to data collected from international literature references.

Rodrigues et al. (2015) applied a survey in the city of São Paulo in order to obtain data pertaining to the stock and age of appliances, but they still adopted lifespan averages from the international literature in past decades. Araujo et al. (2017) also applied a population survey to estimate WEEE in the Brazilian city of Fernando de Noronha obtaining a lot of useful data, but they also adopted lifespan averages from the international literature. Some discrete average lifespans adopted in those Brazilian studies are summarized in Table 1, all obtained from previous literature.

2. Materials and methods

2.1. Scope of the study

This study aims at developing WEEE lifespan and disposal profiles of the city of Sao Jose dos Campos, in the State of Sao Paulo, Brazil, with an estimated population of 695,992 inhabitants in 2016 (IBGE, 2017). This primary data must reflect the contexts of each of the seven major Zones of the city (Table 2).

Table 1
Some WEEE Lifespan averages (in years) adopted in Brazilian studies.

References	Refrigerator	Mobile phone	Washing machine
Araujo et al. (2012)	12	4	10
ABDI (2013)	15	3	11
Rodrigues et al. (2015)	11	4	11
Araujo et al. (2017)	8	2	7

The 16 types of EEE/WEEE appliances in the scope of the study were selected with the aim of covering a wide variety of what is considered to be the most common equipment at households: (a) Refrigerators/Freezers; (b) Air conditioners; (c) CRT TV; (d) LCD TV; (e) landline phones; (f) smartphones; (g) non-smartphone mobiles; (h) notebooks; (i) desktops; (j) LCD monitors; (k) CRT monitors; (l) computer keyboards; (m) washing machines; (n) basic printers; (o) multiuse printers; (p) radios.

2.2. WEEE estimation method

According to the procedural guideline for WEEE estimation developed by Wang et al. (2013), Market Supply (Eq. (1)) is the first recommended method that adopts a variable for lifespan profiles.

$$W(n) = \sum_{t=0}^n POM(t) \cdot L^p(t, n) \quad (1)$$

where: $W(n)$ = Amount (units of appliances) of WEEE generated in year n ; $POM(t)$ = Products on Market (acquired) in year t ; $L^p(t, n)$ = Probability (%) that a product acquired in year t will become waste in year n .

One advantage of this method is that the lifespan values are represented as probabilistic distributions, rather than discrete average values (as adopted in most of the studies cited in Section 1). This allows for a better representation of the various WEEE disposal ages which depend on social, technological and economic aspects. Thus, the lifespan variable in this model ($L^p(t, n)$ in Eq. (1)) is expressed as the probabilities (%) of each possible lifespan duration of appliances acquired in past years.

The other variable in this model is $POM(t)$, which represents the number of products acquired by the population in each year. Besides the historical data on the acquisition of products, this variable can also include estimates produced by market forecasting studies which focus on product acquisition in future years, thus providing a more comprehensive estimation of WEEE generation for the future with the use of the Market Supply method. This variable can be more easily quantified at the national level or at the level of organizations, where there is a higher control on EEE purchases (or sales) and imports. However, survey research is a recommended solution to collect data for such variables, as discussed in Section 1.1.

2.3. Survey and analysis method

In order to collect data for estimating WEEE generation in São José dos Campos, a survey research was carried out with a sample of the population. The survey was also useful to collect information on WEEE disposal behaviours. Each question was applied to each type of appliance listed in Section 2.1. The questions adopted in the research are listed in Appendix 1. It is worth noticing that Question 5 is for obtaining values for $POM(t)$ in the Market Supply Method, whilst Question 10 is for determining the lifespan distributions ($L^p(t, n)$) for each appliance.

In order to have an accurate representation of the social and economic profiles in the different municipal Zones (Table 2), the selected population sampling method were: stratified sampling to estimate a proportion of a finite population (Eq. (2)), and stratified sampling to estimate an average of a finite population (Eq. (3)). Both methods were selected because the survey form contains both quantitative and qualitative variables. The largest sample size between both Equations was adopted for each Zone.

$$n = \frac{\sum_{i=1}^k \left(\frac{N_i^2 \hat{p}_i (1 - \hat{p}_i)}{w_i} \right)}{N^2 D + \sum_{i=1}^k N_i \hat{p}_i (1 - \hat{p}_i)} \quad (2)$$

Table 2Population density in the Zones of Sao Jose dos Campos, Brazil, year 2015. Source: [Sao Jose dos Campos \(2016\)](#).

City zone	No. households	% Households	No. Population	% Population	Avg. family income	Municipal HDI
Center (ZC)	27,855	13.3%	80,779	11.7%	High	0.860
North (ZN)	19,702	9.5%	66,984	9.7%	Low	0.744
East (ZE)	53,039	25.3%	180,331	26.2%	Low	0.779
Southeast (ZSE)	14,251	6.9%	51,303	7.5%	Low	0.744
South (ZS)	76,939	37.4%	261,592	38.0%	Mid-Low	0.795
West (ZW)	14,874	7.3%	46,109	6.7%	High	0.850
S.F. Xavier district	486	0.2%	1504	0.2%	–	0.747
Total	207,146	100%	688,602	100%	–	0.787

$$n = \frac{\sum_{i=1}^k \left(\frac{N_i^2 \sigma_i^2}{w_i} \right)}{N^2 D + \sum_{i=1}^k N_i \sigma_i^2} \quad (3)$$

where

n = Sample size of the stratified zone

N_i = Population size of stratum i

σ_i^2 = Population variance of stratum i

$w_i = N_i/n$

$D = d^2/Z^2$, where d = sample error (adopted: 5%), and Z = abscissa of the Normal distribution

$\hat{p}_i$ = Estimate of the true proportion of stratum (when there is no previous estimate, adopt $\hat{p}_i = 0.50$ to obtain the highest possible sample size for i).

Eq. (2) was applied first because it does not depend on variance, with a sample error of 5%, which resulted in a sample size of 399 households. The survey began in August 2016 and was applied in each city zone alternately in order to adjust the sample proportions to the true proportions of the households in each particular zone (Table 2). After the first 100 responses, the variance values obtained were then applied to Eq. (3), adjusting the sample size to 401 households. After 8 months of survey with the work of six students using two tablets, a sample comprised of 459 interviews was obtained, with maximum variation of 3% from the true household proportions in Table 1. The S.F. Xavier District was not considered in this sample because its calculated sample size = 1 would not reflect a minimum acceptable statistical distribution.

3. Results and discussion

3.1. EEE lifespan profiles

Table 3 shows the results for sample sizes of discarded equipment regarding the average lifespans and respective standard deviations of each type of appliance obtained from a sample of 459 interviewed households. From Table 3, it is possible to notice that the equipment that is most discarded by the population since 2010 are smartphones, refrigerators or freezers, and washing machines. For this reason, further analyses of the results presented herein is focused on those aforementioned appliances. As the acquisition of second-hand goods, either from donation or direct negotiation with past users is a common practice in Brazil, Table 2 also distinguishes the lifespan profiles of the total sample (new + second-hand appliances acquired) from just new appliances acquired (as clarified by Question 6 in Section 2.3).

In comparison to Table 1, it is possible to notice that the lifespan averages currently adopted in Brazilian studies are often much higher than those obtained in Table 3, indicating that those previous studies are either adopting data that may not reflect Brazilian reality, or that they may have become obsolete. This highlights the importance of studies like the present one to produce representa-

tive local data. For instance, if such higher lifespan averages were to be adopted when planning WEEE take-back systems in Sao Jose dos Campos, that would lead to the underestimation of WEEE generation in the short-term, thus putting at risk the system efficiency and efficacy.

Another interesting observation derived from Table 3 is that smartphones have an average lifespan of less than two years, reflecting their fast obsolescence, and have been discarded at a rate of 2.8 appliances per household, indicating high usage and disposal rates by all household family members. On the other hand, the highest lifespans are those for CRT monitors (10 years, but for a very small sample), refrigerators and freezers (7.8 years), and CRT TVs (6.9 years).

Observing Table 3, it is also of foremost importance to highlight the relatively high standard deviations for lifespans of all appliances. This result reinforces the recommendation of preferably adopting probabilistic distributions for lifespans instead of discrete averages.

For the selected WEEE estimation method (Market Supply), the lifespan profiles are represented by a probabilistic distribution, i.e. the % of discarded appliances purchased in year t after 1, 2, ..., n years. In the present study, the lifespan probabilistic distributions obtained for some appliances are presented in Fig. 1. Please note that Fig. 1 distinguishes the lifespans of new acquired appliances from the total lifespan of both new and second-hand appliances (according to Question 6 in Section 2.3).

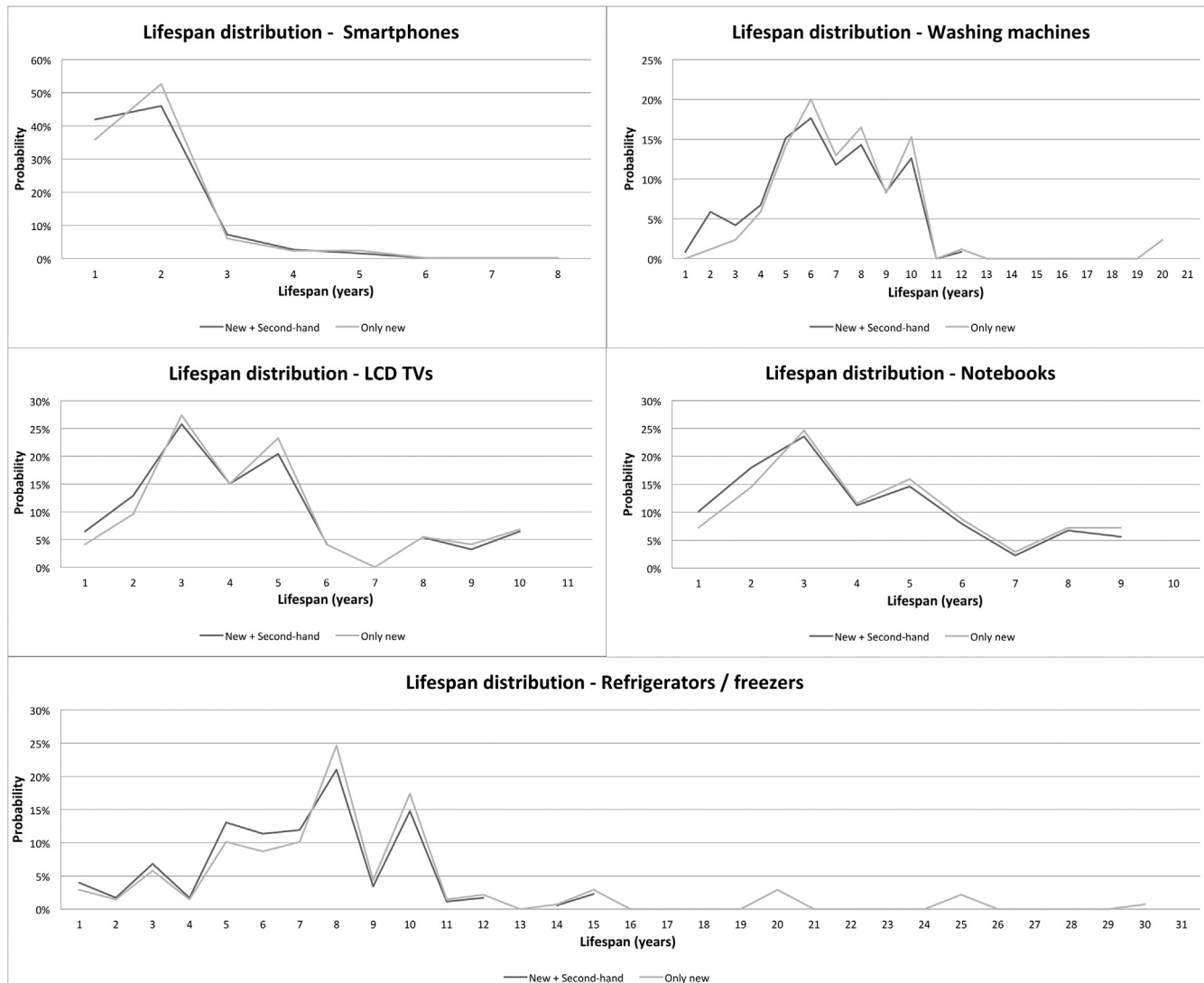
The lifespan profiles and disposal estimates of smartphones for each city zone are illustrated in Fig. 2 (where n^* represents the total number of discarded smartphones by the sample population in each city Zone). Fig. 2 shows that in some zones the lifespan of smartphones is mostly 1 year, whilst in most other zones is 2 years. Interestingly, the zones where the smartphones most likely have a 1-year lifespan are low-income areas (North and East, see Table 2), which may indicate that: both zones are main receivers of second-hand mobiles; high obsolescence and low-quality products acquired by consumers from these areas. Lifespan variations among the different zones also occurred to other types of appliances.

Obtaining other specific lifespan distributions like Fig. 2 for other types of appliances per each city Zone, with the current data survey, is limited by the small number of discarded appliances (n^*) in each Zone (e.g. discarded washing machines is $n^* = 7$ in the North Zone), what results in inexpressive distributions for such specific lifespans. Thus, besides the representative sample of total interviews (n), the number of households within the sample who had discarded determined appliances (n^*) in the 5-years period in a single Zone was low for most types of appliances. For stratified lifespan distributions per appliance and per Zone, the sample size n^* of discarded appliances is more relevant than the sample size n of total interviews. An ideal approach would be to iteratively recalculate sample size n until the sample size n^* of each type of appliance in each city Zone produces a statistically representative distribution, but this would lead to an unfeasibly high number of

Table 3

Sample sizes, average lifespans and standard deviations for each type of discarded appliance (from year 2010).

WEEE	Discarded since year 2010: % households	Avg. discarded/ household	New + Second-hand		Only new	
			Average lifespan (years)	Lifespan standard deviation	Average lifespan (years)	Lifespan standard deviation
Refrigerator/Freezer	28,3%	0.39	7.78	4.43	8.98	3.22
Air conditioner	0%	0	–	–	–	–
CRT TV	5.4%	0.08	6.94	2.51	7.24	2.36
LCD TV	13.7%	0.21	4.34	2.38	4.87	2.87
Landline phone	4,6%	0.12	3.30	2.49	3.39	2.79
Smartphone	71.7%	2.77	1.78	0.91	1.98	0.75
Non-Smart. mobile	5.2%	0.11	2.29	1.24	2.46	1.65
Notebook	13.7%	0.20	4.02	2.25	4.54	1.97
Desktop	2.0%	0.04	5.94	3.07	6.78	3.29
LCD monitor	1.7%	0.03	5.11	2.94	5.33	2.85
CRT monitor	0.2%	0.01	10	0	10	0
Keyboard	2.0%	0.04	5.93	3.07	5.94	3.07
Washing machine	25.1%	0.27	6.76	2.92	7.03	2.64
Basic printer	1.3%	0.04	2.94	1.39	3.22	1.13
Multiuse printer	3.9%	0.07	4.31	2.56	4.65	2.78
Radio	1.1%	0.02	4.14	0.69	4.68	0.44

**Fig. 1.** Lifespan distributions of some appliances.

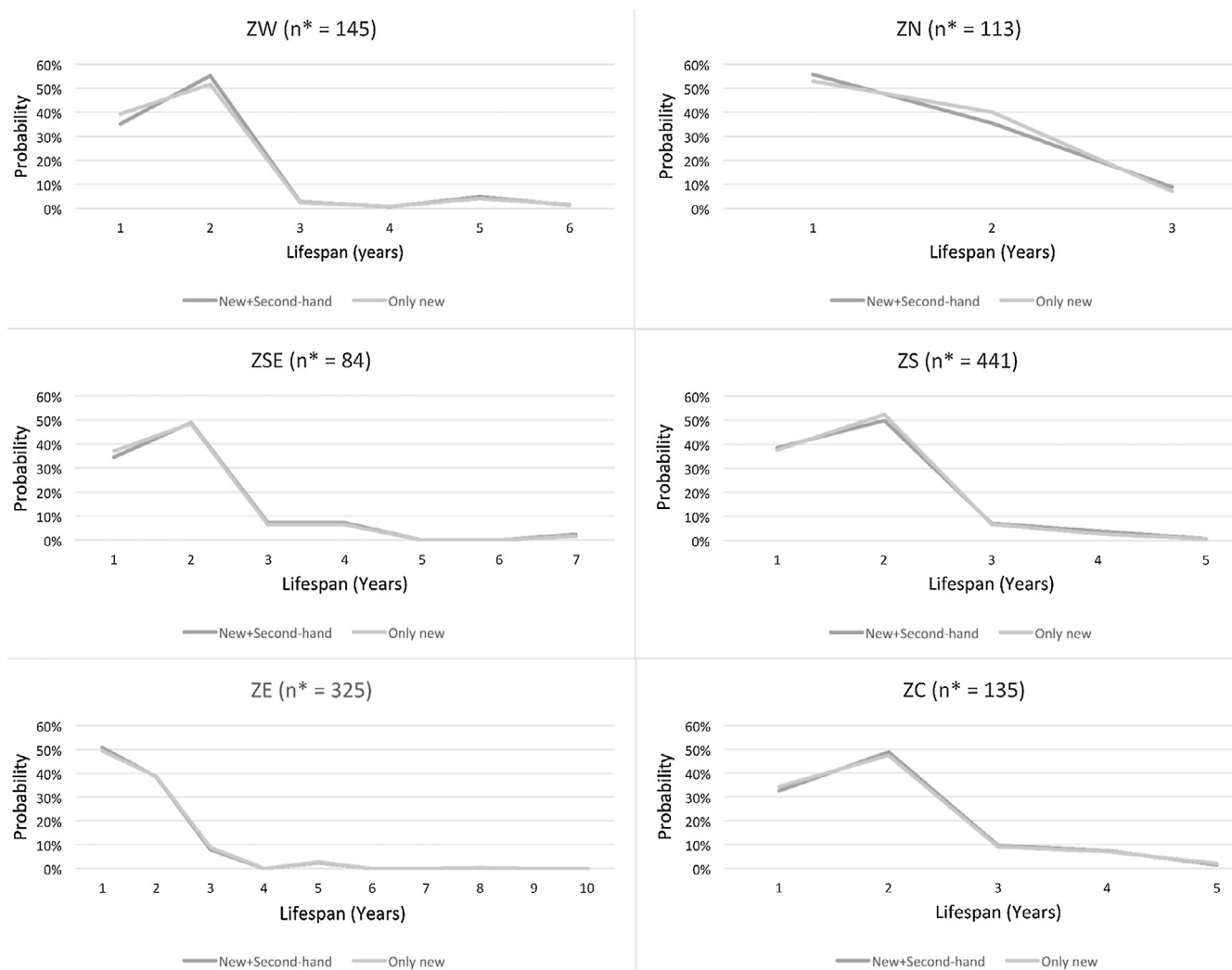


Fig. 2. Probabilistic lifespan profiles for smartphones in the different city Zones (n^* = number of appliances discarded by the sample households in the previous 5 years).

total interviews n . It is still necessary to check the likelihood of Figs. 1 and 2 distributions to correspond to known statistical distributions, like Normal, Weibull etc. Overall, the lifespan profiles in Figs. 1 and 2 can be a reference to other studies in Brazil or other locations with a similar social-economic characteristic that of São José dos Campos.

3.2. WEEE generation estimates

In order to estimate the WEEE generation from the current EEE in stock (acquired EEE as answered by population sample and projected to total households), the Market Supply method was applied, and the result for smartphones is presented in Fig. 3. It is necessary to highlight that the results in Fig. 3 only refer to products acquired or in stock (in use) until December 2016. Therefore, the adopted POM(t) values for the results in Fig. 3 are the ones obtained just from the survey research. This means that products acquired in following years (POM(t) for $t > 2016$) are not included in such analysis, as WEEE generation estimates for further years requires market projection studies, what is presented and discussed in Section 3.2.1.

In Fig. 3, it is possible to observe that smartphones waste generation increased exponentially reaching a peak in 2017, and this is possibly a trend for the following years, whenever specific POM(t) data from market studies in the city becomes available.

The estimated amount of approximately 167 k smartphones waste produced in 2017 in São José dos Campos corresponds to 1 waste equipment for every 4 citizens (Table 4), considering the estimated population of approx. 703 k people in 2017 (IBGE, 2017). In that year, the highest share of waste smartphones produced corresponds to appliances acquired in 2016 and 2015 (Fig. 3), in accordance with the probabilistic lifespan in Figs. 1 and 2.

Regarding the generation of waste refrigerators and freezers (Fig. 4), due to their lifespan profile, the highest expected amount of WEEE produced in the city from the POM(t) until 2016 will be in the year 2019, with the highest share corresponding to appliances acquired in 2009 (average lifespan = 10 years), followed by 2011 (average lifespan = 8 years), which are the peak probabilistic lifespans of such equipment, according to Fig. 1. Due to the sampling method adopted (Section 2.3), it is also possible to estimate the WEEE generation in the different city Zones (Fig. 5). This allows for a more detailed planning of take-back systems, e.g. the location and capacity of WEEE dismantling units, and WEEE collection routing.

With the estimates of WEEE generation for the most representative types of appliances in the sample (see Table 2), it is possible to determine the expected composition of WEEE generated in São José dos Campos in 2017, corresponding to a total of 4 kg/capita in the year (Table 4). This result is comparable to per capita averages for annual WEEE generation estimated by other Brazilian studies:

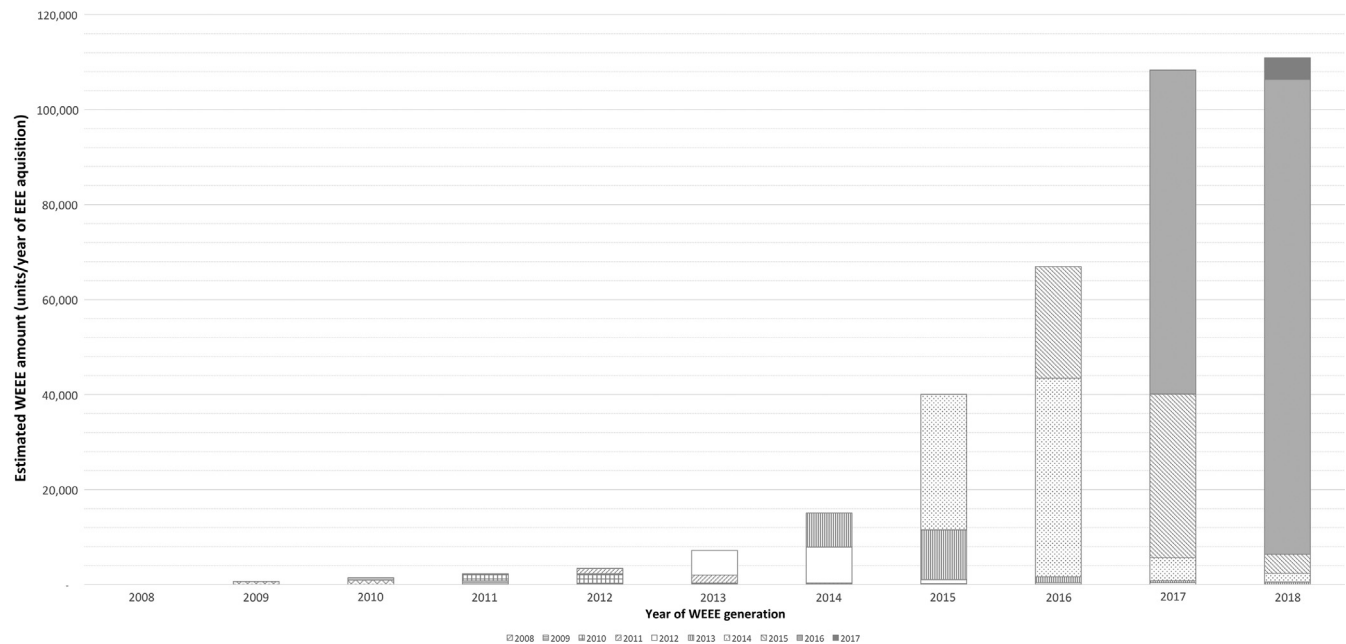


Fig. 3. Waste smartphones generation in São José dos Campos estimat based on POM(t) until 2016 (Note: x label = WEEE generation year; stacked columns = years of appliances acquisition).

Table 4

Estimated WEEE amount and per capita generation in 2017 in São José dos Campos.

WEEE	Estimated amount	Avg. weight (Kg) ^a	Amount WEEE/capita (units)	WEEE per Capita (Kg/inhab.)	WEEE per Capita (%w)
Smartphones	167,481	0.3	0.24	0.071	1.76%
Non-Smartphone mobiles	5202	0.3	0.01	0.002	0.05%
Landline phones	9032	0.2	0.01	0.003	0.07%
Desktop computers	6125	9.5	0.01	0.083	2.06%
LCD TVs	36,977	5	0.05	0.263	6.52%
CRT TVs	12,980	39	0.02	0.720	17.86%
Basic printers	3588	4.5	0.01	0.023	0.57%
Multiuse printers	11,738	4.5	0.02	0.075	1.86%
Refrigerators/freezers	19,969	56.5	0.03	1.604	39.78%
Washing machines	18,019	41.5	0.03	1.063	26.36%
Notebooks	31,379	2.8	0.04	0.125	3.10%
Total	322,490	–	–	4.032	99.99%

^a Average weights based on Spitzbart et al. (2016) and own Internet research by the Authors.

ranging from 2 to 3.1 kg per capita in São Paulo city (Rodrigues et al., 2015); and 3.77 kg per capita in the Brazilian average (Araujo et al., 2012). Thus, the figures in Table 4 can be useful for other WEEE estimation studies in Brazil or in countries with similar social-economical profiles to those present in the city studied in this paper.

It can be observed that, in absolute amounts of units, smartphones are the most generated WEEE type in 2017, whilst refrigerators/freezers and washing machines are the most representative in terms of weight. It is also possible to assess WEEE generation per capita in the different city Zones. Table 5 shows that, for the most common types of WEEE in this study, the West Zone (ZW), a high-income region of Sao Jose dos Campos, produces, in average, a much higher amount of WEEE than the other Zones.

3.2.1. WEEE generation estimates based on projections for POM(t) for the following years

Estimation of WEEE generation in São José dos Campos based on the data produced with the applied survey is limited to appliances purchased until year 2016, as discussed previously and shown in Figs. 3–5. In order to make projections of WEEE generation for the following years, considering not just the informed equipment

acquired by consumers until 2016, but also consumption projections for the next years, future market studies need to be carried out. With such market forecasts, it is possible to obtain values for the variable POM(t) in Eq. (1), with t representing not only past but also future years. Analysis of the applicable and most adequate market projection methods is beyond the scope of the present study. It is important to mention that such analysis also needs to take into consideration the differences between mature and non-mature markets (please refer to Araujo et al., 2012), what may also influence the tendencies for future years.

As previously commented, market projections for EEE consumption in the next years are currently not available neither in the municipal scale of São José dos Campos city, nor at the level of the State of São Paulo, or countrywide. This has been informed to the Authors by representatives of the Brazilian Association of EEE Industries (ABINEE), the Commerce Federation of the State of São Paulo (FECOMERCIO-SP), and the Board of Trade of São José dos Campos city. Therefore, the only possible ways to project the variable POM(t) for future years and thus estimate WEEE generation based on such products are: (1) to analyse and extrapolate available historical series of EEE consumption at the national level (ABINEE, 2017); or (2) to analyse and extrapolate EEE consumption

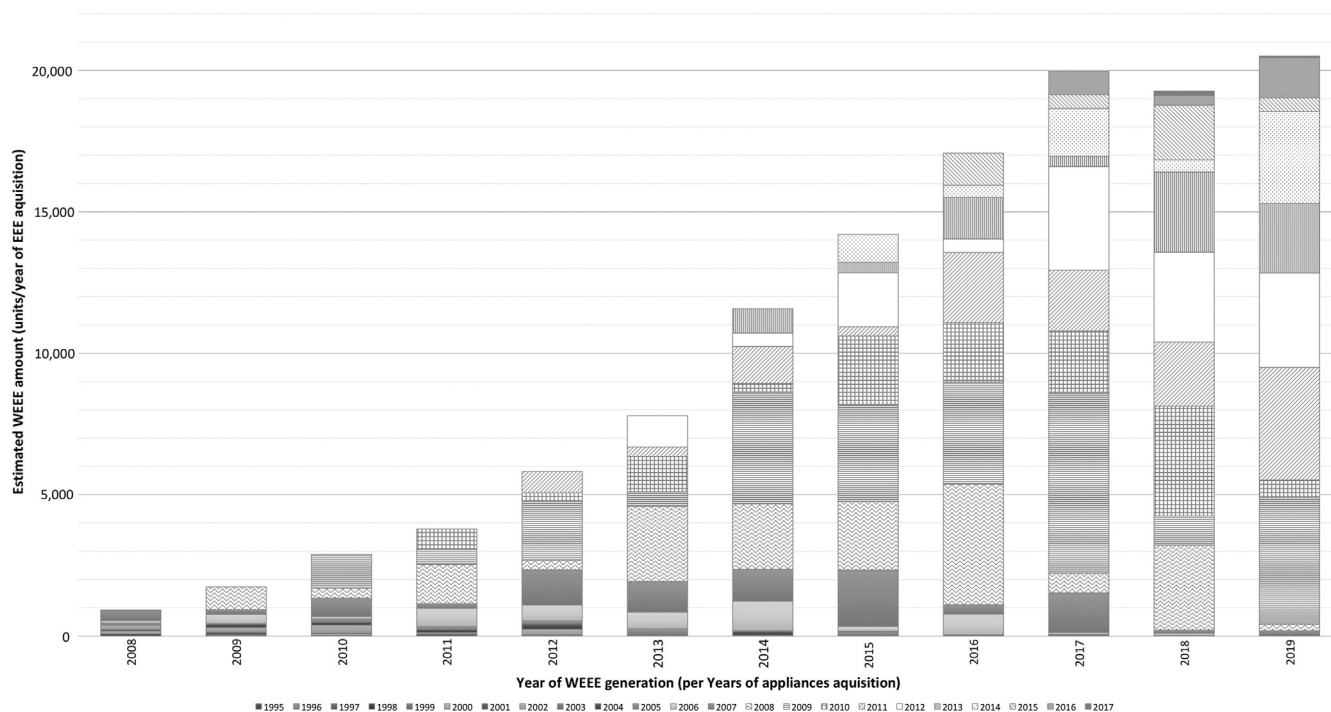


Fig. 4. Waste refrigerators/freezers generation in São José dos Campos estimated based on POM(t) until 2016 (Note: x label = WEEE generation year; stacked columns = years of appliances acquisition).

data obtained in the survey. In the first case, the following steps are necessary: (i) to determine the EEE consumption per capita in Brazil in each of the previous years; (ii) to apply a regression analysis in order to identify the function with best correlation index; and (iii) to extrapolate this to the estimated population growth of São José dos Campos city. A similar approach was applied to the data collected by the survey, considering appliances currently in use (in stock) and their respective years of purchase. The result of such analyses is presented in Fig. 6. It is possible to observe that WEEE generation estimates in each year can vary significantly, depending on the adopted approach for the POM(t) variable projections. This highlights the necessity of EEE market projections in Brazil as a support for WEEE management planning.

The weakness of the first approach, which extrapolates the historical series of EEE consumption countrywide to the particular context of São José dos Campos, is that the city may not follow the country's average consumption patterns. Considering its relative high GDP and purchasing power of the population (PPP), the tendency is that it should consume more EEE than the Brazilian average. The weakness of the second approach for POM(t) projections based on the survey data is that it comprehends a short number of previous years, and that the data is informed by the population, therefore being subject to uncertainties. In both cases, the approaches are based on future projections after a correlation and regression analysis with historical series data, what may not be the most adequate market projection technique. Nevertheless, the correlation indexes obtained for EEE consumption projections in both cases are relatively high (above 0.95). The Brazilian historical series by ABINEE can be a good potential source of data for studies in other Brazilian cities, or overall Brazilian WEEE generation projections for future years, once the most adequate market projection methods are analysed and selected.

Results shown in Fig. 6 indicate that the adopted market projection method can influence significantly the estimation of WEEE generated. The estimated EEE consumption for the city is much higher when considering the survey results than when adopting the National historical series and extrapolating it to the city under

study. Such variation ranges from approx. 0.15 to more than 0.5 smartphones per capita to be purchased in the city in 2025. This causes a large variation in the waste smartphones generation estimate for that same year, from less than 40 k to more than 160 k waste appliances.

The currently existing Brazilian studies on WEEE generation estimates did not make projections for future years, as they are restricted to estimations for a single year, thus not allowing for comparison. The estimated growth of waste mobile phones generation in Brazil in the period of 2000–2008 is around 0.35% (Araujo et al., 2012). The variation in waste smartphones generation in the present study for a similar interval, from 2014 to 2022, ranges from 2.4% to 9.1%. Several reasons can explain such differences: The adopted WEEE generation estimates method; the sources and quality of the data; the adopted lifespan profiles; the local profile of São José dos Campos compared to Brazilian average; and a different behaviour of the smartphones market (current study) compared to overall mobile phones market (Araujo et al., 2012).

3.3. WEEE disposal behaviours

The survey also sought to describe what is the behaviour of the population regarding WEEE disposal in São José dos Campos. Table 6 presents the results (total and per zone). It is worth noting that São José dos Campos is considered to be a benchmark for selective collection of recyclable waste schemes in Brazil, as in the last 10 years this service has had a coverage of 90%+ in the city (CEMPRE, 2016), which is by far higher than the Brazilian average coverage of 15%, indicating that citizens in this city have a historical tradition of separating waste. It can also be seen that the East Zone, a low-income region, has the highest disposal rate on the sidewalks, and the lowest delivery to the selective collection system. The highest rate of selective and special collection was in the North Zone, a high-income region in the city.

Table 7 presents the usual beneficiaries of second-hand WEEE donations. The most common ones are acquaintances (71% donations), followed by waste pickers (11%). The highest percentage

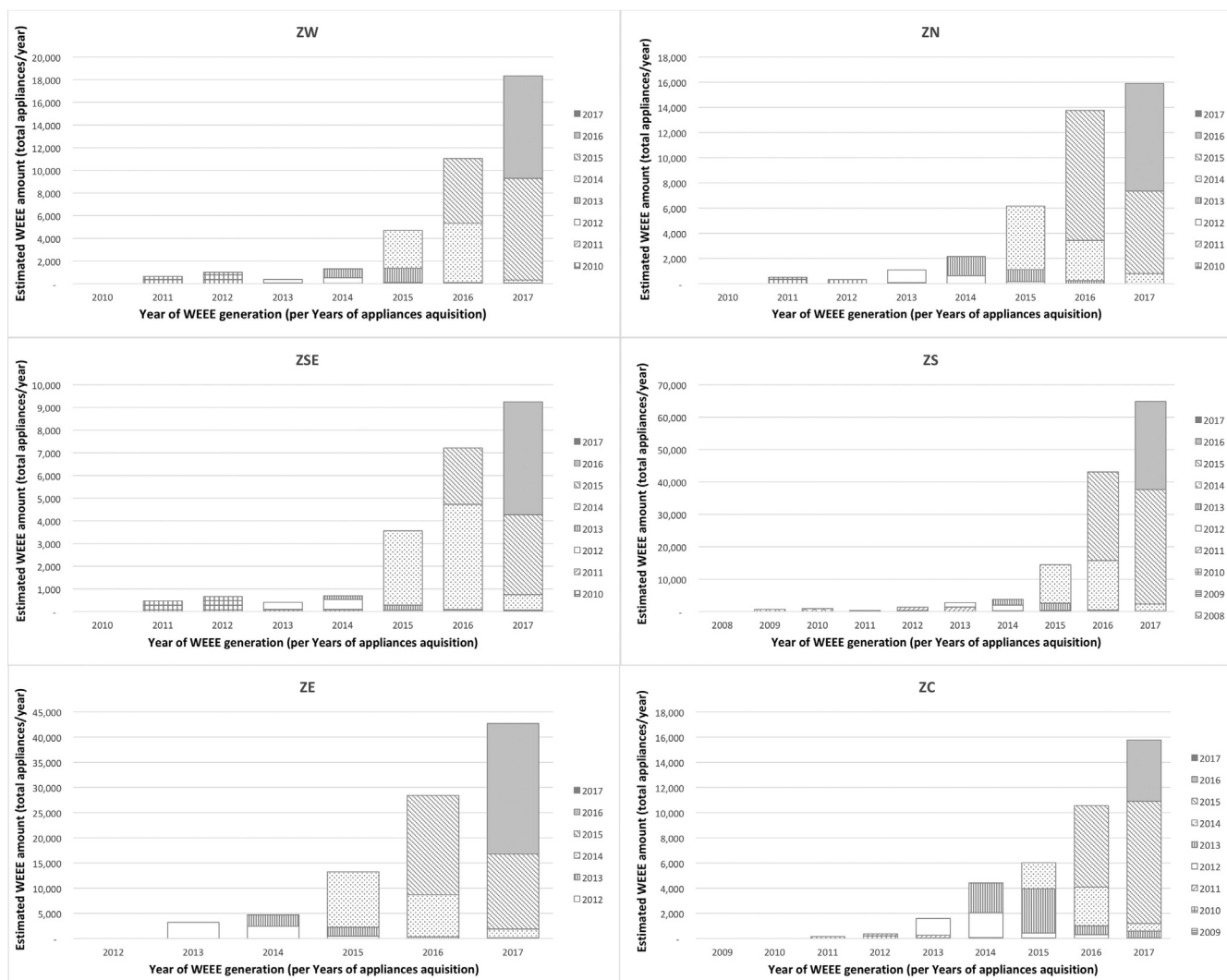


Fig. 5. Waste smartphones generation in different Zones of São José dos Campos estimated based on POM(t) until 2016 (Note: x label = WEEE generation year; stacked columns = years of appliances acquisition).

Table 5
Estimated WEEE generation (kg/capita) of most common appliances in 2017, by city Zones.

Zone	Smartphones	Refrigerators/freezers	Washing machines	Sum
Center (ZC)	0.057	2.215	0.995	3.267
North (ZN)	0.070	1.414	1.292	2.775
East (ZE)	0.070	2.026	1.120	3.216
Southeast (ZSE)	0.053	1.553	0.783	2.388
South (ZS)	0.073	1.491	0.991	2.555
West (ZW)	0.117	2.605	1.841	5.563

of donations to acquaintances was in the South Zone, a low-income area in the city.

4. Conclusions and recommendations

This study aimed at using a method to estimate WEEE generation in municipalities which takes into consideration the different social and economic profiles in the cities and not just relies on secondary data from literature, by means of applying a survey. In summary, we can conclude that:

- Considering the current Brazilian studies on WEEE generation estimates, which adopt discrete average lifespan values from previous literature, may lead to significantly different lifespan

profiles than the ones actually occurring in the regions of study, and thus mislead local WEEE generation estimates for future years; lifespan distributions from updated data are more representative than discrete averages from secondary data.

- Surveys are a resourceful tool to produce local primary data on EEE consumption, WEEE lifespan profiles and other variables of WEEE estimate models;
- Local market projections for EEE sales are needed to produce data for POM(n) variable, thus considering the future products acquisitions by citizens. This has a significant influence on WEEE generation estimates for future years, and cannot be neglected;
- The city of Sao Jose dos Campos showed significant variation in WEEE lifespan profiles and disposal behaviour among the differ-

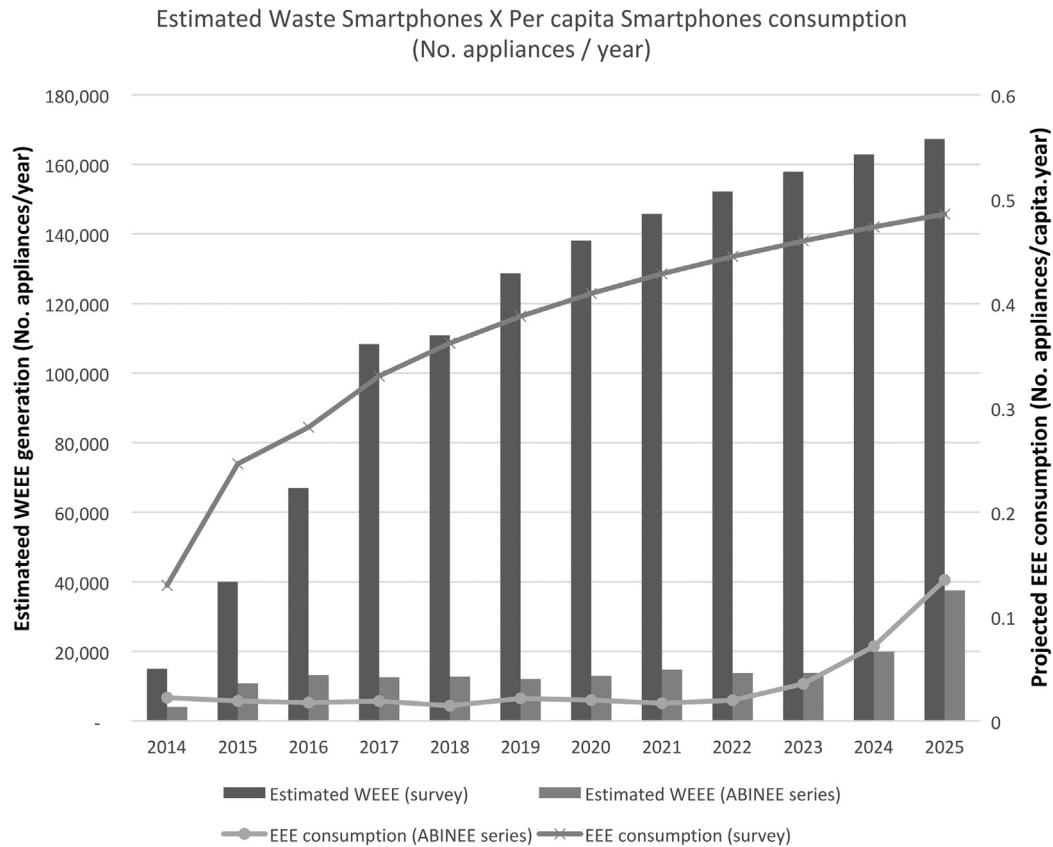


Fig. 6. WEEE generation estimates of Smartphones for future years, based on two approaches for POM(t) projections – regression from Brazilian EEE sales historical data, and from the survey data.

Table 6

WEEE disposal behaviour in S.J. dos Campos, Brazil (total and per Zones).

Disposal	Total	ZC	ZS	ZE	ZN	ZW	ZSE
Walkway	8.0%	5.9%	8.4%	12.7%	9.8%	3.4%	7.8%
Special WEEE collection	6.75%	8.1%	5.9%	7.5%	4.5%	8.9%	5.5%
Selective collection (door-to-door + drop-off)	75.5%	73.9%	77.5%	70.5%	79.3%	78.6%	73.4%
Mixed with MSW	5.8%	7.25%	5.6%	5.8%	4.2%	3.3%	8.6%
Others	3.9%	4.9%	2.6%	3.5%	2.2%	5.8%	4.5%

Table 7

WEEE donation receivers in S.J. dos Campos, Brazil (total and per Zones).

Beneficiaries	Total	ZC	ZS	ZE	ZN	ZW	ZSE
Waste pickers	11.0%	6.4%	9.3%	14.7%	9.2%	17.5%	8.9%
Acquaintances	71.3%	68.0%	74.8%	68.9%	71.3%	71.3%	73.5%
Cooperatives	4.2%	5.9%	3.9%	5.4%	3.2%	2.2%	4.3%
NGOs	9.5%	14.6%	9.1%	7.7%	9.9%	6.1%	9.9%
Others	4.0%	5.1%	2.9%	3.5%	6.4%	2.8%	3.5%

ent Zones, reflecting the influences of diverse social and economic aspects.

Some downsides of this approach are:

- The required effort and workforce to apply a survey comprehending a representative sample of the population, which may take several months;
- Uncertainties on the replies, as they rely on the memory of interviewees, which is especially critical for certain types of EEE that were acquired many years ago (e.g. refrigerators).

Further steps of this research are:

- To apply the Market Supply model using future market estimations for POM(n) in the next years;
- To check the probability of lifespan distributions against known statistical distributions (e.g. Weibull, Normal, etc.);
- To size and plan WEEE take-back schemes based on WEEE generation estimates in the city; and
- To estimate the WEEE recycling and hazardousness potentials - amounts of valuable and hazardous components in the WEEE composition - in the city.

5. Supplementary material

Additional material from this research is available in Portuguese at the link below. <https://goo.gl/f99EoF>. Password: WEEE_Brazil.

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Appendix 1

Survey Questions (original form is available as Supplementary Material):

- General questions:
 - (1) What municipal Zone?
 - (2) How many people living at household?
- Current stock (for each type of appliance mentioned in Section 2.1):
 - (3) How many do you actually have?
 - (4) How many are working?
 - (5) In which year did you acquire the appliances?
 - (6) How many were new when acquired?
 - (7) Do you intend to change, fix or donate any of them?
 - (8) When do you intend to change/fix/donate?
- Previous stock (for each type of appliance mentioned in Section 2.1):
 - (9) How many did you change or disposed of since the year 2010?
 - (10) How many years did it last?
 - (11) How many were working?
 - (12) How many did you change or dispose of since the year 2010?
- Disposal:
 - (13) Where do you usually dispose of WEEE? (on walkways; mixed MSW; selective collection; cooperative; special collection; others)
 - (14) To whom do you usually donate second-hand EEE? (Acquaintances; NGOs; waste pickers; recycling cooperatives; others).

References

- ABDI, 2012. Logística Reversa de Equipamentos Eletroeletrônicos Análise de Viabilidade Técnica e Econômica. Agência Bras. Desenvolv. Ind. - ABDI.
- ABINEE, 2017. Mercado Oficial de Telefones Celulares - Em Unidades - IDC. Available at: <<http://www.abinee.org.br>>. Access in: 28/09/2018.
- ABRELPE, 2016. Panorama dos Resíduos Sólidos no Brasil 2016. ABRELPE, São Paulo.
- Araújo, M.G., Magrini, A., Mahler, C.F., Bilitewski, B., 2012. A model for estimation of potential generation of waste electrical and electronic equipment in Brazil. Waste Manage. 32, 335–342. <https://doi.org/10.1016/j.wasman.2011.09.020>.
- Araujo, D.R.R., de Oliveira, J.D., Selva, V.F., Silva, M.M., Santos, S.M., 2017. Generation of domestic waste electrical and electronic equipment on Fernando de Noronha Island: qualitative and quantitative aspects. Environ. Sci. Pollut. Res. 24, 19703–19713. <https://doi.org/10.1007/s11356-017-9648-3>.
- Baldé, C.P., Wang, F., Kuehr, R., Huisman, J., 2015. The Global E-waste Monitor – 2014. United Nations University, IAS – SCYCLE, Bonn, Germany.
- Brasil, 2010a. Law No 12.305, 2010. Establishes the Brazilian National Solid Waste Policy.
- Brasil, 2010b. Decree No 7.404, 2010. Regulates Law No 12.305, 2010.
- Cabral Neto, J., Silva, M.M., Santos, S.M., 2016. A time series model for estimating the generation of lead acid battery scrap. Clean Technol. Environ. Policy 18, 1931–1943. <https://doi.org/10.1007/s10098-016-1121-3>.
- CEMPRE, 2016. Pesquisa Ciclossoft 2016. Available at: <<http://cempre.org.br/ciclossoft/id/8>>. CEMPRE.
- Chirapat, P., Kittinan, A., Kiattiporn, W., Nawanuch, T., Surus, T., 2012. Development of forecasting model for strategically planning on E-waste management in Thailand. Electronics Goes Green 2012+, ECG 2012 – Joint International Conference and Exhibition, Proceedings.
- de Oliveira, C.R., Bernardes, A.M., Gerbase, A.E., de Oliveira, C.R., Bernardes, A.M., Gerbase, A.E., 2012. Collection and recycling of electronic scrap: a worldwide overview and comparison with the Brazilian situation. Waste Manage. 32, 1592–1610. <https://doi.org/10.1016/j.wasman.2012.04.003>.
- COMLURB, 2013. Caracterizações Gravimétrica e Bacteriológica de Resíduos Sólidos Domiciliares da Cidade do Rio de Janeiro – 2013. Rio de Janeiro.
- COMLURB, 2016. Caracterizações Gravimétrica dos Resíduos Sólidos Domiciliares da Cidade do Rio de Janeiro – 2016. Vasa, Rio de Janeiro.
- Gregory, J., Magalini, F., Kuehr, R., Huisman, J., 2009. E-waste take-back system design and policy approaches. UNEP StEP 3576, 1–24.
- Huisman, J., Delgado, C., Magalini, F., Kuehr, R., 2008. 2008 Review of Directive 2002/96 on Waste Electrical and Electronic Equipment (WEEE). United Nations Univ, Bonn, Germany.
- Ikhlayel, M., 2016. Differences of methods to estimate generation of waste electrical and electronic equipment for developing countries: Jordan as a case study. Resour. Conserv. Recycl. 108, 134–139. <https://doi.org/10.1016/j.resconrec.2016.01.015>.
- IBGE, 2017. IBGE Cidades. Available at: <<http://www.cidades.ibge.gov.br>>. Access: 10/01/2018.
- IPEA, 2012. Diagnóstico sobre Catadores de Resíduos Sólidos: Relatório de Pesquisa. IPEA, Brasília.
- Kim, S., Oguchi, M., Yoshida, A., Terazono, A., 2013. Estimating the amount of WEEE generated in South Korea by using the population balance model. Waste Manage. 33, 474–483. <https://doi.org/10.1016/j.wasman.2012.07.011>.
- Lee, S.-J., Cooper, J., 2008. Estimating regional material flows for LCDs, in: 2008 IEEE International Symposium on Electronics and the Environment, IEEE International Symposium on Electronics and the Environment-ISEE. pp. 320–325.
- Ongondo, F.O., Williams, I.D., Cherrett, T.J., 2011. How are WEEE doing? A global review of the management of electrical and electronic wastes. Waste Manage. 31, 714–730. <https://doi.org/10.1016/j.wasman.2010.10.023>.
- Polak, M., Drapalova, L., 2012. Estimation of end of life mobile phones generation: the case study of the Czech Republic. WASTE Manage. 32, 1583–1591. <https://doi.org/10.1016/j.wasman.2012.03.028>.
- Rodrigues, A.C., Gunther, W.M.R., Boscov, M.E.G., 2015. Estimativa da geração de resíduos de equipamentos elétricos e eletrônicos de origem domiciliar: proposição de método e aplicação ao município de São Paulo, São Paulo, Brasil. Eng. Sanit. e Ambient. 20, 437–447. <https://doi.org/10.1590/S1413-41522015020000133701>.
- Sao Jose dos Campos, 2016. <<http://www.sjc.sp.gov.br/>>. Accessed in 10/01/2018.
- De Souza, R.G., Climaco, J.N., Sant’Anna, A.P., Rocha, T.B., Valle, R., Quelhas, O.L., 2016. Sustainability assessment and prioritisation of e-waste management options in Brazil. Waste Manage. <https://doi.org/10.1016/j.wasman.2016.01.034>.
- Spitzbart, M., Herbeck, E., Schluep, M., 2016. Business Plan Calculation Tool for Manual E-waste Dismantling Facilities. United Nations University/StEP Initiative.
- StEP, 2017. StEP E-Waste World Map [WWW Document]. URL <<http://www.step-initiative.org/step-e-waste-world-map.html>> (accessed 12.15.17).
- Steubing, B., Böni, H., Schluep, M., Silva, U., Ludwig, C., 2010. Assessing computer waste generation in Chile using material flow analysis. Waste Manage. 30, 473–482. <https://doi.org/10.1016/j.wasman.2009.09.007>.
- UNEP, 2007. E-waste Volume I: Inventory Assessment Manual. United Nations Environ. Program. 127. <<https://doi.org/DTI/0989/PA>>.
- UNEP, 2012. E-Waste Volume III: E-waste Take-back system”. Unep 146.
- Wang, F., Huisman, J., Meskers, C.E.M., Schluep, M., Stevels, A., Hagelüken, C., 2012. The Best-of-2-Worlds philosophy: developing local dismantling and global infrastructure network for sustainable e-waste treatment in emerging economies. Waste Manage. 32, 2134–2146. <https://doi.org/10.1016/j.wasman.2012.03.029>.
- Wang, F., Huisman, J., Stevels, A., Baldé, C.P., 2013. Enhancing e-waste estimates: improving data quality by multivariate input-output analysis. Waste Manage. 33, 2397–2407. <https://doi.org/10.1016/j.wasman.2013.07.005>.

3.7 Artigo 7 – Circular Economy and Solid Waste Management: Challenges and Opportunities in Brazil *

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ABSTRACT

Brazil has a vast territory and regional diversity, which turn the promotion of circular economy (or its improvement) into a complex challenge. This paper aims to analyse these challenges and solid waste management opportunities as a driving force for the circular economy in Brazil. Therefore, two significant challenges are discussed in this paper: increasing industrial waste recovery and improving the selective collection of municipal solid waste. Industries are concentrated in the country's South and Southeast regions, where 56% of the population live and where the best logistic infrastructure is also located. These features can help the circular economy related to industrial and municipal solid waste in those regions but bring challenges concerning the waste generated in the other part, such as the Brazilian Amazon. According to legislation, the productive sector is responsible for its products before and after use (waste prevention actions). One of the national laws establishes reverse logistic systems for several items, such as agrochemical packaging and electrical and electronic waste, but some sectors have better results than others in terms of circular economy. The cement industry in Brazil is highlighted as a case of relative success. On the other hand, the mining industry in Brazil has challenges related to the large amount of waste generated with high environmental and social risks, which caused several large accidents in the last years.

Keywords: Municipal solid waste; Industry; Recycling; Reverse Logistics; Amazon.



Circular Economy and Solid Waste Management: Challenges and Opportunities in Brazil

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Abstract

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Keywords Municipal solid waste · Industry · Recycling · Reverse logistics · Amazon

Introduction

The circular economy aims, among other aspects, to minimise the environmental, social and economic damages and impacts of solid waste as well as to minimise the waste itself.

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Therefore, governments and companies can apply integrated solid waste management concepts, especially the solid waste hierarchy [1, 2].

Waste prevention is the first option in the solid waste hierarchy and may enlarge the circular economy concept by defining more precisely how to deal in case of a non-waste status of a product through a life cycle thinking [3]. Products towards waste prevention and circular economy should be designed considering dematerialisation, longevity, durability, biocompatibility, interdependency and sharing uses. This conception brings several challenges for the whole society, from industry to consumer.

When prevention does not occur, integrated solid waste management can help to reduce waste. Nevertheless, it may require a complex system of procedures and actions to decrease the amount of waste produced and increase the recovering of the waste generated, leaving the small amount possible to final disposition [4]. This search for a circular economy in solid waste management may even mean interesting business opportunities.

Developing countries, such as Brazil, usually have a lot to improve towards a circular economy and therefore towards the best solid waste management practices. Like any other country, Brazil has its particularities that can be seen as challenges and opportunities to spread circular economy practices nationwide. These challenges and opportunities exist due to the country's natural, social, economic, cultural, technological and infrastructure diversities. This paper aims to present some of the main challenges and opportunities for the circular economy in solid waste management in Brazil, observed by the authors during years of research in the field.

Brazil's primary challenge is to develop the National Policy on Solid Waste (NPSW) in its full potential. The NPSW is an advanced and comprehensive Brazilian law from 2010 about solid waste [5], but the implementation and regulation of some of its guidelines are still being structured. The law has different aspects that make the circular economy's achievement easier. It states that reusable and recyclable solid waste is an economic good with a social value that can create jobs and income and promote citizenship (Article 6).

In the present paper, some aspects of the National Policy on Solid Waste that can contribute to the circular economy in solid waste management are addressed, highlighting Brazil's problems and strengths to increase the flux of materials to be recycled.

The first part of this paper presents the integrated solid waste management (Article 10 NPSW) to solve solid waste problems considering Brazil's political, economic, environmental, cultural and social dimensions. Different issues will be addressed, as the role of waste pickers' cooperatives in this Policy (Article 8 NPSW). The Brazilian government estimates 600,000 waste pickers in the country who collect recyclable waste in the streets and sell them back to the industries [6]. The association of these workers in cooperatives ensures them higher social protection and selling power when dealing with scrap buyers. Nevertheless, aspects that hinder the formal expansion of this activity will be explored.

The second part of this paper addresses aspects related to the industrial sector. In NPSW, the manufacturer is responsible for making products and packaging that cause less environmental impacts and are also easier to reuse or recycle (Articles 31 and 32 NPSW). Manufacturers, importers, distributors and retailers of six products must promote reverse logistics for their waste: pesticides and their packaging, batteries, tyres, lubricant oils and their packaging, fluorescent lamps, and electrical and electronic waste and their components (Article 33 NPSW). Two of them are detailed in this paper: (1) pesticide packaging, which was regulated in 2000, before NPSW and is well organised throughout the country, and (2) electrical and electronic waste, which was regulated only in 2020 and is in its very beginning. The cement

and mining industry are also presented, as positive and negative cases, respectively, of circular economy in solid waste management. The mining industry is under pressure to minimise the risks related to its waste, which caused severe accidents in the last years. Materials recycling in Brazil (mainly plastics) is also presented.

Part 1 — The Circular Economy and the Municipal Solid Waste Management in Brazil

The Potential of Solid Waste Management to Promote the Circular Economy in Brazil

Brazil generated 79 million tonnes of municipal solid waste (MSW) in 2018, representing $1.039 \text{ kg (person}\cdot\text{day)}^{-1}$. Only 2.2% of this waste were recovered in sorting facilities, whilst 24.4% went to inadequate final destinations, and 59.5% went to landfills operating in Brazil [7–9]. There are 621 operating landfills in Brazil, but only 22 (3.5%) have biogas recovery associated with electricity generation, reaching 179 MW of installed capacity (0.08% of the total, considering all electricity sources) [8, 10]. There was no information for the rest 13.9% of the MSW in Brazil, suggesting that its final destination was also inadequate [7–9]. Although the situation is not good, an improvement from 1989 data can be noticed when the first national survey about the subject was done. In that year, there was 0.9% of recycling, and the inadequate final disposition reached 87.8% of the total [11]

A comparison between the integrated solid waste management of developed countries with the one practised in Brazil was made by Oliveira Neto and Otávio [12]. The authors listed some of Brazil's problems and challenges to improve its solid waste management, increasing the recyclable material flow to industries and, therefore, promoting the circular economy:

- Recycling/composting: in 2014, Germany had 67.3% of composting/recycling, whilst in Brazil, the estimates for 2020 are 2.2% of recycling and 0.2% of composting [7, 13];
- Awareness: in Barcelona (Spain), the awareness campaign budget reached 7% of the total spent on the solid waste management of the municipality, i.e. the same percentage spent on municipal solid waste transport. In Brazil, the values invested in these campaigns are low, which helps to explain why municipal solid waste management does not improve substantially in the country [12];
- No waste-to-energy facilities in Brazil: in 2017, there were 492 waste-to-energy facilities only in Europe, which treated thermally 96 million tonnes of waste in that year [14];
- Diversion from landfills: from 1995 to 2018, Europe decreased the amount of municipal solid waste destined to landfills by 57% [15], employing recycling, composting and waste-to-energy in 74% of the MSW in 2018. In Brazil, as already mentioned, this percentage was 2.2% in 2020. Besides, 2408 open dumpsites were still operating in 2020 in Brazil [7]. The end of open dumpsites, at first scheduled to 2014 in the National Policy on Solid Waste, was postponed to 2024 [5, 16];
- No modern conceptions of integrated solid waste management, such as Ecoparc, in Barcelona, which treats about 240 thousand tonnes of organic matter from the selective collection and residual waste per year, producing 4 million m^3 of methane and 13 million kWh of electricity [17];
- Financial resources employed in solid waste management: the revenue with recycling, composting and waste-to-energy in countries with structured management systems is about

50% of the expenditure in this management, whilst there is no revenue in Brazil. The little sorting in the country demands public resources to work, and the meagre revenues belong to the waste pickers [12].

There is a strong relationship between the activities of an integrated municipal solid waste management system and the practices of the circular economy, divided into technical and biological cycles according to Fig. 1, adapted from Paes et al. [18]. It can be seen that consumers and the government can contribute to the recovery of materials to new cycles (biological and technical), postponing the end of their life cycle and their final disposition, and promoting their return to the economy.

According to official estimates, the municipal solid waste in Brazil comprises 51.4% of organic matter (biological cycles), 2.9% of metal, 13.1 of paper, 13.5% of plastic and 2.4% of glass (technical cycles), as well as 16.7% of other materials [7, 18]. Therefore, 83.3% of the waste can be recovered via biological (composting or anaerobic digestion) and technical approaches (recovered, repaired, reused, remanufactured or recycled).

Despite a large amount of organic matter discarded in the Brazilian municipal solid waste, the recovery is incipient. Only 127,000 tonnes of municipal solid waste (0.16% of the MSW) went to composting centres in 2018 in Brazil [7], and in 1989, this number was 351,000 tonnes (1%) [11]. It should be considered that the Brazilian population almost doubled from 1989 to 2018, which means a considerable reduction in composting as a final destination of MSW in the country. Some composting centres work with organic matter from specific clients (such as industries), but the full data are not available. There is no record of anaerobic digestion from MSW in the country.

Ferraz et al. [19] presented an example of how the integrated solid waste management can help develop the circular economy on a municipal scale. The authors follow the changes in the

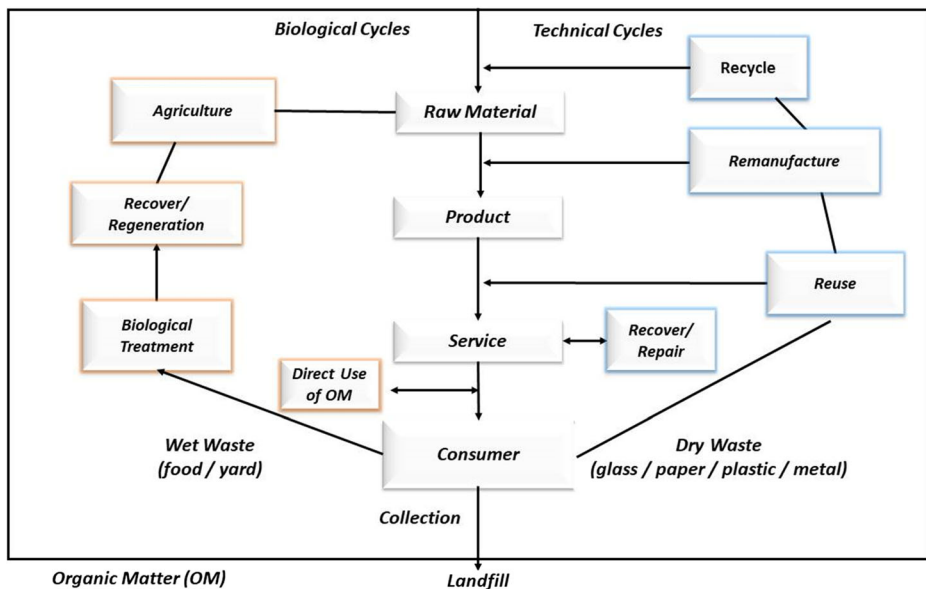


Fig. 1 Interactions between the concepts of integrated solid waste management and circular economy. Adapted from Paes et al. [18]

municipal solid waste management of a Brazilian municipality over 5 years. In this period, the index that assessed only the municipal solid waste sorting and treatment increased from 0.26 to 0.52 (the maximum value possible was 1). This improvement was due to some measures taken by the municipality administration, such as renting a new warehouse for the waste pickers' cooperative (which was 50% larger than the last one) and investments in health and safety of the waste pickers, materials storage and the selective collection of the municipality.

Contrasts of an Developing Country with Continental Dimensions to Promote Circular Economy Using Integrated Municipal Solid Waste Management

Brazil is a country with many and marked social, economic, cultural, demographic and environmental contrasts. These contrasts can be barriers to municipalities' solid waste management to increase the amount of waste recovered and, therefore, promote the circular economy.

Brazilian territory has an area of 8.5 million km², divided into 26 states and the Federal District, where is the capital, Brasília (Fig. 2). The country's population in 2020 was estimated at 212 million inhabitants [20]

Two regions will be highlighted since their contrasts can influence solid waste management and the circular economy: North and Southeast regions. The North region presents the largest area, 3.85 million km² or 45.2% of the Brazilian territory. This region has around 18.7 million inhabitants or 8.8% of the Brazilian population, which means a population density of 4.8 inhabitants per square kilometre. Seven states comprise the North region: Acre (AC), Amazonas (AM), Amapá (AP), Pará (PA), Rondônia (RO), Roraima (RR) and Tocantins (TO) (Fig. 1). The recyclable materials market in the north region is limited to the cities of Manaus (the

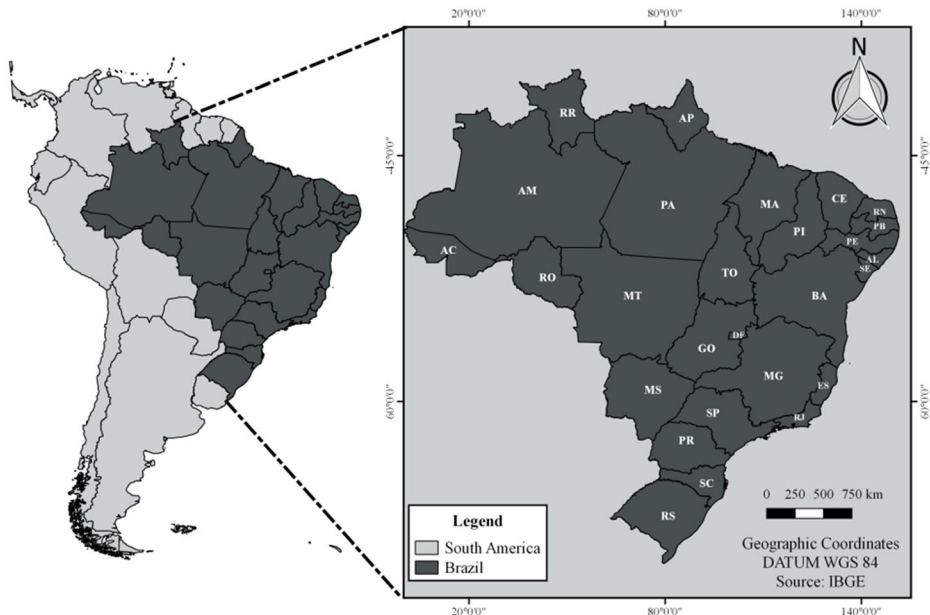


Fig. 2 Political map of Brazil, divided into 26 states and the Federal District (DF) [21]

capital of the Amazonas State) and Belém (the capital of the Pará State), the largest cities of the region with more than one million inhabitants each [20].

The Southeast region comprises the States of São Paulo (SP), Minas Gerais (MG), Rio de Janeiro (RJ) and Espírito Santo (ES) (Fig. 1). This region has an area of 925,000 km² or 10.9% of the national territory. In the Southeast region live around 89 million people, or 42% of the Brazilian population, i.e. 96.3 inhabitants per square kilometre [20].

Therefore, the first contrast that affects Brazil's circular economy is the difference in the population density, which is related to the municipal solid waste generation and the materials that the industries can recover. The municipalities' average area in the North region is 8577 km², varying from 103 to 159,533 km². In the Southeast region, the municipalities' average area is 554 km², varying from 3.6 to 10,727 km² [22]. This reality influences the municipal solid waste logistic due to the solid waste's geographic spread in each municipality territory.

Official data about solid waste management, available on The National Survey of Information on Sanitation (SNIS in Portuguese), only consider the urban population. However, this population represents 85% of all country's total population, and 9 % has no solid waste collection [8]. Based on the last published report of SNIS, in the Brazilian North region, the amount of municipal solid waste collected in 2018 reached 1.05 kg (person·day)⁻¹ or 5.1 million tonnes, which means 8.2% of all MSW collected in the country. In the Southeast region, these numbers were 0.92 kg (person·day)⁻¹ and 27.4 million tonnes in 2018, i.e. 43.2% of the municipal solid waste collected in Brazil in that year.

Information related to municipal solid waste management and of great interest to the circular economy is presented in Table 1. This information was obtained from 2012 to 2019 in the SNIS database, and the selected municipalities were those that uploaded their data about MSW management in this period voluntarily and more consistently. The nomenclature of the indicators follows the one adopted by SNIS, and the most important for circular economy purposes is IN031, which is the recovery rate of dry recyclable materials in relation to the total MSW collected.

The results in Table 1 show that the IN031 indicator grows during the period considered in both regions, except for 2016, probably due to the worsening of Brazil's economic and political crisis. From 2012 to 2019, IN031 grew 200% in the North region and 74% in the Southeast region. The numbers in the Southeast region are better due to different factors, including selective collection schemes in the municipalities, an essential raw material source to promote circular economy from municipal solid waste.

Only 24 municipalities of the North region (10.3% of all municipalities in the region) had a selective collection in 2018, reaching 11.8% of the urban population at most. In the Southeast region, 473 municipalities claimed to have selective collection schemes (39.4%), reaching 42.8% of the urban population at most [8]. The fact that a selective collection scheme exists does not mean that all the population is reached. Fehr et al. [24] concluded that no Brazilian municipality diverted from landfill more than 15% of the total municipal solid waste mass, even though several models of selective collection have been tested.

Different factors contribute to this contrast in selective collection numbers, besides those related to population density, already mentioned. These factors are related to logistic infrastructure, weather conditions, distance to the recycling markets and environmental education, among others [18]. However, there is another important territorial aspect to be considered in the circular economy, especially in Brazil: the lands destined to preserve the way of life and the culture of the Indigenous groups. In Brazil, Indigenous reservations have 1.06 million km², and 87% of these lands are in the North region (only 0.1% in the Southeast region). About

Table 1 Municipal solid waste management indicators of Brazilian North and Southeast regions, based on information voluntarily uploaded by the municipalities [23]

Indicator	Unit	2012	2013	2014	2015	2016	2017	2018	2019
North region									
CO116	t	917,316	1,011,169	1,277,644	1,235,651	1,178,741	1,052,656	1,053,071	1,188,610
CO117	t	2,513,288	2,745,458	2,702,527	2,776,929	2,564,558	2,759,866	2,819,462	2,323,907
CO142	t	129,944	59,880	70,416	80,611	49,039	191,957	61,001	35,173
CS048	t	5,369	49,028	19,338	40,434	18,723	32,731	23,405	37,847
CS009	t	11,710	27,428	24,622	41,574	26,362	28,360	32,361	35,879
IN031	%	0.33	0.71	0.60	1.01	0.69	0.70	0.82	1.00
Southeast region									
CO116	t	7,154,298	7,394,124	7,561,710	8,786,857	7,656,211	7,456,089	7,050,754	7,961,560
CO117	t	15,522,405	16,128,437	18,313,979	15,846,039	16,207,102	16,476,157	17,325,149	17,372,630
CO142	t	588,778	729,382	786,932	390,737	288,284	338,464	445,337	611,897
CS048	t	275,640	273,765	354,219	249,512	199,610	260,709	226,319	277,545
CS009	t	209,320	319,380	401,870	358,268	321,298	395,994	393,572	406,360
IN031	%	0.89	1.30	1.49	1.42	1.32	1.61	1.57	1.55

Source: SNIS database [23]. IN031: recovery rate of dry recyclable materials in relation to the total MSW collected by the municipalities; CS009 total amount of dry recyclable materials recovered; CO116 total amount of MSW collected by the public agent; CO117 total amount of MSW collected by particular agents; CO142 total amount of MSW collected by other agents; CS048 total quantity of MSW collected in the selective collection performed by associations or cooperatives of waste pickers with support of the municipalities

24% of the North region's territory comprise Indigenous reservations, which have some restrictions to infrastructure works such as roads and waterways [25].

The logistic structure also affects Brazil's circular economy since almost all transport in Brazil happens by roads. There are 140,028 km of roads in the North region (0.036 km of road/km²). However, only 22,389 are paved (0.006 km/km²), and just 22% of these paved roads (5000 km) were classified as good or very good. The Southeast region has 533,795 km of roads or 0.57 km of roads/km², and 62,520 km is paved (0.06 km·km⁻²). Besides, 34,386 km of paved roads (55%) is considered good or very good [26]. This scenario explains the best conditions for transporting recyclable materials in the Southeast region compared to the North region.

The North region has the highest annual average precipitation in the country (1900 to 3100 mm per year) [27], which impairs the road maintenance (especially those non-paved) and, therefore, the transport of solid waste. This is one reason why transport by rivers is much more common in the northern region than in the Southeast region.

Logistics limitations make unfeasible the materials flow from some municipalities in the North region to the recycling markets of Manaus (AM) and Belém (PA) Pará, according to Oliveira et al. [28]. Part of the recyclable material collected in the States of Acre and Rondônia and in the south of the State of Amazonas is sent, by road, to the recycling market in São Paulo, in the Southeast region approximately 3000 km away [28]. In São Paulo, the prices of some materials are better than those in the Manaus market. In July 2019, aluminium was sold in Manaus for US\$ 1.42/kg, cardboard for US\$ 0.06/kg, paper for US\$ 0.12/kg and PET for US\$ 0.22/kg. In São Paulo city, also in July 2019, aluminium was sold for US\$ 1.39/kg, cardboard and paper for US\$ 0.17/kg and PET for US\$ 0.66/kg. Glass and long-life packaging are also traded in São Paulo, whilst there is no market for these materials in Manaus.

The higher number of waste pickers' cooperatives is another contrast between the two regions. As already mentioned in 'Introduction', this social group has vital importance in collecting and sorting recyclable materials in the country. Brazil has 1232 waste pickers' cooperatives, being 4.3% in the North region and 41.6% in the Southeast region. The better structure of the recycling market and the higher industries' concentration in the Southeast region help to explain this situation [8].

Although the number of cooperatives in the North region is small, they diverted from landfill 38,000 tonnes of MSW in 2019, according to data in Table 1 (CS048 indicator). This amount represents 1% of municipalities' total waste; the same percentage diverted from landfill by the Southeast region's cooperatives. However, the amount diverted by cooperatives in the Southeast represents 2/3 of all dry recyclable waste sent for recycling from MSW in this region. Therefore, the development of the recycling market, still incipient in the North region, is an important driving force for the management of municipal solid waste and the circular economy.

Figure 3 compares the evolution of the IN031 indicator—the recovery rate of dry recyclable materials in relation to the total MSW collected—in the States of Amazonas (AM) and São Paulo (SP). Both states show a growth trend in the indicator, which is higher in the State of Amazonas (0.16% per year) than in São Paulo (0.09% per year). Although the recycling market is restricted to the city of Manaus, the selective collection carried out by cooperatives and supported by the city government (CS048) was the factor that influenced most the growth of IN031 in the period. In 2012, there were no records of a selective collection in Amazonas, whilst in 2019, 11,946 tons of dry recyclable materials were collected by waste pickers' cooperatives, of which 11,900 tons (99.6%)

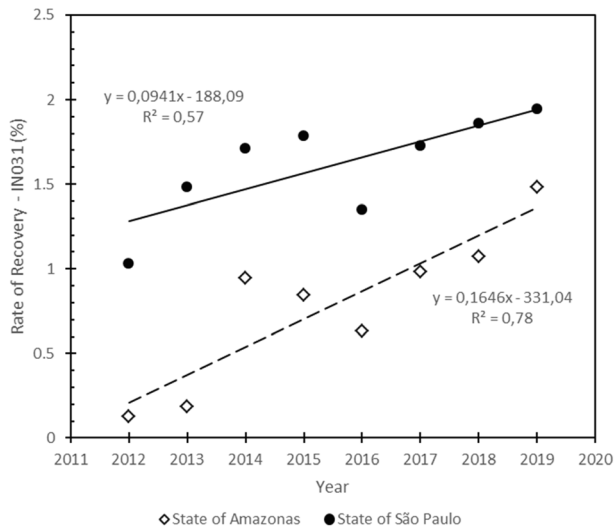


Fig. 3 Rate of recovery of dry recyclable materials in relation to the total MSW collected by municipalities in the States of Amazonas and São Paulo from 2012 to 2019

only in the metropolitan region of Manaus. This evolution can be due to the National Policy on Solid Waste (NSPW), enacted in 2010.

Brazil's contrasts show the difficulties of the North region regarding shared responsibility, reverse logistics and circular economy. Paes et al. [29] suggested that the lack of logistic structure and the low industrialisation in this region explain the higher percentages of recyclable waste (paper, plastic and metal) disposed of in landfills or open dumpsites.

At the present moment, a transition to the circular economy in Brazil is easier to happen in the Southeast and South regions (States of Paraná [PR], Santa Catarina [SC] and Rio Grande do Sul [RS]), since these regions are the most developed and industrialised in the country. However, even in more developed states and cities, there are significant economic and social diversities that could stimulate new consumption and discard patterns, improving the access of all the Brazilian population to products and diminishing the amount of waste.

Paes et al. [18] suggest public policies and solid waste management practices, such as integrating the informal sector in the recycling chain and services related to repair clothes, shoes, furniture and electronic devices. A large part of the efforts towards a circular economy in municipal solid waste management in Brazil is made by the informal sector, which has a strong presence in Brazilian cities. The authors also suggest home and community composting to prevent waste and to decentralise food production. These actions could be stimulated by legal and financial instruments, capacity building and carbon credits projects.

Another practice that could be stimulated is sharing the function, service and value of a physical product by groups and communities. This sharing use of products can decrease the need for individual purchases and, consequently, improve products' longevity and performance since it will require them to endure more uses in different contexts. In collaborative consumption, which encompasses sharing economy, multiple individuals have access to a product by dividing the ownership cost without a mediator. Thus, it becomes imperative that new legislation and infrastructure be developed to enable the implementation of waste prevention in its full potential to, then, successfully achieve a circular economy.

Part 2 — The Circular Economy and the Brazilian Industry

Brazil's industry has a long tradition of practices similar to the circular economy concept. For example, since the 1950s, the petrochemical sector has been building big complexes in cities of the States of Bahia (BA), São Paulo (SP), Rio de Janeiro (RJ) and Rio Grande do Sul (RS). These first-generation petrochemicals basically transform the naphtha from refineries into ethylene and propylene. Afterwards, the second-generation petrochemicals, which produce plastics and elastomers, and the third-generation petrochemicals, which make products such as films, pipes and bottles, were installed surrounding the former ones. This industrial park attracted recyclers companies, first working with industrial waste and later with post-consumer waste [30].

Several industries in Brazil (mostly multinational) adopt the 'Zero Waste to Landfill' policy. Internal strategies force them to search for alternatives for their waste, avoid final disposition in landfills and circulate waste back into several production chains. However, a common alternative for Brazilian industrial waste is the final disposition, which increases the waste of resources and brings other problems.

The Different Stages of the Brazilian Mining and Cement Industries Concerning the Circular Economy

One of Brazil's less circular economic sectors is the mining industry, responsible for 4% of the Brazilian Gross Domestic Product (GDP) in 2018 [31]. The country exported 409 million tonnes of minerals in 2018 and 2019, such as iron ore (68% of the sector's exports), gold (9%), niobium (7%), copper (9%), bauxite (1%) and manganese (1%), among others [32].

A consequence of this production is a large amount of waste obtained during ore exploration and its processing. The ore exploration rejects are usually inert and can be employed as gravel in the construction sector or are left in the same mines when the activity has close. The ore processing ends up generating a large amount of waste that is stacked if it is dry. If this waste is wet or liquid, it goes to tailing dams, which must be monitored continuously and mean costs and risks. In Brazil, only the company Vale, one of the largest global mining companies, generated 626 million tonnes of mining waste in 2019, and 47% of this total was related to iron ore exploration and processing [33]. This amount of mining waste was eight times larger than the amount of municipal solid waste generated in Brazil in 2018 ('The Potential of Solid Waste Management to Promote the Circular Economy in Brazil') [7–9].

There are environmental, health, safety and social risks related to the mining waste management, as observed in the last decade in Brazil when two significant accidents occurred, both in the State of Minas Gerais (MG): in 2015, in the municipality of Mariana and, in 2019, in the municipality of Brumadinho. These accidents affected the soil and water bodies since the mud from iron ore processing reached the Atlantic Ocean. Besides, around 270 people died, and thousands lost their homes. The Vale company invested US\$ 100 million, only in 2019, trying to hold and remove the rejects of Brumadinho's accident, and these investments must reach US\$ 340 million until 2023 [30]. In 2020, Brazil had 858 tailing dams with mining waste, and 50 were classified as high risk of failure [34].

These accidents and the risk of others and the enormous amount of potential raw material wasted increase the interest of the mining industry to be more circular. This challenge demands advances in research and development and laws that stimulate the different sectors of society involved.

Construction is one of the leading destinations of mining waste since this sector consumes a large amount of raw material. Brazilian studies have evaluated the use of waste from iron ore processing [35, 36] and its first stages of industrialisation [37, 38], as well as the waste from bauxite processing [39–41] to produce cement, mortars and concrete [37, 38]. Other studies with this waste have proposed the use in the production of bricks, tiles, flooring etc. [42–45]. The waste of ornamental stones, such as marble, also has shown potential in producing mortars, concrete and bricks [46–48].

As addressed in Part 1 for the municipal solid waste, the country's continental dimensions are a barrier to the reverse logistics of the industrial waste and, therefore, for the development of extensive systems in Brazil. However, the geographical concentration of industries in the South and Southeast regions is an opportunity for them to trade their waste (as in the petrochemical complexes). This practice is difficult for the other regions' industries due to logistic problems, which are even more complicated for the mining industries since their major is located in remote places, usually near forests and preservation areas.

Despite the Brazilian mining sector's problems, maybe the cement industry can be considered the most circular in the country. In this scenario, the main driving force is the great need for raw material and energy required to produce cement. Part of the natural raw material, usually obtained by mining, can be replaced by some waste types. For example, slag from the steel industry, rich in iron and calcium (among other elements), is commonly employed worldwide to produce clinker. In 2015, a Brazilian steel mill opened a cement factory that uses the blast furnace slag generated by the company, which turned it into one of the most competitive players in the country's cement industry [49].

The co-processing of waste in cement factories' kilns, replacing part of the fossil fuels employed (which are, to some extent, imported by Brazil) is another example of the circular economy in the cement industry. There are 38 plants, representing about 70% of the country's cement production capacity, with one or more kilns with the environmental permit to co-process waste. According to the industries' estimates, there are 15% of thermal substitution and the alternative fuels most used are tyres (53%) and hazardous waste (36%). The remaining 11% is divided into sawdust with oil, used oil and solvents, contaminated soils and, more recently, refused-derived fuel [50, 51].

The cement industry was responsible for the success of the logistic reverse of tyres in Brazil, which started in 1999 for environmental and public health reasons: tyres with rainwater were good places for the mosquitoes that transmitted the virus of dengue fever [52]. From that moment, more than 5.7 million tonnes of wasted tyres were correctly destined, i.e. about 1 billion ride tyres. From 2001 to 2014, the correct destination varied from 50 thousand tonnes to 450 thousand tonnes, and from 2015 to 2019, this amount grew to 471 thousand tonnes, the higher quantity ever. In 2019, 70% of the wasted tyres collected were co-processed in cement factories, whilst the other 30% were recovered as rubberised asphalt and rubber products, such as industrial floorings and car floor mats. The Brazilian association responsible for the reverse logistics of tyres states that the target of one tyre correctly destined for each tyre sold (excluding the ones in new vehicles) is achieved. Tyres are delivered in 1053 collection points around the country and managed by the municipalities. The association collects the tyres and sends them to the final correct destination [53]. Another market is supplied by tyres shops, which commercialise worn tyres mainly to rubberised asphalt and rubber products companies.

The Circular Economy in Brazil and the Agrobusiness

In 2019, agribusiness was responsible for about 21% of the Brazilian GDP [54]. Therefore, agriculture in Brazil receives a large amount of investment, which aims to increase rural productivity. The most used resources to reach this goal are waste recovery and the use of pesticides.

There is a scarcity of data concerning the waste recovery in all Brazilian crops. One exception is the sugarcane crop, which in the last harvest (2019/2020) reached 642 million tonnes [55]. The amount of sugarcane bagasse is stabilised since 2010, around 160 million tonnes, all of that sent to heat or electricity generation. It is noteworthy that the amount of sugarcane bagasse sent to electricity generation is growing through the years, from 19 thousand tonnes in 2010 to almost 30 thousand tonnes in 2019 [56]. The National Agency of Electricity recorded, in 2021, 409 thermal power plants with sugarcane bagasse as fuel. Two other thermal power plants are fuelled with elephant grass (bovine feed), 13 with rice peels, 60 with forest waste and 18 with black liquor (these last two are from the paper and cellulose industry). Altogether, they comprehend 15 GW of installed capacity, about 7% of the total Brazilian installed capacity of electricity generation. Other 41 MW (0.02% of the total) are from 20 thermal power plants based on biogas produced with agribusiness waste (15 from animal waste, 4 from vegetal waste and 1 from forest waste) [10]. Some small unities do mulching, composting or anaerobic digestion and use the final product (mulch, compost or biogas).

As already mentioned in [‘Introduction’](#), the Brazilian National Policy of Solid Waste, from 2010, obliged manufacturers, importers, distributors and retailers of six types of products to promote reverse logistics for their waste. The reverse logistics system of pesticide packaging, created and managed by the agrochemical industry, can be considered the most successful of all six, despite other cases with good performance (as tyres, with the majority being co-processed in cement factories — [‘The Different Stages of the Brazilian Mining and Cement Industries Concerning the Circular Economy’](#)).

In 2018, according to Food and Agriculture Organization of the United Nations, Brazil was the third-largest consumer of pesticides in the world, with more than 377,000 tonnes (the world consumption was more than 4 million tonnes) [57].

One of the issues that this consumption arises is related to the pesticide packaging collection and its waste. Until 2000, Brazil’s pesticides packaging was mostly incinerated or landfilled, depending on the pesticide and material’s packaging. However, it was common to see these packaging in rivers, vacant lots and household waste, threatening the environment and public health [58, 59].

A Brazilian 2000 law (number 9.974/00) tried to regulate the management of pesticides packaging. From that moment, the management of this waste was the responsibility of the manufacturer [60]. In 2002, the pesticide manufacturers created the Brazilian Institute for Empty Packaging Processing (InpEV) to organise reverse logistics.

Since then, InpEV installed 304 stations and 107 central locations to receive and treat pesticide packaging collected all over the country. The stations are the smallest places and receive pesticide packaging direct from the farmers, separate the ones that are considered contaminated from the non-contaminated and send all of them to the central locations, which are the largest places. The central locations separate the packaging according to the material: cardboard, metals, low-density polyethylene (LDPE), high-density polyethylene (HDPE) and polypropylene (PP). According to InpEV, in 2019, more than 45,000 tonnes of pesticide packaging were correctly destined by farmers, about 94% of all pesticide packaging legally

commercialised in Brazil. Of this amount, 2700 tonnes were incinerated, whilst 42,300 t were recycled [59, 61]. The main products from these materials recycling were agrochemical and lubricant oil packaging, wastewater and electricity pipes and battery cases [62].

The system is financed by the 112 manufacturers of pesticides in Brazil, with more than 4500 retailers and 1,800,000 rural properties. The farmers deliver their empty packaging in the 411 stations or central locations distributed throughout the 26 states and the Federal District [62]. The reverse logistics of empty packaging of pesticides in Brazil generated about US\$ 80 million in 2019, considering the investments, the values traded by recyclers, the 1500 direct jobs and others.

However, the system has a lot to improve since there is a known market for illegal, falsified and smuggled pesticides in the country. This market is a threat to the environment and public health, not only because of the product but also because of the packaging, not accepted in the reverse logistics system [62]. Another point is that, as already mentioned, 6% of the packaging commercialised do not have the correct final destination, which means that more than 2900 tonnes per year are disposed of without control and probably impacting the environment.

The country's continental dimensions and the concentration of industries in the South and Southeast regions, already addressed in other sections ('Contrasts of an Underdeveloped Country With Continental Dimensions to Promote Circular Economy Using Integrated Municipal Solid Waste Management' and 'The Different Stages of the Brazilian Mining and Cement Industries concerning the Circular Economy'), also hinder the reverse logistics of empty packaging of pesticides. In Brazil, the whole reverse logistics of these packaging, since the rural properties to the final destination, is done through roads, usually by lorries. All the packaging is transported to only 10 recyclers and 4 incinerators with environmental permits to treat this kind of waste. However, 9 of these companies are in the State of São Paulo (7 recyclers and 2 incinerators) [61, 63]. It should be highlighted that the higher planted areas of grains in Brazil are out of the São Paulo State, like rice (68% of the production is in Rio Grande do Sul State), cotton (89% of the production is in the States of Mato Grosso [MT] and Bahia [BA]), corn (46% of the production is also in MT and BA), soybean (44% of the production is in the States of Mato Grosso and Paraná [PR]) and coffee (55% is in Minas Gerais State [MG]). The State of São Paulo produces 51% of the sugar cane and 63% of Brazil's orange [64]. In 2019, the most considerable amount of empty packaging was produced in the States of Mato Grosso (25% of the total) and Paraná (13%), and there is only one authorised recycler in each state and no incinerator [64].

The Circular Economy in Brazil and the Electrical and Electronic Sector: the Challenges of e-waste in the Information Society

Before National Policy on Solid Waste, different laws aimed to implement reverse logistics systems for five out of the six items comprehended by the 2010 law, among them is the empty packaging of pesticides addressed in 'The Circular Economy in Brazil and the Agrobusiness'. The only exception was the electrical and electronic waste (or only e-waste from now on), and because of that, a reverse logistics system of this waste had to be designed, developed and implemented only from 2010.

The inclusion of e-waste in the law reflects that Brazil is considered the leading e-waste producer in Latin America, with approximately 2.1 million tonnes per year [58]. However, even in 2020 (10 years after the law), there is poor control of the country's current e-waste flows, with uncertainties regarding their real quantities, composition and destination [65–67].

The information society has demanded more electrical and electronic equipment and generated a large amount of e-waste, such as information technology (IT) appliances, TVs and monitors, cooling and freezing equipment, technical instruments and tools. The size, weight and chemical composition of e-waste can vary significantly depending on the product. On the one hand, e-waste contains valuable materials for recycling, like precious metals, rare earth elements, plastics and other metals, which requires efficient take-back, sorting and recycling capacity. On the other hand, some of the elements inside e-waste can be contaminants, causing environmental and human health risks, especially in inadequate e-waste management [68, 69].

In Brazil, the municipal solid waste management scenario (Part I) also applies to e-waste, with a large amount of waste going to open dumpsites, low selective collection and informal sector participation. Despite the lack of data, there is evidence that e-waste has been dumped in open areas and processed by informal actors to extract valuable materials. The most common practices are dismantling, burning cables and acid leaching of printed circuit boards [70].

Because of the economic interest, there is collection and processing of e-waste and their parts from private companies, waste pickers' cooperatives and scrap dealers. However, many of them operate without the required documentation and working conditions. Some of these businesses have failed due to a short or uncertain supply of good-quality material caused by the formal waste streams' competition and inefficiency. Another critical factor for the feasibility of such systems is the lack of fiscal stimulus for recycling.

After years of negotiation, producers, importers, distributors and retailers of electrical and electronic equipment (EEE) signed with the Federal Government in 2019 a contract called Sectoral Agreement (SA), following the National Solid Waste Policy. The agreement details operational procedures, technical requirements, responsibilities and other rules and criteria for implementing and operating a reverse logistics system of e-waste in Brazil [71]. It was the basis for an Act of 2020, which sets progressive targets for the country's e-waste recovery, reaching 17% in 2025 [72]. Such progressive implementation starts in the largest cities and more developed regions, expanding to the other areas after that. The SA also defines rules for cooperatives and Municipal Solid Waste entities' participation and isolated take-back initiatives (voluntary and unlinked to the official system ruled by the SA). A critical definition in the SA and the law is that e-waste is not considered hazardous waste unless it is dismantled to enable transport and pre-sorting operations without bureaucracy and costs.

According to the National Policy on Solid Waste and the Sectorial Agreement, the implementation and operation of a reverse logistics system must be executed by one or more Management Entities (ME), assembled by producers, importers, distributors and retailers. Currently, there is one official ME in the country, called Green Eletron. By late 2020, Green Eletron had installed e-waste delivery stations across the States of Sao Paulo (SP), Rio de Janeiro (RJ) and the Federal District (DF), but the coverage is still low. Recent studies have assessed the need for an adequate number of e-waste delivery points in strategic positions [73, 74].

The ME can subcontract specialised companies to operate the reverse logistics system and the recycling and destination processes. Currently, the core e-waste recycling operator for the official reverse logistics system is Sintronics™, a company founded by an electronic equipment producer located in the State of Sao Paulo (SP). This company executes the stages of pre-sorting, dismantling and destination of materials for recycling since it adopts the 'zero waste to landfill' policy. Sintronics™ is also specialised in recycling plastics from e-waste, producing, among other products, specific components for printers, following the producers' standards.

As mentioned before, Brazil's Southeast region concentrates the largest number of industries in the country, which is also true for recycling plants. Thus, most of the e-waste collected across the country needs to reach this region, especially São Paulo State, to be recycled. Distances from Brazilian State capitals in the North or Northeast regions to São Paulo may reach 3000 km. Besides, Brazilian logistics depends almost totally on road transport, and many roads that connect such regions are in poor conditions ('[The Potential of Solid Waste Management to Promote the Circular Economy in Brazil](#)'). Another obstacle for efficient reverse logistics is each State's legislation. Some states prohibit transboundary transport of hazardous materials, which can prejudice some e-waste components' destination to proper treatment in another state.

Brazil can recycle several e-waste components, but the most valuable ones, namely special metals (precious and rare earth elements), are mostly exported or extracted by irregular activities. Some companies implemented capacity for hydrometallurgical extraction of such metals before, but these plants shut down due to the lack of e-waste supply. There is an increasing number of research projects trying to develop national technology at feasible costs. A promising solution is biomining. For example, the project Rematronic, carried out by a Federal technological institute, presented good efficiency in extracting some valuable metals with bacteria [75]. This technology was recently transferred to a private waste management company operating in Brazil.

In summary, some fundamental factors for a better e-waste circular economy in Brazil are:

- The coverage of formal e-waste delivery points needs to improve;
- The irregular activities need to be inhibited;
- The informal sector needs to be integrated with the formal systems. The most recommended solution for such integration is the assembling of waste pickers' cooperatives. However, these organisations require financial support, capacity building, adequate processes and documentation;
- The waste management and the recycling economy sector need to be stimulated in all Brazilian regions, by special funding schemes, tax exemptions, National and States legislation updating and capacity building, for example;
- Feasible national technologies to extract special metals from e-waste and treat the hazardous components need to be developed and implemented.

The Circular Economy and the Materials Industry

Brazil has good numbers related to the recycling of some materials (Table 2), despite the low levels of municipal solid waste recovery already mentioned in Part I. The reason for this difference is that a large amount of waste is not recorded. The data presented in Table 1 (CS048 indicator) refer to dry solid waste recovered by waste pickers' cooperatives supported by the municipalities. Nevertheless, there are informal waste pickers, scrap dealers and waste management companies, not linked to cooperatives, that also collect recyclable waste in households, retailers, service providers and industries. Table 2 presents the results of the recycling rate of some materials in Brazil, based on reports from industry associations such as The Brazilian Association of PET Industry (ABIPET), Brazilian Industry Commitment to Recycling (CEMPRE) and Stimulus Plan to Plastic Chain (PICPLAST) [76–78]. It is possible

Table 2 Recycling rate of some materials in Brazil [76–78]

Material	Recycling rate (%)	
	More recent	Previous
Plastic (all) [77, 78]	24.0 (2019)	21.7 (2011)
PET [78, 79] — poly (ethylene terephthalate)	51.0 (2015)	17.5 (2004)
Steel can [78]	46.7 (2015)	47.0 (2012)
Aluminium can [78]	97.6 (2019)	97.9 (2015)
Long-life packaging [78]	31.3 (2019)	21.7 (2015)
Paper (all) [78]	66.9 (2019)	31.2 (2014)
Cardboard [78]	85.0 (2019)	84.7 (2014)
Glass [78]	47.0 (2011)	33.0 (1994)

to see an evolution of the recycling rates considering the most recent data and the previous ones.

Economic data about how much recycling means to the country are found to plastics for 2018 and 2019 and can be seen in Table 3 [76]. Table 4 shows data about plastic waste recycled in 2019 in Brazil [77].

Table 3 shows that recycled plastic production had slight growth in the period, despite the number of industries and employees decreased from 2018 to 2019. About 18 thousand Brazilians were working in 700 plastic recycling industries, which had a revenue of near US\$ 0.5 billion. All this structure recycled 24% of the plastic waste generated in the country in 2019.

It can be observed in Table 4 that half of the plastic waste sent to recycling in Brazil was from households, 28% was from industries and the rest was from retailers or service providers. Only 12% was sent from waste pickers' cooperatives directly to recycling industries. However, a part of the material sold by scrap dealers to recyclers is bought from cooperatives and/or informal waste pickers, but there is no information about the percentage. A reasonable amount of plastic waste (17%) came from processing industries where waste can be shredded, washed, dried, extruded and/or granulated before selling the processed material to industries that will transform them into new products. Waste management companies and informal waste pickers, not-associated with cooperatives, sell waste directly to recycling industries. Recyclers also buy recyclable materials direct from the waste source and after sorting in open dumps.

Table 4 also shows that almost half of the plastic waste sent to recycling was from the Southeast region in 2019. In contrast, the North region sent only 4%, which corroborates the

Table 3 Economic data related to plastic recycling in Brazil in 2018 and 2019 [76]

	Unit	2019	2018
Plastic recycling industries	Number	695	716
Direct jobs	Number	17,919	18,662
Revenue	US\$	454,000,000	436,000,000
Installed capacity	Tonnes	1,848,000	1,802,000
Excess capacity	%	38	39
Amount of plastic consumed in recycling	Tonnes	1,320,000	1,230,000
Amount of recycled plastic produced	Tonnes	838,000	757,000
Amount of plastic waste generated	Tonnes	3,500,000	3,430,000
Recycling rate	%	24.0	22.0

Table 4 Information about plastic waste recycled in Brazil in 2019 [77]

	%
Type of waste	
Pre-consumer (industries)	28.0
Post-consumer (households)	52.5
Post-consumer (not from households)	19.5
Source of waste	
Industries	28.0
Scrap dealers	26.0
Processing industries	17.0
Waste pickers' cooperatives	12.0
Waste management companies	8.0
Informal waste pickers	4.0
Direct from source	3.0
Open dumps	2.0
Brazilian region where the waste was generated	
Southeast	48.5
South	27.5
Northeast	12.0
Middle-west	6.0
North	4.0
Imported from other countries	2.0

discussions in item 1.2 about the Southeast region's best structure (industrial and logistic, among others). It is also noteworthy that the plastic recycling industry imported 2% of its raw material from neighbouring countries. For comparison, in 2019, 56% of the Brazilian recycled plastic was produced by the Southeast region, whilst the North region produced only 1.4%.

Conclusions and Final Remarks

The present paper aimed to show an overview of Brazil's municipal and industrial solid waste management's challenges and opportunities to promote the circular economy. Some convergence points can be established between these two solid waste management:

- (a) The industries are concentrated in the South and Southeast of Brazil, favouring recycling in these parts of the country. For instance, the Southeast region was responsible for collecting almost half of the plastic waste sent to recycling and producing 56% of the recycled plastic in Brazil in 2019.
- (b) There are only two cities with recycling industries in the North region, which comprises 43% of the Brazilian territory. However, these two cities positively affect the circular economy in the North region. The rate of recovery of dry recyclable materials generated in this region increased in the last years, based on the work of waste pickers.
- (c) The best logistics structure (roads) is also concentrated in Brazil's South and Southeast regions. The North region has high precipitation and a significant part of its territory with Indigenous and forest areas, contributing to problematic logistics.
- (d) Scientific and technological development is more favoured in the South and Southeast regions, especially when considering solid waste management. Those regions have more universities, research centres, technical personnel and financial resources than the Brazilian Amazon [25].

- (e) The large amount of waste produced by the mining industry in Brazil requires scientific and technological development to optimise processes that make the recovery of materials feasible. Simultaneously, researches have to be conducted, also, to improve ore exploration and processing, aiming to prevent and reduce waste generation and its impacts on the environment.
- (f) The reverse logistics of empty packaging of pesticides, which started in 2000, is the best in the country, although improvements can be made in the transport and final destination. However, scientific and technological development can promote the circular economy in agricultural commodities in different ways. One example is the controlled delivery of pesticides based on nanotechnology, which can improve the product's efficiency and, therefore, decrease the amount of pesticide needed [79] and empty packaging generated. Studies can also achieve plants more resistant to diseases, microorganisms replacing synthetic molecules in disease control, best practices in agriculture and less impacting packaging.
- (g) Society is becoming more and more concerned about human activities' environmental impacts and will demand that industries provide products that promote waste prevention and the circular economy. Researches with that purpose can change many segments, such as the electrical and electronic industry, which is responsible for a growing waste generation due to the rapid obsolescence of the technologies in this sector.

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Declarations

Conflict of Interest The authors declare no competing interests.

References

1. Paes MX, Mancini SD, Medeiros GA, Bortoleto AP, Kulay LA (2018) Life cycle assessment as a diagnostic and planning tool for waste management—a case study in a Brazilian municipality. *J Solid Waste Technol Manag* 44:259–269. <https://doi.org/10.5276/JSWTM.2018.259>
2. Lima NSS, Mancini SD (2017) Integration of informal recycling sector in Brazil and the case of Sorocaba city. *Waste Manag Res* 35:721–729. <https://doi.org/10.1177/0734242X17708050>
3. Bortoleto AP (2014) Waste prevention policy and behaviour: new approaches to reducing waste generation and its environmental impacts. Routledge, New York
4. Zanin M, Mancini SD (2004) Resíduos Plásticos e Reciclagem: Aspectos Gerais e Tecnológicos [Plastic waste and recycling: general and technological aspects]. EduFSCar, São Carlos, SP, Brasil.
5. Brasil (2010). Lei nº 12.305, de 02 de agosto de 2010. Política Nacional de Resíduos Sólidos. Brasília [National policy for solid waste], DF. http://www.planalto.gov.br/ccivil_03/_Ato2007-2010/2010/Lei/L12305.htm
6. MMA Ministério do Meio Ambiente (2011). Plano Nacional de Resíduos Sólidos: versão preliminar para consulta pública. [National Solid Waste Plan: preliminary version for public consultation]. MMA, Brasília.
7. MMA - Ministério do Meio Ambiente. Versão Preliminar do Plano Nacional de Resíduos Sólidos – PLANARES [Preliminary version of the National Solid Waste Plan – PLANARES]. Brasília: MMA, 2020. <http://consultaspublicas.mma.gov.br/planares/wp-content/uploads/2020/07/Plano-Nacional-de-Res%C3%ADduos-S%C3%B3lidos-Consulta-P%C3%BAblica.pdf>.

8. SNIS - Sistema Nacional de Informações em Saneamento Básico (2019). Diagnostico do Manejo de Resíduos Sólidos, 2018 [Diagnosis of Solid Waste Management, 2018]. SNIS/MDR, Brasília. <http://www.snis.gov.br/diagnostico-anual-residuos-solidos/diagnostico-do-manejo-de-residuos-solidos-urbanos-2018>.
9. ABRELPE - Associação das Empresas de Limpeza Pública (2019). Panorama dos Resíduos Sólidos no Brasil 2018/2019 [Panorama of Solid Waste in Brazil 2018/2019]. São Paulo: Abrelpe, 2019. <https://abrelpe.org.br/panorama/>. Accessed 26 november 2020.
10. ANEEL – Agência Nacional de Energia Elétrica [Brazilian Agency of Electricity]. Sistema de Informações de Geração (Information System for Electricity Generation). Brasília: ANEEL, 2021. <https://bit.ly/2IGf4Q0>. Accessed 06 march 2021.
11. IBGE - Instituto Brasileiro de Geografia e Estatística [Brazilian Institute of Geography and Statistics]. Pesquisa Nacional de Saneamento Básico 1989 (National Survey on Basic Sanitation 1989). Rio de Janeiro: IBGE, 1990.
12. Oliveira Neto R, Souza LE, Otávio PC (2014) Avaliação da gestão integrada de resíduos sólidos urbanos no Brasil em comparação com países desenvolvidos [Evaluation of integrated municipal solid waste in Brazil compared to developed countries]. Rev Monografias Amb – REMOA. <https://doi.org/10.5902/2236130814684>
13. EUROSTAT (2020) Recycling rate of municipal waste. https://ec.europa.eu/eurostat/databrowser/view/sdg_11_60/default/table?lang=en. Accessed 05 October 2020.
14. CEWEP –Confederation of European Waste-to-Energy Plants (2017). Waste to energy plants in Europe in 2017. <https://www.cewep.eu/waste-to-energy-plants-in-europe-in-2017/>. Accessed 05 October 2020.
15. EUROSTAT (2018) Municipal waste statistics. https://ec.europa.eu/eurostat/statistics-explained/index.php/Municipal_waste_statistics#Municipal_waste_treatment Accessed 05 October 2020.
16. Brasil (2020) Atualização do Marco Legal do Saneamento Básico [Update of the Basic Framework for Basic Sanitation]. http://www.planalto.gov.br/ccivil_03/_ato2019-2022/2020/lei/114026.htm. Accessed 05 October 2020.
17. Ecoparc (2020) Ecoparc de Barcelona. <http://ecoparcbcn.com/>. Accessed 05 October 2020.
18. Paes MX, Medeiros GA, Mancini SD, Ribeiro FM, Oliveira JAP (2019) Transition to circular economy in Brazil: a look of the municipal solid waste management in the State of São Paulo. Manag Decis. <https://doi.org/10.1108/MD-09-2018-1053>
19. Ferraz JL, Mancini SD, Paes MX, Medeiros GA, Bizzo WA Proposal and test of an assessment method for municipal solid waste management systems. J Solid Waste Technol Manag. To be published in Aug 2021.
20. IBGE - Instituto Brasileiro de Geografia e Estatística [Brazilian Institute of Geography and Statistics] (2020) Cidades@ [Cities]. <https://cidades.ibge.gov.br/>. Accessed 02 October 2020.
21. IBGE - Instituto Brasileiro de Geografia e Estatística (2020) Portal de mapas [Map portal] <https://portaldemapas.ibge.gov.br/portaal.php#homepage>. Accessed 02 October 2020.
22. IBGE - Instituto Brasileiro de Geografia e Estatística (2020). Áreas territoriais [Territorial areas]. <https://www.ibge.gov.br/geociencias/organizacao-do-territorio/estrutura-territorial/15761-areas-dos-municipios.html?=&t=acesso-ao-produto>. Accessed 02 October 2020.
23. SNIS – Sistema Nacional de Informações sobre Saneamento [National Survey of Information on Sanitation]. SNIS Série Histórica [SNIS Historical Series]. <http://app4.mdr.gov.br/serieHistorica/>. Accessed 24 february 2021. Accessed 05 October 2020.
24. Fehr M, Sousa KA, Santos MQ, Domingues MMO (2009) Using test communities to demonstrate household waste separation procedures in Brazil. Environ Pract 11:25–31. <https://doi.org/10.1017/S1466046609090036>
25. FUNAI - Fundação Nacional do Índio (2020) Índios no Brasil: terras indígenas [Indians in Brazil: indigenous lands] <http://www.funai.gov.br/index.php/indios-no-brasil/terras-indigenas>. Accessed 15 October 2020.
26. CNT – Confederação Nacional do Transporte (2020). Pesquisa de Rodovias [Road Search]. <https://pesquisarodovias.cnt.org.br/>. Accessed 15 October 2020.
27. Alvares CA, Stape JL, Sentelhas PC, Gonçalves JLM, Sparovek G (2013) Köppen’s climate classification map for Brazil. Meteorol Z 22:711–728. <https://doi.org/10.1127/0941-2948/2013/0507>
28. Oliveira BOS, Medeiros GA, Paes MX, Mancini SD (2021) Integrated municipal and solid waste management in the Amazon: addressing barriers and challenges in using the Delphi method. Int J Environ Impacts 4:49–61. <https://doi.org/10.2495/EI-V4-N1-49-61>
29. Paes MX, Medeiros GA, Mancini SD, Gasol C, Rieradevall-Pons J, Gabarrell-Durany X (2020) Transition towards eco-efficiency in municipal solid waste management to reduce GHG emissions: the case of Brazil. J Clean Prod 263:121370. <https://doi.org/10.1016/j.jclepro.2020.121370>
30. EPE – Empresa de Pesquisa Energética (2018). Panorama do refino e da petroquímica no Brasil [Panorama of refining and petrochemicals in Brazil]. Brasília, EPE. <https://www.epe.gov.br/sites-pt/publicacoes-dados->

- abertos/publicacoes/PublicacoesArquivos/publicacao/-topico-412/NT%20Refino%20e%20Petroqu%C3%ADmica_2018.11.01.pdf. Accessed 10 October 2020.
31. Ministério de Minas e Energia, Secretaria de Geologia, Mineração e Transformação Mineral (SGM) Boletim do Setor Mineral Janeiro 2020 [Mineral Sector Bulletin January 2020]. <http://www.mme.gov.br/docs/documents/36108/406012/Boletim+Informativo+do+Setor+Mineral+2019.pdf/ef8e7327-8022-458e-1b1e-039c73ffed18>. Accessed 10 October 2020.
 32. IBRAM – Instituto Brasileiro de Mineração (2019) Relatório Anual de atividades - Julho 2018 a junho de 2019 [Annual Activity Report - July 2018 to June 2019]. <http://portaldamineracao.com.br/ibram/wp-content/uploads/2019/07/relatorio-anual-2018-2019.pdf>. Accessed 10 October 2020.
 33. VALE - Companhia Vale do Rio Doce (2020) Relatório de sustentabilidade 2019 [Sustainability report 2019]. VALE, Rio de Janeiro, Brasil. <http://www.vale.com/brasil/PT/sustainability/relatorio-de-sustentabilidade-2019/Paginas/default.aspx>. Accessed 10 October 2020.
 34. ANM – Agência Nacional de Mineração (2020). Barragens: boletim semanal [Dams: weekly newsletter]. <https://www.gov.br/anm/pt-br/assuntos/barragens/boletim-semanal-de-barragens-de-mineracao/arquivos/report-semanal-2020-11-09.pdf/view>. Accessed 10 October 2020.
 35. Carrasco EVM, Magalhães MDC, Santos WJD, Alves RC, Mantilla JNR (2017) Characterisation of mortars with iron ore tailings using destructive and nondestructive tests. *Const Build Mater* 131:31–38. <https://doi.org/10.1016/j.conbuildmat.2016.11.065>
 36. Fontes WC, Mendes JC, Silva SN, Peixoto RAF (2016) Mortars for laying and coating produced with iron ore tailings from tailing dams. *Constr Build Mater* 112:988–995. <https://doi.org/10.1016/j.conbuildmat.2016.03.027>
 37. Carvalho SZ, Vernilli F, Almeida B, Oliveira MDS, Silva SN (2017) The recycling effect of BOF slag in the portland cement properties. *Resour Conserv Recycl* 127:216–220. <https://doi.org/10.1016/j.resconrec.2017.08.021>
 38. Carvalho SZ, Vernilli F, Almeida B, Oliveira MDS, Silva SN (2018) Reducing environmental impacts: the use of basic oxygen furnace slag in portland cement. *J Clean Prod* 172:385–390. <https://doi.org/10.1016/j.jclepro.2017.10.130>
 39. Romano RCO, Bernardo HM, Maciel MH, Pileggi RG, Cincotto MA (2018) Hydration of Portland cement with red mud as mineral addition. *J Therm Anal Calorim* 131:2477–2490. <https://doi.org/10.1007/s10973-017-6794-2>
 40. Fortes GM, Lourenço RR, Montini M, Gallo JB, Rodrigues JA (2016) Synthesis and mechanical characterisation of iron oxide rich sulfobelite cements prepared using bauxite residue. *Mater Res* 19:276–284. <https://doi.org/10.1590/1980-5373-MR-2015-0180>
 41. Brito WS, Silva AMF, Santa RAAB, Svensson K, Souza JAS, Pöllmann H, Riella HG (2018) An assessment of reuse of light ash from bayer process fluidised bed boilers in geopolymer synthesis at ambient temperature. *Adv Mater Sci Eng* 2018:1–6. <https://doi.org/10.1155/2018/1287243>
 42. Mendes BC, Pedrotti LG, Fontes MPF, Ribeiro JCL, Vieira CMF, Pacheco AA, Azevedo ARG (2019) Technical and environmental assessment of the incorporation of iron ore tailings in construction clay bricks. *Constr Build Mater* 227:116669. <https://doi.org/10.1016/j.conbuildmat.2019.08.050>
 43. Babisk MP, Amaral LF, Ribeiro LS, Vieira CMF, Prado US, Gadioli MCB, Oliveira MS, Luz FS, Monteiro SN, Garcia Filho FC (2020) Evaluation and application of sintered red mud and its incorporated clay ceramics as materials for building construction. *J Mater Res Technol* 9:2186–2195. <https://doi.org/10.1016/j.jmrt.2019.12.049>
 44. Barreto IAR, Costa ML (2018) Viability of Belterra clay, a widespread bauxite cover in the Amazon, as a low-cost raw material for the production of red ceramics. *Appl Clay Sci* 162:252–260. <https://doi.org/10.1016/j.clay.2018.06.010>
 45. Monteiro SN, Vieira CMF (2014) On the production of fired clay bricks from waste materials: a critical update. *Constr Build Mater* 68:599–610. <https://doi.org/10.1016/j.conbuildmat.2014.07.006>
 46. Marvila MT, Azevedo ARG, Alexandre J, Colorado H, Antunes MLP, Vieira CMF (2020) Circular economy in cementitious ceramics: replacement of hydrated lime with a stoichiometric balanced combination of clay and marble waste. *Appl Ceram Technol* 18:192–202. <https://doi.org/10.1111/ijac.13634>
 47. Simão L, Hotza D, Ribeiro MJ, Novais RM, Montedo ORK, Raupp-Pereira F (2020) Development of new geopolymers based on stone cutting waste. *Constr Build Mater* 257:119525. <https://doi.org/10.1016/j.conbuildmat.2020.119525>
 48. Bezerra WBA, Garcia Filho FC, Pereira AC, Monteiro SN, Marvila MT, Antunes MLP, Alexandre J, Azevedo ARG, Rocha HA (2020) Effect of the incorporation of marble waste in the properties of clay ceramic bricks. *Mater Sci Forum* 1012:250–255. <https://doi.org/10.4028/www.scientific.net/msf.1012.250>
 49. CSN – Companhia Siderúrgica Nacional (2020) Cimento [Cement]. https://www.csn.com.br/conteudo_pti.asp?idioma=0&conta=45&tipo=59653. Accessed 10 October 2020.

50. ABCP - Associação Brasileira do Cimento Portland (2020) Panorama do Coprocessamento 2020 – Ano Base 2018 [Panorama of Coprocessing 2020 - Base Year 2018]. https://coprocessamento.org.br/wp-content/uploads/2020/10/Panoramaco_processamento_2020_bx.pdf. Accessed 10 October 2020.
51. VC - Votorantim Cimentos (2020) Resíduos urbanos como fonte de energia para nossas fábricas [Urban waste as an energy source for our factories] https://coprocessamento.org.br/wp-content/uploads/2020/10/Panoramaco_processamento_2020_bx.pdf. Accessed 10 October 2020.
52. Lutinski JA, Quadros SO, Tiburski J, Bedim CF (2018) Gestão dos pneus inservíveis e sua contribuição para o programa de prevenção à Dengue, febre Chicungunya e Zika vírus no município de Chapecó SC [Waste tyres management and their contribution to the Dengue, Chickungunya fever and Zika virus prevention program in the Chapeco (SC) municipality]. *Revista Interdisciplinar de Estudos em Saúde* [Interdisciplinary Journal of Health Studies]. Doi. <https://doi.org/10.33362/ries.v7i1.1238>
53. RECICLANIP [National Association of Tyre Industry]. Principais destinações [Main destinations]. <https://www.reciclanip.org.br/>. Accessed 02 March 2021.
54. CNA – Confederação da Agricultura e Pecuária do Brasil (Brazilian Association of Agriculture and Livestock) (2020) Panorama do Agro (Survey of Agribusiness in Brazil). Available at <https://www.cnabrazil.org.br/cna/panorama-do-agro#:~:text=Em%202019%2C%20a%20soma%20de,R%24%20494%2C8%20bilh%C3%B5es>. Accessed 10 October 2020. Accessed 10 October 2020.
55. CONAB – Companhia Nacional de Abastecimento [National Company of Supply]. Boletim da safra de Cana-de-açúcar [Sugarcane harvest report]. <https://www.conab.gov.br/info-agro/safras/cana/boletim-da-safra-de-cana-de-acucar>. Accessed 06 March 2021
56. EPE – Empresa de Pesquisa Energética. Balanço Energético Nacional [Brazilian Energy Balance]. https://www.epe.gov.br/sites-pt/publicacoes-dados-abertos/publicacoes/PublicacoesArquivos/publicacao-479/topico-528/BEN2020_sp.pdf. Accessed 04 March 2021.
57. FAO – Food and Agriculture Organization of the United Nations (2020) Pesticides use. <http://www.fao.org/faostat/en/#data/RP/visualize>. Accessed 14 September 2020.
58. Carvalho MMX, Nodari ES, Nodari RO (2015) “Defensives” or “pesticides”? A history of the use and perception of pesticides in the State of Santa Catarina, Brazil, 1950–2002. *Hist cienc saude-Manguinhos* 24: 75–91. <https://doi.org/10.1590/s0104-59702017000100002>
59. Mancini SD, Oliveira AB, Gianelli BF (2019) Reverse logistics of agrochemical packaging in Brazil: case study. In: *Proceedings of 34th International Conference on Solid Waste Technology and Management*. Widener University, Annapolis-USA.
60. Brasil (2000) Lei nº 9974, de 06 de junho de 2000. Lei dos Agrotóxicos [Agrochemicals Law]. Brasília, DF. http://www.planalto.gov.br/ccivil_03/leis/19974.htm. Accessed 08 november.
61. INPEV - National Institute for Processing Empty Packages (2020) INPEV em números (Numbers of INPEV). <https://inpev.org.br/sistema-campo-limpo/em-numeros/>. Accessed 14 September 2020.
62. INPEV (2019) Relatório de Sustentabilidade (Sustainability Report). <https://inpev.org.br/Sistemas/Saiba-Mais/Relatorio/inpEV-RS2019pdf> Accessed 14 September 2020.
63. Oliveira AB (2019) Avaliação do ciclo de vida de cenários de logística reversa de embalagens de agrotóxicos no Brasil (Life cycle assessment of scenarios of reverse logistic of agrochemicals packaging in Brazil). Ph.D. Thesis. São Paulo State University, Sorocaba.
64. MAPA - Ministério da Agricultura, Pecuária e Abastecimento (2018) Projeções do Agronegócio: Brasil 2017/2018 a 2027/2028 projeções de longo prazo [Agribusiness Projections: Brazil 2017/2018 to 2027/2028 long term projections] Brasília, MAPA. https://www.gov.br/agricultura/pt-br/assuntos/politica-agricola/todas-publicacoes-de-politica-agricola/projecoes-do-agronegocio/banner_site-03-03-1.png/@download/file/projecoes-do-agronegocio-2018.pdf. Accessed 14 September 2020.
65. Forti V, Baldé CP, Kuehr R, Bel G (2020) The Global E-waste Monitor 2020: quantities, flows and the circular economy potential. United Nations University (UNU)/United Nations Institute for Training and Research (UNITAR) – co-hosted SCYCLE Programme, International Telecommunication Union (ITU) & International Solid Waste Association (ISWA), Bonn/Geneva/Rotterdam. http://ewastemonitor.info/wp-content/uploads/2020/07/GEM_2020_def_july1_low.pdf#. Accessed 14 September 2020.
66. Magalini F, Kuehr R, Baldé CP (2015) eWaste in Latin America: statistical analysis and policy recommendations. GSMA-UNU. https://www.gsma.com/publicpolicy/wp-content/uploads/2015/11/GSMA2015_Report_eWasteInLatinAmerica_English.pdf. Accessed 14 September 2020.
67. de Souza RG, Climaco JCN, Sant’Anna AP, Rocha TB, Valle RAB, Quelhas OLG (2016) Sustainability assessment and prioritisation of e-waste management options in Brazil. *Waste Manag* 57:46–56. <https://doi.org/10.1016/j.wasman.2016.01.034>
68. Bachér J, Mrotzek A, Wahlström M (2015) Mechanical pre-treatment of mobile phones and its effect on the Printed Circuit Assemblies (PCAs). *Waste Manag* 45:235–245. <https://doi.org/10.1016/j.wasman.2015.06.009>

69. Kiddee P, Naidu R, Wong MH (2013) Electronic waste management approaches: an overview. *Waste Manag* 33:1237–1250. <https://doi.org/10.1016/j.wasman.2013.01.006>
70. Prasad MNV, Vithanage M, Borthakur A (eds) (2020) Handbook of electronic waste management: international best practices and case studies. Elsevier, Butterworth-Heinemann
71. MMA- Ministério do Meio Ambiente (2020). Acordo Setorial de Eletroeletrônicos [Electrical and Electronics Sectorial Agreement]. Brasília, DF. <https://sinir.gov.br/component/content/article/2-sem-categoria/474-acordo-setorial-de-eletroeletronicos>. Accessed 14 September 2020
72. Brasil (2020) Decreto 10.240, de 12 de fevereiro de 2020. Regulamenta o de sistema de logística reversa de produtos eletroeletrônicos e seus componentes de uso doméstico [Regulates the reverse logistics system for electronic products and their components for domestic use]. Brasília, DF. Available at https://www.planalto.gov.br/ccivil_03/_Ato2019-2022/2020/Decreto/D10240.htm. Accessed 08 november 2020
73. Abbondanza MNM, Souza RG (2019) Estimating the generation of household e-waste in municipalities using primary data from surveys: a case study of Sao Jose dos Campos, Brazil. *Waste Manage* 85:374–384. <https://doi.org/10.1016/j.wasman.2018.12.040>
74. Ottoni M, Dias P, Helena L (2020) A circular approach to the e-waste valorisation through urban mining in Rio de Janeiro. *Brazil J Clean Prod* 261:120990. <https://doi.org/10.1016/j.jclepro.2020.120990>
75. CTI - Centro de Tecnologia da Informação Renato Archer (2020) Rematronic – Recuperações de Materiais Eletrônicos [Rematronic - Electronic Material Recoveries]. CTI, Campinas-SP. <https://www.cti.gov.br/pt-br/rematronic>. Accessed 14 September 2020.
76. PICPLAST – Plano de Incentivo à Cadeia do Plástico [Stimulus Plan to Plastic Chain]. Índice de plástico reciclado pós-consumo cresceu em 2019 segundo estudo encomendado pelo PIC PLAST [Rate of recycled post-consumer plastic increased in 2019 according to study commissioned by PICPLAST]. <http://www.picplast.com.br/detalhe-noticia/indice-de-plastico-reciclado-pos-consumo-cresceu-em-2019-segundo-estudo-encomendado-pelo-picplast>. Accessed 28 february 2021.
77. CEMPRE – Compromisso Empresarial para a Reciclagem [Brazilian Industry Commitment to Recycling]. Taxas de Reciclagem [Recycling Rates]. <https://cempre.org.br/taxas-de-reciclagem/>. Accessed 28 February 2021.
78. ABIPET – Associação Brasileira da Indústria do PET [Brazilian Association of PET Industry]. Censo da Reciclagem de PET no Brasil [Reporto f PET recycling in Brazil]. <http://www.abipet.org.br/index.html?method=mostrarDownloads&categoria.id=3>. Accessed 28 February 2021.
79. Fraceto LF, Grillo R, Medeiros GA, Scognamiglio V, Rea G, Bartolucci C (2016) Nanotechnology in Agriculture: which innovation potential does it have? *Front Environ Sci* 4. <https://doi.org/10.3389/fenvs.2016.00020>

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3.8 Artigo 8 – Social evaluation of municipal solid waste management systems from a life cycle perspective: a systematic literature review *

* Costa, A. M., Mancini, S. D., Paes, M. X., Ugaya, C. M. L., de Medeiros, G. A., Souza, R. G. Social evaluation of municipal solid waste management systems from a life cycle perspective: a systematic literature review. *Int J Life Cycle Assess.* 2022; 27(5): 719–739. doi: n°. 10.1007/s11367-022-02057-6

ABSTRACT

Purpose: The social aspects of municipal solid waste management (MSWM) systems are underpinning their sustainability and effectiveness. The assessment of these systems from a life cycle perspective is widespread throughout environmental life cycle assessment (LCA), but few studies have used social life cycle assessment (S-LCA). The present study is an innovative review with the objective to analyse and describe the current level of development of S-LCA applications in MSWM, and to identify the main methodological challenges and best practices, aiming at recommending approaches to harmonise future S-LCA applications in MSWM. Materials and methods: A systematic review of the literature found 36 relevant scientific articles. These were submitted to bibliometric and content analysis, which includes an analysis of how methodological aspects of the four phases of S-LCA were applied in comparison with best practice and existing guidelines. Results and discussion: There was a predominance of case studies in developing countries (59%) and evaluation of the stages of collection/transportation, pre-processing (sorting) and landfilling (55%). There were more studies focusing on stakeholders, “workers” and “local communities” and in the impact subcategories “employment”, “working hours”, “health and safety/working conditions”, “community involvement/participation” and “health and safety/living conditions of community”. There was great variability in the application of the method (47% of the studies included methodological developments). However, the 39% based on UNEP guidelines were closer to a methodological consensus. Conclusion: In general, studies need more detail and clarity in describing the methodological decisions used. Improvements are needed for issues that limit the S-LCA method, including the difficulties of covering the entire life cycle, relating impacts to the functional unit, standardizing impact assessment methods, addressing allocation and data quality issues and interpretation of results and their limitations. Improvements can be achieved by using participatory methods in the selection of categories, subcategories and impact indicators, as well as by clarifying the definition of a product system and detailing “cut-off criteria” of processes/organizations and the impact of these decisions on results.

Keywords: S-LCA; Social life cycle assessment; Social impact; Societal LCA; Life cycle assessment; Waste management; Municipal solid waste.



Social evaluation of municipal solid waste management systems from a life cycle perspective: a systematic literature review

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Abstract

Purpose The social aspects of municipal solid waste management (MSWM) systems are underpinning their sustainability and effectiveness. The assessment of these systems from a life cycle perspective is widespread throughout environmental life cycle assessment (LCA), but few studies have used social life cycle assessment (S-LCA). The present study is an innovative review with the objective to analyse and describe the current level of development of S-LCA applications in MSWM, and to identify the main methodological challenges and best practices, aiming at recommending approaches to harmonise future S-LCA applications in MSWM.

Materials and methods A systematic review of the literature found 36 relevant scientific articles. These were submitted to bibliometric and content analysis, which includes an analysis of how methodological aspects of the four phases of S-LCA were applied in comparison with best practice and existing guidelines.

Results and discussion There was a predominance of case studies in developing countries (59%) and evaluation of the stages of collection/transportation, pre-processing (sorting) and landfilling (55%). There were more studies focusing on stakeholders, “workers” and “local communities” and in the impact subcategories “employment”, “working hours”, “health and safety/working conditions”, “community involvement/participation” and “health and safety/living conditions of community”. There was great variability in the application of the method (47% of the studies included methodological developments). However, the 39% based on UNEP guidelines were closer to a methodological consensus.

Conclusion In general, studies need more detail and clarity in describing the methodological decisions used. Improvements are needed for issues that limit the S-LCA method, including the difficulties of covering the entire life cycle, relating impacts to the functional unit, standardizing impact assessment methods, addressing allocation and data quality issues and interpretation of results and their limitations. Improvements can be achieved by using participatory methods in the selection of categories, subcategories and impact indicators, as well as by clarifying the definition of a product system and detailing “cut-off criteria” of processes/organizations and the impact of these decisions on results.

Keywords S-LCA · Social life cycle assessment · Social impact · Societal LCA · Life cycle assessment · Waste management · Municipal solid waste

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1 Introduction and objective

There are several definitions of waste (White et al. 1995; Laurent et al. 2014). They can be classified by their physical state, material, origin, safety, among others (White et al. 1995). This review concentrates on Municipal Solid Waste, which includes non-hazardous solid waste from households, industries, commercial and institutional establishments (e.g., hospitals and market waste), yard waste and street sweepings (Schübeler 1996).

The social aspects linked to municipal solid waste management (MSWM) are becoming ever more relevant to attempt to make the process more sustainable and effective (Ma and Hipel 2016; Deus et al. 2019). For society to participate in MSWM, it must be well-accepted by the overall population. In Irkutsk, Russia, the scenario with the greatest social acceptance is composed of source separation, recycling, biological processing and disposal in the sanitary landfill (Tulokhonova and Ulanova 2013). In Paionia, Greece, composting is the most accepted technology, with more than 70% of residents willing to participate in source separation in an MSWM system based on biological treatments (Handakas et al. 2017). Thus, decision-making processes must consider social factors that may interfere with acceptance of the process, including demographic aspects, income patterns and lifestyle habits (Ezeah and Roberts 2012; Nie et al. 2018). Additionally, stakeholders need to be included in decision-making processes for effective governance, as this generates political legitimacy, transparency and public participation (Deus et al. 2019).

MSWM can lead to such positive social impacts as poverty reduction and economic development via employment and income generation (Menikpura et al. 2012a; Umair et al. 2015; Azimi et al. 2020b), mainly in such labour-intensive stages as collection, selective collection and recycling (Azimi et al. 2020b). For example, selective collection and recycling generate 83% of jobs in the Portuguese integrated packaging waste management system (Ferrão et al. 2014).

Improperly implemented MSWM can impact vulnerable populations due to lack of access to the service. Data suggests that around 2 billion people worldwide lack access to the regular waste collection (Wilson and Velis 2015). Additionally, damage to human health occurs through diseases spread by insects and rodents or respiratory diseases associated with different forms of air emissions (Erses Yay 2015; Khandelwal et al. 2019b). For example, De Feo et al. (2021) estimates that in Naples, Italy, each citizen loses the corresponding to 24 min of life (measured in DALY—disability-adjusted life year) due to emissions from paper and cardboard present in the unsorted residual MSW, and that 30 potential jobs are not generated.

MSWM also can induce occupational and health risks to workers (Ma and Hipel 2016) due to poor working conditions (Umair et al. 2015; Ibáñez-Forés et al. 2019). In some cases, there is a precarious relationship between the waste management service providers and the community that can pose risks to the health and safety of the community (Azimi et al. 2020a). The worst impacts are found in scenarios where the informal sector predominates (Yıldız-Geyhan et al. 2017). On the other hand, in some cities, it is in this sector that the highest recycling rates are also found (Yıldız-Geyhan et al. 2019) and, in some cases, it may be the most accepted by the community and generate greater socioeconomic repercussions (e.g. Yıldız-Geyhan et al. 2017 for Maltepe, Turkey). Therefore, due to poor working conditions, some authors classify scenarios with incineration (Aleisa and Al-Jarallah 2018; Zhou et al. 2019), mechanical biological treatment (Fuss et al. 2020) and mechanical recycling (Foolmaun and Ramjeeawon 2013) as preferable instead of selective collection and landfilling, as these generate a greater percentage of skilled jobs, and consequent improvements in health, safety and income (Zhou et al. 2019; Fuss et al. 2020). Other researchers point out that a scenario of integration of the informal sector with the formal sector is more suitable to generate social benefits (Yıldız-Geyhan et al. 2019; Azimi et al. 2020a), as it provides improvements in psychological and physical working conditions, as well as offering social benefits and education (Aparcana and Salhofer 2013a).

Deus et al. (2019) analysed the current level of the conceptual development of the indicators used in MSWM and found significant use of indicators related to the life cycle, such as environmental (LCA), economic (financial costs of MSWM) and energy (emergy tools), but little use of social indicators. However, there is a contemporary tendency to try for a complete assessment of goods and services through life cycle thinking and the three pillars of sustainable development (economic, environmental and social). The life cycle sustainability assessment (LCSA) is a methodology based on these concepts (European Commission 2011; Visentin et al. 2020).

Life cycle assessment (LCA) has been increasingly used and disseminated by scholars worldwide to analyse the environmental performance of solid waste management systems (Laurent et al. 2014; Christensen et al. 2020). More than 350 scientific articles have been published in the last twenty years in this area (Christensen et al. 2020). However, studies on social life cycle assessment (S-LCA) applied to MSWM are scarce (Yıldız-Geyhan et al. 2019). There is a need for future LCA studies to integrate social aspects (Paes et al. 2020; Zhang et al. 2021). When analysing sustainability, studies have shown that improved social indicators can reduce system efficiency (Vinyes et al. 2013) and increase costs (Harijani et al. 2017; Fattahi 2020; Iodice et al. 2021).

S-LCA is a technique for evaluating positive and negative impacts, including the potential social and socio-economic impacts of products, while considering their entire life cycle (UNEP 2020). The technique is still in development, with the first steps being taken between 1996 and 2009. From then until 2012, there were many uncertainties regarding the technique. However, the period from 2013 to 2016 was one of intense development in this field (Ramos Huarachi et al. 2020), probably due to the publication of the “*The methodological sheets for subcategories in social life cycle assessment (S-LCA)*” (Benoît et al. 2013), which complemented the publication “*Guidelines for social life cycle assessment of products*” (Benoît and Mazijn 2009). As of 2017, the number and scope of proposals for S-LCA methodologies with practical applications have grown. However, the circumstances to which they are now applied are so broad that there is currently a search for standardization (Ramos Huarachi et al. 2020). Iofrida et al. (2018) note that this variability in the S-LCA format stems from an absence of a strong epistemological foundation. For complete and rigorous development, it is necessary to ground S-LCA in the science of sustainability and social sciences (Iofrida et al. 2018).

S-LCA should be based on ISO 14040 and ISO 14044 (2006a and 2006b) standards (focused on Environmental LCA), but the wide variety of methods and different applications make its development a challenge. Therefore, research to clarify the current state of the art and identify future directions is required (Arcese et al. 2018). Several S-LCA review studies include treatments that address general aspects of the method (Wu et al. 2014; Chhipi-Shrestha et al. 2015; Petti et al. 2018; Bonilla-Alicea and Fu 2019; Huertas-Valdivia et al. 2020; Ramos Huarachi et al. 2020), while other studies cover specific issues, such as the development of indicators (Kühnen and Hahn 2017); establishing system boundaries (Dubois-Iorgulescu et al. 2018); in S-LCA type I (Russo Garrido et al. 2018); and implementation of the UNEP guidelines (Tokede and Traverso 2020). In addition, there are also reviews on the application of S-LCA to specific sectors of the economy, such as biodiesel (Macombe et al. 2013), mining (Mancini and Sala 2018) and urban mobility (Gompf et al. 2020). However, the literature search conducted for the current study found no reviews of S-LCA applied to municipal solid waste management. Therefore, there is a research gap, despite the concern about the social impacts of this topic (Ersees Yay 2015).

Consequently, this study aims to (i) analyse and describe the existing literature on S-LCA as applied to municipal solid waste management (MSWM), (ii) understand the current conceptual development level (iii) and identify the main methodological challenges and standardization requirements. In order to contribute to both the advancement of methodology and the harmonization of its application in

future studies. This can facilitate the interpretation and comparison and enhance the practical application of S-LCA to the improvement of MSWM.

1.1 S-LCA methodological background

The first phase in an S-LCA is the goal and scope definition. The study's goal, that is, why it is being conducted and the intended use of any results. This must be clearly defined, as several framing aspects that are decided on at this stage needs to be related to it, including the definition of the product system, the stakeholders and impact categories to be analysed (UNEP 2020). The scope determines the methodological framework (UNEP 2020). Some elements that should be defined in this phase are functional unit, reference flow, product system and the system boundaries, stakeholders, impact categories and subcategories, data collection strategies, and quality requirements (UNEP 2020).

The study object is quantitatively defined through a functional unit (FU) and should include the product's functions and specifications (UNEP 2020). The reference flow expresses the FU into specific product flows (UNEP 2020).

The definition of product system should seek to consider the product's entire life cycle, which includes the upstream (use of resources, purchase of goods and services) and downstream (distribution, use and end-of-life) processes. However, the system boundaries can be narrowed according to the S-LCA goal, available resources and data. In this case, the reasons must be duly justified (UNEP 2020). There are two different approaches adopted to determine the product system: the organization and the process perspectives. In the first approach, social impacts are more related to the behaviour of organizations where the processes occur rather than the processes themselves. Thus, the product system is composed of the organization within which the processes occur (Dreyer et al. 2006; Jørgensen et al. 2008; Macombe et al. 2011). In the second, a focus on the process forms the basis for the assessment, as used in environmental LCAs (Dreyer et al. 2006; Hunkeler 2006; Macombe et al. 2011). However, the two approaches are complementary (Paragahawewa et al. 2009). When using the organizational perspective, problems arise if the company produces more than one product, as it may become difficult to determine the portion of the total social impacts to be attributed to the respective products (Dreyer et al. 2006). Some authors suggest using allocation based on value creation or the number of work hours spent to resolve this problem (Dreyer et al. 2006; Martínez-Blanco et al. 2014).

S-LCA deals with impacts that affect stakeholders, these being the groups of social actors that have shared interests with organizations due to their relationships with the system under analysis and who can affect or be affected by the achievement of a given organizational purpose (UNEP

2020; Russi et al. 2012). They are classified into five main categories: workers, local community, society (national and global), consumers (along all stages of the value chain), value chain actors (e.g. providers) and children (UNEP 2020). Other categories or subgroups can be added according to the study and local characteristics: NGOs, governments, future generations, shareholders, management, business partners, etc. (UNEP 2020).

The impact categories associated with such social interest groups include human rights, working conditions, health and safety, cultural heritage, governance and socio-economic repercussions. These are examples of endpoint impacts (Benoît et al. 2013). The groups involved are abstract, general and heterogeneous, and their aggregation into scientifically valid impact categories is difficult to achieve (UNEP 2020). Therefore, they may be broken down into impact subcategories or midpoint impacts based on socially significant themes or attributes derived from international agreements and conventions (Dreyer et al. 2006; UNEP 2020). Finally, there are the inventory indicators. These are measured from the inventory data. Examples of this logic would include impact category: working conditions; subcategory: social security and benefits; and inventory indicators: (a) the percentage of employees covered by health insurance, (b) the percentage of employees covered by retirement insurance, (c) the percentage of employees covered by paid maternity leave, etc. (Benoît and Mazijn 2009; UNEP 2020).

The social life cycle inventory (S-LCI) mostly relates to data collection and quality. This can be a challenge with S-LCA, since social impacts are related to the behaviour of organizations and their interaction with stakeholders. Two organizations producing exactly the same product/service using the same processes may have quite different social impacts (Dreyer et al. 2006; Jorgensen et al. 2008). Therefore, when conducting an S-LCA, site-specific data is essential to guarantee enhanced representativeness of the results (Vinyes et al. 2013). However, obtaining data may be resource-intensive and is not always feasible (Kloepffer 2008). As a result, various solutions have been proposed to accommodate this issue, including using generic data (e.g. country or sector) to identify hotspots and determine the points to be evaluated with specific data (Dubois-Iorgulescu et al. 2018); or combining both data types, that is, use specific data for foreground processes and more generic data for background ones (Ciroth and Franze 2011; Martínez-Blanco et al. 2014). This permits a more comprehensive assessment of the life cycle because if it is not possible to assess certain processes/organizations based on primary and site-specific data; these can be assessed based on secondary sector/country data or databases.

The social life cycle impact assessment (S-LCIA) phase comprises the classification, aggregation and characterization steps (UNEP 2020). During classification, inventory

results are assigned to a specific stakeholder category and/or impact category (UNEP 2020). In the characterization step, S-LCI results are converted into mutually-compatible units and aggregated within the same impact category (ISO 14044 2006b). In S-LCA, the characterization is not always mathematical, as in environmental LCA, where it can be a basic aggregation step of qualitative data or involve comparison with performance benchmarks. Another aspect of S-LCA is the scoring and weighting (optional step) processes that may form part of the characterization stage or interpretation (UNEP 2020). There are two types of characterization approaches, Type I and Type II. The first assesses performance by scoring the inventory database on relative relevance (determined, for example, based on benchmarks, e.g. international agreements), then aggregating these using a scoring system. The second approach assesses the cause and effect of impact pathways, using characterization factors that have the results translated into a midpoint or endpoint impact category (UNEP 2020; Martínez-Blanco et al. 2014; Sureau et al. 2018; Tokede and Traverso 2020). There is no consensus concerning the S-LCIA phase (Yildiz-Geyhan et al. 2017; Ibáñez-forés et al. 2019). There are no characterization models between impact subcategories and categories in the Type I approach, which is the most used by LCA practitioners, and the impact pathway (Type II) models in S-LCA are not well developed (UNEP 2020). The update of the UNEP guidelines (UNEP 2020) does, however, cover this step, and it is expected that the current variability will be mitigated in future studies. However, there is currently no widely accepted method for the impact assessment phase (Souza et al. 2021).

The interpretation phase has been little explored in S-LCA studies, as many methodological obstacles still exist before this aspect can be considered complete and robust (Martínez-Blanco et al. 2014). According to UNEP guidelines (Benoît and Mazijn 2009), the interpretation phase should consist of (a) identification of significant problems; (b) assessment of completeness and consistency issues; (c) analysis of the level of engagement with stakeholders; and (d) conclusions, recommendations and reports. The UNEP guidelines (UNEP 2020) adds two steps: (e) sensitivity and data quality assessment; (f) materiality assessment.

2 Materials and methods

This study is characterized by a systematic literature review combining bibliometric and content analyses. These methods are complementary (Homrich et al. 2018). Bibliometric analysis is a quantitative method of analysing scientific production indicators used to describe the state of the art of an area of knowledge (Zupic and Čater 2015; Visentin et al. 2020; Donthu et al. 2021). Content analysis is a systematic

method for retrieving meaningful information from texts (Du et al. 2019); its steps are summarized in (a) planning the review, establishing of research questions, research strategy and coding; (b) conducting the review (frequency counts and cross-tabulations); and (c) interpretation of results, reporting and disseminating (Homrich et al. 2018). They were applied in this study to obtain an overview of how the studied research field has developed and to obtain a detailed view of how social assessment has been applied in MSWM in a life cycle perspective.

2.1 Sampling procedures

Firstly, the relevant keywords were defined, and the initial searches for articles to integrate the bibliographic portfolio were conducted. The keywords were defined based on the bibliometric study by Ramos Huarachi et al. (2020), who pointed out that the most used in S-LCA studies are: “SLCA”, “social impact”, and “social impact assessment”, and by Tsai et al. (2020), who reported “municipal solid waste” and “waste management” to be the keywords that appeared most frequently in studies on waste management. It was noticed that many studies encountered in the literature analysis used a form of life cycle sustainability assessment (LCSA) that combined social aspects with environmental and economic considerations. Such studies were also included in the study, and a new search was carried out on the databases using the keywords “life cycle sustainability assessment” AND (“waste management” OR “municipal solid waste”). Table 1 summarizes the combinations of keywords used in the searches.

The search was conducted in the *Web of Science* and *Scopus* databases. The first indexes all journals with an impact factor calculated in the Journal Citation Reports (JCR) (Paes et al. 2019), while the second one is the largest database of peer-reviewed literature (Homrich et al. 2018), and has significant coverage in engineering sciences (Larsen and von Ins 2010; Mostafa Hatami et al. 2021). The combination of keywords and boolean operators was delimited to “article title, abstract, and keyword” without date restriction. Only articles published in journals, in English, and up until 2021 were considered. A total of 273 articles were found on Scopus and 208 articles on the Web of Science (WoS).

After eliminating duplicates, the initial list was reduced to 365 articles. This secondary list of references then underwent an inclusion/exclusion analysis based on reading their titles and abstracts. This selection sought to exclude articles outside the scope of the study. Only articles containing case studies or methodology development that evaluated social aspects from a life cycle perspective applied to municipal solid waste management (MSWM) were included. In this way were studies included that:

- Claim to evaluate social aspects based on an LCA-like framework, as used by Jørgensen et al. (2008). Therefore, studies that had developed other methods with the LCA were included, such as cost–benefit analysis (CBA), multi-criteria decision analysis (MCDA), life cycle sustainability assessment (LCSA) and socio-technical analysis. Studies that used any of these methods but outside the LCA perspective were not considered. Those that analysed only social costs (as in LCC) were also excluded.
- Encompass municipal solid waste management systems. We also excluded studies that only focussed on upstream activities prior to waste generation (with the exception of waste prevention studies) or on post-waste management system aspects (how studies with a focus on secondary waste management systems aspects, such as the use of biomass/biowaste feedstock for energy or material production) (Laurent et al. 2014).

At this step, 304 articles were excluded, and the remainder 61 were subject to complete reading to verify the same inclusion/exclusion criteria, which eliminated a further 26 articles. As a result of reading the articles, one further relevant study was added (Foolmaun and Ramjeeawon 2013). This was present in the reference lists of articles being analysed but had not been retrieved during the database searches. This is probably because it did not use the keywords established, as the study focused on PET waste and did not use the keywords “waste management” OR “municipal solid waste”.

The steps for selecting the bibliographic portfolio are summarized in Fig. 1. By the end of the process, the bibliographic portfolio had 36 selected articles. These are

Table 1 Keywords used in the search for articles and results obtained

Keywords combination	Search results (No. of articles)	
	Scopus	Web of Science
1 (“SLCA” OR “social impact assessment” OR (“social” AND “life cycle assessment”)) AND (“municipal solid waste” OR “waste management”)	259	199
2 “life cycle sustainability assessment” AND (“waste management” OR “municipal solid waste”)	14	09

Search for articles on databases by Keywords combination

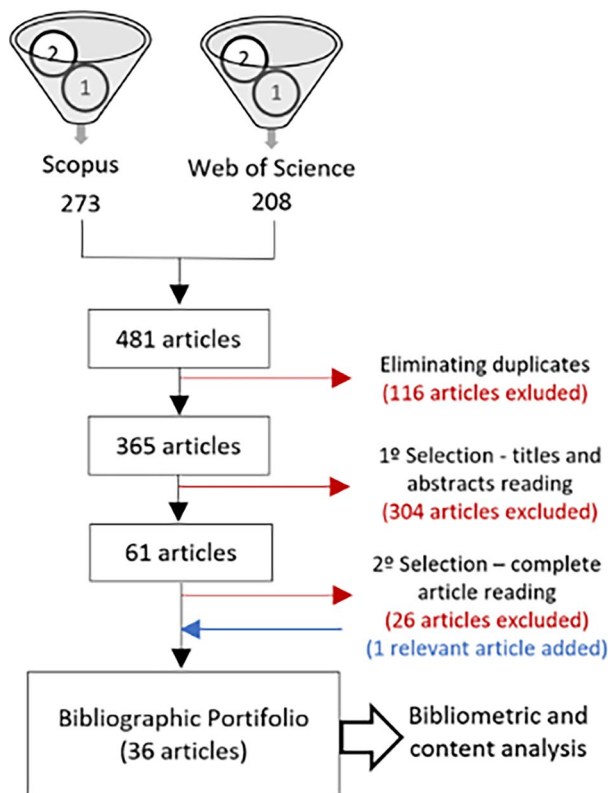


Fig. 1 Steps for the bibliographical portfolio selection

displayed in Table 2, which also includes a summary of the objective and method used in each study.

2.2 Data analysis

The selected articles were subjected to bibliometric and content analysis. In the sample of selected articles were analysed: the amount of publications over time; which sustainability dimensions were studied, that is, which areas belonging to the triple bottom line (environmental, social and economic) were evaluated; sources of publications in which journals publication occurred; the type of waste studied; and the geographic area of distribution of the case studies.

It was also recorded how the methodological aspects of the four phases of the S-LCA (definition of objective and scope, inventory analysis, impact assessment and interpretation) were applied in comparison to the requirements of ISO 14040 and 14044 (2006a; 2006b), and the recommendations of the UNEP guidelines (Benoît and Mazijn 2009; Benoît et al. 2013; UNEP 2020). It was sought to extract from the texts of the analysed articles information about: goal definition, function and functional unit, scope, stakeholders, impact categories and subcategories, quality and data source, method of impact assessment and interpretation.

3 Results and discussion

3.1 Literature panorama

The earliest study on the assessment of social impacts in the life cycle of municipal solid waste management was that of Kijak and Moy (2004). The authors propose a decision-making method that integrates environmental LCA with other tools to assess social and economic impacts. In subsequent years, no publications were identified within the scope of this study (explained in item 2.1), until 2011, when the first S-LCA study using the UNEP guidelines (Benoît and Mazijn 2009) was identified. From 2012 onwards, the number of publications began to increase, driven mainly by studies with LCSA approaches, probably inspired by the UNEP publication “Towards a life cycle sustainability assessment. Making informed choices on products” (Valdivia et al. 2011). In 2013, there was a significant increase in references to S-LCA (Fig. 2), most likely as a result of the publication of “The methodological sheets for subcategories in social life cycle assessment (S-LCA)” (Benoît et al. 2013), which detailed the impact subcategories and the construction of the inventory. From 2011 onwards, the overall trend was for a year-on-year increase in the number of publications related to the evaluation of social aspects within the perspective of the life cycle applied to MSWM, despite a decrease in some years (such as 2014 and 2018, which had 1 and 2 publications, respectively).

Of the 36 analysed studies, some 25% (9 studies) were devoted exclusively to the assessment of social impacts, and the majority (61%, that is, 22 studies) evaluated social considerations associated with environmental and economic aspects. The remaining studies (14% of the total) combined social and environmental aspects evaluations. These results indicate that S-LCA is gradually being incorporated into LCA studies of MSWM as part of a search for a method for complete sustainability analysis. This demonstrates the importance of the approach of the current study, where not only research devoted to S-LCA is considered, but also other studies that include social assessments using the life cycle perspective. Therefore, articles that would be otherwise considered beyond the scope of the study can be included, making it possible to employ a broad view of how social aspects have been evaluated in MSWM studies from a life cycle perspective.

The journals that published the most articles on the subject were “Resources, Conservation & Recycling”, “Journal of Cleaner Production” and the “International Journal of Life Cycle Assessment”, which tied in first place with 6 publications each. The journals that had publications in this area can be seen in Fig. 3. Additionally, Table 2 includes the journal in which each analysed article was published.

Table 2 Overview of the 36 selected studies

ID	Article	Journal	Study objective	Method/framework
1	Kijak and Moy (2004)	JIE	To develop a decision support tool to achieve sustainability in solid waste management by balancing environmental, social and economic impacts at global, regional and local levels	Integrates Environmental LCA, pressure-state-response model to measure social and economic aspects and multi-attribute utility theory
2	Lehmann et al. (2011)	S	To discuss the selection of social impact subcategories for integration with decision-making software in waste management packaging in Spain and Portugal	Employs Social Impact Assessment and S-LCA
3	Menikpura et al. (2012a)	WM&R	To assess the sustainability of MSW system in Nonthaburi, Thailand	Own LCSA framework combining Environmental LCA and socioeconomic indicators
4	Menikpura et al. (2012b)	JMCW	To assess the impacts of existing open dumping situation in Sri Lanka via environmental, economic and social indicators, using a life cycle approach, and compare with a landfill with gas recovery scenario	Employs Environmental LCA, Life Cycle Costing and Social Impact Assessment
5	Aparcana and Salhofer (2013a)	IJLCA	To assess the social impacts of three Peruvian recycling systems based on two formalisation approaches and determine the feasibility of an S-LCA method developed by author's previous work	Own S-LCA method developed by author's previous work based on different methodologies and case studies on Social Impact Assessment and S-LCA
6	Aparcana and Salhofer (2013b)	IJLCA	To develop a methodological procedure for assessing the social impacts of formalized versus informal recycling systems in low-income countries	Own S-LCA method based on different methodologies and case studies on Social Impact Assessment and S-LCA
7	Foolmaun and Ramjeeawon (2013)	IJLCA	To investigate and compare the environmental and social impacts of four waste disposal alternatives to PET bottles in Mauritius	Employs Environmental LCA and S-LCA
8	Hu et al. (2013)	IJLCA	To propose a method of LCSA application in a concrete recycling case study	Employs an LCSA method that combines Environmental LCA, Life Cycle Costing and S-LCA
9	Menikpura et al. (2013)	WBV	To assess the sustainability of both recycling and sanitary landfilling in Nonthaburi, Thailand and to estimate the contributions of recycling to improving the sustainability	Own LCSA framework combining Environmental LCA and socioeconomic indicators
10	Tulokhonova and Ulanova (2013)	WM&R	To assess and compare ecological, economic and social aspects of four life cycle model-based integrated MSW management scenarios in Irkutsk, Russia	Integrates ecological, economic, and social aspects using a life cycle-based integrated waste management model
11	Vinyes et al. (2013)	IJLCA	To assess and compare the sustainability of three domestic waste oil collection systems in a 10,000-inhabitant neighbourhood in Barcelona, Spain, to determine which systems should be promoted for the authorities	Employs an LCSA method that combines Environmental LCA, Life Cycle Costing and S-LCA
12	Ferrão et al. (2014)	RCR	To develop a methodological approach and assess the environmental, economic and social impacts of packaging waste management system in Portugal	Combines Environmental LCA, economic Input–Output Analysis, and employment characteristics generated by the system
13	Souza et al. (2015)	JCP	To develop and apply a methodology for selecting social and economic impact categories with stakeholder consultation	Own methodology based on a combination of theories drawn from LCA and decision science concepts and tools
14	Umair et al. (2015)	RCR	To investigate the social impacts of informal electronic waste (WEEE) recycling in Pakistan to provide decision-making support	Employs S-LCA using the UNEP guidelines
15	Souza et al. (2016)	WM	To assess the sustainability and prioritise electronic waste management systems alternatives in Rio de Janeiro, Brazil	Combines Environmental LCA, qualitative assessment of social and economic indicators, and multi-criteria analysis

Table 2 (continued)

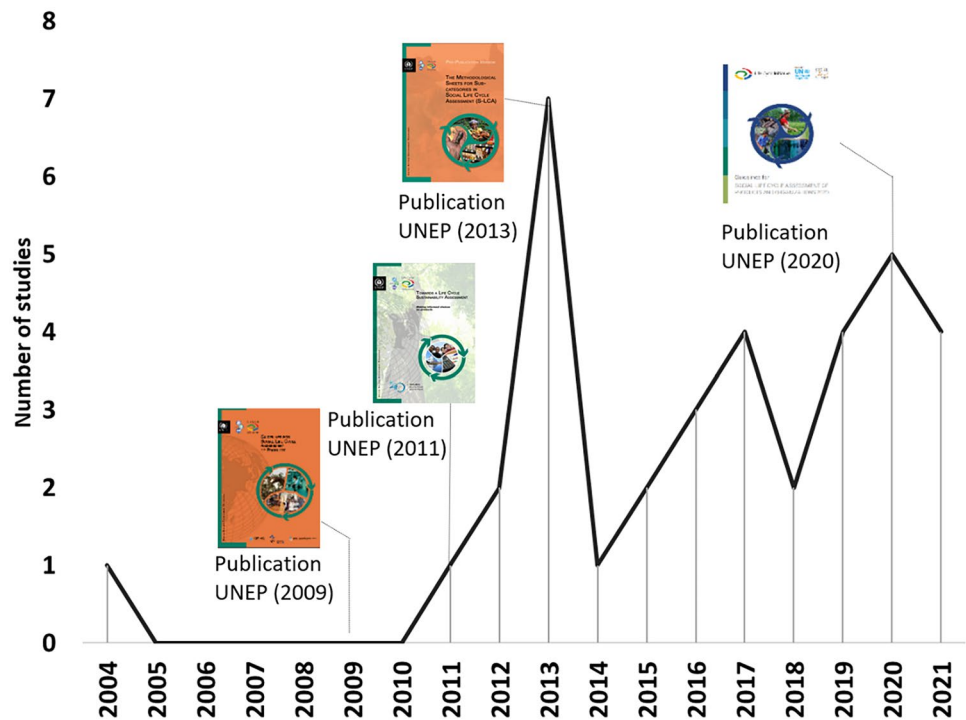
ID	Article	Journal	Study objective	Method/framework
16	Sou et al. (2016)	WM&R	To assess the sustainability of incineration ash management scenarios in Macao, and determine the most suitable treatment method	Integrates localized health risk assessment, Environmental, Cost-benefit Analysis, Fuzzy theory and Analytical Hierarchy Process
17	Lu et al. (2017)	S	To identify and raise materiality issues to improve Green House Gases (GHG) management of incineration plants, and to determine potential issues in the improvement of management via qualitative analysis of national laws and regulations	Employs life cycle inventory analysis, S-LCA and materiality analysis
18	Harijani et al. (2017)	JCP	To propose a systematic approach to facilitate decision-making in choosing an ideal and sustainable MSW recycling and disposal system	Integrates a multiperiod mixed integer linear programming model with an extended form of S-LCA
19	Yildiz-Geyhan et al. (2017)	RCR	To assess the social impacts of packaging waste management systems scenarios in Istanbul, Turkey	Employs S-LCA method
20	Aleisa and Al-Jarallah (2018)	IJLCA	To extend Environmental LCA to economic and social aspects for MSW system development in Kuwait	Integrates Environmental LCA, statistical costing method, social indicators and analytical hierarchy process
21	Nie et al. (2018)	WM	To propose a Decision Support System (DSS) model that considers the three pillars of sustainability when assessing MSW classification scenarios in Pudong, China	Integrates Cost-benefit analysis (CBA), Environmental LCA and analytical hierarchical process
22	Ibáñez-Forés et al. (2019)	EI	To propose and apply a set of social indicators to assess the socio-economic and working conditions of different actors involved in the life cycle of an MSW management system in developing countries	Employs social performance assessment based on S-LCA
23	Santos et al. (2019)	JCP	To propose an S-LCA-based management conceptual model for areas degraded by irregular solid waste disposal	Own S-LCA methodology, based on UNEP guidelines and open participatory management tools
24	Yildiz-Geyhan et al. (2019)	RCR	To assess the environmental and social impacts of packaging waste management systems scenarios in Turkey	Combines Environmental LCA with S-LCA
25	Zhou et al. (2019)	JCP	To extend and apply the previous 3E model towards a 3E + S model to assess the sustainability of current waste treatment systems in China	Integrates Environmental LCA, Life Cycle Costing, S-LCA through its 3E + S model (environmental, energy, economic and social) and applies Multi-criteria Decision Analysis that integrates Analytical Hierarchy Process (AHP) with an entropy weighting method
26	Azimi et al. (2020a)	S	To assess the social performance of the MSWM system of Kabul, Afghanistan	Employs S-LCA method
27	Azimi et al. (2020b)	S	To assess the environmental economic and social aspects of MSWM scenarios in Kabul, Afghanistan	Combines Environmental LCA, Life Cycle Costing, and job creation measurement for each scenario
28	Fattahi (2020)	AOP	To propose and apply a data-driven two-stage stochastic programming model to estimate the social performance of MSW treatment facilities in Tehran, Iran	Integrates a two-stage stochastic mathematical model with S-LCA
29	Fuss et al. (2020)	RCR	To contribute to an understanding of how to construct sociotechnical scenarios for future orientation when implementing Mechanical Biological Treatment (MBT) in countries with emerging economies	Combines Environmental LCA with socio-technical analysis

Table 2 (continued)

ID	Article	Journal	Study objective	Method/framework
30	Taelman et al. (2020)	RCR	To develop an operational sustainability framework stakeholder-inclusive to assess MSW system sustainability	Own LCSA methodology which uses the ELECTRE II (Elimination and Choice of Translating Reality) method in an environmental and economic LCSA with local social impact assessment and multi-criteria decision analysis (MCDA)
31	De Feo et al. (2021)	JCP	To develop a method for evaluating the potential environmental, social, and economic benefits of paper and cardboard discarded in the unsorted municipal solid waste (MSW) fraction	Own method that integrates six LCSA-based indicators (carbon footprint, ecological footprint, health footprint, occupational benefits, economic profit and economic saving)
32	Handakas et al. (2017)	FEB	To assess alternative scenarios for municipal solid waste management (MSWM) in the municipality of Paionia, Greece	Combines Environmental LCSA, Cost-benefit analysis, SWOT analysis and societal acceptability
33	Iodice et al. (2021)	WM	To assess the sustainability of the Construction and Demolition Waste (CDW) management in the Campania Region, Italy	Employs the sustainability framework developed in Taelman et al. (2020), and further complements with Societal Life Cycle Costing
34	Magrini et al. (2021)	STE	To propose a framework for the sustainability assessment and prioritisation of waste prevention measures at consumption level	Integrates Environmental LCSA, Life Cycle Costing, S-LCA, and efficiency indicators
35	Sanjuan-Delmás et al. (2021)	WM	To assess the social, environmental and economic impacts of the organic fraction of municipal solid waste (OFMSW) in three EU cities and compare them	Employ the sustainability framework from Taelman et al. (2020)
36	Arena et al. (2016)	JCP	To assess the technical feasibility, environmental compatibility and social aspects of a recycling process of absorbent hygiene products	Combines assessment of technological feasibility with Environmental LCSA and S-LCA

IJLCA Journals International Journal of Life Cycle Assessment, *RCR* Resources, Conservation & Recycling, *JCP* Journal of Cleaner Production, *S* Sustainability, *WM&R* Waste Management & Research, *WM* Waste Management, *AOP* Annals of Operations Research, *EI* Ecological Indicators, *JIE* Journal of Industrial Ecology, *JMCW* Journal of Material Cycles Waste Management, *WBV* Waste Biomass Valor, *FE* Presentius Environmental Bulletin, *STE* Science of the Total Environment

Fig. 2 The number of studies for year and marks of UNEP's guidelines

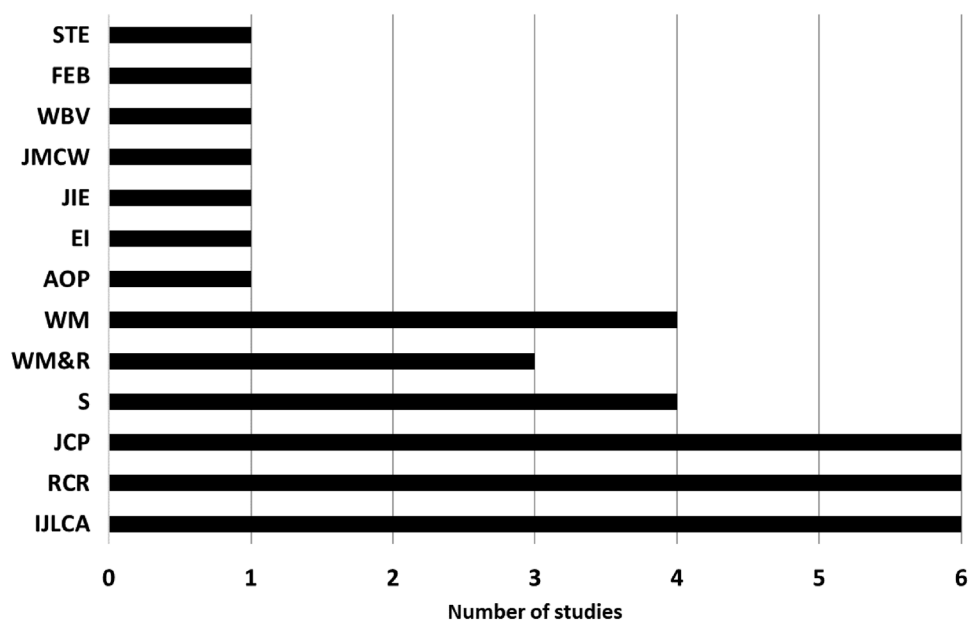


Most of the studies (20) focused on a specific waste type, while the remainder (16) evaluated municipal solid waste management. Within the first group, recyclable waste (composed mainly of plastic, paper, glass and metal) was evaluated in 10 studies, waste from electrical and electronic equipment (WEEE) in 3 and construction and demolition waste (CDW) in 2. The forms grouped in Fig. 4 under “Others” appeared in 1 study each: MSW incineration ash; evaluation of waste and industrial waste

incineration plants; cooking oil; absorbent hygiene products; and organic fraction.

Analysis of the countries in which the case studies occurred took place for 32 references, with 4 studies excluded as they focused exclusively on methodology development. Two of the references evaluated more than one country, and these were added to the count, giving a total of 35 locations and 22 different countries. Figure 5 shows the geographic distribution of case study publications. It can

Fig. 3 Distribution of publications across journals. *IJLCA* Journals International Journal of Life Cycle Assessment, *RCR* Resources, Conservation & Recycling, *JCP* Journal of Cleaner Production, *S* Sustainability, *WM&R* Waste Management & Research, *WM* Waste Management, *AOP* Annals of Operations Research, *EI* Ecological Indicators, *JIE* Journal of Industrial Ecology, *JMCW* Journal of Material Cycles Waste Management, *WBV* Waste Biomass Valor, *FE* Fresenius Environmental Bulletin, *STE* Science of the Total Environment



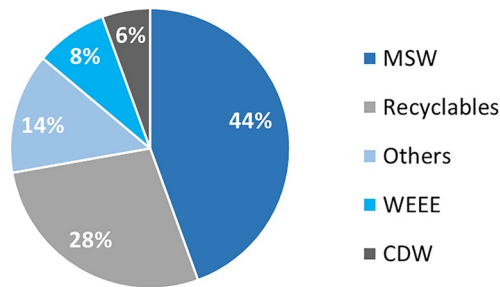


Fig. 4 Typology of waste evaluated in the selected studies

be seen that Italy was the country with the highest number of publications (4 studies) followed by Brazil and China (3 publications each).

Most studies (19, i.e. 59%) were carried out in upper-middle (13), lower-middle (4) or low (2) income countries (The World Bank 2021). While 13 studies (that is, 41%) took place in high-income countries. In Environmental LCAs studies, there was a predominance of studies from high-income countries (Khandelwal et al. 2019a, b), which was not observed in the analysed S-LCA studies. This indicates a greater concern of researchers with the social impacts of MSWM in middle and low-income countries than in high-income countries.

The literature search revealed a scarcity of S-LCA studies in MSWM, especially when compared to Environmental LCA in waste management studies, on which hundreds of

articles have been published in the past decade (Christensen et al. 2020). Consequently, there is a need to conduct more S-LCA studies in MSWM and, in addition, to expand study coverage across the globe. Studies in countries in Africa, Latin America and the Caribbean and South Asia are recommended since countries in these regions are known to be facing MSWM problems (Wilson et al. 2015; Bui et al. 2022). In addition, studies in high-income countries that explore social issues and identify potential critical social aspects of MSWM would also be interesting, as this type of study in these countries is little explored.

3.2 S-LCA's methodological issues applied to MSWM

The methods used by studies to conduct S-LCA were varied. This is because, of the 36 studies retrieved, 47% carried out methodological development. Of those 13 were methodological developments with case studies; and 4 were exclusively methodological developments. The rest of the studies were exclusively case studies (19, that is, 53%). However, the majority (14, that is, 39%) used the UNEP guidelines as a basis, and about 36% (13 studies) claim to have developed their own method. This included Aparcana and Salhofer (2013b), who developed their own method based on different methodologies and case studies on social impact (including S-LCA), among them UNEP guidelines, and Zhou et al. (2019), who developed their own LCSA method with the 3E + S model (environmental, energy, economic and social).

Country	Studies	Income
Italy	4	H
Brazil	3	UM
China	3	UM
Portugal	2	H
Spain	2	H
Afghanistan	2	L
Iran	2	LM
Thailand	2	UM
Turkey	2	UM
Australia	1	H
Kuwait	1	H
Netherlands	1	H
Taiwan	1	H
Pakistan	1	LM
Sri Lanka	1	LM
Mauritius	1	UM
Peru	1	UM
Russia	1	UM
Belgium	1	H
Germany	1	H
Hungary	1	H
Greece	1	H

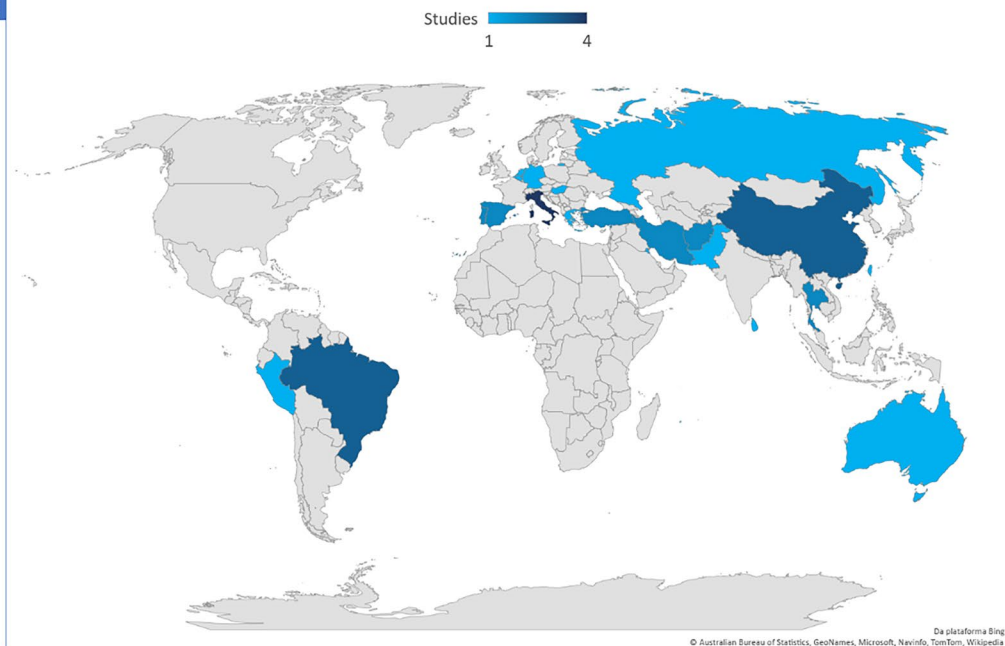


Fig. 5 Geographical distribution of case studies based on locations of waste management systems under study (total of 26 studies). Income: Low (L); Lower-middle (LM); Upper-middle (UM); High (H)

About 28% (that is 10 studies) used multi-criteria decision analysis (MCDA) in the phases of assessment (e.g. Harijani et al. 2017), weighting (e.g. Aleisa and Al-Jarallah 2018) or aggregation (e.g. Zhou et al. 2019). MCDA is often used in waste management studies (Hoang and Fogarassy 2020). The advantage is that this enables the application of both quantitative and qualitative criteria to evaluate the sustainability of alternatives with the cooperation of various groups of stakeholders (such as specialists or decision-makers) (Hoang and Fogarassy 2020). This indicates that waste management researchers are already used to using this tool, which may explain its choice for integration with S-LCA and other methods (e.g. LCSA or CBA).

The remaining studies used other methods or a combination of methods: Lu et al. (2017) used green house gases (GHG) inventory analysis with S-LCA based on UNEP guidelines and materiality analysis; while Nie et al. (2018) used cost–benefit analysis (CBA), environmental LCA and analytical hierarchical process (AHP). These findings are in line with the observation of Jørgensen et al. (2008) that S-LCA is developing in a scientific field with many parallels to methodological discussions of cost–benefit analysis (CBA), social impact assessment (SIA), social accounting and others. The methods and objectives of the analysed studies are briefly presented in Table 2, since it is not the aim of this study to analyse the general methods used in the articles, but cover only those relating to the S-LCA section of such studies.

3.2.1 Goal and scope (G&S) definition

Two main classes of objectives were found in the analysed articles: (a) support public policies and assist decision-making (15 articles), and (b) identify and measure impacts (11 articles). Another series of studies (8 articles) did not specify an S-LCA based goal. Comparison of product/process/organization, which is a commonly used objective in S-LCA (Jørgensen et al. 2008), was present only in de Souza et al. (2016) and Vinyes et al. (2013). The latter deserves to be highlighted, as it compares three types of cooking oil waste collection systems and establishes the system's function, functional unit and reference flow according to its efficiency. The lack of studies with a comparison-based objective is probably due to the experimental nature of the research (almost half focused on methodological development). The difficulty in relating analysed impacts to the functional unit is that inputs and outputs must be standardized to ensure comparable results (Tokede and Traverso 2020).

The difficulty in correlating the results of the indicators to the functional unit was almost ubiquitous, as studies are often based on qualitative or semi-quantitative data, which complicates their correlation with the functional unit used (Umair et al. 2015; Zhou et al. 2019). Of the total number

of studies evaluated, only 8 studies (22%) relate the impacts to functional units: Menikpura et al. (2012a, 2012b, 2013), Vinyes et al. (2013), Taelman et al. (2020), De Feo et al. (2021), Iodice et al. (2021) and Sanjuan-Delmás et al. (2021). All adopted quantitative impact categories, with Vinyes et al. (2013) also adding a scoring system approach. The impact assessment phase of these studies will be explored in more detail in Sect. 3.2.2.

Three main types of functional units were used in the evaluated studies: 1 ton of a given waste type (13 studies); the total amount of a given type of waste in a region (4 studies); or the total amount of waste managed by a given actor over a defined period (8 studies). One study used 500 kg and another 10 studies did not specify the functional unit.

In the articles analysed, the division between organizational and process approaches to determine the product system was not always clear. In most cases, the municipality is responsible for managing the MSW and is its sole executor, which results in an overlap between the process and the organization formats. Another potential methodological problem involves the definition of the life cycle parts and the processes or organizations that lie within the scope of the studies, which were often not very detailed. The studies established the processes that were part of the product system (e.g. collection and final disposal). However, they did not explain the exclusion criteria for other processes, whether it is irrelevant due to the purpose of the study because of the low impact of a particular process, lack of data etc.

Even the studies that carried out an Environmental LCA together with S-LCA clearly defined the system limits for the Environmental LCA, did not use the same limits for the S-LCA and did not consider the life cycle perspective for all social indicators used in the study. This was noted previously by Lehmann et al. (2011). Several reasons might explain this phenomenon, including differences in the various stages in relevance, data availability and practical interest to decision-makers (Lehmann et al. 2011). This should be justified in any future studies, and the possible outcomes of these choices indicated. For example, Menikpura et al. (2012b) designed a product system that ranged from oil extraction to the replacement of co-products (in this case, landfill gas energy reuse) and analysed the social impact categories “impact on human health” and “land occupation” for all life cycle stages. However, for the category “employment opportunities”, the system was limited to the stages of collection, transport and final disposal since the other stages lacked relevance in social terms.

The lack of systematization and transparency in the delimitation of system boundaries can reduce the robustness of a study, its potential for application elsewhere and comparison (Dubois-Iorgulescu et al. 2018), as well as increasing the level of uncertainty and the risk of neglecting indirect

effects or externalities (Lagarde and Macombe 2013). It is recommended that studies clearly define the product system and detail both the “cut-off criteria” that led to the exclusion of certain processes/organizations and the impact of these decisions on the study results.

The analysed publications were classified according to their scope in terms of the evaluated life cycle stages. The waste collection was considered the initial stage since in MSWM Environmental LCAs it is common to adopt a “zero burden assumption”, which states that the waste enters the system without an environmental load (Laurent et al. 2014). Thus, cradle-to-cradle was considered from the collection of waste to the substitution of co-products in the production chain (e.g. secondary/recycled raw material); cradle-to-grave corresponded to the phase from collection to landfill rejects; cradle-to-gate included from collection to a treatment unit; and gate-to-gate covered from waste input to output at the treatment unit.

This classification showed that the main approach was cradle-to-grave (12 studies), followed by cradle-to-gate (7 studies), cradle-to-cradle (8 studies), gate-to-gate (4 studies), gate-to-grave (2 studies), totalling 33 articles. The other articles only addressed methodology development and did not determine a product system (Souza et al. 2015; Santos et al. 2019; Taelman et al. 2020), leaving each study responsible for applying the methodology according to the local conditions and purpose of the study.

The most analysed MSWM stages in the articles were collection/transport, pre-treatment in recovery facilities (identified in some studies as MRF, “material recovery facility”), and the final disposal of waste. This was because there is greater job generation at the collection and separation stage, hence the concern regarding working conditions (Azimi et al. 2020b). In landfills, the concern

lies with how such installations affect local communities (Menikpura et al. 2012b). In these stages, separation, formal collection and final disposal in the landfill were the most frequently analysed aspects (in 24, 22 and 19 studies, respectively). It is noted that, as a field, S-LCA studies have been moving steadily towards including the assessment of critical social issues in waste management, such as informal collection (6 studies), and open dumping (3 studies), reflecting the impact of conducting research in developing countries in Asia and South America. However, there is room to increase studies in these sectors. Of the treatment technologies, incineration was the one most analysed (16 studies), followed by mechanical recycling (14 studies) and anaerobic digestion (8 studies). Figure 6 shows the frequency at which each stage of MSWM was analysed in the studies.

There is a need to expand these studies and evaluate other issues, such as waste segregation at source (e.g. Tulokhonova and Ulanova 2013), and waste generation reduction/prevention (e.g. Magrini et al. 2021 and Sajuan-Delmás et al. 2021), which are also critical social issues. For example, waste segregation at the source is influenced by stakeholder preferences (Nie et al. 2018), and millions of vulnerable people earn their livelihoods through informal waste management (Wilson et al. 2006; Linzner and Lange 2013).

3.2.1.1 Stakeholders From the analysed studies, it was possible to identify which stakeholder category (SC) is involved in MSWM, examples of actors belonging to each SC and the most evaluated impact subcategories for each SC (Table 3). This list can be used by future studies in the area when deciding which stakeholder groups need to be considered. However, it is not exhaustive, and other actors may need

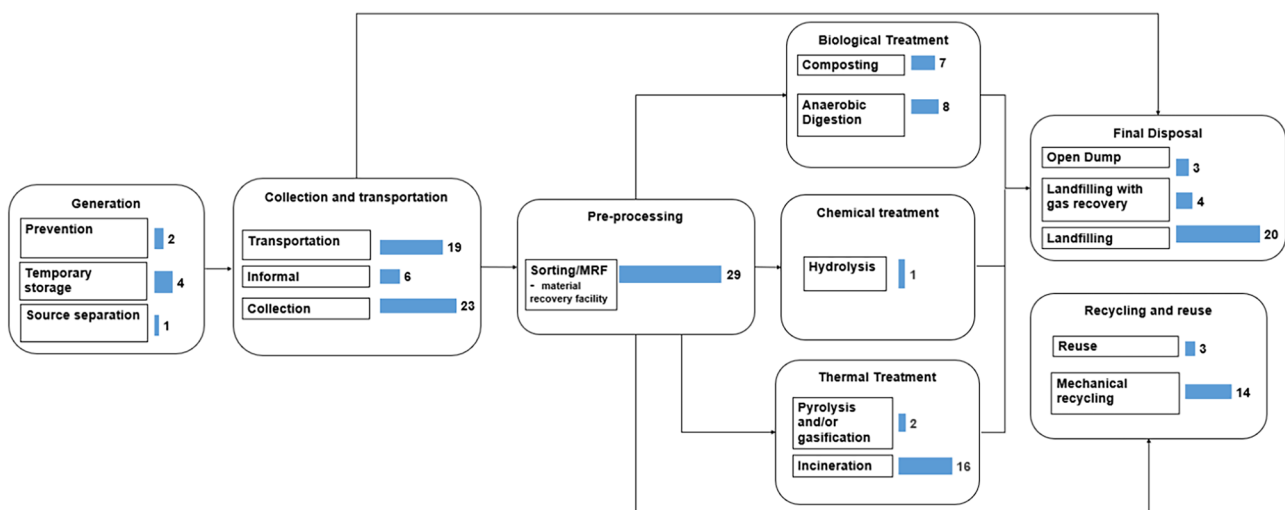


Fig. 6 Frequency of analysis of waste management processes addressed in analysed articles

Table 3 Stakeholders listed by selected studies and impact subcategories assessed

Stakeholder category	Examples of actors	Percentage of studies that assessed the SC	Most evaluated impact subcategories for respective SC
Workers	Administrative and operational, formal and informal, public and private workers involved in waste management (e.g., drivers, collectors, waste pickers, traders, employees of waste management administration; employees of waste treatment units, etc.)	89%	Health and safety/working conditions; work hours; Equal opportunities and discrimination; Child labour; Fair wages; Social benefits; Professional development/training
Local community	City residents; people living near waste management facilities	89%	Employment/local employment; Community engagement and participation; Safe and healthy living conditions; Odour
Society	City population; country population	47%	Contribution to economic development; Public commitment to sustainability; Social acceptance; Land use/urban space
Consumers/Users	Populations served by the waste management service; waste generators (e.g., entire population of a municipality); citizens participating in waste collection	42%	Consumer health and safety; Waste management quality/ service satisfaction; Comfort/Convenience/Complexity
Workers families	Children, spouses and dependents of sectoral workers	14%	Socio-economic conditions and income generation
Value chain	Manufacturers; importers; owners and employees of waste-related businesses; waste picker cooperatives and associations	11%	Relationship with suppliers and stakeholders
Government	Local authorities; municipal public management; decision-makers; planning sector	8%	Legislation

to be added or removed depending on the local situation of the system under study. It is noted that, in addition to the main stakeholders' categories provided by UNEP guidelines (Benoît and Mazijn 2009; Benoît et al. 2013; UNEP 2020), categories of workers' families, government and third sector are added here.

Stakeholders are important for social assessment, as S-LCA results can vary according to their perceptions (Jørgensen et al. 2008). Their consultation can help to understand the impacts and social processes relevant to each case study (Tokede and Traverso 2020). Indeed, stakeholders' involvement can improve the definition of impact categories, subcategories and indicators, as well as serve as a source of information for primary/specific data inventory (Ibáñez-Forés et al. 2019).

Most (64%) of the analysed studies did not describe the reasons that supported the selection of stakeholders. The rest of the studies cited the way of choosing the stakeholders, mostly following the UNEP guidelines list. From this, the authors of each paper would generally select stakeholders based on the objective of the study, based on the relevance for the studied system, as determined by the initial literature review each study had made. Studies were also restricted to stakeholders directly connected with the system under study (for example, waste collection workers or the community adjacent to a landfill).

Half the studies did not specify the stakeholders despite adopting them, but it was possible to infer which stakeholders were being analysed from the adopted impact categories and subcategories. Workers and local community were analysed in 32 of the 36 studies (89%); society in 17 (47%), consumer in 15 (42%), worker's families in 5 (14%), value chain in 4 (11%) and government in 3 (8%) (Table 3).

This distribution reveals a concern regarding MSWM impacts on both sector workers and those local communities directly impacted by the system. It also indicates a need for more studies on "consumers" and "value chain actors" to promote a better understanding of the role of these actors in MSWM and its potential impacts. This is also the conclusion of the review by Tokede and Traverso (2020). Table A1 (Online Resource) provides information on which stakeholders are evaluated in each analysed article.

3.2.2 Categories, subcategories and impact indicators

As previously mentioned, in an S-LCA, the recognition of social impacts is variable. Thus, a challenge for any method is to define which impact categories should be included in the assessment and how to measure social impacts (Jørgensen et al. 2008; Benoît et al. 2010). The approaches most used by the authors of the 36 analysed articles when selecting the impact categories/subcategories were based on (a) consultation with specialists in the area, decision-makers and/or

stakeholders (28%); (b) the UNEP guideline list (Benoît and Mazijn 2009; Benoît et al. 2013) (28%); (c) reviewing the literature on the main social aspects relevant to the topic of study and previous research related to S-LCA practice (17%); (d) author judgments on topics relevant to the study context (17%); and (e) study goal (14%) (f) availability and reliability of data (11%). Half of the selected articles combined the use of more than one approach.

A total of 67 different impact categories/subcategories were represented in the analysed studies (see Table A2 in Online Resource). This relatively high number demonstrates the general lack of consensus on which categories to evaluate in a MSWM S-LCA. Furthermore, there is a lack of standardization and consensus in using the terms for the impact categories, subcategories and indicators and their meanings. Something that is used in one study as an impact category may be deployed in another as an indicator. For example, “working hours” is used as a social indicator in Ibáñez-Forés et al. (2019) and by Vinyes et al. (2013), but as an impact subcategory in Aparcana and Salhofer (2013a, b) and by Umair et al. (2015). Therefore, these terms are recommended to be standardized based on UNEP guidelines (Benoît and Mazijn 2009; Benoît et al. 2013; UNEP 2020).

The impact subcategories employment/local employment and health and safety/working conditions are considered the most relevant for S-LCA studies of MSWM, being used in 61% and 50% of the studies, respectively. Other subcategories used frequently in the studies were (a) working hours; (b) community involvement/participation; (c) safe and healthy living conditions in the community (40% each); (d) equal opportunities/discrimination/social inclusion; (e) contribution to economic development (37% each); (f) child labour (33%); (g) fair salary; (h) benefits/social security; and (i) professional development/education/training (30% each).

Only 7 studies used endpoint impact categories: Menikpura et al. (2012a; 2012b; 2013), Taelman et al. (2020), De Feo et al. (2021), Iodice et al. (2021) and Sanjuan-Delmás et al. (2021), the latter two use the methodology by Taelman et al. (2020). These all quantitatively assessed the harm to human/public health and community/human well-being categories. This is explored in more detail in Sect. 3.2.2.

These results corroborate those of the stakeholder analysis, that social issues involving workers and the local community are topic hotspots in an MSWM S-LCA. These are directly involved with the MSWM, a powerful generator of jobs and economic development, but with concerns regarding working conditions (Umair et al. 2015; Yıldız-Geyhan et al. 2017). In addition, MSWM is a service provided to improve the community’s living conditions and, at the same time, it requires community acceptance and collaboration to operate efficiently (Menikpura et al. 2012b; Nie et al. 2018).

Another factor that may favour the prevalence of the use of these stakeholders is the dissemination of approaches to evaluate the indicators of the impact subcategory. For example, as part of the stakeholder “workers”, is evaluated the impact subcategory “employment/local employment” observing the indicator “number of employees”. In contrast, it is not clear for “value chain actors”, for example, how to evaluate the relationship with suppliers.

The impact categories are analysed using indicators. These can be qualitative (descriptive), semi-quantitative (e.g. scores) or quantitative (e.g. physical units) (Tokede and Traverso 2020). In the analysed studies, there was a predominance of semi-quantitative indicators (20 studies, 55%), followed by quantitative (11 studies, 30%), while a smaller number of studies used qualitative indicators (e.g. Ferrão et al. 2014; Umair et al. 2015; Azimi et al. 2020b). A combination of different indicators was also found, as with Arena et al. (2016) and Fuss et al. (2020). Table A2 (Online Resource) shows impact categories and the type of indicator used by the 36 analysed studies, which can guide future authors when choosing their studies.

The large number and variety of indicators and inventory data used by the evaluated articles made it impossible to detail them in this study. However, to illustrate the variety, we can cite the work of Azimi et al. (2020a) which used 90 inventory indicators to represent 20 impact categories derived from 8 stakeholders; and Ibáñez-Forés et al. (2019), who analysed 29 inventory data derived from 22 indicators and 12 impact categories.

The high number of indicators can also make it difficult to carry out S-LCA, due to the greater time and resources required for data collection. In addition, there is an increasing chance of double counting, making it difficult to interpret and communicate the results and their application and for decision-makers to define policies. Therefore, it is recommended that studies limit the indicators to those really relevant to the existing study context. The authors Lehmann et al. (2013), De Luca et al. (2015) and Tokede and Traverso (2020) also emphasize the importance of context. Prioritisation can be done based on previous work, using the multi-criteria analysis method, involving stakeholders and experts in the field. Examples of such approach include Bouillass et al. (2021), Gompf et al. (2020), Santos et al. (2019) and Souza et al. (2015), the last two of which involve MSWM. Similarly, Sureau et al. (2018) recommend using a participatory approach method when selecting impact categories and choosing indicators to reflect stakeholder values. Such measures can help ensure relevance, robustness and practicality in selected indicators (Tokede and Traverso 2020). In this sense, the UNEP (2020) improves on this by suggesting approaches for defining the stakeholders and impact subcategories.

3.2.3 Social life cycle inventory

In the analysed studies of S-LCA applied to MSWM, those based on local and primary data predominated (59% of the studies), followed by mixed local data associated with generic and primary with secondary (28%) information. The sources for obtaining such data are mostly interviews and/or field observation (63%), though documents and reports (31%) and expert opinion (19%) were also used. It is common to use more than one of these sources in combination. Use of S-LCA databases (e.g. PSILCA and SHDB) did not occur for any of the analysed studies.

Interviews and field observations were the primary data source for 20 studies, but only 7 showed concern with the sample size and covered this step in the methods. Triangulation for data validation was performed in 5 studies, from the crossing between information obtained in interviews and field observations or with the answers provided by different actors in the interviews (e.g. workers and managers) and secondary data. Data quality was discussed in 8 studies, 5 of which evaluated their data quality descriptively, mentioning the means of obtaining data and factors that could reduce data quality, while 4 studies performed consistency, uncertainty or sensitivity analysis. Details on data collection methods and quality can be found in Table A3 (Online Resource).

3.2.4 Social life cycle impact assessment

The type I approach was the most commonly used for the studies analysed in this review (25 studies). The type II approach was adopted in 6 studies (Menikpura et al. 2012a, 2013; De Feo et al. 2021; Umair et al. 2015; Taelman et al. 2020; Iodice et al. 2021; Sanjuan-Delmás et al. 2021). Menikpura et al. (2012b) and De Feo et al. (2021) mixed the two approaches.

The social life cycle impact assessment (S-LCIA) on MSWM was primarily based on scoring systems that convert qualitative inventory data into quantitative data according to the degree of compliance with criteria defined for analysed indicators (16 studies, i.e. 48%). For example, Aparcana and Salhofer (2013a) transformed the survey responses into a scoring system of 0 (non-fulfilment) and 1 (fulfilment) for each compliance indicator. Ibáñez-Forés et al. (2019) used a hybrid scoring system: conversion of yes and no answers to 0 and 100; percentage of respondents; others calculated by linear regression; or compared to performance requirements. These scores are converted on a scale from 0 to 100 (worst to best performance), with all categories having equal weight.

The characterization model from environmental LCA was adopted by 4 studies (i.e. 4, Menikpura et al. 2013; De Feo et al. 2021). They use the Swedish EPS characterization model, which derives correlations between environmental emissions and damage to human health expressed as DALY/

year (which can be thought of as one lost year of “healthy” life). Menikpura et al. (2012a; 2013) also used the impact category “community well-being”, expressed as the number of individuals who can have a “good life” as a result of the opportunities generated by the MSWM system calculated using the method given in Hunkeler (2006).

A qualitative/descriptive assessment of collected information was adopted in 4 studies (i.e. 12%, e.g. Umair et al. 2015; Azimi et al. 2020b). Umair et al. (2015) compared the inventory data with international norms and standards or, where norms were non-existent, with data from previous studies to classify the performance of an impact category, which can be negative (when it does not meet the norms), indifferent (when meeting the standards) or positive (only for genuinely positive categories that improve the situation). Then they plot the results graphically using a method that colour codes the performance impact categories. These authors also developed a system for relating the subcategories of impacts to potential impacts by analysing their cause-and-effect relationship (direct relationship, indirect or unrelated), a system based on the previous work of Citroth and Franze (2011).

Multi-criteria decision-making (MCDM) was used in the S-LCIA phase in 4 studies (i.e. 12%, e.g. Souza et al. (2016); Harijani et al. 2017; Nie et al. 2018; Fattahi 2020). For example, Harijani et al. (2017) calculated the social score of waste facilities for each inventory indicator using a fuzzy analytic hierarchy process (AHP) approach with the help of experts.

Another 3 studies (i.e. 9%) used the method developed by Taelman et al. (2020) which combines a variety of both traditional environmental LCA methods with local impact assessment methods proposed by miscellaneous references. For example, (i) risks related to work and linked to the workers measured by the number of fatal and non-fatal accidents/FU (functional unit), based on Citroth and Eisfeldt (2016); (ii) urban space consumption of the waste collection and treatment system, that considers the space used for WMS (including facilities) against the total area of the region (micro), expressed by m^2/m^2 (area occupied by public waste collection points and treatment facilities per FU/total land area of the region), based on den Boer et al. (2007). Mid-point impact categories are then aggregated into area-of-protections (AoPs) (e.g. AoPs human well-being and AoPs human health) using multi-criteria decision analysis based on the implementation of the ELECTRE II method.

Lastly, two studies used a single quantitative indicator (Handakas et al. 2017; Fuss et al. 2020). The first presents survey results as a percentage of respondents who answered questions regarding the indicator in a certain way. The second establishes the number of job openings and training required for each scenario.

This classification was made based on the approach that prevailed in the studies, but in some cases, these were mixed.

The three remaining studies did not carry out an impact assessment. Lehmann et al. (2011) and Souza et al. (2015) focused on the selection of social impact subcategories, and Hu et al. (2013) on the goal and scope definition. There was a significant methodological variation in the S-LCIA phase of the studies applied to MSWM. This occurred because this is the least methodologically well-developed S-LCA phase (Sureau et al. 2018; Souza et al. 2021).

3.2.5 Social life cycle interpretation

This phase is little explored in the analysed studies. The authors limited themselves to presenting the scores of the indicators, some adding a more detailed description of the aspects that justified the scores (as in Umair et al. 2015) and indicating the most significant impact subcategories that should receive greater attention from decision-makers. As mentioned above, only 8 studies addressed the assessment of data quality.

4 Limitations of the study

Expanding the scope of this review to include studies that evaluated social aspects from a life cycle perspective of the MSWM (e.g. LCSA), despite allowing a broad view of studies in this area, made it difficult to group, summarize and detail the information contained in the articles on the methodological aspects adopted for S-LCA. Indeed, it was common for articles not to detail aspects of S-LCA and to focus methodological details on other methods used in the study (e.g. environmental LCA, LCC or aggregation of results). As case studies of S-LCA in MSWM emerge, a new study could be developed to re-examine methodological aspects such as scope definition, the impact assessment and interpretation phases. Future studies could also extend the analysis to other waste types (e.g. hazardous waste, industrial waste) to see if there are specifics in applying S-LCA (e.g. differences in stakeholders and impact categories assessed).

5 Conclusions and recommendations

The number of studies involving the Social Life Cycle Assessment of Municipal Solid Waste Management has increased over the years as attempts continue to completely assess the sustainability of such systems. This growth is mainly driven by the assessment of social issues related to MSWM in developing countries. The most analysed MSWM stages were collection/transport, pre-treatment/sorting in recovery facilities, and the final disposal in landfills. Emphasis was given on how the impacts of these MSWM stages affect the stakeholders, “workers” and “local communities”. The main impacts analysed are employment generation, working conditions, workers’

health and safety, the local community’s safe and healthy living conditions, and its participation in the MSWM.

This article has provided a general description of how S-LCA has been applied in MSWM. It has been found that there is no methodological consensus for conducting the studies, but those based on UNEP guidelines are closest to a methodological harmonization.

As S-LCA is a technique still under development, it is not surprising that 47% of the analysed studies involved methodological development. Overall, the methodology needs improvement: the need to consider the entire product life cycle; relating impacts to the functional unit; addressing allocation and data quality issues; standardizing impact assessment methods; and the interpretation of results and their limitations. However, some of these issues, such as relating the impacts to the functional unit, the life cycle impact assessment stage and allocation issues, are limitations of the S-LCA method itself. These still need to be developed in the general context before they can be applied specifically to MSWM.

This study contributes to the search for a consensus on how to apply S-LCA in MSWM. In addition, it can serve as a guide for future research, as it summarizes the scope of the method’s applications, how this has advanced and the existing limitations. From this, it has been possible to draw up a series of recommendations for future studies:

- The use of detailed methodological descriptions should be a key focus for future studies. Greater transparency is essential to enable correct interpretation, verification of results reliability and comparison between studies.
- It is important to use participatory methodologies to choose categories, subcategories and impact indicators relevant to the study; limit those used to the most significant; reduce inconsistencies in S-LCA results; and increase practical applicability.
- It is necessary to clarify the definition of a product system and life cycle stages within the analysis, detailing “cut-off criteria” that led to the exclusion of particular processes/organizations and the impact of these decisions on the study results.

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Declarations

Conflict of interest The authors declare no conflict of interest.

References

- Aleisa E, Al-Jarallah R (2018) A triple bottom line evaluation of solid waste management strategies: a case study for an arid Gulf State. *Kuwait Int J Life Cycle Assess* 23:1460–1475. <https://doi.org/10.1007/s11367-017-1410-z>
- Aparcana S, Salhofer S (2013a) Application of a methodology for the social life cycle assessment of recycling systems in low income countries: three Peruvian case studies. *Int J Life Cycle Assess* 18:1116–1128. <https://doi.org/10.1007/s11367-013-0559-3>
- Aparcana S, Salhofer S (2013b) Development of a social impact assessment methodology for recycling systems in low-income countries. *Int J Life Cycle Assess* 18:1106–1115. <https://doi.org/10.1007/s11367-013-0546-8>
- Arcese G, Lucchetti MC, Massa I, Valente C (2018) State of the art in S-LCA: integrating literature review and automatic text analysis. *Int J Life Cycle Assess* 23:394–405. <https://doi.org/10.1007/s11367-016-1082-0>
- Arena U, Ardolino F, Di Gregorio F (2016) Technological, environmental and social aspects of a recycling process of post-consumer absorbent hygiene products. *J Clean Prod* 127:289–301. <https://doi.org/10.1016/j.jclepro.2016.03.164>
- Azimi AN, Dente SMR, Hashimoto S (2020a) Social lifecycle assessment of household waste management system in Kabul city. Sustainability. <https://doi.org/10.3390/SU12083217>
- Azimi AN, Dente SMR, Hashimoto S (2020b) Analyzing waste management system alternatives for Kabul City, Afghanistan: Considering social, environmental, and economic aspects. *Sustain* 12:1–16. <https://doi.org/10.3390/su12239872>
- Benoît NC, Franze J, Valdivia S, Traverso M, Vickery G, Siddharth P (2013) The methodological sheets for subcategories in social life cycle assessment (S-LCA). Available at https://www.lifecycleinitiative.org/wp-content/uploads/2013/11/SLCA_methodological_sheets_11.11.13.pdf. Accessed 13 Jan 2022
- Benoît C, Norris GA, Valdivia S, Ciroth A, Moberg A, Bos U, Prakash S, Ugaya C, Beck T (2010) The guidelines for social life cycle assessment of products: just in time! *Int. J Life Cycle Assess* 15:156–163. <https://doi.org/10.1007/s11367-009-0147-8>
- Benoît C, Mazijn B (2009) (eds) Guidelines for social life cycle assessment of products. Paris. p. 104, ISBN: 978–92–807–3021–0. Available at <https://www.lifecycleinitiative.org/wp-content/uploads/2012/12/2009%20-%20Guidelines%20for%20sLCA%20-%20EN.pdf>. Accessed 14 Jan 2022
- Bonilla-Alicea RJ, Fu K (2019) Systematic map of the social impact assessment field. *Sustain* 11. <https://doi.org/10.3390/su11154106>
- Bouillass G, Blanc I, Perez-Lopez P (2021) Step-by-step social life cycle assessment framework: a participatory approach for the identification and prioritization of impact subcategories applied to mobility scenarios. *The International Journal of Life Cycle Assessment*. <https://doi.org/10.1007/s11367-021-01988-w>
- Bui T, Tseng J, Tseng M, Lim MK (2022) Resources, conservation & recycling opportunities and challenges for solid waste reuse and recycling in emerging economies: a hybrid analysis. *Resour Conserv Recycl* 177:105968. <https://doi.org/10.1016/j.resconrec.2021.105968>
- Ciroth A, Eisfeldt F (2016) PSILCA – A Product Social Impact Life Cycle Assessment database. Database version 1.0. Available at: https://www.openlca.org/wp-content/uploads/2016/08/PSILCA_documentation_v1.1.pdf. Accessed 26 Apr 2022
- Ciroth A, Franze J (2011) LCA of an ecolabeled notebook, consideration of social and environmental impacts along the entire life cycle. *GreenDelta*. Available at: https://www.greendelta.com/wp-content/uploads/2017/03/LCA_laptop_final.pdf. Accessed 31 Jan 2022
- Chhipi-Shrestha GK, Hewage K, Sadiq R (2015) “Socializing” sustainability: a critical review on current development status of social life cycle impact assessment method. *Clean Technol Environ Policy* 17:579–596. <https://doi.org/10.1007/s10098-014-0841-5>
- Christensen TH, Damgaard A, Levis J, Zhao Y, Björklund A, Arena U, Barlaz MA, Starostina V, Boldrin A, Astrup TF, Bisinella V (2020) Application of LCA modelling in integrated waste management. *Waste Manag* 118:313–322. <https://doi.org/10.1016/j.wasman.2020.08.034>
- De Feo G, D’Argenio F, Ferrara C, Grosso A (2021) A procedure to assess the environmental, social and economic benefits wasted in the paper and cardboard fraction of the unsorted residual waste. *J Clean Prod* 296:126566. <https://doi.org/10.1016/j.jclepro.2021.126566>
- De Luca AI, Iofrida N, Strano A, Falcone G, Gulisano G (2015) Social life cycle assessment and participatory approaches: a methodological proposal applied to citrus farming in Southern Italy. *Integr Environ Assess Manag* 11:383–396. <https://doi.org/10.1002/ieam.1611>
- den Boer J, den Boer E, Jager J (2007) LCA-IWM: A decision support tool for sustainability assessment of waste management systems. *Waste Management* 27(8):1032–1045. <https://doi.org/10.1016/j.wasman.2007.02.022>
- Deus RM, Bezerra BS, Battistelle RAG (2019) Solid waste indicators and their implications for management practice. *Int J Environ Sci Technol* 16:1129–1144. <https://doi.org/10.1007/s13762-018-2163-3>
- Donthu N, Kumar S, Mukherjee D, Pandey N, Lim WM (2021) How to conduct a bibliometric analysis: an overview and guidelines. *J Bus Res* 133:285–296. <https://doi.org/10.1016/j.jbusres.2021.04.070>
- Dreyer LC, Hauschild MZ, Schierbeck J (2006) A framework for social life cycle impact assessment. *Int J Life Cycle Assess* 11:88–97. <https://doi.org/10.1065/lca2005.08.223>
- Du C, Ugaya C, Freire F, Dias LC, Clift R (2019) Enriching the results of screening social life cycle assessment using content analysis: a case study of sugarcane in Brazil. *Int J Life Cycle Assess* 24:781–793. <https://doi.org/10.1007/s11367-018-1490-4>
- Dubois-Iorgulescu A-M, Saraiva AKEB, Valle R, Rodrigues LM (2018) How to define the system in social life cycle assessments? A critical review of the state of the art and identification of needed developments. *Int J Life Cycle Assess* 23:507–518. <https://doi.org/10.1007/s11367-016-1181-y>
- Erses Yay AS (2015) Application of life cycle assessment (LCA) for municipal solid waste management: a case study of Sakarya. *J Clean Prod* 94:284–293. <https://doi.org/10.1016/j.jclepro.2015.01.089>
- European Commission (EC) (2011) Supporting environmentally sound decisions for waste management. <https://doi.org/10.2788/52632>. Available at: <https://op.europa.eu/en/publication-detail/-/publication/7cb97094-01d3-46af-8593-d42737759f0f/language-en>. Accessed 31 Jan 2022
- Ezeah C, Roberts CL (2012) Analysis of barriers and success factors affecting the adoption of sustainable management of municipal solid waste in Nigeria. *J Environ Manage* 103:9–14. <https://doi.org/10.1016/j.jenvman.2012.02.027>
- Fattahi M (2020) A data-driven approach for supply chain network design. *Ann Oper Res* 288:265–284. <https://doi.org/10.1007/s10479-020-03532-9>
- Ferrão P, Ribeiro P, Rodrigues J, Marques A, Preto M, Amaral M, Domingos T, Lopes A, Costa EI (2014) Environmental, economic and social costs and benefits of a packaging waste management system: a Portuguese case study. *Resour Conserv Recycl* 85:67–78. <https://doi.org/10.1016/j.resconrec.2013.10.020>
- Foolmaun RK, Ramjeeawon T (2013) Comparative life cycle assessment and social life cycle assessment of used polyethylene

- terephthalate (PET) bottles in Mauritius. *Int J Life Cycle Assess* 18:155–171. <https://doi.org/10.1007/s11367-012-0447-2>
- Fuss M, Vergara-Araya M, Barros RTV, Poganietz WR (2020) Implementing mechanical biological treatment in an emerging waste management system predominated by waste pickers: a Brazilian case study. *Resour Conserv Recycl* 162:105031. <https://doi.org/10.1016/j.resconrec.2020.105031>
- Gompf K, Traverso M, Hetterich J (2020) Towards social life cycle assessment of mobility services: systematic literature review and the way forward. *Int J Life Cycle Assess* 25:1883–1909. <https://doi.org/10.1007/s11367-020-01788-8>
- Handakas E, Manariotis I, Yannopoulos P, Zarkadas I, Sarigiannis D (2017) Decision support tool for urban solid waste management. *Fresenius Environ Bull* 26:405–412
- Harijani MA, Mansour S, Karimi B, Lee C-G (2017) Multi-period sustainable and integrated recycling network for municipal solid waste—a case study in Tehran. *J Clean Prod* 151:96–108. <https://doi.org/10.1016/j.jclepro.2017.03.030>
- Hoang NH, Fogarassy C (2020) Sustainability evaluation of municipal solid waste management system for hanoi (Vietnam)—why to choose the “waste-to-energy” concept. *Sustain* 12:1–20. <https://doi.org/10.3390/su12031085>
- Homrich AS, Galvão G, Abadia LG, Carvalho MM (2018) The circular economy umbrella: trends and gaps on integrating pathways. *J Clean Prod*. <https://doi.org/10.1016/j.jclepro.2017.11.064>
- Hu M, Kleijn R, Bozhilova-Kisheva KP, Di Maio F (2013) An approach to LCSA: The case of concrete recycling. *Int J Life Cycle Assess* 18:1793–1803. <https://doi.org/10.1007/s11367-013-0599-8>
- Huertas-Valdivia I, Ferrari AM, Settembre-Blundo D, García-Muñia FE (2020) Social life-cycle assessment: a review by bibliometric analysis. *Sustainability* 12:1–25. <https://doi.org/10.3390/su12156211>
- Hunkeler D (2006) Societal LCA methodology and case study. *Int J Life Cycle Assess* 11(6):371–382. <https://doi.org/10.1065/lca2006.08.261>
- Ibáñez-Forés V, Bovea MD, Coutinho-Nóbrega C, de Medeiros HR (2019) Assessing the social performance of municipal solid waste management systems in developing countries: Proposal of indicators and a case study. *Ecol Indic* 98:164–178. <https://doi.org/10.1016/j.ecolind.2018.10.031>
- Iodice S, Garbarino E, Cerreta M, Tonini D (2021) Sustainability assessment of construction and demolition waste management applied to an Italian case. *Waste Manag* 128:83–98. <https://doi.org/10.1016/j.wasman.2021.04.031>
- Iofrida N, Strano A, Gulisano G, De Luca AI (2018) Why social life cycle assessment is struggling in development? *Int J Life Cycle Assess* 23:201–203. <https://doi.org/10.1007/s11367-017-1381-0>
- ISO (2006a) ISO 14040 International Standard. In: environmental management—life cycle assessment—principles and framework
- ISO (2006b) ISO 14044 International Standard. In: environmental management—life cycle assessment—requirements and guidelines
- Jørgensen A, Le Bocq A, Nazarkina L, Hauschild M (2008) Methodologies for social life cycle assessment. *Int J Life Cycle Assess* 13:96–103. <https://doi.org/10.1065/lca2007.11.367>
- Kloepffer W (2008) Life cycle sustainability assessment of products. *Int J Life Cycle Assess* 13, 89–95. <https://doi.org/10.1065/lca2008.02.376>
- Khandelwal H, Dhar H, Thalla AK, Kumar S (2019a) Application of life cycle assessment in municipal solid waste management: a worldwide critical review. *J Clean Prod* 209:630–654. <https://doi.org/10.1016/j.jclepro.2018.10.233>
- Khandelwal H, Thalla AK, Kumar S, Kumar R (2019b) Life cycle assessment of municipal solid waste management options for India. *Bioresour Technol* 288:121515. <https://doi.org/10.1016/j.biortech.2019.121515>
- Kijak R, Moy D (2004) A decision support framework for sustainable waste management. *J Ind Ecol* 8:33–50. <https://doi.org/10.1162/1088198042442306>
- Kühnen M, Hahn R (2017) Indicators in social life cycle assessment: a review of frameworks, Theories, and Empirical Experience. *J Ind Ecol* 21:1547–1565. <https://doi.org/10.1111/jiec.12663>
- Lagarde V, MacOmbe C (2013) Designing the social life cycle of products from the systematic competitive model. *Int J Life Cycle Assess* 18:172–184. <https://doi.org/10.1007/s11367-012-0448-1>
- Larsen PO, von Ins M (2010) The rate of growth in scientific publication and the decline in coverage provided by science citation index. *Scientometrics* 84:575–603. <https://doi.org/10.1007/s11192-010-0202-z>
- Laurent A, Bakas I, Clavreul J, Bernstad A, Niero M, Gentil E, Hauschild MZ, Christensen TH (2014) Review of LCA studies of solid waste management systems—part I: lessons learned and perspectives. *Waste Manag* 34:573–588. <https://doi.org/10.1016/j.wasman.2013.10.045>
- Lehmann A, Russi D, Bala A, Finkbeiner M, Fullana-i-Palmer P (2011) Integration of social aspects in decision support, based on life cycle thinking. *Sustainability* 3:562–577. <https://doi.org/10.3390/su3040562>
- Lehmann A, Zschieschang E, Traverso M, Finkbeiner M, Schebek L (2013) Social aspects for sustainability assessment of technologies—challenges for social life cycle assessment (SLCA). *Int J Life Cycle Assess* 18:1581–1592. <https://doi.org/10.1007/s11367-013-0594-0>
- Linzner R, Lange U (2013) Role and size of informal sector in waste management—a review. *Proceedings of the Institution of Civil Engineers. Waste Resourc Manag* 166:69–83. <https://doi.org/10.1680/warm.12.00012>
- Lu YT, Lee YM, Hong CY (2017) Inventory analysis and social life cycle assessment of greenhouse gas emissions from waste-to-energy incineration in Taiwan. *Sustain* 9. <https://doi.org/10.3390/su9111959>
- Ma J, Hipel KW (2016) Exploring social dimensions of municipal solid waste management around the globe—a systematic literature review. *Waste Manag* 56:3–12. <https://doi.org/10.1016/j.wasman.2016.06.041>
- Macombe C, Feschet P, Garrabé M, Loeillet D (2011) 2nd International seminar in social life cycle assessment—recent developments in assessing the social impacts of product life cycles. *Int J Life Cycle Assess* 16:940–943. <https://doi.org/10.1007/s11367-011-0331-5>
- Macombe C, Leskinen P, Feschet P, Antikainen R (2013) Social life cycle assessment of biodiesel production at three levels: a literature review and development needs. *J Clean Prod* 52:205–216. <https://doi.org/10.1016/j.jclepro.2013.03.026>
- Magrini C, Degli Esposti A, De Marco E, Bonoli A (2021) A framework for sustainability assessment and prioritisation of urban waste prevention measures. *Sci Total Environ* 776:145773. <https://doi.org/10.1016/j.scitotenv.2021.145773>
- Mancini L, Sala S (2018) Social impact assessment in the mining sector: review and comparison of indicators frameworks. *Resour Policy* 57:98–111. <https://doi.org/10.1016/j.resourpol.2018.02.002>
- Martínez-Blanco J, Lehmann A, Muñoz P, Antón A, Traverso M, Rieradevall J, Finkbeiner M (2014) Application challenges for the social Life Cycle Assessment of fertilizers within life cycle sustainability assessment. *J Clean Prod* 69:34–48. <https://doi.org/10.1016/j.jclepro.2014.01.044>
- Menikpura SNM, Gheewala SH, Bonnet S (2012a) Framework for life cycle sustainability assessment of municipal solid waste management systems with an application to a case study in Thailand. *Waste Manag Res* 30:708–719. <https://doi.org/10.1177/0734242X12444896>
- Menikpura SNM, Gheewala SH, Bonnet S (2012b) Sustainability assessment of municipal solid waste management in Sri Lanka:

- problems and prospects. *J Mater Cycles Waste Manag* 14:181–192. <https://doi.org/10.1007/s10163-012-0055-z>
- Menikpura SNM, Gheewala SH, Bonnet S, Chiemchaisri C (2013) Evaluation of the effect of recycling on sustainability of municipal solid waste management in Thailand. *Waste and Biomass Valorization* 4:237–257. <https://doi.org/10.1007/s12649-012-9119-5>
- Mostafa Hatami A, Sabour MR, Nikravan M (2021) A systematic analysis of research trends on incineration during 2000–2019. *Int J Environ Sci Technol* 18:353–364. <https://doi.org/10.1007/s13762-020-02794-x>
- Nie Y, Wu Y, Zhao J, Zhao J, Chen X, Maraseni T, Qian G (2018) Is the finer the better for municipal solid waste (MSW) classification in view of recyclable constituents? A comprehensive social, economic and environmental analysis. *Waste Manag* 79:472–480. <https://doi.org/10.1016/j.wasman.2018.08.016>
- Paes LAB, Bezerra BS, Deus RM, Jugend D, Battistelle RAG (2019) Organic solid waste management in a circular economy perspective—a systematic review and SWOT analysis. *J Clean Prod* 239:118086. <https://doi.org/10.1016/j.jclepro.2019.118086>
- Paes MX, de Medeiros GA, Mancini SD, Bortoleto AP, Puppim de Oliveira JA, Kulay LA (2020) Municipal solid waste management: Integrated analysis of environmental and economic indicators based on life cycle assessment. *J Clean Prod* 254:119848. <https://doi.org/10.1016/j.jclepro.2019.119848>
- Paragahawewa U, Blackett P, Small B (2009) Social life cycle analysis (S-LCA): some methodological issues and potential application to cheese production in New Zealand. Report by Agresearch Parent. Available at: https://saipatform.org/uploads/Library/SocialLCA-FinalReport_July2009.pdf. Accessed 26 Apr 2022
- Petti L, Serreli M, Di Cesare S (2018) Systematic literature review in social life cycle assessment. *Int J Life Cycle Assess* 23:422–431. <https://doi.org/10.1007/s11367-016-1135-4>
- Ramos Huarachi DA, Piekarski CM, Puglieri FN, de Francisco AC (2020) Past and future of social life cycle assessment: historical evolution and research trends. *J Clean Prod* 264:121506. <https://doi.org/10.1016/j.jclepro.2020.121506>
- Russi D, Ayuso S, Palmer PF (2012) El Análisis Social del Ciclo de Vida como instrumento de RSC: el punto de vista de las empresas españolas. Càtedra Mango de Responsabilidad Social Corporativa. Available at: <http://mango.esci.upf.edu/DOCS/Documents-de-treball/12-ACVS.pdf>. Accessed 26 Apr 2022
- Russo Garrido S, Parent J, Beaulieu L, Revéret JP (2018) A literature review of type I SLCA—making the logic underlying methodological choices explicit. *Int J Life Cycle Assess* 23:432–444. <https://doi.org/10.1007/s11367-016-1067-z>
- Sanjuan-Delmás D, Taelman SE, Arlati A, Obersteg A, Vér C, Óvári Á, Tonini D, Dewulf J (2021) Sustainability assessment of organic waste management in three EU Cities: Analysing stakeholder-based solutions. *Waste Manag* 132:44–55. <https://doi.org/10.1016/j.wasman.2021.07.013>
- Santos AC, Mendes P, Ribau Teixeira M (2019) Social life cycle analysis as a tool for sustainable management of illegal waste dumping in municipal services. *J Clean Prod* 210:1141–1149. <https://doi.org/10.1016/j.jclepro.2018.11.042>
- Schübeler P (1996) Conceptual framework for municipal solid waste management in low-income countries. Urban Management Infrastructure. Work. Pap. No. 9. Available at: <http://inswa.or.id/wp-content/uploads/2012/07/conceptualframework-swm-schuebeler.pdf>. Accessed 30 Jan 2021
- Sou WI, Chu A, Chiueh PT (2016) Sustainability assessment and prioritization of bottom ash management in Macao. *Waste Manag Res* 34:1275–1282. <https://doi.org/10.1177/0734242X16665914>
- Souza RG, Clímaco JCN, Sant’Anna AP, Rocha TB, do Valle R, de AB, Quelhas OLG (2016) Sustainability assessment and prioritization of e-waste management options in Brazil. *Waste Manag* 57, 46–56. <https://doi.org/10.1016/j.wasman.2016.01.034>
- Souza RG, Rosenhead J, Salhofer SP, Valle RAB, Lins MPE (2015) Definition of sustainability impact categories based on stakeholder perspectives. *J Cleaner Prod* 105:41–51. <https://doi.org/10.1016/j.jclepro.2014.09.051>
- Souza A, Watanabe MDB, Cavalett O, Cunha M, Ugaya CML, Bonomi A (2021) A novel social life cycle assessment method for determining workers’ human development: a case study of the sugarcane biorefineries in Brazil. *Int J Life Cycle Assess*. <https://doi.org/10.1007/s11367-021-01936-8>
- Sureau S, Mazijn B, Garrido SR, Achten WMJ (2018) Social life-cycle assessment frameworks: a review of criteria and indicators proposed to assess social and socio-economic impacts. *Int J Life Cycle Assess* 23:904–920. <https://doi.org/10.1007/s11367-017-1336-5>
- Taelman S, Sanjuan-Delmás D, Tonini D, Dewulf J (2020) An operational framework for sustainability assessment including local to global impacts: focus on waste management systems. *Resour Conserv Recycl* 162:104964. <https://doi.org/10.1016/j.resconrec.2020.104964>
- The World Bank (2021) World Bank Country and Lending Groups. <https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups>. Accessed 14 Oct 2021
- Tokede O, Traverso M (2020) Implementing the guidelines for social life cycle assessment: past, present, and future. *Int J Life Cycle Assess* 25:1910–1929. <https://doi.org/10.1007/s11367-020-01814-9>
- Tsai FM, Bui TD, Tseng ML, Lim MK, Hu J (2020) Municipal solid waste management in a circular economy: A data-driven bibliometric analysis. *J Clean Prod* 275:124132. <https://doi.org/10.1016/j.jclepro.2020.124132>
- Tulokhonova A, Ulanova O (2013) Assessment of municipal solid waste management scenarios in Irkutsk (Russia) using a life cycle assessment-integrated waste management model. *Waste Manag Res* 31:475–484. <https://doi.org/10.1177/0734242X13476745>
- Umair S, Björklund A, Petersen EE (2015) Social impact assessment of informal recycling of electronic ICT waste in Pakistan using UNEP SETAC guidelines. *Resour Conserv Recycl* 95:46–57. <https://doi.org/10.1016/j.resconrec.2014.11.008>
- UNEP (2020) Guidelines for Social Life Cycle Assessment of Products and Organizations. Benoît Norris, C., Traverso, M., Neugebauer, S., Ekener, E., Schaubroeck, T., Russo Garrido, S., Berger, M., Valdivia, S., Lehmann, A., Finkbeiner, M., Arcese, G. (eds.). United Nations Environment Programme (UNEP). Available at <https://www.lifecycleinitiative.org/wp-content/uploads/2021/01/Guidelines-for-Social-Life-Cycle-Assessment-of-Products-and-Organizations-2020-22.1.21sml.pdf>. Accessed 14 Jan 2022
- Valdivia S, Ugaya CML, Sonnemann G, Hildenbrand J (2011) (eds) Towards a life cycle sustainability assessment. Making informed choices on products. Paris, ISBN: 978-92-807-3175-0. Available at: <https://www.lifecycleinitiative.org/wp-content/uploads/2012/12/2011%20-%20Towards%20LCSA.pdf>. Accessed 14 Jan 2022
- Vinyes E, Oliver-Solà J, Ugaya C, Rieradevall J, Gasol CM (2013) Application of LCSA to used cooking oil waste management. *Int J Life Cycle Assess* 18:445–455. <https://doi.org/10.1007/s11367-012-0482-z>
- Visentin C, Trentin AW, da S, Braun AV Thomé A (2020) Life cycle sustainability assessment: a systematic literature review through the application perspective, indicators, and methodologies. *J Clean Prod* 270. <https://doi.org/10.1016/j.jclepro.2020.122509>
- White P, Franke M, Hindle P (1995) Integrated solid waste management: a lifecycle inventory. Springer, US, Boston, MA. <https://doi.org/10.1007/978-1-4684-6705-5>

- Wilson DC, Velis CA (2015) Waste management—still a global challenge in the 21st century: an evidence-based call for action. *Waste Manag Res* 33:1049–1051. <https://doi.org/10.1177/0734242X15616055>
- Wilson DC, Velis C, Cheeseman C (2006) Role of informal sector recycling in waste management in developing countries. *Habitat Int* 30:797–808. <https://doi.org/10.1016/j.habitatint.2005.09.005>
- Wilson DC, Rodic L, Modak P, Soos R, Rogero AC, Velis C (2015) Global Waste Management Outlook, UNEP, United Nations Environment Programme ISWA, International Solid Waste Association. Available at: <https://www.unep.org/resources/report/global-waste-management-outlook>. Accessed 31 Jan 2022
- Wu R, Yang D, Chen J (2014) Social Life Cycle Assessment Revisited Sustainability 6(7):4200–4226. <https://doi.org/10.3390/su6074200>
- Yıldız-Geyhan E, Altun-Çiftçioğlu GA, Kadirgan MAN (2017) Social life cycle assessment of different packaging waste collection system. *Resour Conserv Recycl* 124:1–12. <https://doi.org/10.1016/j.resconrec.2017.04.003>
- Yıldız-Geyhan E, Yılan G, Altun-Çiftçioğlu GA, Kadirgan MAN (2019) Environmental and social life cycle sustainability assessment of different packaging waste collection systems. *Resour Conserv Recycl* 143:119–132. <https://doi.org/10.1016/j.resconrec.2018.12.028>
- Zhang J, Qin Q, Li G, Tseng C (2021) Sustainable municipal waste management strategies through life cycle assessment method: a review. *J Environ Manage* 287:112238. <https://doi.org/10.1016/j.jenvman.2021.112238>
- Zhou Z, Chi Y, Dong J, Tang Y, Ni M (2019) Model development of sustainability assessment from a life cycle perspective: a case study on waste management systems in China. *J Clean Prod* 210:1005–1014. <https://doi.org/10.1016/j.jclepro.2018.11.074>
- Zupic I, Čater T (2015) Bibliometric methods in management and organization. *Organ Res Methods* 18:429–472. <https://doi.org/10.1177/1094428114562629>

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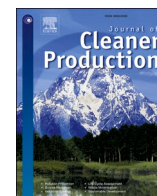
3.9 Artigo 9 – Life cycle sustainability impact categories for sustainable procurement *

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ABSTRACT

Sustainable procurement depends on the definition of evaluation criteria when structuring selection processes for the most sustainable suppliers and products. Since a product or service may be associated with several sustainability impact categories throughout its life cycle, the procurement process may consider them as evaluation criteria. A recommended approach for such product evaluation is Life Cycle Sustainability Assessment (LCSA). The main objective of this study is to identify and catalogue general sustainability impact categories to be used as evaluation criteria in sustainable procurement. This research applied cognitive mapping, a tool in Problem Structuring Methods, based on a systematic literature review and interviews with 10 experts in the field. They also collaborated in the categorization and validation of the results. This study suggests a set of 58 midpoint and 14 endpoint sustainability categories, providing a contribution for different actors in society and supply chains, allowing them to evaluate supply options in sustainable procurement. The participation of experts enriched the understanding of the problem, allowing to observe a fourth sustainability dimension for procurement, comprising Supplier Governance and Compliance.

Keywords: Sustainable procurement; Life cycle sustainability assessment; Sustainability impact categories; Supplier selection problem; Problem structuring methods; Cognitive mapping.



Life cycle sustainability impact categories for sustainable procurement

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ABSTRACT

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1. Introduction

Sustainable Procurement is the activity of purchasing goods or services with the most positive impact throughout the life cycle of a product (ISO, 2017). The context of supplier selection should address the sustainability issue (Hervani et al., 2005; Igarashi et al., 2013).

Supplier selection involves multiple decision criteria, with diverse natures: qualitative, quantitative, and other subjective and specific characteristics of each product or business, which makes the decision process a challenge (Govindan et al., 2015; Igarashi et al., 2013). A decision criterion is a performance measure used to assess the impact of each decision alternative on the objective sought (Franco and Montibeller, 2010). Studies are developing in supplier selection, especially regarding the use of quantitative methods and in considering social aspects (Galo et al., 2018; Govindan et al., 2015).

Sustainable procurement is recognized as an important tool to promote sustainable production and consumption practices (Molin et al., 2021). It is consistent with the principles of sustainable development, and is increasingly present in the agenda of procurement managers seeking to implement corporate responsibility in their supply chains (Walker et al., 2012; Walker and Brammer, 2009). The importance of

adopting effective tools, policies, and assessment models to promote sustainable consumption comes from its primary objective for global sustainability, as well as handling its cultural and operational barriers (Delmonico et al., 2018; Kucukvar et al., 2014).

Assessment of sustainability criteria in procurement processes needs to observe a life cycle perspective. Life Cycle Thinking (LCT) is a systemic approach that provides an evaluation of products and services considering cumulative impacts associated with the flows of energy, substances and materials in all stages of a product's or service's life cycle, from extraction of raw materials to final disposal (Sala et al., 2013). LCT is aligned with the sustainability assessment of products in a procurement process (Alhola et al., 2019; Jenssen and de Boer, 2019).

Life Cycle Sustainability Assessment (LCSA) is a methodology to assess sustainability from a life cycle perspective (Costa et al., 2019; Jørgensen et al., 2013). It supports studies in sustainable supply chain management (van Kempen et al., 2017). Currently, the most used LCSA approaches consider the joint verification of environmental, economic and social perspectives of a product or process throughout its life cycle, starting by defining objective, scope and impact categories (Costa et al., 2019). Impact categories refer to logical groupings of LCSA results, related to sustainability issues of interest to stakeholders and

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decision-makers along the life cycle (van Kempen et al., 2017).

There is a need for a methodological framework that satisfies the buyers' sustainability objectives for sustainable procurement evaluation (Ponce et al., 2019). The methodological framework should consider the evaluation of procurement options by sustainability criteria observing the product's life cycle (Laosirihongthong et al., 2019). LCSA impact categories can be used as decision criteria to assess the impact of each decision alternative, to fulfil the sustainability goals (Alhola et al., 2019; Souza et al., 2015; van Kempen et al., 2017). In order to apply LCSA to sustainable procurement, it is necessary to clarify what life cycle sustainability impact categories can be used as criteria for evaluation and selection of suppliers and products (Alhola et al., 2019; Butt et al., 2015; Jenssen and de Boer, 2019).

Sustainability assessment studies are often supported by Problem Structuring Methods – PSMs (Franco and Montibeller, 2010; Souza et al., 2015). PSMs are models provided for cognitive structuring of complex problems (Mingers and Rosenhead, 2004; Pereira and Morais, 2020). In some PSMs, cognitive mapping is a technique applied to structure confusing or complex data, which aids the interview process, increases understanding, and allows for managing qualitative data extracted from document analysis (Assunção et al., 2020; Mingers and Rosenhead, 2004). Sustainable supplier and product selection is a complex problem (Govindan et al., 2015; Hervani et al., 2005). Considering that the impact categories in a LCSA can be seen as decision criteria (Costa et al., 2019; Souza et al., 2015), there is an opportunity to use cognitive mapping as a method for investigating and identifying potential impact categories for sustainable procurement.

The main objective of this study is to define LCSA impact categories for sustainable procurement, based on literature review and consultation of experts. The study combines cognitive mapping and LCSA in a sustainable procurement decision-support approach.

2. Literature review

2.1. Sustainable procurement and LCT

Significant recent changes in international policies evidence the need to develop procurement processes to promote sustainability in a life cycle perspective. There are developments in India (Mansi, 2015) and in the European Union (EU), particularly with the Directive 2014/24/EU on public procurement (Alhola et al., 2019). The European Commission's Product Environmental Footprint Guide (PEF) established a life cycle-based multi-criteria calculation of the environmental footprint of goods or services. Another life cycle-based tool is the Environmental Product Declaration (EPD), whose demand has increased in recent years. EPD refers to a voluntary declaration to communicate the environmental performance of a product (Del Borghi et al., 2020). "Procura +" is a campaign promoted by the Group Local Governments for Sustainability – ICLEI, which fosters projects and initiatives in sustainable procurement. The Brazilian Environmental Agenda in Public Administration (A3P) is the largest sustainable procurement program in Latin America (Delmonico et al., 2018) and promotes the use of sustainability criteria in public administration (Trigo et al., 2017). Examples of general sustainability criteria commonly adopted by these approaches are: organisational governance, consumer issues, community involvement and development, labour practices, human rights (also involving accessibility and social inclusion), climate change, use of resources, generation of waste, and fair operations.

Advances in sustainable procurement put pressure on organisations to evaluate sustainability of their products and services (Patala et al., 2016). However, there are obstacles to enhance sustainability in procurement. The main obstacle is that there are productive activities outside the control of organisations, across supply chains (Nawaz and Koc, 2019). Other barriers, especially in emerging economies, involve: aspects of organisational culture, such as lack of awareness and resources; operational aspects, such as lack of training and time

availability; and methodological aspects (Delmonico et al., 2018). These methodological challenges concern how to balance different aspects involved in sustainability, the trade-offs between them, and the measurement of different aspects, aligned with the evaluation systems adopted by market actors (Ponce et al., 2019; Walker et al., 2012).

The selection of sustainability criteria is significant as a fundamental aspect of sustainable procurement (Laosirihongthong et al., 2019). It is not obvious how procurement officers can operationalise them as part of their strategies (Jenssen and de Boer, 2019). Few articles assess the sustainability of procurement processes using quantitative or qualitative methods, what indicates a need for further studies and guidelines to measure the procurement performance (Molin et al., 2021). Specifically in procurement of food in organisations, the FOODSCALE is an innovative tool which covers the three dimensions of sustainability, treated as interdependent and coexisting, and considers the entire food life cycle. It presents 11 categories to assess food sustainability, such as: organic food, seasonality, fairly traded product, food waste, origin of food, consumer engagement, and engaging with smaller producers and local communities (Goggins and Rau, 2016).

ISO 20400:2017 (Sustainable Procurement - Guidelines) is the first international standard for sustainable procurement (Harris and Divakarla, 2017). The Standard presents key issues, such as the need for "defining sustainability impacts", but does not describe in detail what the sustainability impacts would be, and how they must be defined (ISO, 2017). There is an opportunity to explore which sustainability impacts could be considered to support the process and decisions of sustainable procurement (Harris and Divakarla, 2017).

Models based on LCT were highlighted to promote sustainable consumption, and have gained enormous interest worldwide to quantify sustainability impacts in a systematic way (Kucukvar et al., 2014). LCT implementation involves policy and strategy decisions on a higher level, but still gets limited coverage in the supplier selection literature (Jenssen and de Boer, 2019). The application of life cycle models requires skilled professionals of multidisciplinary background. A full integration of the sustainability aspects (e.g. environmental and costing) still needs a lot of interdisciplinary efforts (Luttenberger and Luttenberger, 2017; Pullman and Wikoff, 2017). Among the three traditional pillars of sustainability, the environmental one is the most present in scientific publications in the area of sustainable procurement, which in this case is often called "green procurement" (Miemczyk et al., 2012).

Among LCSA frameworks under development, the proposal by the United Nations Environment Program (UNEP/SETAC) is a major reference (Costa et al., 2019; van Kempen et al., 2017; Zamagni, 2012). This framework uses ISO Standard 14040 (Environmental Life Cycle Assessment - LCA) as a basis, and consists of four phases: (i) objective and scope definition, (ii) inventory analysis, (iii) impact assessment, and (iv) interpretation. LCSA considers the three pillars traditionally contained in the definition of sustainability (Kloepffer, 2008; van Kempen et al., 2017):

- Life Cycle Assessment (LCA) corresponds to the environmental module. Its methodological structure is standardised in the series of international standards ISO 14040 to 14044 and widely developed in the International Life Cycle Data Reference System – ILCD (Souza et al., 2015).
- Life Cycle Costing (LCC) corresponds to the cost assessment. Its methodological structure is standardised in the international standard IEC 60300-3-3 (IEC, 2004). It has suggested methodological guidelines for application in sustainable procurement in the ISO 20400 Standard (ISO, 2017) and in the "Procura +" Manual for the implementation of sustainable procurement (Waris et al., 2019).
- Social Life Cycle Assessment (SLCA): corresponding to the social pillar of sustainability. Its methodological basis was registered by the United Nations Environment Program (UNEP, 2020), based on the structure of the LCA.

Previous studies provide a structured literature review on sustainable procurement and supplier selection addressing multiple sustainability criteria, eg. (Govindan et al., 2015; Jenssen and de Boer, 2019; Miemczyk et al., 2012). The studies show that LCA associated with sustainable procurement is a tool to achieve more sustainable practices, representing rapidly growing literature streams. However, empirical descriptions of LCA applied to procurement are limited. The perspective most taken in literature is concerned with sustainability as a scientific concept. Although previous studies have contributed with the exploitation of viability and technical possibilities of LCA in specific contexts of procurement decisions (e.g. food and construction) the use of sustainability impact assessment approaches (e.g. LCSA) in a wider procurement context could be further explored. Analyses would be directed towards how sustainability is measured in sustainable procurement, and how better sustainability measuring could translate this into useful information for decision-makers. Examples of evaluation criteria adopted in previous studies are: waste generation; recycling fee (including re-use and energy recovery approaches); cost of the supplied component and cost of disposal; emission of air and water pollution; verification of supplier compliance with legislation, social norms and standards; equality in employment among suppliers in terms of diversity, gender, etc.; implementation of codes of practice and conduct (Govindan et al., 2015; Jenssen and de Boer, 2019; Miemczyk et al., 2012).

LCSA assesses sustainability impacts at midpoint or endpoint categories (van Kempen et al., 2017). Midpoint impacts are intermediate consequences of inventory flows that produce aggregated impacts, such as acidification, human toxicity, eutrophication etc. (Ismaeel, 2018). Endpoint impacts represent an area of concern affected by midpoint impacts, like damages to the ecosystem or human health (ISO, 2006). Individually or in combination, midpoint impacts cause the effects described by endpoints, as illustrated in Fig. 1.

The selection of relevant life cycle impact categories is a crucial step in conducting LCSA (Ismaeel, 2018; Kloeppfer, 2008). One approach to defining impact categories is elaborating them by consultation of stakeholders and experts involved with each problem (Souza et al., 2015; van Kempen et al., 2017). One proposal for the combination of the categories to provide an integrative sustainability assessment is the use of Multi-Criteria Decision Analysis (MCDA) (UNEP/SETAC, 2011; Zamagni, 2012), which is often structured by PSMs (Souza et al., 2015). MCDA comprehends methodologies to support decision making when multiple objectives have to be pursued (Franco and Montibeller, 2010). The identification of life cycle sustainability impact categories for sustainable procurement represents the structuring of multiple decision criteria for supplier and product selection problems (Costa et al., 2019; Galo et al., 2018). MCDA methods, such as AHP (Analytic Hierarchy Process) and TOPSIS (Technique for Order Preference by Similarity to Ideal Solution), are particularly widespread and used in Supplier Selection Problems – SSPs (Galo et al., 2018; Silva and Belderrain, 2019).

2.2. Cognitive mapping and its application to LCSA

PSMs are often called “soft-operational research” (soft-OR), usually adopted as complementary approaches to mathematical, “hard-OR” methods (Eden and Ackermann, 2018). The most prominent PSMs are: Soft Systems Methodology (SSM), used for systems redesign; Strategic

Choice Approach (SCA), a planning method focused on managing uncertainty; Robustness Analysis (RA), which focuses on maintaining flexibility under uncertainty; and Strategic Options Development and Analysis (SODA), which uses cognitive mapping as a modelling tool to elicit individual and collective views on a problem (Cunha and Morais, 2019; Mingers and Rosenhead, 2004). Structuring cognitive maps is relevant to visually represent the perception of stakeholders and experts about the context under analysis, being used to support reasoning in various areas of knowledge. It performs well when used to systematise semi-structured interviews (Assunção et al., 2020; Mingers and Rosenhead, 2004).

Cognitive maps are structures with means and ends representing cause-effect relations, modelling the issues involved in a problem, creating a network (graph) of nodes connected by unidirectional arrows (Cunha and Morais, 2019). The effects (ends) are usually represented at the top of a map, with the causes (means) to the bottom (Ackermann et al., 1992; Santos et al., 2011). The top of the maps contain the broader strategic objectives, which depend on operational (more measurable) objectives as intermediate steps. The operational objectives can be translated into performance indicators and defined as criteria for evaluating decision alternatives, measuring the extent of achievement of the strategic objectives (Franco and Montibeller, 2010). Fig. 2 illustrates a cognitive map structure with some cause-effect relations, representing a case study on e-waste in Brazil.

A merged map aggregates the views in the individual maps, integrating the results (Abuabara et al., 2018; Mingers and Rosenhead, 2004). The results of the literature survey can also be translated into a merged cognitive map (Ackermann et al., 1992; Santos et al., 2011). The cause-effect hierarchy of the final merged map allows for the identification of potential impact categories according to their topological position as midpoint or endpoint (Guarnieri et al., 2016; Souza et al., 2015). Endpoints are found at the end of causal chains (destinations of arrows), at the upper areas of the map, while midpoints are positioned immediately below them (Souza et al., 2015). The merged map makes it possible to identify the results that are distributed in the different individual maps, and to complete their explanations through new reflections and consultations with experts (Eden and Ackermann, 2018; Souza et al., 2015).

There is a correlation between LCSA impact pathways (ex. Fig. 1) and the causal structure of cognitive maps (Fig. 2), since both represent cause-effect networks. Both structures present endpoint and midpoint effects according to their topological position in the represented network (Souza et al., 2015).

Studies have used cognitive mapping in the context of urban sustainability strategies (Assunção et al., 2020; Awasthi and Omrani, 2018). An example is the analysis of socioeconomic implications in implementing bio-waste refineries (Kokkinos et al., 2018). Cognitive mapping studies commonly adopt a joint consideration of bibliographic records and expert opinion (Ackermann et al., 1992; Farsari et al., 2010). Specifically in support to LCSA, there are applications of cognitive maps to define sustainability impact categories of Brazilian e-waste management (Guarnieri et al., 2016; Souza et al., 2015). Falcone and Imbert (2018) used literature review integrated with semi-structured interviews to identify social categories and indicators relevant to LCSA for promoting the market uptake of bio-based products. Ruparathna and

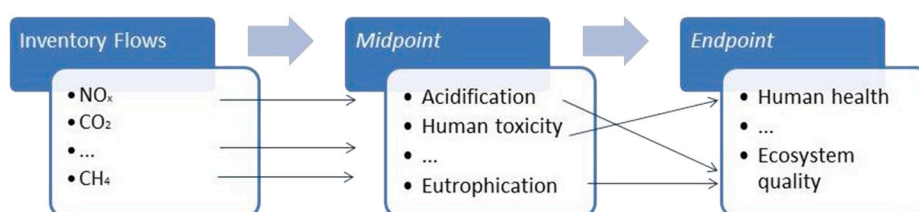


Fig. 1. Example impact analysis framework for LCSA. Source: adapted from (UNEP/SETAC, 2011).

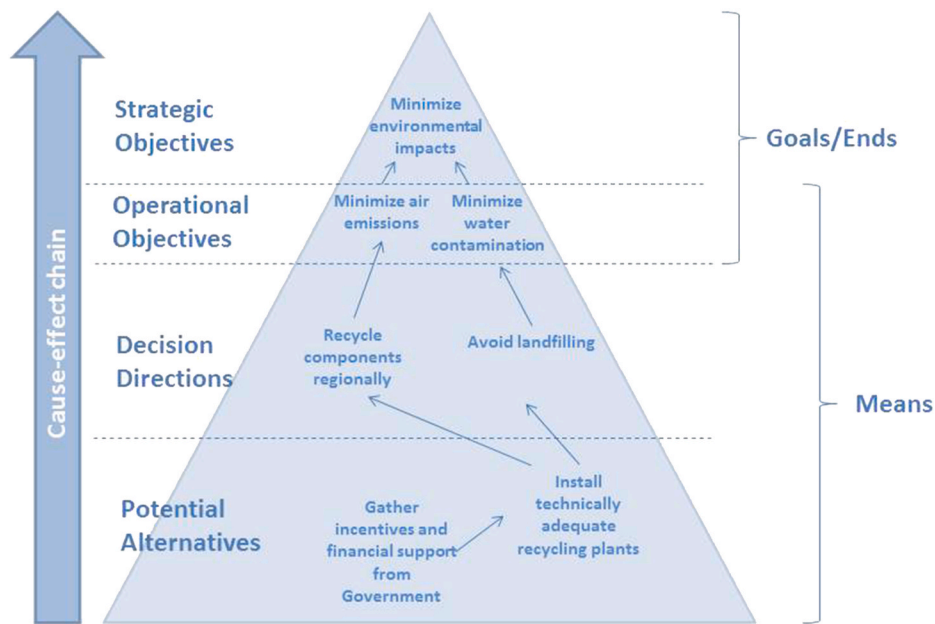


Fig. 2. Structure of a cognitive map. Source: adapted from (Souza et al., 2015).

Hewage (2015) used a similar approach to survey the practices and opportunities for sustainable procurement in the Canadian construction industry. The use of several information sources improves the validity of the results, promotes the reduction of their bias, and allows for the observation of hidden trends that may have been overlooked using a single source (Ruparathna and Hewage, 2015).

Despite the wide applicability and proven usefulness of PSMs in tackling various complex problems (Ackermann et al., 2020), they are occasionally seen as imprecise, having a negative connotation to some operations research (OR) practitioners (Eden and Ackermann, 2018). Reasons for this include a lack of knowledge of the PSM and their benefits, and a perception that they lack mathematical robustness (Ackermann et al., 2020). Progressive PSM studies can aid the lack of knowledge and difficulties in learning, and motivate appreciation of the importance of adopting soft-OR to complex problems. These are mechanisms to leverage competence and confidence in PSMs (Ackermann et al., 2020).

3. Research method

In order to achieve the research objective, the methodology consisted of: systematic literature review; semi-structured interviews; cognitive mapping; interpretation and validation of the LCSA categories. Fig. 3 illustrates the methodology, which is detailed below.

3.1. Reviewing the bibliographic content

Firstly, the authors conducted a systematic literature review (SLR), recommended for choosing impact categories in life cycle studies (Ismaeel, 2018; Langlois et al., 2014). The SLR starts from peer-reviewed articles from scientific journals, allowing for qualitative exploration of

underlying themes relevant to the research topic (Mejías et al., 2016). The SLR only included literature published in international, peer-reviewed scientific journals (excluding books and research reports), ensuring the quality of the content. The SLR searched the Scopus and Web of Science databases. The authors paired keywords (“procurement” or “purchasing” or “supplier selection”) with (“life cycle sustainability assessment” or “life cycle assessment” or “life cycle costing” or “social life cycle assessment” or “life cycle analysis”), connected by “AND”, and further with “AND (“impact categories” or “sustainability categories” or “sustainability criteria” or “sustainability impacts” or “drivers” or “barriers” or “practices”)). The searches covered the period from 2000 to 2019 to ensure contemporaneity. The SLR protocol is in accordance with the methodology proposed by Govindan and Hasanagic (2018).

3.2. Conduct interviews, build individual and merged cognitive maps

The second step of the methodology involved semi-structured interviews for the consultation with experts. The use of semi-structured interviews involves the formulation of open questions, which allows for wide elicitation of the topic under study, and is consistent with the goal of identifying the impact categories based on the knowledge of experts (Michelsen and de Boer, 2009; Ruparathna and Hewage, 2015; Souza et al., 2015). The authors conducted ten real-time interviews via the internet. The Covid-19 pandemic limited in-person interviewing, however, conducting the interviews online did not detract from the quality of interaction with the interviewees. The criterion for the number of interviewees was the saturation principle of qualitative research. This principle determines that the sample size is sufficient when conducting new interviews does not clarify or raise more content on the researched theme (Mason, 2010; Ruparathna and Hewage, 2015). The tenth interview reached the saturation of results.

The interviewees are professionals and researchers with large experience in sustainable procurement. A shortlist of experts supported the choice of interviewees by identifying their individual and collective involvement with LCSA and Sustainable Procurement studies and applications. The shortlist considered experts who are active in renowned institutions or with vast bibliographic production on the research topic. Table 1 shows the profiles of the interviewed experts.

The interviews started with an “icebreaker” talk explaining the

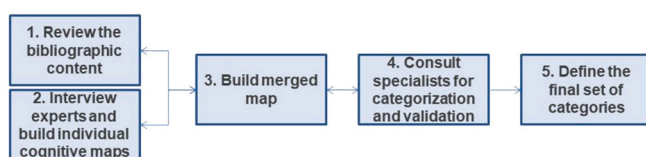


Fig. 3. Methodological structure of the research. Based on: (Falcone and Imbert, 2018; Ruparathna and Hewage, 2015; Souza et al., 2015).

Table 1

Profile of the experts interviewed. Source: prepared by the authors.

Name initials	Profile
D.D.	PhD researcher on Sustainable Procurement (SP).
L.M.	PhD researcher. Public server with large experience in SP and LCT.
T.V.	PhD. Coordinator of the National Sustainability Chamber at a federal institution in Brazil. Author of books on public sustainable bidding.
L.B.	PhD. researcher on SP and LCSA. Author of books and guides on SP.
D.C.	Public server with large experience in SP, Waste Management and Environmental Labelling.
C.T.	Master and MBA in Sustainability. Large experience in sustainability labelling.
R.R.	Master's student in Sustainable Development. MBA in Environmental Management. Sustainability Management member at a non-profit organisation in Brazil.
A.J.	PhD. Experienced in cooperation projects with the European Union and United Nations on environmental labelling for SP.
T.C.	PhD. Lawyer with international experience in LCA and waste management.
P.O.	Expert in SP. Sustainability Commission Coordinator at a federal research institute in Brazil.

context of the research. Then, there were questions of the kind: “what (is/are/explains/are the reasons/happens if ... ?)”, “why (is it important/does it happen ...)?” or “how (does it happen/is this related to that/should it be ...)?”. Examples of questions are: “what are the main selection criteria in sustainable procurement?”; and “what are the sustainability aspects evaluated for a purchasing decision?” This allowed respondents to broadly explain a variety of different aspects related to the object of study. The facilitators also asked questions like: “how is (effect) A achieved?”, and “how does issue A relate to issue B?”, seeking for detailed explanations of possible cause-effect chains associated with each identified potential impact category. Such a methodological approach was defined in accordance with [Ruparathna and Hewage \(2015\)](#) and [Souza et al. \(2015\)](#).

The interviewees' speeches based the further construction of individual cognitive maps. The authors conducted the interviews with each expert individually, building their particular cognitive maps. In a collaborative way with the experts, the mapping process allowed to uproot and elicit the concepts and respective relations based on their knowledge. Each expert reviewed and validated the final version of their respective individual map.

Once having the results of the SLR and the maps of the individual interviews, the authors compiled all information into a single map, by overlapping and merging the partial maps. The overlap occurred through the union of similar concepts (nodes) that contain the same meaning and by connections (arrows) between them.

3.3. Consult experts for categorization and define the final set of categories

The appreciation and discussion by experts ensured that the impact categories (decision criteria) are consistent with respect to enabling procurement evaluation. This set of decision criteria needs to satisfy some properties that MCDA models require ([Franco and Montibeller, 2010](#)). The impact categories must also be consistent with the main issues related to the life cycle analysis framework ([Cherubini et al., 2019](#); [EC-JRC, 2010](#)). Despite terminology differences, there is a high degree of convergence between decision science and life cycle analysis regarding the expected characteristics of a coherent set of decision criteria ([Souza et al., 2015](#)). [Table 2](#) compiles such properties.

The facilitators (authors), in an interactive process with the experts, suggested a redefinition of initially identified categories (operational objectives, mid-top of the maps), adapting them to better satisfy the properties in [Table 2](#). The merged map was used as the main supporting reference in this step. The experts reflected on this suggested set of categories and redefined it as much as necessary, until it satisfied all the

Table 2Desirable properties of decision criteria and life cycle impact categories. Source: adapted from [Souza et al. \(2015\)](#).

Properties	Decision criteria (Franco and Montibeller, 2010).	Life cycle impact categories (EC-JRC, 2010 ; ISO, 2006).
Essential	Consider all essential objectives in the decision. Clear and unambiguous meaning for all members of the decision group. The set should comprise the smallest number of objectives required for the analysis.	Consider the essential objectives/goals. Understandable, considered relevant, and globally accepted.
Understandable, concise and precise		
Operational	It should be possible to measure the performance of decision alternatives against each of the decision criteria.	Impact categories should allow aggregation of inventory data from the product system through indicators. Midpoint categories should be tangible and measurable.
Non-redundant; preferably independent	They should not measure the same concern twice. Decision criteria should offer decision alternatives a performance measurement on one criterion disregarding all other criteria, allowing the use of an aggregation function.	Avoid double counting. Aggregation of impacts.

properties in [Table 2](#). This redefinition also involved merging duplicate or dependent impact categories into one, sharing respective performance indicators. When a new category was suggested, the cause-effect chain was discussed and revised to understand the new causal connections and hierarchy resulting from this change. Such a methodological approach was defined in accordance with [Souza et al. \(2015\)](#).

The same professionals and experts participating in the semi-structured interviews (Section 3.1) evaluated and validated the results. The choice of the same participants was because they were sufficient in the saturation criterion of the sample of interviewees. Also, they already had a link with the research and the content of the maps. The validation with experts led to compiling the final results, allowing for the definition of the final set of general LCSA impact categories for sustainable procurement.

4. Results and discussion

Results and discussion are presented simultaneously, starting from the compilation of SLR results, to the analyses of cognitive maps and the compilation of recommended impact categories.

4.1. SLR results

[Appendix 1](#) contains the resulting papers considered in the SLR. [Table 3](#) presents the compilation of the impact categories extracted literally from these papers.

The papers pointed out that the key methodological references are ISO 14040 LCA Standard ([ISO, 2006](#)), UNEP/SETAC framework ([UNEP/SETAC, 2011](#)) and the International Life Cycle Data Reference System – ILCD ([EC-JRC, 2010](#)). There is convergence in all papers regarding the impact categories according to these seminal references.

[Table 3](#) shows a greater coverage of environmental categories compared to costing and social ones. This confirms that the environmental approach is the most widespread (and structured) within LCT ([Costa et al., 2019](#)). The papers (in accordance with ISO 14040:2006, ILCD and UNEP/SETAC) converged to the following Areas of Protection (endpoints) for environmental assessment: human health, ecosystem

Table 3

List of LCSA impact categories for procurement identified in the papers. Source: adapted from the cited papers.

#	Impact category	Dimension	Papers (# ref. Appendix 1)
1	Climate change	Environmental	1-2; 5-7; 9-10; 14-16; 21-22; 24-27; 29-34; 37-38
2	Global warming	Environmental	1-2; 6; 21; 24-26; 30-32; 35; 38
3	Ozone depletion	Environmental	1-2; 5-7; 9-10; 14-15; 21-22; 24-27; 31-32; 37-38
4	Human toxicity	Environmental	1-2; 6-7; 10; 14; 16; 25-27; 29; 38
5	Eco-toxicity	Environmental	1-2; 6; 14; 21; 25-27; 29-31; 33; 37-38
6	Human toxicity - Cancer effects	Environmental	9; 15; 21; 24; 30-31; 33; 37
7	Human toxicity - Non-Cancer effects	Environmental	9; 15; 21; 31; 37;
8	Fresh water toxicity	Environmental	9; 15; 24
9	Abiotic depletion	Environmental	21; 38
10	Inorganic respiratory particles	Environmental	1-2; 5-7; 9-10; 14-16; 24-27; 30; 33; 37
11	Ionising radiation	Environmental	2; 6; 9; 14; 25; 27
12	Photochemical ozone formation	Environmental	2; 5-7; 9-10; 14-16; 21-22; 24-25; 27; 30-32; 34; 38
13	Terrestrial acidification	Environmental	1; 2; 5-7; 9-10; 14-16; 21-22; 24-27; 29-33; 37-38
14	Eutrophication	Environmental	1-2; 5-7; 10; 14; 16; 21-22; 25-27; 29-34; 37-38
15	Terrestrial eutrophication	Environmental	9; 15; 24
16	Freshwater eutrophication	Environmental	9; 15; 24
17	Marine water eutrophication	Environmental	9; 15; 24
18	Land use	Environmental	2; 5; 9; 14-15; 25; 27; 33-34; 37
19	Depletion of resources (water)	Environmental	2; 5; 9-10; 15; 24; 34; 37
20	Depletion of mineral resources	Environmental	2; 6-7; 9-10; 14-16; 21; 24-26; 31; 33-34; 37
21	Use of non-renewable energy (fossil fuels)	Environmental	1-2; 9; 14
22	Water use	Environmental	2; 6
23	Impacts on human health	Environmental	1; 6-7; 10; 18; 25; 27
24	Smoke formation	Environmental	1; 26; 37
25	Cumulative energy demand	Environmental	5; 15; 32
26	Ecosystem quality	Environmental	6; 10; 18; 25; 27
27	Resource depletion	Environmental	6; 10; 18; 25; 27
28	Labour cost	Cost	1
29	Material acquisition cost	Cost	1; 26-27
30	Waste disposal cost	Cost	1; 26-27
31	Production cost	Cost	18
32	Transportation costs	Cost	18; 26
33	Health and safety	Social	12
34	Transparency and feedback	Social	12
35	Product security	Social	12
36	Employment	Social	12
37	Income	Social	12
38	Human rights	Social	12
39	Gender issues and discrimination	Social	12
40	Access to material resources and land use change	Social	12
41	Governance	Social	12
42	Socio-economic repercussions	Social	12
43	Cultural heritage	Social	12
44	Working conditions	Social	12
45	Damage to workers	Social	12; 18
46	Damage to local community	Social	12; 18
47	Damage to other actors in value chain	Social	12; 18
48	Damage to consumer	Social	12; 18
49	Damage to society	Social	12; 18

quality, and resource depletion.

The costing impact categories cover the costs of labour, material acquisition, waste disposal, production and transportation. These categories comprehend a composition of different costs involved (direct and externalities), such as investments, wastewater, taxes, and fuel ([Sanchez et al., 2020](#); [van Kempen et al., 2017](#)).

Regarding social impacts, [Table 3](#) allows identifying specific stakeholders: workers; local community; other actors of the value chain; consumers; and society, based on ([Falcone and Imbert, 2018](#); [Fan et al., 2018](#)). Such stakeholder impact categories correspond to areas of concern, or endpoints ([UNEP, 2020](#)).

4.2. Cognitive maps of the interviews

[Fig. 4](#) shows an individual map from one of the ten interviews, illustrating the results of cognitive mapping of the interviews.

In [Fig. 4](#), node 27 (social impacts) is an end effect of node 28 (stimulus to race leadership). This set of end nodes that are influenced and hierarchically superior (more strategic than operational) constitute lines of argumentation in the interviewee's discourse. An example of argumentation in [Fig. 4](#) is the sequence of nodes: 20-24-18-0. The systematisation of the decision making process and the identification of key nodes in the argumentation are practical contributions of the study. These cognitive maps aim to provide guidelines on how to assess a product based on sustainability criteria and indicators. The cause and effect chains in the maps describe a set of potential categories and indicators that are similar to other tools, such as GRI (Global Reporting Initiative) and ISO 26,000 (Guidance in social responsibility). The application of this method is also possible in non-academic environments that demand the analysis of complex problems. For example, in business environments, from a trained facilitator and knowledgeable in cognitive mapping, it is possible, with the collaboration of the people involved in the context, to structure the problem by creating a chain of alternatives and group consensus.

[Fig. 4](#) brings relevant issues that emerged exclusively from that interview, which were not in the results of the literature review ([Table 3](#)). Examples are some social aspects linked to encouragement of movements and new social issues, like: initiatives for "former inmates of the prison system", "women victims of domestic violence", "family agriculture", "national industry", "race leadership". Other innovative issues revealed by the interviews were: "adoption of quality and environmental management systems"; "organisational culture focused on sustainability" and "governance model with a systemic vision of sustainability".

The major concerns of the interviewees were the need for the suppliers to present guarantees of sustainable production and governance of production chains. They focused on the constitutive aspects of the suppliers' internal processes and organisational culture, including governance criteria and the adoption of practices to ensure sustainable production. Some practices emerged (action level, bottom of the maps) such as the use of certification and environmental labelling, implementing reverse logistics, and the maturity of environmental management systems (in accordance with previous studies, e.g.: [Harris and Divakarla, 2017](#); [Ponce et al., 2019](#)). Specifically about environmental labelling, the experts' discourse has conformity with the use of Environmental Product Declaration (EPD).

4.3. Merged map

A merged map compiled all the content of the previous results. [Fig. 5](#) presents it. [Fig. 5](#) enabled to identify midpoint/endpoint hierarchies for issues of concern. The merged map unfolds the main objective (sustainable procurement) down to strategic and operational objectives. Concepts such as legitimacy (legality), governance, compliance, and management processes did not fit into the three traditional sustainability dimensions and converged in a fourth one: Supplier Governance

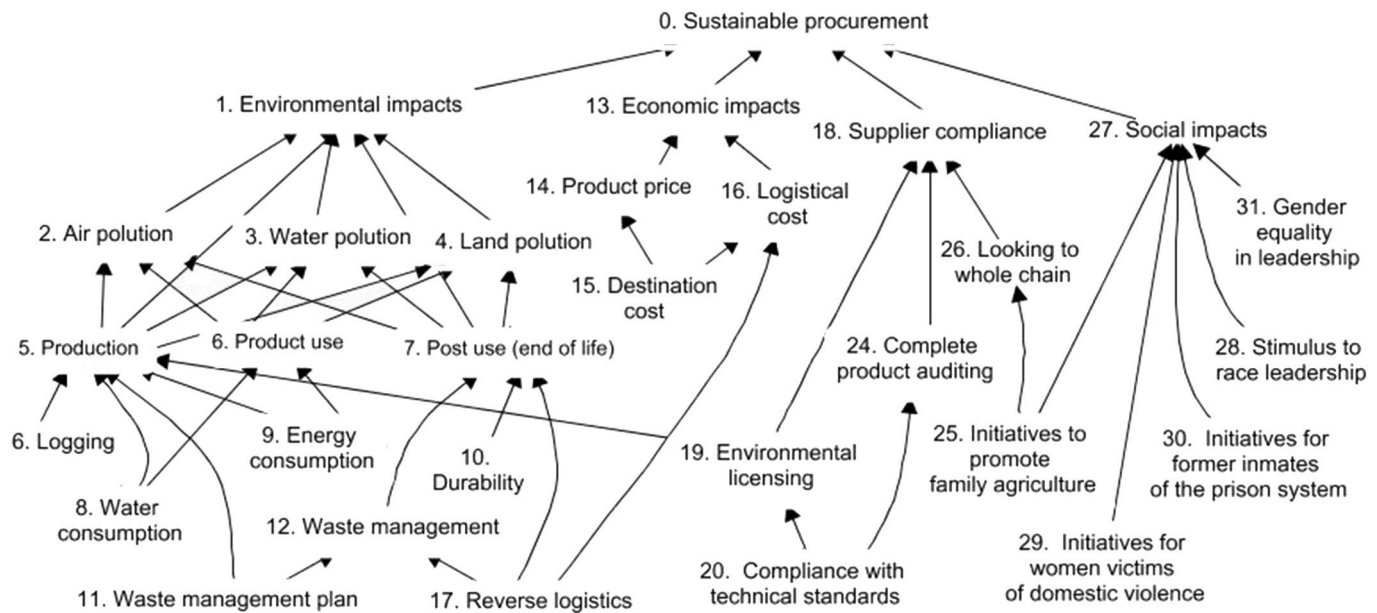


Fig. 4. Individual map from an interview. Source: prepared by the authors.

and Compliance. Besides the SLR results (Table 3) and UNEP/SETAC framework consider “Governance” as a social impact category, there was a necessity of considering it as a strategic sustainability dimension involving the impact of supplier governance for sustainable procurement problems. That dimension is not included in the three LCSA constitutive approaches, but has significant impact on the whole production chain and all the stakeholders involved (Harris and Divakarla, 2017; Laosirihongthong et al., 2019). A previous study in e-waste problem (Guarnieri et al., 2016) has already classified “governance and compliance” as a “strategic dimension” distinct from the three traditional ones.

Fig. 5 shows four dimensions (Environmental, Economic, Social and Governance) as key strategic directions for sustainable procurement (nodes 1 to 4). The mid-bottom contents in the map represent particular aspects of each dimension. The strategic objectives identified for each dimension represent the endpoints for LCSA (nodes 5 to 18). The operational objectives, located in a topological position below strategic objectives, represent midpoints.

The collaboration of the experts during the interactive construction of the merged map was essential to expand and compare the elicited issues. Some appearing to be more specific in the individual maps, in the merged map turned up more strategical. For example, the concept of “waste management” in Fig. 4 was topologically distant to the main objective (sustainable procurement). In Fig. 5, “waste management” was considered at the midpoint level, connected directly to the “management system maturity (management impact)” (endpoint). The experts considered that “waste management” is more related to sustainability management of the suppliers, than properly to the broader environmental cause-effect chain.

The experts found that their main suggestions of decision criteria (impact categories) were more related to aspects of sustainability management, which were allocated to the cluster of supplier governance and compliance (Fig. 5, [B]). In the environmental dimension, the discussion of the merged map led to a consolidation of the already established criteria (LCA), indicating its sufficiency for this dimension. The environmental endpoints elicited by the merged map are aligned to the overall environmental concerns (human health, ecosystem quality, resource depletion). In the economic cluster, two aspects from the interviews stood out, consisting of potential midpoints: operating cost (while using the product) and costs before and after the use of the product. These criteria are connected to the life cycle costing

externalities, and positioned at the strategic level. There was a link between traditional and innovative evaluation criteria. While the costs of ownership comprises traditional midpoints (like “production”, and “transportation” seen in Table 3), the externalities costs were configured by new proposed midpoints: “operating cost” (while using the product), and the “cost outside the use of the product”. Social impacts like encouragement of movements and new social issues (involving initiatives for “former inmates of the prison system”, “women victims of domestic violence”, “family agriculture”; “national industry”; “race leadership”) were a new contribution from the experts (e.g. Fig. 4).

4.4. Refinement and compilation of LCSA impact categories for sustainable procurement

The experts discussed and refined the potential set of (midpoint and endpoint) impact categories until it satisfied the expected properties presented in Table 2. Table 4 shows the final list of impact categories. Categories marked with an asterisk (*) emerged exclusively from the interviews. Appendix 2 contains this table and suggests 220 category indicators.

The study identified 58 midpoint and 14 endpoint categories, linked to the 4 dimensions of sustainable procurement. Different actors in society can use this compilation for evaluation of procurement options. The fourth dimension of sustainability in procurement - Supplier Governance and Compliance - is one of the main contributions for studies using methodological approaches to evaluate procurement alternatives. The experts interpreted that new proposed dimension as a clear and univocal cluster in the merged map. They indicated that the three traditional sustainability approaches are insufficient to address aspects like: systemic vision, management processes, and third-party certifications, which were considered essential in the evaluation for sustainable procurement. This result aligns with the conclusions of previous studies, that indicated the relevance of the supplier’s governance and compliance component for sustainable procurement, and the need for a strategy factor through LCT (Ahmed et al., 2020; Guarnieri et al., 2016).

The catalogue of impact categories presented in Table 4 offers a contribution for LCSA community, procurement officers and society in general. It opens opportunities for a thorough procurement assessment considering innovative and important aspects, like the categories marked with an asterisk (*) in Table 4. The study also suggests specific

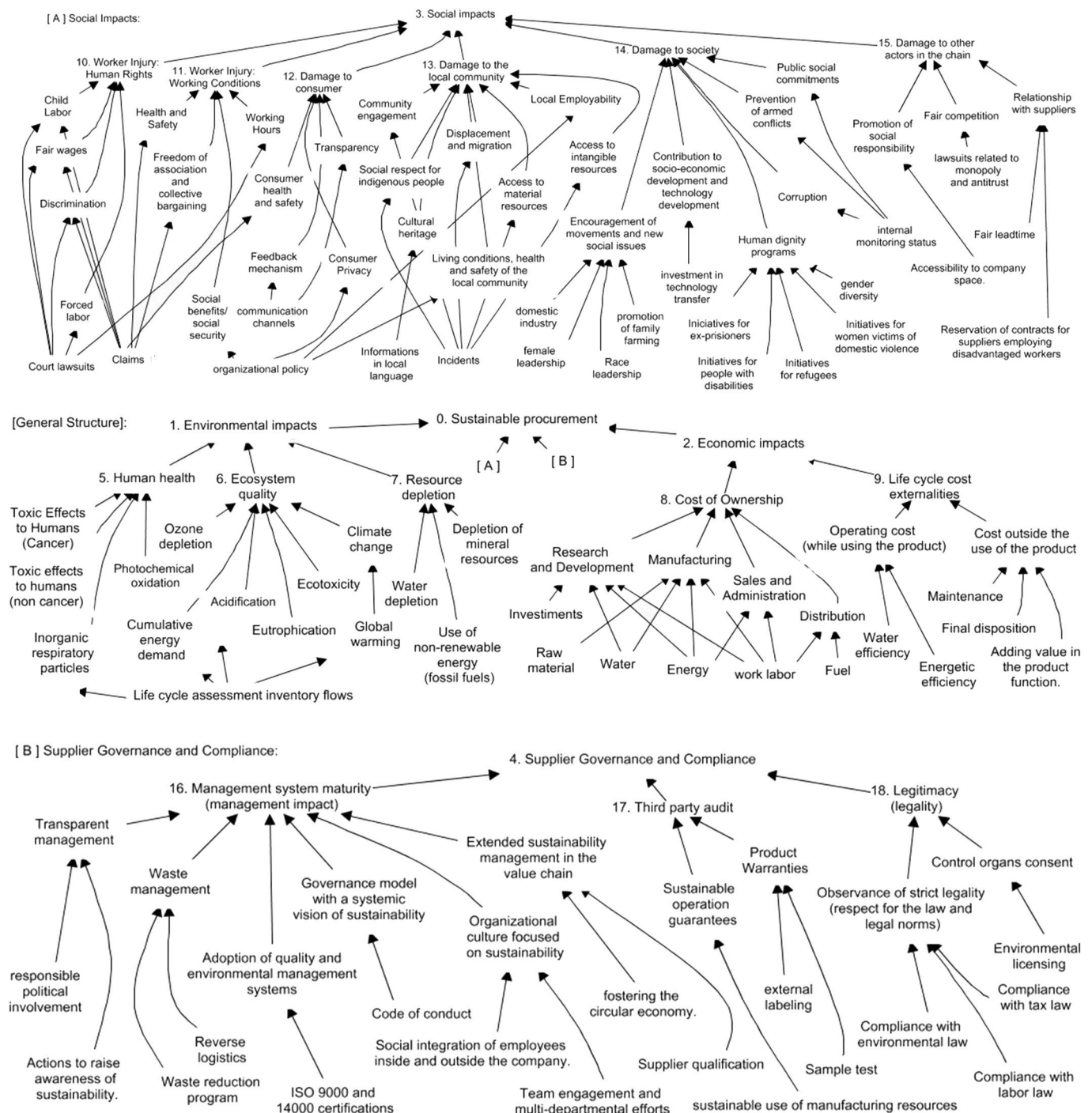


Fig. 5. Merged map. Source: prepared by the authors.

category indicators (Appendix 2) that may enhance sustainable procurement assessments. For the LCSA community, the proposal to consider supplier governance and compliance as a new dimension of sustainability is a novelty that provides an important insight for future work.

There are some issues of concern when trying to operationalise LCSA in a practical procurement context. There are limitations in terms of quantifying the impact categories in order to allow the operationalisation of the method, which can be aided by using the indicators suggested in Appendix 2. The experts concluded that, in their opinion, the indicators in Appendix 2 are assessable and possible to be operationalised in practice. However, implementation of such a set of indicators has still

not been done in known real procurement projects. Future applications of the proposed approach (Appendix 2) in real projects can help to identify issues and to improve this model. Operational limitations can be addressed through the use of methodological and information systems, such as ISO 14040 and ILCD (for LCA), as also IEC 60300-3-3 for LCC. Also, it is possible to use primary, site-specific data and to involve local stakeholders in the definition of the indicators. Site-specific indicators involve: site visit or site-specific audit; interviews with community members, employees and management; organisation-specific reports, such as GRI reports; and organisation-specific audits, such as AA 1000 and SA8000 (UNEP, 2020). The UNEP/SETAC databases for SLCA and LCSA are currently available to support the methodological

Table 4

Catalogue of life cycle sustainability impact categories for sustainable procurement.

Sustainability dimension for procurement	Life cycle sustainability impact categories in sustainable procurement (evaluation criteria)		
	Endpoints	Midpoints	
Environmental	Human Health	Toxic Effects to Humans (Cancer)	
		Toxic effects to humans (non-cancer)	
	Ecosystem quality	Photochemical oxidation	
		Inorganic respiratory particles	
		Ozone depletion	
		Cumulative energy demand	
		Eco-toxicity	
	Resource depletion	Acidification	
		Eutrophication	
		Climate change	
		Water depletion	
Economic	Costs of Ownership	Depletion of mineral resources	
		Use of non-renewable energy (fossil fuels)	
	Life cycle costing externalities	Research and Development	
		Manufacturing	
	Social	Damages to society	Sales and Administration
			Distribution
		Workers: Human Rights	Operating cost (while using the product) ^a
			Cost outside the use of the product ^b
		Workers: Working Conditions	Public social commitments
			Contribution to socio-economic development and technology development
		Supplier Governance and Compliance ^a	Damages to consumers
Corruption			
Damages to the local community	Encouragement of movements and new social issues ^a		
	Human dignity programs		
Damages to other value chain actors	Child labour		
	Forced labour		
Management system maturity (management impact) ^a	Fair wages		
	Discrimination		
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a		Health and Safety
			Working Hours
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a		Social benefits/social security
		Freedom of association and collective bargaining	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Consumer health and safety	
		Feedback mechanism	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Consumer Privacy	
		Transparency	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Displacement and migration	
		Community engagement	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Cultural heritage	
		Social respect for indigenous people	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Local Employability	
		Access to intangible resources	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Access to material resources	
		Living conditions, health and safety of the local community	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Fair competition	
		Promotion of social responsibility	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Relationship with suppliers	
		Transparent management ^a	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Organisational culture focused on sustainability ^a	
		Waste management ^a	
Supplier Governance and Compliance ^a	Management system maturity (management impact) ^a	Adoption of quality and environmental management systems ^a	
		Governance model with a systemic vision of sustainability ^a	

Table 4 (continued)

Sustainability dimension for procurement	Life cycle sustainability impact categories in sustainable procurement (evaluation criteria)	
	Endpoints	Midpoints
	Third party auditing ^a	Extended sustainability management in the value chain ^a
		Product Warranties ^a
	Legitimacy (legality) ^a	Sustainable operation guarantees ^a
		Control organs consent ^a
		Observance of strict legality (respect for the law and legal norms) ^a

^a Categories emerged exclusively from the interviews.

implementation by procurement officers and end consumers. It is recommended that the procurement agent check the legal mechanisms in their context to ensure compliance. Mainly in public procurement, normally linked to a set of legal restrictions, requirements and regulations for bidding, indicators and assessment bases must be established in accordance with current regulations.

Future practical applications can provide new indicators and impact categories to reflect changes in external environments and different local contexts. As mentioned by Goggins and Rau (2016), concepts of sustainability are changing all the time, with new evidence continually emerging on how to improve sustainability in procurement. For example, damages to the local community can be adjusted to be representative of a specific geographical area. As the practical application of the catalogue of categories and indicators will be disseminated and expanded, the sharing of experience by the agents who applied it can generate improvements in the approach, whether in the list of categories used, as well as in the indicators and their forms of measurement.

The catalogue of impact categories can assist companies to better manage suppliers or improve buyer supplier relation. Since a key issue in the client-supplier relationship in sustainable management is communication (Walker and Brammer, 2012), the clear establishment of the evaluation criteria by the purchaser makes the dialogue around procurement alternatives more objective. Communication ensures collaboration between organisations, sharing initiatives, data and information along the supply chain, and meeting the companies' sustainability goals, which strengthens the search for establishing long-term supply partnerships. Long-term partnership asserts the importance of moral values across supply relationships, and the relevance of collaboration between supply chain agents, which can become a differential in business partnership and allow for the exploration of new market opportunities.

Each type of product and different manufacturers in each sector has different production baselines (Butt et al., 2015; EC-JRC, 2010). To cover the plurality of specificities involved, the compiled categories needed to be generalist. Depending on the characteristics of the product or even on the conditions of the buyer or supplier, there may be limitations and economic infeasibility to apply and measure some of the proposed criteria. For example, for suppliers that have no direct relationship with indigenous communities, there is a limitation to consider the midpoint "social respect for indigenous people". One common solution is to prioritise categories that best represent or are considered most important for each dimension (e.g. in a study of e-waste management in Brazil by Souza et al., 2016). Such prioritisation of decision criteria is often supported by the application of MCDA methods (Costa et al., 2019; Galo et al., 2018). Depending on the evaluation system adopted in each procurement problem, it is possible to suppress impact categories that are irrelevant to a given context (Patala et al., 2016). In the context of Supplier Selection Problems, the criteria specifically related to suppliers and their management are highlighted (Alhola et al., 2019; Igarashi et al., 2013; Pullman and Wikoff, 2017). Thus, "supplier governance and compliance" dimension criteria are potentially

prioritised in the catalogue (Table 4).

To guide what indicators are considered relevant, it is important to consider that each organisation may have different objectives and available resources, where bolder purposes lead to a greater amount of criteria under evaluation. For example, there is a significant variation in objectives and available resources between private and public procurement. The adoption of sustainable procurement across the sectors with the private sector registers higher levels than their public counterparts. The literature shows public sector organisations being bureaucratic, whose managers are less committed to their organisation, which is in contrast to their private sector counterparts (Delmonico et al., 2018). For both public and private sectors, in green procurement, the more complex the procurement, the greater the organisation's ability to apply LCA (Jenssen and de Boer, 2019). The application of LCA can also be adjusted to different levels of decision by type of purchase. For example, a specific (one-time) purchase may have simpler system limits for its LCA compared to a strategic purchase, which incurs long-term impacts for stakeholders (Butt et al., 2015).

The experts' validation indicated that the proposed criteria are sufficient and valid to evaluate sustainable procurement options. This indicates that the model is operational (measurable), with clear and unambiguous criteria (i.e., satisfies Table 2 requisites), showing that these LCSA impact categories can be adopted as criteria for evaluating options in sustainable procurement. The final results are also in accordance with literature impact categories (e.g. human rights, climate change, working conditions; as in Falcone and Imbert, 2018; UNEP/SETAC, 2011), but also present innovative impact categories, like organisational culture focused on sustainability; governance model with a systemic vision of sustainability; sustainable operation guarantees; observance of strict legality. Both the convergence with the literature and the experts' validation of the categories support that it is possible to use LCSA in a decision support methodology for sustainable procurement (ex. MCDA methods). It also confirms that the innovative combination of Cognitive Mapping, SLR, and Semi-Structured Interviews is a valid multimethodology to define and analyse sustainability impact categories in sustainable procurement and other sustainability-related decision problems.

4.5. Advantages and limitations of the methodology

The use of cognitive mapping was fundamental to structure the decision problem of sustainable procurement. The technique was able to provide the elicitation of new aspects and facilitated the convergence to a collective understanding (as supported by Assunção et al., 2020; Cunha and Morais, 2019). The use of the saturation criterion was a methodological advantage for determining when it would be time to stop the interviews (as supported by Ruparathna and Hewage, 2015). Other works reached saturation between 06 and 10 interviews (Mason, 2010; Ruparathna and Hewage, 2015); this one achieved it in the tenth.

While offering significant support to LCSA, Sustainable Procurement and Supplier Selection Problems, the methodology has some limitations, especially in the use of cognitive mapping as a means of problem structuring:

- i. The task of determining the cause-effect structure in the maps is demanding for the facilitator and experts (Souza et al., 2015).
- ii. The construction of maps and cataloguing of potential impact categories involves subjectivity, and qualitative understanding may vary depending on each interviewee (Assunção et al., 2020). The merged map building process facilitates consensus, but other techniques or quantitative approaches (e.g. MCDA) could facilitate a more objective consensus building (Souza et al., 2016).
- iii. Although there is a hierarchical relation between endpoints and midpoints, within the total set of categories, this work did not determine which categories are more important or priorities.

MCDA or a quantitative approach could support their ranking or prioritisation, in each decision problem (Marttunen et al., 2017).

5. Conclusions and recommendations

The main result of the research is the compilation of 58 midpoint categories (Table 4), relating to 14 endpoint sustainability impact categories, which offers a contribution to different actors in supply chains in evaluating their supply options with sustainable procurement. The cause-effect relationships between the elucidated sustainability aspects for sustainable procurement (Fig. 5) allowed for classifying them into midpoints and endpoints. The consultation of experts enriched the understanding of the problem, eliciting a fourth sustainability dimension for procurement, comprising "Supplier Governance and Compliance". Eliciting the impact categories permitted to suggest 220 midpoint indicators (Appendix 2) to assess the compiled decision criteria. The paper makes a major contribution to diverse actors in different markets by offering a catalogue of categories and suggested indicators to guide the assessment of options in sustainable procurement. Procurement agents can use the catalogue of impact categories as a roadmap for assessing the sustainability of procurement, and through the proposed indicators, take the measurements of each category, which is configured as an objective criterion for the sustainability assessment.

The "Supplier Governance and Compliance" dimension consisted of the endpoints "Management system maturity (management impact)", "Third party auditing" and "Legitimacy (legality)". It involves new perspectives of criteria for sustainability assessment in supplier selection problems, including midpoints such as: "organisational culture focused on sustainability"; "governance model with a systemic vision of sustainability"; "extended sustainability management in the value chain"; "observance of strict legality (respect for the law and legal norms)"; and "product warranties".

Experts involvement revealed well established impact categories in LCT/LCSA frameworks, such as "climate change", "resource depletion" and "ecosystem quality" but also other innovative ones, like: "encouragement of movements and new social issues", which comprises initiatives for: "former inmates of the prison system", "women victims of domestic violence", "family agriculture"; "national industry"; and "race leaderships".

In the economic dimension, the results structured the evaluation criteria based on two endpoints: "costs of ownership" and "life cycle costing externalities". While the costs of ownership comprise traditional costs like "manufacturing", "sales and administration" and "distribution", the costing externalities were configured by new proposed midpoints: "operating costs (while using the product)", and the "costs before/after the use of the product".

Recommendations for future studies include:

- i. Prioritisation of procurement decision criteria using methods like MCDA.
- ii. Performing the integrated prioritisation of the impact categories through MCDA.
- iii. Applying the compiled decision criteria in sustainable procurement problems.

Supplementary material

Appendix 1: Papers analysed in the SLR.

Appendix 2: Catalogue of Life Cycle Sustainability Impact Categories for Sustainable Procurement with the proposed category indicators.

CRedit authorship contribution statement

Daniel Felipe de Oliveira: Conceptualization, Formal analysis, Investigation, Resources, Visualization. **Ricardo Gabbay de Souza:** Supervision, Validation.

Declaration of competing interest

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

Data availability

No data was used for the research described in the article.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jclepro.2022.135448>.

References

- Abuabara, L., Paucar-Caceres, A., Belderrain, M.C.N., Burrowes-Cromwell, T., 2018. A systemic framework based on Soft or approaches to support teamwork strategy: an aviation manufacturer Brazilian company case. *J. Oper. Res. Soc.* 69, 220–234. <https://doi.org/10.1057/s41274-017-0204-9>.
- Ackermann, F., Alexander, J., Stephens, A., Pincombe, B., 2020. In defence of soft OR: reflections on teaching soft OR. *J. Oper. Res. Soc.* 71, 1–15. <https://doi.org/10.1080/01605682.2018.1542960>.
- Ackermann, F., Eden, C., Cropper, S., 1992. Getting started with cognitive mapping. In: *7th Young OR Conference*, pp. 65–82.
- Ahmed, S.U., Ali, A., Memon, A.H., Hussain, A., 2020. Policy framework for implementation of sustainable procurement practices in healthcare industry. *Environ. Dev. Sustain.* 23, 3929–3945. <https://doi.org/10.1007/s10668-020-00750-w>.
- Alhola, K., Ryding, S.O., Salmenperä, H., Busch, N.J., 2019. Exploiting the potential of public procurement: opportunities for circular economy. *J. Ind. Ecol.* 23, 96–109. <https://doi.org/10.1111/jiec.12770>.
- Assunção, E.R.G.T.R., Ferreira, F.A.F., Meidutė-Kavaliauskienė, I., Zopounidis, C., Pereira, L.F., Correia, R.J.C., 2020. Rethinking urban sustainability using fuzzy cognitive mapping and system dynamics. *Int. J. Sustain. Dev. World Ecol.* 27, 261–275. <https://doi.org/10.1080/13504509.2020.1722973>.
- Awasthi, A., Omrani, H., 2018. A scenario simulation approach for sustainable mobility project evaluation based on fuzzy cognitive maps. *Int. J. Model. Simulat.* 38, 262–272. <https://doi.org/10.1080/02286203.2018.1451711>.
- Butt, A.A., Toller, S., Birgisson, B., 2015. Life cycle assessment for the green procurement of roads: a way forward. *J. Clean. Prod.* 90, 163–170. <https://doi.org/10.1016/j.jclepro.2014.11.068>.
- Cherubini, E., Zanghelini, G.M., Piemonte, D., Muller, N.B., Dias, R., Kabe, Y.H.O., Soto, J., 2019. Environmental sustainability for highways operation: comparative analysis of plastic and steel screen anti-glare systems. *J. Clean. Prod.* 240 <https://doi.org/10.1016/j.jclepro.2019.118152>.
- Costa, D., Quinteiro, P., Dias, A.C., 2019. A systematic review of life cycle sustainability assessment: current state, methodological challenges, and implementation issues. *Sci. Total Environ.* 686, 774–787. <https://doi.org/10.1016/j.scitotenv.2019.05.435>.
- Cunha, A., Morais, D., 2019. Problem structuring methods in group decision making: a comparative study of their application. *Int. J. Oper. Res.* 19, 1081–1100. <https://doi.org/10.1007/s12351-017-0310-0>.
- Del Borghi, A., Moreschi, L., Gallo, M., 2020. Communication through ecolabels: how discrepancies between the EU PEF and EPD schemes could affect outcome consistency. *Int. J. Life Cycle Assess.* 25, 905–920. <https://doi.org/10.1007/s11367-019-01609-7>.
- Delmonico, D., Jabbour, C.J.C., Pereira, S.C.F., de Sousa Jabbour, A.B.L., Renwick, D.W.S., Thomé, A.M.T., 2018. Unveiling barriers to sustainable public procurement in emerging economies: evidence from a leading sustainable supply chain initiative in Latin America. *Resour. Conserv. Recycl.* 134, 70–79. <https://doi.org/10.1016/j.resconrec.2018.02.033>.
- EC-JRC, 2010. International reference life cycle data system (ILCD) Handbook – general guide for life cycle assessment – detailed guidance. JRC. <https://doi.org/10.2788/38479>.
- Eden, C., Ackermann, F., 2018. Theory into practice, practice to theory: action research in method development. *Eur. J. Oper. Res.* 271, 1145–1155. <https://doi.org/10.1016/j.ejor.2018.05.061>.
- Falcone, P.M., Imbert, E., 2018. Social life cycle approach as a tool for promoting the market uptake of bio-based products from a consumer perspective. *Sustain.* 10. <https://doi.org/10.3390/su10041031>.
- Fan, L., Pang, B., Zhang, Y., Zhang, X., Sun, Y., Wang, Y., 2018. Evaluation for social and humanity demand on green residential districts in China based on SLCA. *Int. J. Life Cycle Assess.* 23, 640–650. <https://doi.org/10.1007/s11367-016-1166-x>.
- Farsari, I., Butler, R.W., Szivas, E., 2010. The use of cognitive mapping in analysing sustainable tourism policy: methodological implications. *Tour. Recreat. Res.* 35, 145–160. <https://doi.org/10.1080/02508281.2010.11081630>.
- Franco, L.A., Montibeller, G., 2010. Problem structuring for multicriteria decision analysis interventions. *Wiley Encycl. Oper. Res. Manag. Sci.* <https://doi.org/10.1002/9780470400531.eorms0683>.
- Galo, N.R., Calache, L.D.D.R., Carpinetti, L.C.R., 2018. A group decision approach for supplier categorization based on hesitant fuzzy and ELECTRE TRI. *Int. J. Prod. Econ.* 202, 182–196. <https://doi.org/10.1016/j.ijpe.2018.05.023>.
- Goggins, G., Rau, H., 2016. Beyond calorie counting: assessing the sustainability of food provided for public consumption. *J. Clean. Prod.* 112, 257–266. <https://doi.org/10.1016/j.jclepro.2015.06.035>.
- Govindan, K., Hasanagic, M., 2018. A systematic review on drivers, barriers, and practices towards circular economy: a supply chain perspective. *Int. J. Prod. Res.* 56, 278–311. <https://doi.org/10.1080/00207543.2017.1402141>.
- Govindan, K., Rajendran, S., Sarkis, J., Murugesan, P., 2015. Multi criteria decision making approaches for green supplier evaluation and selection: a literature review. *J. Clean. Prod.* 98, 66–83. <https://doi.org/10.1016/j.jclepro.2013.06.046>.
- Guarnieri, P., Silva, L.C., Levino, N.A., 2016. Analysis of electronic waste reverse logistics decisions using Strategic Options Development Analysis methodology: a Brazilian case. *J. Clean. Prod.* 133, 1105–1117. <https://doi.org/10.1016/j.jclepro.2016.06.025>.
- Harris, K., Divakarla, S., 2017. Supply chain risk to reward: responsible procurement and the role of ecolabels. *Procedia Eng.* 180, 1603–1611. <https://doi.org/10.1016/j.proeng.2017.04.323>.
- Hervani, A., Helms, M.M., Sarkis, J., 2005. Performance measurement for green supply chain management. *Benchmark Int. J.* 12, 330–353. <https://doi.org/10.1108/14635770510609015>.
- IEC, 2004. Application Guide – Life Cycle Costing. International Electrotechnical Commission (IEC), Geneva.
- Igarashi, M., De Boer, L., Fet, A.M., 2013. What is required for greener supplier selection? A literature review and conceptual model development. *J. Purch. Supply Manag.* 19, 247–263. <https://doi.org/10.1016/j.pursup.2013.06.001>.
- Ismael, W.S.E., 2018. Midpoint and endpoint impact categories in Green building rating systems. *J. Clean. Prod.* 182, 783–793. <https://doi.org/10.1016/j.jclepro.2018.01.217>.
- ISO, 2017. ISO 20400:2017 Sustainable Procurement — Guidance, first ed.
- ISO, 2006. ISO 14040:2006 Environmental Management — Life Cycle Assessment — Principles and Framework, second ed.
- Jenssen, M.M., de Boer, L., 2019. Implementing life cycle assessment in green supplier selection: a systematic review and conceptual model. *J. Clean. Prod.* 229, 1198–1210. <https://doi.org/10.1016/j.jclepro.2019.04.335>.
- Jørgensen, A., Herrmann, I.T., Bjørn, A., 2013. Analysis of the link between a definition of sustainability and the life cycle methodologies. *Int. J. Life Cycle Assess.* 18, 1440–1449. <https://doi.org/10.1007/s11367-013-0617-x>.
- Kloepffer, W., 2008. Life cycle sustainability assessment of products. *Int. J. Life Cycle Assess.* 13, 95. <https://doi.org/10.1065/lca2008.02.376>.
- Kokkios, K., Lakioti, E., Papageorgiou, E., Moustakas, K., Karayannis, V., 2018. Fuzzy cognitive map-based modeling of social engagement to overcome uncertainties in establishing waste biorefinery facilities. *Front. Energy Res.* 6, 1–17. <https://doi.org/10.3389/fenrg.2018.00112>.
- Kucukvar, M., Eglimez, G., Tatari, O., 2014. Sustainability assessment of U.S. final consumption and investments: triple-bottom-line input-output analysis. *J. Clean. Prod.* 81, 234–243. <https://doi.org/10.1016/j.jclepro.2014.06.033>.
- Langlois, J., Fréon, P., Steyer, J.P., Delgenes, J.P., Hélias, A., 2014. Sea-use impact category in life cycle assessment: state of the art and perspectives. *Int. J. Life Cycle Assess.* 19, 994–1006. <https://doi.org/10.1007/s11367-014-0700-y>.
- Laosirihongthong, T., Samaranayake, P., Nagalingam, S., 2019. A holistic approach to supplier evaluation and order allocation towards sustainable procurement. *Benchmarking* 26, 2543–2573. <https://doi.org/10.1108/BLJ-11-2018-0360>.
- Luttenberger, A., Luttenberger, L.R., 2017. Sustainable procurement and environmental life-cycle costing in maritime transport. *WMU J. Marit. Aff.* 16, 219–231. <https://doi.org/10.1007/s13437-016-0116-6>.
- Mansi, M., 2015. Sustainable procurement disclosure practices in central public sector enterprises: evidence from India. *J. Purch. Supply Manag.* 21, 125–137. <https://doi.org/10.1016/j.pursup.2014.12.002>.
- Marttunen, M., Lienert, J., Belton, V., 2017. Structuring problems for Multi-Criteria Decision Analysis in practice: a literature review of method combinations. *Eur. J. Oper. Res.* 263, 1–17. <https://doi.org/10.1016/j.ejor.2017.04.041>.
- Mason, M., 2010. Sample size and saturation in PhD studies using qualitative interviews. *Forum Qual. Sozialforsch.* 11 <https://doi.org/10.17169/fqs-11.3.1428>.
- Mejías, A.M., Paz, E., Pardo, J.E., 2016. Efficiency and sustainability through the best practices in the Logistics Social Responsibility framework. *Int. J. Oper. Prod. Manag.* 36, 164–199. <https://doi.org/10.1108/IJOPM-07-2014-0301>.
- Michelsen, O., de Boer, L., 2009. Green procurement in Norway: a survey of practices at the municipal and county level. *J. Environ. Manag.* 91, 160–167. <https://doi.org/10.1016/j.jenvman.2009.08.001>.
- Miemczyk, J., Johnsen, T.E., Macquet, M., 2012. Sustainable purchasing and supply management: a structured literature review of definitions and measures at the dyad, chain and network levels. *Supply Chain Manag. An Int. J.* 17, 478–496. <https://doi.org/10.1108/13598541211258564>.
- Mingers, J., Rosenhead, J., 2004. Problem structuring methods in action. *Eur. J. Oper. Res.* 152, 530–554. [https://doi.org/10.1016/S0377-2217\(03\)00056-0](https://doi.org/10.1016/S0377-2217(03)00056-0).
- Molin, E., Martin, M., Björklund, A., 2021. Addressing sustainability within public procurement of food: a systematic literature review. *Sustain.* 13. <https://doi.org/10.3390/su132313395>.
- Nawaz, W., Koç, M., 2019. Exploring organizational sustainability: themes, functional areas, and best practices. *Sustain.* 11. <https://doi.org/10.3390/su11164307>.
- Patala, S., Jalkala, A., Keränen, J., Väisänen, S., Tuominen, V., Soukka, R., 2016. Sustainable value propositions: framework and implications for technology suppliers. *Ind. Market. Manag.* 59, 144–156. <https://doi.org/10.1016/j.indmarman.2016.03.001>.

- Pereira, L. de S., Morais, D.C., 2020. Multicriteria decision model to establish maintenance priorities for wells in a groundwater system. *Water Resour. Manag.* 34, 377–392. <https://doi.org/10.1007/s11269-019-02457-8>.
- Ponce, H.G., Gil, M.T.N., Durán, M.P., 2019. Responsible public procurement. Design of measurement indicators. *CIRIEC - España, Rev. Econ. Pública, Soc. Cooperativa* 253–280. <https://doi.org/10.7203/CIRIEC-E.96.12627>.
- Pullman, M., Wikoff, R., 2017. Institutional sustainable purchasing priorities: stakeholder perceptions vs environmental reality. *Int. J. Oper. Prod. Manag.* 37, 162–181. <https://doi.org/10.1108/IJOPM-07-2014-0348>.
- Ruparathna, R., Hewage, K., 2015. Sustainable procurement in the Canadian construction industry: current practices, drivers and opportunities. *J. Clean. Prod.* 109, 305–314. <https://doi.org/10.1016/j.jclepro.2015.07.007>.
- Sala, S., Farioli, F., Zamagni, A., 2013. Progress in sustainability science: lessons learnt from current methodologies for sustainability assessment: Part 1. *Int. J. Life Cycle Assess.* 18, 1653–1672. <https://doi.org/10.1007/s11367-012-0508-6>.
- Sanchez, S.A., Eckelman, M.J., Sherman, J.D., 2020. Environmental and economic comparison of reusable and disposable blood pressure cuffs in multiple clinical settings. *Resour. Conserv. Recycl.* 155, 104643 <https://doi.org/10.1016/j.resconrec.2019.104643>.
- Santos, P.R., Curo, R.S.G., Belderrain, M.C.N., 2011. Aplicação do mapa cognitivo a um problema de decisão do setor aeroespacial de defesa do Brasil. *J. Aero. Technol. Manag.* 3, 215–226. <https://doi.org/10.5028/jatm.2011.03021211>.
- Silva, G.F.P.da, Belderrain, M.C.N., 2019. Prioritization of strategic initiatives in the context of natural disaster prevention. *Brazilian J. Oper. Prod. Manag.* 16, 473–489. <https://doi.org/10.14488/bjopm.2019.v16.n3.a10>.
- Souza, R.G., Clímaco, J.C.N., Sant'Anna, A.P., Rocha, T.B., do Valle, R. de A.B., Quelhas, O.L.G., 2016. Sustainability assessment and prioritisation of e-waste management options in Brazil. *Waste Manag.* 57, 46–56. <https://doi.org/10.1016/j.wasman.2016.01.034>.
- Souza, R.G., Rosenhead, J., Salhofer, S.P., Valle, R.A.B., Lins, M.P.E., 2015. Definition of sustainability impact categories based on stakeholder perspectives. *J. Clean. Prod.* 105, 41–51. <https://doi.org/10.1016/j.jclepro.2014.09.051>.
- Trigo, A.M., Nunes, C., Maruyama, Ú., Pinzón, I.A.R., Martínez, P., 2017. Environmental change in sustainable management approach in Brazilian public education: multiple case study. In: *Proc. Int. Conf. Ind. Eng. Oper. Manag.* 2017, pp. 1046–1055.
- UNEP/SETAC, 2011. Towards a Life Cycle Sustainability Assessment.
- UNEP, 2020. Guidelines for Social Life Cycle Assessment of Products and Organizations. United Nations Environment Programme.
- van Kempen, E.A., Spiliotopoulou, E., Stojanovski, G., de Leeuw, S., 2017. Using life cycle sustainability assessment to trade off sourcing strategies for humanitarian relief items. *Int. J. Life Cycle Assess.* 22, 1718–1730. <https://doi.org/10.1007/s11367-016-1245-z>.
- Walker, H., Brammer, S., 2012. The relationship between sustainable procurement and e-procurement in the public sector. *Int. J. Prod. Econ.* 140, 256–268. <https://doi.org/10.1016/j.ijpe.2012.01.008>.
- Walker, H., Brammer, S., 2009. Sustainable procurement in the United Kingdom public sector. *Supply Chain Manag.* 14, 128–137. <https://doi.org/10.1108/13598540910941993>.
- Walker, H., Miemczyk, J., Johnsen, T., Spencer, R., 2012. Sustainable procurement: past, present and future. *J. Purch. Supply Manag.* 18, 201–206. <https://doi.org/10.1016/j.pursup.2012.11.003>.
- Waris, M., Panigrahi, S., Mengal, A., Soomro, M.I., Mirjat, N.H., Ullah, M., Azlan, Z.S., Khan, A., 2019. An application of analytic hierarchy process (ahp) for sustainable procurement of construction equipment: multicriteria-based decision framework for Malaysia. *Math. Probl. Eng.* <https://doi.org/10.1155/2019/6391431>, 2019.
- Zamagni, A., 2012. Life cycle sustainability assessment. *Int. J. Life Cycle Assess.* 1–13. https://doi.org/10.1007/978-3-319-71058-7_18-1.

3.10 Artigo 10 – Life Cycle Sustainability Assessment (LCSA) framework for Lithium-Ion batteries (LIBs) based on the Sustainable Development Goals (SDGs)*

* Domingues, A., Franzoni, M., Padoan, F., Souza, R.G., Silva, J.R. Life Cycle Sustainability Assessment (LCSA) framework for Lithium-Ion batteries (LIBs) based on the Sustainable Development Goals (SDGs). Submetido ao J Ind Ecol, em 12/11/2022.

ABSTRACT

The Sustainable Development Goals (SDGs) of the United Nations (UN) provide targets aimed at contributing to a more prosperous life in the world. The priorities include protecting the environment through the use of renewable energy sources and reducing poverty. One of the challenges associated with SDGs is to assess how much and how the targets are impacted. The Life Cycle Sustainability Assessment (LCSA) approach has the potential to identify critical aspects of the SDGs related to products and their respective life cycles, as it provides a holistic framework for comprehending their environmental, social, and economic impacts. This paper aims to develop a framework for evaluating the life cycle sustainability impacts of Lithium-Ion Batteries (LIBs), based on the integration of LCSA and SDG. The study applied a multimethodology approach comprehending Systematic Literature Review (SLR), Decision Sciences, and Expert consultation. The main results present a complete framework of recommended LCSA impact categories and indicators for the life cycle of LIBs, based on the SDGs and the SLR. The proposed framework can support productive and end-of-life sectors within the LIBs life cycles to become more sustainable and help in the fulfilment of the SDG.

Keywords: Life Cycle Sustainability Assessment; Li-Ion batteries; Sustainable Development Goals; Industrial Ecology; Lithium-Ion batteries (LIBs).

Article Type: Research and analysis

Title: Life Cycle Sustainability Assessment (LCSA) framework for Lithium-Ion batteries (LIBs) based on the Sustainable Development Goals (SDGs)

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Data Availability Statement: The data that supports the findings of this study are available in the supporting information of this article.

Keywords: Life Cycle Sustainability Assessment, Sustainable Development Goals, Industrial Ecology, Li-ion battery, Lithium-Ion batteries (LIBs), Sustainable products.

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Abstract: The Sustainable Development Goals (SDGs) of the United Nations (UN) provide targets aimed at contributing to a more prosperous life in the world. The priorities include protecting the environment through the use of renewable energy sources and reducing poverty. One of the challenges associated with SDGs is to assess how much and how the targets are impacted. The Life Cycle Sustainability Assessment (LCSA) approach has the potential to identify critical aspects of the SDGs related to products and their respective life cycles, as it provides a holistic framework comprehending their environmental, social, and economic impacts. This paper aims to develop a framework for evaluating the life cycle sustainability impacts of Lithium-Ion Batteries (LIBs), based on the integration of LCSA and SDG. The study applied a multimethodology approach comprehending Systematic Literature Review (SLR), Decision Sciences, and Expert consultation. The main results present a complete framework of recommended LCSA impact categories and indicators for the life cycle of LIBs, based on the SDGs and the SLR. The proposed framework can support productive and end-of-life sectors within the LIBs life cycles to become more sustainable and help in the fulfillment of the SDG.

1 INTRODUCTION

Lithium-Ion battery (LIB) is a product with increasing attention due to its role in the transition to renewable energy and transport electrification, key initiatives in decarbonization and mitigation of climate change (Christensen et al., 2021; Sommerville et al., 2021). LIBs largely supply energy to electrical and electronic equipment (EEE), and to electric vehicles (Harper et al., 2019). Despite the importance of LIBs for Sustainable Development (SD), their increasing demand leads to challenges for environmental, social and economic sustainability (Barke et al., 2021; Rajaeifar et al., 2022), as in the depletion of metallic resources (Liu et al., 2022). The extraction

of raw materials and production of LIBs use large amounts of resources, harming the environment and human health (Bielewski et al., 2021). LIBs supply chains involve social problems such as poverty, disrespect for human rights, child labor, and health and safety risks (Amnesty International, 2017; Harper et al., 2019; Thies et al., 2019).

Improving LIBs sustainability by minimizing environmental and social footprints is critical (Baars et al., 2020; Bielewski et al., 2021; Nature Materials, 2022). Several studies have assessed the life cycle environmental impacts of LIBs (e.g., Gaines & Dunn, 2014, Cusenza et al., 2019), with little attention to social and economic life cycle impacts (Thies et al., 2019; Barke et al., 2021). LIBs sustainability requires considering environmental, social and economic aspects of all stages of supply chain in a life cycle approach integrated with the Sustainable Development Goals (SDGs) (Bielewski et al., 2021; Nature Materials, 2022).

In 2015, the UN introduced the seventeen SDGs, which are presented in a global indicators framework; each goal includes a list of targets that are measured using 230 indicators (United Nations, 2015). The success of the SDGs depends on coordinated and complementary actions between governments, companies, civil society, and science (Sachs et al., 2019). Some stakeholders do not have a shared understanding of how to operationalize the SDGs (Spaiser et al., 2016; Caiado et al., 2018), a serious problem that compromises SD (Subramaniam et al., 2020). One reason is that the SDGs are conceived at the country level, making national governments responsible for the actions (Montiel et al., 2021). This hinders the planning and execution of actions by other entities in society, like companies, that have an important role in promoting development, but whose role in the SDGs is unclear (Redman, 2018; Mio, Panfilo & Blundo, 2020; Villiers et al., 2021). The subjectivity, complexity and conflicting interests toward SD limit the integration of SDGs within business strategies (Kuhnen & Hahn, 2017; Sullivan, Thomas &

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Rosano, 2018). The SDG framework is criticized for not providing practical business directions, as the broad scope inhibits concrete action (Mio, Panfilo & Blundo, 2020; Montiel et al., 2021). Making the SDGs relevant to companies is an emerging critical issue (UNGC, 2015; Redman, 2018). There is a knowledge gap in practical approaches for product level assessments of the SDGs by companies (UNEP, 2020; Montiel et al., 2021).

Assessing and managing sustainability considering the SDG framework requires a life cycle perspective and tools that avoid problem shifting from one country to another, or from one dimension of sustainability to another, such as the Life Cycle Assessment (LCA) (Laurent et al., 2019). LCA is a robust approach to evaluate environmental impacts from product systems (Souza et al., 2015; Valdivia et al., 2021), covering all life cycle stages, from raw material extraction to end-of-life (EoL) disposal (Wulf et al., 2019). LCA was initially designed to assess the potential environmental impacts by the life cycles, but developments (Kloepffer, 2008; Traverso et al., 2012, UNEP, 2011) have expanded the framework to include potential social and economic impacts, giving rise to Life Cycle Sustainability Assessment (LCSA). LCSA has the potential to identify the critical environmental, social and economic aspects of the SDGs and how each is impacted, pointing out how to address each aspect (UNEP 2020). Further developments are still needed for the practical application of life cycle approaches in the evaluation of SDGs, such as indicators adapted to the level of companies and products (Laurent et al., 2019).

Recent studies proposed linking the LCSA with the SDGs (Maier et al., 2016; Wulf et al., 2018; UNEP, 2020; Hannouf et al., 2022), using three main approaches: i) linking existing LCSA indicators and categories with SDG targets (e.g. Wulf et al., 2018; UNEP, 2020; Hannouf et al., 2022); ii) development of LCSA impact pathways based on the SDGs (e.g. Chandrakumar & McLaren, 2018; UNEP, 2020); and iii) screening SDGs targets to identify and prioritize

sustainability impact categories, not proposing an LCSA framework (Maier et al., 2016). The first approach is challenging as, with the SDGs covering sustainability at the macro level and LCSA at the micro level, there are many gaps in linking LCSA with SDGs, making it difficult to measure the extent to which a product system contributes to the SDGs in a life cycle perspective (Hannouf et al. 2022). The mapping of LCSA impact pathways by UNEP is in early stages of development (UNEP, 2020), still not presenting a framework with impact categories and their respective indicators. The third approach does not seek an LCSA framework based on the SDGs, but rather uses the SDGs to identify groups of priority sustainability themes within a life cycle. There are still gaps in developing SDG assessment based on life cycle approaches, with current efforts are focused on mapping the links of existing UNEP LCSA indicators with the SDGs, but not the other way around, i.e. to provide an SDG-based LCSA framework (Backes & Traverso, 2022).

None of these previous attempts to link SDGs to LCSA focused on the LIBs life cycle. This paper aims to develop a framework for assessing the life cycle sustainability of LIBs based on the integration between LCSA and the SDGs. A set of sustainability categories and indicators was defined based on the SDGs, literature review and expert consultation.

2 METHODS

The study adopted a multimethodology including a Systematic Literature Review (SLR), Problem Structuring Methods (PSMs) and experts consultation. Two different SLRs were carried out to acknowledge the state-of-the-art in the: i) integration of LCSA with SDGs, and ii) application of LCSA to LIBs. A PSM, cause-effect mapping, was adopted for being an adequate analytical approach for structuring complex problems (Rosenhead, 2016), as is the nature of sustainability decisions (Valdívia et al., 2021; Schaubroeck & Rugani, 2017). Causal maps help to portray complex problems, using an interactive approach to build a graphical representation that

characterizes the relationships between system elements, helping to identify strategic and operational objectives (Ackermann, 2012; Eden & Ackermann, 2018). PSMs are suitable to support decisions toward sustainability (Souza et al., 2015). In this study, causal mapping was used for structuring a causal hierarchy linking the identified LCSA impact categories and indicators to the SDGs.

The prioritization of SDGs targets, and further definition of impact categories and indicators, was conducted to meet the desirable properties for a coherent set of decision criteria, that also apply to LCA impact categories (Franco & Montibeller, 2010; Souza et al., 2015). These properties are:

1. Essential: considers all essential objectives;
2. Understandable: clear meaning to all members of the decision group;
3. Operational: the performance of decision alternatives is measurable for each of the essential objectives;
4. Non-redundant: not measuring the same concern twice;
5. Concise: the least number of criteria needed for the analysis of all essential goals;
6. Preferentially independent: the performance on an objective does not depend directly on other objectives (Franco & Montibeller, 2010; Souza et al., 2015).

The following methodological steps were taken:

i) Mapping the life cycle of LIBs: the entire LIBs product system, from cradle to grave, was mapped, identifying the materials, resources, and processes, as well as the relevant locations of each pre-use life cycle stage (extraction and production of LIBs components). A clear definition of system boundaries is necessary to judge which impact categories are relevant to the assessment

framework (Buchmayr et al., 2021). In this study, this step was based on SLR of peer-reviewed literature, searched in the Scopus and Web of Science databases (e.g. Olivetti et al., 2017; Mancini et al., 2021) and gray literature (e.g. U.S. Geological Survey; Joint Research Centre - European Commission). Data on the location of operations are essential for building life cycle inventories, mainly relating to social and economic issues, as living and working conditions are very different between countries (Havardi-Burger et al., 2021). The complete list of studies can be found in the supplementary material S1.

ii) Qualitative analysis of SDG targets related to the LIBs' life cycle: a thorough qualitative analysis was performed to verify which SDG targets are more directly related to the LIBs life cycle. This was done by analyzing each of the 150 targets from SDGs 1 to 16. SDG 17 was excluded, as this is considered an instrument for achieving the other SDGs (Singh et al., 2018; Weber & Weber, 2020). In a spreadsheet, each of the 150 targets was crosslinked to all stages of LIBs life cycle. The study adopted a visual scale (red, yellow, and green) reflecting how strong is the relation of each target to each life cycle stage (see supplementary material S2).

Initially, each researcher evaluated the associations individually. The inclusion or exclusion of each goal was justified using materiality based on the peer-reviewed literature or international reports (e.g. Amnesty International, International Labor Organization) containing information on the LIBs life cycle. Materiality assessment is widely used in sustainability reports (Calabrese et al., 2016), being important in identifying stakeholders and issues that relate to an organization's impacts (Farooq et al., 2021). It determines the relevance of an issue for an analyzed context (Beske et al., 2020). In this paper, a target was considered material when it had a strong ("green") or weak ("yellow") association with the LIBs life cycle, supported by the literature. For example, target 3.5 - Strengthen the prevention and treatment of substance abuse, including narcotic drug

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5 164 is justification in the literature, such as: “the presence of hundreds of miners and assistants was
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11 167 1.5) were strong associated because all stages of the LIBs' supply chain have significant potential
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17 169 To reduce individual bias, the group performed a consensual validation. Then the targets strongly
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23 172 property 1 (Essential), as all essential objectives for the sustainability assessment of the life cycle
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25 173 of LIBs are included in the framework.
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29 174 **iii) Clustering of sustainability themes:** selected SDG targets were grouped, redefined, or
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31 175 eliminated by clustering. Clustering finds and categorizes groups with similar themes in a dataset
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33 176 (Tchupo & Macht, 2022; Rocchi & Pescatore, 2022). It was used to group selected SDG targets
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35 177 by similarity. This step seeks to meet two desirable properties: 4 (Non-redundant) and 5 (Concise).
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37 178 For example, in this step targets 1.1, 1.2, 2.1, 10.1, and 15.3 were grouped in the cluster theme
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39 179 called Poverty, and target 1.3 was eliminated by redundancy. Grouping similar themes was used
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41 180 in previous sustainability assessment frameworks, representing topics (e.g., Voegeli et al., 2019),
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43 181 impact categories (e.g. Maier et al., 2016) or sustainability themes (e.g. Havardi-Burger et al.,
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45 182 2021).
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50 183 **iv) Causal mapping for analysis and definition of the LIBs LCSA impact categories:** after
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52 184 grouping the targets, causal mapping was used to analyze the hierarchical cause-effect
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54 185 relationships between the targets in each cluster. Each cause-effect relationship was derived from
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the SLR. The complete list of studies is in the supplementary material S3. Causal mapping allows the identification of broader objectives, whose intermediate steps are operational objectives, usually translated into performance indicators (Françoze & Belderrain, 2022). Based on the final map's cause-effect hierarchy, potential LCSA impact categories can be identified, according to their topological position as midpoint or endpoint effects (Souza et al., 2015). Causal mapping aids to structure LCSA impact pathways and identifying midpoint impact categories where indicators are more measurable (Souza et al., 2015). Causal approaches are considered adequate to holistically and comprehensively understand sustainability impacts, as well as to identify impact pathways (Voegeli et al., 2019).

v) Proposal of LCSA categories and indicators for LIBs: based on the causal map, a Vertical Diagram was organized consisting of core axes, expressing the strategic dimensions represented by the clusters and cause-effect streams. Based on the axes, respective impact categories were identified, and an LCSA structure for LIBs was developed. For each impact category, measurement indicators were adapted for the assessment of the LIBs life cycle. This step seeks to meet two other desirable properties: 2 (Understandable) and 3 (Operational).

vi) Review and validation by experts of the proposed framework: external validation of the developed framework was carried out with a consultation of experts in the areas of LCA, LCSA and sustainability consulting from different countries. The experts' contributions were collected through an online survey, disseminated through professional networks and contacted by e-mail addresses identified in relevant papers. The experts evaluated the adequacy of the proposed impact categories in the context of the LIBs life cycle, according to the six desirable properties for coherent decision criteria (Franco & Montibeller, 2010), using a 5-point Likert scale. The survey

form is available in the supplementary material S11. The results of the expert consultation were used to refine the proposed framework.

vii) SLR 1 (LCSA/LCA + SDGs): following the feedback from the experts, an SLR was conducted to map the literature that interrelates LCSA/LCA with the SDGs. This SLR was conducted to enrich the cause-effect map based on the SDGs, to guarantee that relevant impact categories were not ignored, as well as to detail the linking between LCSA and the SDGs. The search strategy used the following term, in Web of Science and Scopus: ("life cycle sustainability assessment" OR "life cycle assessment" AND "sustainable development goals").

The initial search obtained 344 documents (174 from Scopus; 170 from Web of Science). The study further filtered: i) only documents of the type “original article”; ii) articles written in English. After this screening, and by eliminating duplicity, the SLR obtained 156 papers. Next, a screening of the titles and abstracts helped to select articles that crossed the LCSA or LCA themes with the SDGs. Articles that did not provide such integration were excluded stage, for example, when the article performed an LCA but did not use these results to assess the contribution to the SDGs, or did not present an interconnection of the LCA or LCSA categories with the SDGs targets (e.g. Colangelo et al., 2021; Sulis et al., 2021). As a result, 78 articles were selected as the final sample for content analysis. These articles (see supplemental material S4) were thoroughly studied to provide detailed knowledge.

Besides bibliometrics data, the content analysis looked for:

- What is the type of product, process, or waste analyzed by LCA/LCSA;
- What type of LCA was conducted: environmental LCA, social LCA, economic LCA, or LCSA;

- How have these techniques been used to measure progress towards the SDGs, and which SDGs have been associated with each type of LCA;
- Identification of impact categories assessed: this allowed to assess whether the proposed structure satisfied sustainability impact categories from the literature.

viii) SLR 2 (LCSA/LCA + LIBs): another SLR was conducted to map the literature that applies LCSA/LCA to the LIBs life cycle, aiming to enrich the cause-effect map based on the SDGs, and to guarantee that possible impact categories were not being ignored. The search strategy used the following term, in Web of Science and Scopus: ("libs" OR "lithium-ion batter*" or "lithium-ion batter*") AND ("sustainable assessment" or "sustainability assessment" or "social life cycle assessment" or "social assessment" or "social sustainability" or "life cycle sustainability assessment"). The search identified only 19 publications for content analysis (see supplemental material S5).

In content analysis, a spreadsheet was developed to extract bibliometrics data, and more specific data such as:

- What type of LCA was conducted; and Identification of impact categories assessed (as in step vii).

ix) Refinement of the impact categories: compilation of expert responses and impact categories found in the two SLRs served as a basis for enrichment of the categories in the proposed sustainability assessment framework, allowing to include categories that were absent in the first proposal (from step v). For the full list of impact categories compiled from each study and the refinements made to the map, see the supplementary materials S4, S5 and S6.

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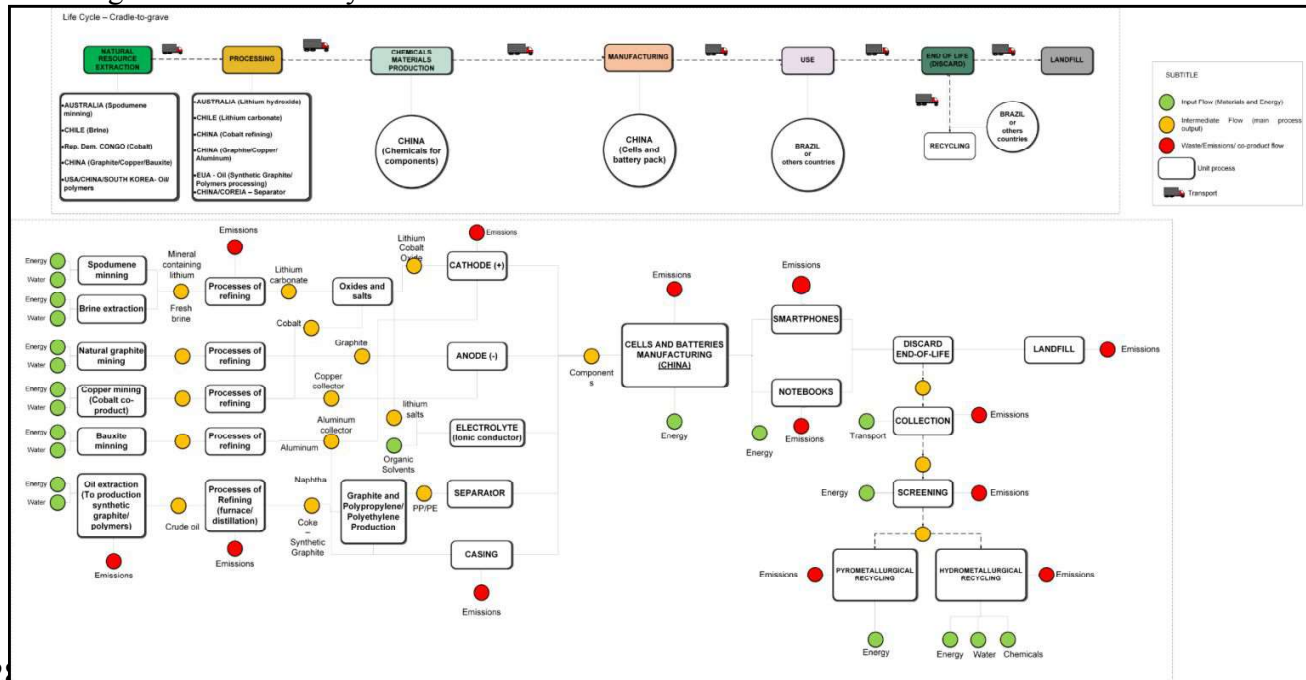
x) Proposition of the LCSA impact pathway framework for LIBs: this step proposes an LCSA structure for the LIBs based on the interpretation of the SDGs to the product level considering the entire life cycle of LIBs; proposes a diagram of the cause-and-effect relationships (impact pathways) integrating the SDGs targets to the final Areas of Protection (AoP), similarly to Souza et al. (2015). The core axes of the structure relate to endpoint categories, whilst the more operational aspects are considered midpoint categories. This step seeks to meet the desirable property 6 (Preferentially independent). All cause-and-effect streams are qualitatively based on the SLRs in multidisciplinary areas. To view the adopted literature, see supplementary material S7.

4 RESULTS AND DISCUSSION

4.1 LIBs life cycle mapping

The mapping of the product system makes it possible to identify Bill of Materials and geographic locations where each stage of the LIBs life cycle LIBs occurs. Figure 1 shows the life cycle system of LIBs from cradle to grave.

Figure 1 - LIBs life cycle.



LIBs are composed of five components (Goodenough, 2018): cathode (positive electrode), anode (negative electrode), electrolyte (ionic conductor), separator and housing. Usually, the cathode is made of lithium cobalt oxide (LiCoO₂) or Nickel-Manganese-Cobalt oxides (NMC), with an aluminum foil as a collector (Swart et al., 2014). The anode is made of graphite and a copper sheet as a collector (Zhang et al., 2014). The electrolyte is a mixture of lithium salt and organic solvents (Fahimi et al., 2022). The separator is usually a membrane made of a porous plastic film (Mossali et al., 2020). Finally, the housing is nickel-plated steel (Crenna et al., 2021). It is observed in Figure 1 that the production of LIBs involves several production chains, namely: mining of metals and carbonaceous materials, extraction of petroleum and its derivatives, chemical industry in refining and production of materials of each component, and assembly of LIBs. As the focus of this study is on LIBs used in consumer electronics, the assembly of these products is included in the system boundaries. The end-of-life processes involve discarding, collection, sorting, recycling and landfilling. Despite the variability and uncertainties associated with the use

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3 299 phase, this was also included in the analysis, as at this stage there is potential for positive and
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5 300 negative impacts from the use of the final products. The use of smartphones and notebooks can
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7 301 favor, on one hand, access to education and foster digital inclusion (Asongu et al., 2019); on the
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9 302 other hand, it can cause impacts related to energy consumption for charging LIBs (Morley et al.,
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14 304 Due to the amount of different materials involved in the production of LIBs, production is
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16 305 also geographically dispersed, involving different countries at each stage of the life cycle (see
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18 306 countries of each stage in Figure 1). For example, the raw material extraction may involve five or
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20 307 more different countries, with Australia and Chile extracting lithium from rocks or brines, the
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22 308 Democratic Republic of Congo (DRC) mining Cobalt, China extracting natural graphite and
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24 309 copper, and the United States and South Korea supplying petroleum and polymers. China is the
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26 310 country that dominates global LIBs production, controlling subsidiaries in the DRC to extract
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28 311 cobalt, refine chemicals and assemble LIBs components and cells (Olivetti et al., 2017; Chen et
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30 312 al., 2020).

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33 313 The mapping results of the LIBs life cycle show that the production of LIBs has the
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35 314 potential to cause several environmental, social and economic impacts (as stated by Harper et al.,
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37 315 2019; Rajaeifar et al., 2022). Current technology targets several critical materials, such as lithium,
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39 316 cobalt and graphite, that are associated with different impacts in their supply chains (Mohr et al.,
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41 317 2020; Thies et al., 2019).

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45 319 **4.2 Goals and targets associated with the LIBs life cycle**

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48 320 The result of the analysis of SDGs targets associated with the LIBs life cycle is summarized
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50 321 in Table 1. Supplementary material S2 presents the full spreadsheet with the colored association

scale and references used to identify whether each target was directly associated or not with the stages of the life cycle. According to this approach, the 95 targets marked with green or yellow colors (in Table 1 and S2) were selected. The 55 targets marked in red were excluded as they were not strongly associated with the life cycle of LIBs.

The 95 targets associated with the life cycle of LIBs are distributed across all 16 SDGs, which suggests that the context of the life cycle of LIBs has the potential to impact the achievement of all relevant aspects of SD. Some objectives stood out in the selection of associated targets (in terms of % of respective targets associated), namely: SDG 1 (71% of the targets were associated), SDG 3 (46% of the targets), SDG 6 (87,5%), SDG 7 (100%), SDG 8 (100%), SDG 9 (100%), SDG 13 (80%), SDG 15 (67%), and SDG 16 (67%). This result can be explained by the characteristics of the materials, extraction and processing sites of raw materials for LIBs. The extraction, processing and assembly of LIBs components and cells are mostly located in underdeveloped countries, such as the DRC, as well as in developing countries, such as China and Chile, which present major social, environmental and economic challenges (Sachs, 2012; Omisore, 2018). Issues such as poverty and inclusion (SDG 1, SDG 8), lack of access to adequate health services (SDG 3), lack of access to safe water and sanitation (SDG 6), energy (SDG 7), disregard for human rights and labor (SDG 8) and environmental problems caused by obsolete and highly polluting industrial processes (SDG 9) are reported in the life cycle of LIBs. The selected targets on climate change (SDG 13) and land use (SDG 15) are directly related to the potential environmental impacts of mining and intensive resource use in these countries, which cause environmental degradation and exacerbate social problems (Kang et al., 2013; Nkulu, 2018; Lebre et al., 2020). The targets of SDG 16 are equally relevant for assessing sustainability of the LIBs. Concerns about corruption, lack of transparency, presence of armed groups, social tensions and conflicts over resources are

problems recognized in the literature, not only in the poorest countries such as the DRC but in all countries in the LIBs life cycle (IISD, 2018; Thies et al., 2019).

Table 1 - Association of SDGs targets with LIBs life cycle.

Goals	SDG1	SDG2	SDG3	SDG4	SDG5	SDG6	SDG7	SDG8	SDG9	SDG10	SDG11	SDG12	SDG13	SDG14	SDG15	SDG16
TARGETS	1.1.	2.1	3.1.	4.1	5.1	6.1	7.1	8.1	9.1	10.1	11.1	12.1	13.1	14.1	15.1	16.1
	1.2.	2.2.	3.2	4.2	5.2	6.2	7.2	8.2	9.2	10.2	11.2	12.2	13.2	14.2	15.2	16.2
	1.3.	2.3.	3.3	4.3	5.3	6.3	7.3	8.3	9.3	10.3	11.3	12.3	13.3	14.3	15.3	16.3
	1.4.	2.4.	3.4	4.4	5.4	6.4	A	8.4	9.4	10.4	11.4	12.4	A	14.4	15.4	16.4
	1.5	2.5	3.5	4.5	5.5	6.5	B	8.5	9.5	10.5	11.5	12.5	B	14.5	15.5	16.5
	A	A	3.6	4.6	5.6	6.6		8.6	A	10.6	11.6	12.6		14.6	15.6	16.6
	B	B	3.7	4.7	A	A		8.7	B	10.7	11.7	12.7		14.7	15.7	16.7
		C	3.8	A	B	B		8.8	C	A	A	12.8		A	15.8	16.8
			3.9	B	C			8.9		B	B	A		B	15.9	16.9
			A	C				8.10		C	C	B		C	A	16.10
			B					A				C			B	A
			C					B							C	B
			D													

In contrast, the 55 excluded targets were predominantly targets with a clear national or international decision scope, where decisions made in the LIBs life cycle could not directly affect, positively or negatively, the achievement of the targets. Seven SDGs stand out in the analysis of excluded targets (SDG 2, SDG 3, SDG 4, SDG 10, SDG 11, SDG 12 and SDG 14). The five targets excluded in SDG 2 were related to policy decisions on seed banks and the agri-food chain, clearly not linked to decisions in the life cycle of LIBs. In SDG 3, the excluded targets relate to public

357 policies for the adoption of international conventions, access to medicines and prevention of road
358 accidents. In SDG 4, the excluded targets were related to public policies for access to education.
359 In SDG 10, the excluded targets were related to the elimination or enactment of laws, public
360 migration policies and adherence to agreements of international bodies such as the World Trade
361 Organization, in which companies with a product system clearly do not have decision-making
362 power. In SDG 11, the excluded targets are related to public policies aimed at infrastructure, for
363 example, Target 11.2 which deals with the access and expansion of safe, accessible and sustainable
364 public transport. Targets excluded from SDG 12 concern national-level public policies for
365 sustainable production and consumption, public procurement, food losses, and sustainable tourism.
366 In SDG 14 the excluded targets are related to illegal fishing, fishing subsidies and economic
367 benefits at the national decision-making level. This result is in agreement with previous studies
368 that recognized the macro dimension of some SDG targets as one of the main challenges when
369 integrating these targets at the micro level of companies and products (UNEP, 2020; Mio, Panfilo
370 & Blundo, 2020; Montiel et al. al., 2021; Hannouf et al., 2022). Furthermore, as the SDG targets
371 cover multidisciplinary themes, involving several different economic sectors (Tremblay et al.,
372 2020), some targets may naturally be out of scope when analyzing the context of a specific
373 industrial sector or product system.

374 According to the methodology applied in this study, the linking of targets with a national
375 scope should not be associated with the product level, because, even if the companies and
376 stakeholders within a product's life cycle have some role in these issues, for example, facilitating
377 the access to basic health services for the local community or influencing public decision-makers,
378 the macro dimension makes it impossible to measure the impact of a single product system on the
379 target, as there is no direct relationship that can be audited. Themes that are very distant from the

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3 380 product and its life cycle are also not suitable to be selected, for example, targets related to agri-
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5 381 food systems or tourism in the case of the life cycle of LIBs. However, from this analysis, another
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7 382 issue emerges: the lack of knowledge about the interrelationship of the different production
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9 383 systems, the trade-offs and synergies between production systems in the same region or country
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11 384 and how they affect the scope of the SDGs together. Life Cycle Thinking applied in a consequential
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13 385 (CLCA) approach to analyze decisions in systems can provide this in-depth insight (Ekvall et al.,
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15 386 2016; Saraiva et al., 2018).

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17 387 Screening targets considering the specific life cycle context is an important step in
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19 388 assessing sustainability based on the SDGs, as different contexts may have different relevant
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21 389 themes as a focus (Souza et al., 2015). For example, for the LIBs product system, targets on
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23 390 Education were considered important to assess the sustainability, whilst for the life cycle of urban
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25 391 surfaces, targets related to early childhood education were not considered relevant, as they were
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27 392 not directly associated with the decision context involved (Henzler et al., 2020).

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29 393 The results of this stage highlight that the mapping of relevant themes from broad issues,
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31 394 such as SDGs, can help to identify aspects that are not always linked to product systems. For
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33 395 example, the degradation of food security in Chinese districts due to pollution of graphite factories,
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35 396 or health problems directly related to alcohol and drug abuse in mining areas, are issues that
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37 397 reinforce or exacerbate existing social problems. In order not to “leave anyone behind” as
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39 398 advocated by the SDGs, supply chains and the LCSA must incorporate mechanisms to assess these
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41 399 issues. It is not an easy task, but recognizing the main issues where the life cycle may be most
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43 400 strongly related is the first step (Almanza & Corona, 2020).

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4.3 Clustering and renaming targets

The selected targets were grouped by similar themes, renamed, or eliminated for remove redundancy, considering the specific context of the life cycle of LIBs (see Table 2). For example, targets 1.1, 1.2, 2.1, 10.1 and 15.3 were grouped in the cluster called Poverty and targets 1.3 and 1.4 were excluded. The cluster called Education grouped all targets involved in any kind of education, including basic (targets 4.1 and 4.2), higher (4.4 and 4.b) and environmental (6.a, 6.b, 12.6, 13.3 and 13.b) education. The targets 7.2 and 7.a were grouped in cluster called Energy and targets 7.1, 7.3 and 7.b were excluded due to redundancy, as they dealt with the same topics already included in the two selected targets (renewable energy, energy efficiency and access to energy). Targets that can be measured with environmental LCA were grouped in cluster 16 called Environmental LCA (targets 6.3 / 6.4 / 6.6 / 8.4 / 12.2 / 12.4 / 14.1 / 14.2 / 14.3 / 15.1 / 15.2 / 15.4 / 15.5). The grouping done for the environmental LCA categories is similar to the environmental categories suggested previously to measure SDGs 6, 8, 12, 14, 15 (Maier et al., 2016). The same rationale was applied to all targets.

Table 2. Grouped targets by an equal or similar theme.

Cluster number	Name	Grouped targets	Excluded targets
1	Poverty	1.1 / 1.2/2.1/ 10.1 /15.3	1.3/ 1.4/1.5/ 2.3 / 2.4 /5.a / 5.b/ 7.1/ 7.3/ 7.b / 8.1/ 8.2 / 8.9/ 8.10/ 9.a / 9.c/ 11.6 / 12. 5 / 12. a / 14.5/ 14.c
2	Education	4.1 / 4.2 / 4.4 / 4.b 6.a / 6.b / 12.6 / 13.3 / 13.b	
3	Violence	5.2 / 16.1 / 16.2	
4	Housing	6.1 / 6.2 / 11.1 / 11.b	
5	Energy	7.2 / 7.a	
6	Youngs	8.6 / 8.b	
7	Rights	5.1 / 5.2 / 5.5 / 10.2 / 16.5 / 16.6 / 16.7 / 16.10	
8	Health and Food	2.1 / 3.1 / 3.2 / 3.3 / 3.4 / 3.5 / 3.9	

9	Economic/Job	8.3 / 8.5 / 8.6 / 8.7 / 8.8 / 9.2 / 9.3 / 15.9 / 15.a / 15.b
10	Infrastructure	9.1 / 9.4
11	R&D investment	9.5 / 9.b
12	Environmental policy	12.6 / 13.2 / 13.3 / 13a
13	Women's	5.1 / 5.5
14	International cooperation	8.a / 16a
15	Compliance	16.10 / 16b
16	Environmental LCA	6.3 / 6.4 / 6.6 / 8.4 / 12.2 / 12.4 / 14.1 / 14.2 / 14.3 / 15.1 / 15.2 / 15.4 / 15.5

4.4 Causal mapping

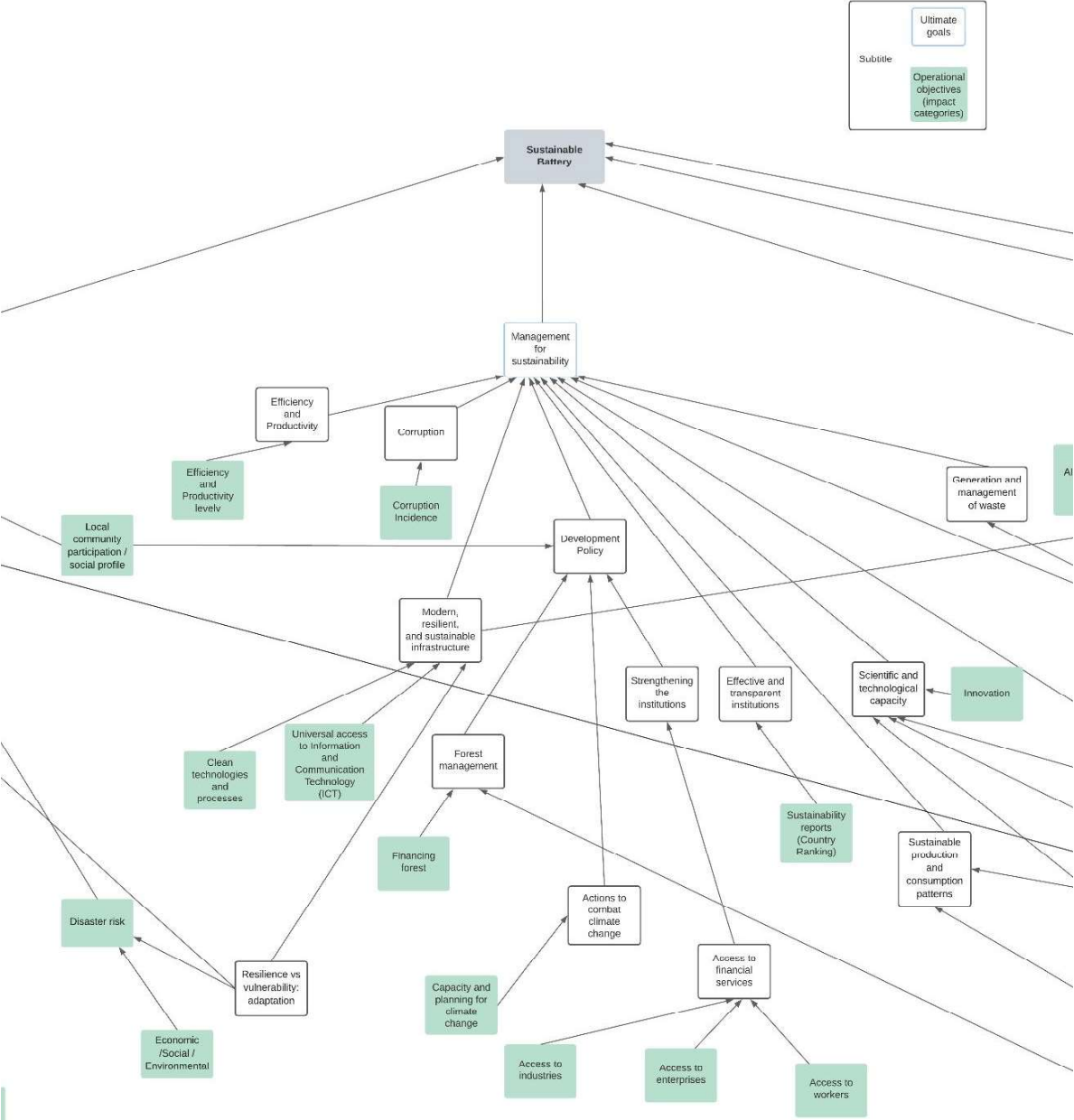
After screening for grouping and elimination of redundant targets, causal mapping was used to analyze the hierarchical cause-effect relationships between the SDG targets selected. Figure 2 exemplifies the construction of the causal map, presenting a part of it to identify the hierarchical relationship of the final objective (endpoint) Management for Sustainability. The complete causal map developed is shown in Figure S8 in the supplementary material.

The final objective Management for sustainability was considered one of the central axes of sustainability assessment for the life cycle of LIBs, considered, in the causal relationship, an effect or final objective in the sustainability structure to make LIBs more sustainable. For this axis to be evaluated, potential main impact categories were identified (midpoints - potential causes for the effect of sustainability management): efficiency and productivity, resilient and sustainable infrastructure, corruption, scientific and technological capacity, development policies, transparent institutions and effective generation and management of chemicals. Following the logic, each of these categories reflects broad themes that need other sub-categories (sub-level 1 and sub-level 2)

to be assessed. For example, Scientific and Technological Capacity is considered a cause, or an important requirement, to generate an effect of management for sustainability (Olawumi & Chan, 2018; Centobelli et al., 2022). Public investment, innovation and the number of R&D workers are factors that affect the scientific and technological capacity (UNESCO, 2020). These subcategories, public spending on R&D, Innovation and R&D workers, are different causes to determine the effect on the scientific and technological capacity of the life cycle of LIBs. The same logic was followed throughout the map. Based on the final map cause-effect hierarchy, potential impact categories can be identified, according to their topological position as midpoint or endpoint effects (Souza et al., 2015). Causal mapping served to identify the midpoint impact categories where indicators can be measured.

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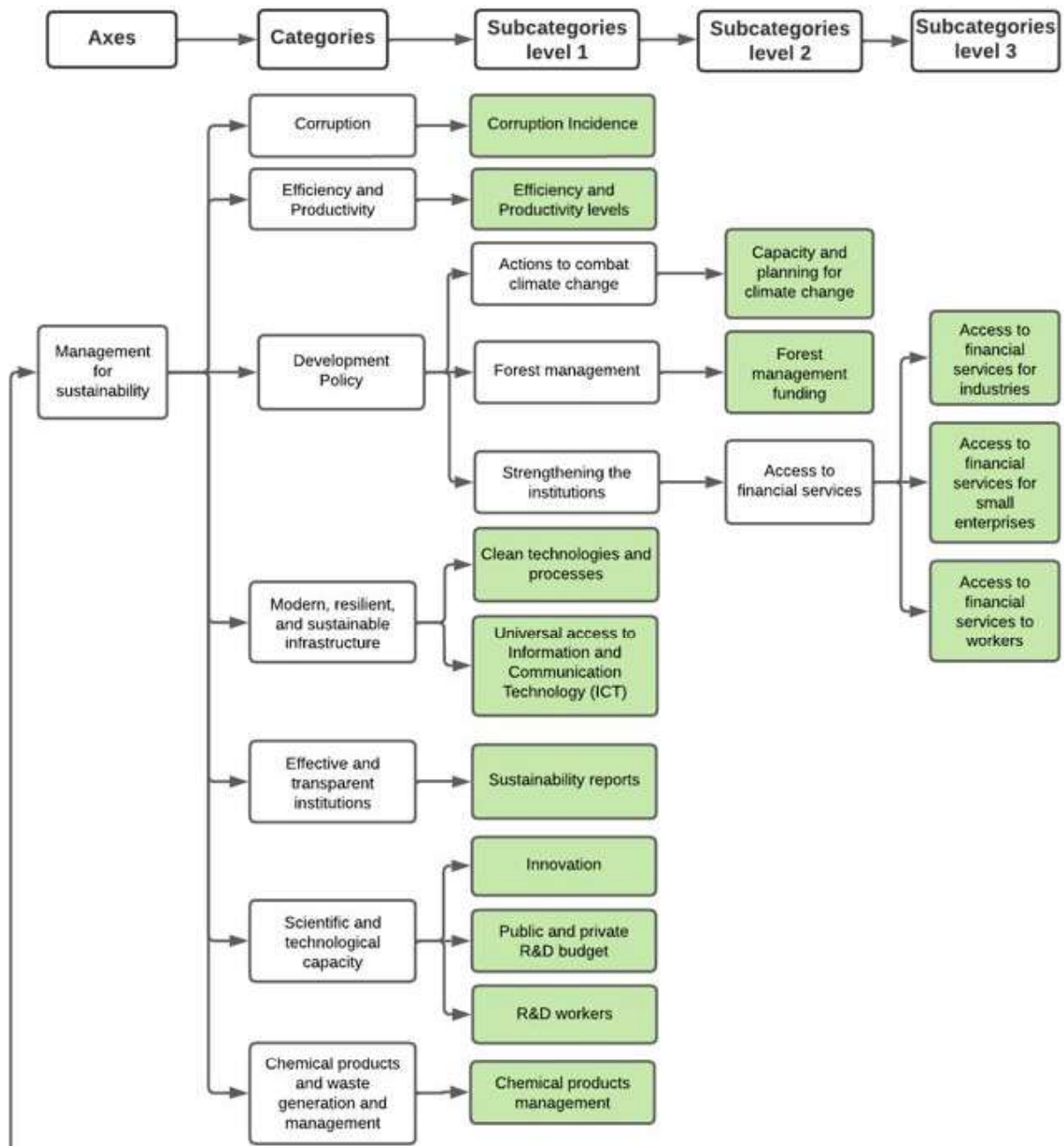
Figure 2 - Part of the causal map built to identify the hierarchical relationship of the final objective
Management for Sustainability.



4.5 Framework for sustainability assessment

Based on the complete causal map, a Vertical Diagram (Figure 3) presents the reinterpretation of the SDGs targets in the context of the LIBs life cycle, structured with five axes detailed in Figure 3: 1. Management for Sustainability, 2. Citizenship and Inclusion, 3. Health, 4. Economic Development and 5. Environment/Environmental impacts, which are considered the final objectives (endpoints) for LIBs sustainability. For each axis, the impact categories and subcategories (midpoints, operational objectives) are marked in green. Unlike other frameworks (Maier et al., 2016; Wang et al., 2018; Henzler et al., 2020; UNEP, 2020) we do not link the axes and categories to the three commonly used dimensions of sustainable development (environmental, social and economic), as these five assessment axes address topics that are permeated by the triple bottom line (i.e., social, economic and environmental dimensions are transversal through the axes). Due to the complexity of the interconnections that involve sustainable development, separating the categories by environmental (biophysical) and social dimensions may not be appropriate, since the latter also covers the economic, technological and institutional spheres (Voegeli et al., 2019).

Figure 3 - Part of Diagram with LCSA axes, categories and subcategories for LIBs. For a complete view of the framework, see the figure included separately in high resolution or pdf format in supplementary material S6.



The proposed LCSA impact assessment framework presents a comprehensive set of impact categories address a wide range of topics for sustainability. This is due to the broad character of

the SDGs. The framework includes the categories already defined in UNEP LCSA guidelines, such as “Human Rights” (gender equality, child and forced labor, right to land), “Education”, “Resources use and efficiency” and “environmental impacts”, but goes further to include sustainability issues such as “access to basic and financial services”, “access to technology”, “disaster risks”, “sustainable transport”, “forest management” and “food security” within the life cycle of LIBs, which are issues that affect all stakeholders involved in the life cycle. This result confirms previous findings that several social issues relevant to the SDGs are not fully covered by the currently developed S-LCA and LCSA frameworks (Almanza & Corona, 2020). There are many challenges to assessing social issues in life cycle thinking. The S-LCA itself is still a young and immature method, which makes it a great challenge to interpret the results of internationally agreed social goals such as the SDGs (Pollok et al., 2021).

There are frameworks developed to assess the sustainability of diverse product systems (Chandrakumar & McLaren, 2018), as well as in the life cycle of LIBs (Mancini & Sala, 2018; Gebhardt et al., 2022). However, all studies found present assessments of existing categories that are widely used in LCA, S-LCA, LCC and LCSA structures proposed by UNEP. Further analysis of the SDGs should be carried out for different types of products (Almanza & Corona, 2020).

The main innovation of the proposed assessment framework is the inclusion of sustainability themes that are specifically relevant to the life cycle of LIBs, considering a broader perspective of sustainability and life cycle stages, which is precisely provided by the SDGs targets. The proposed structure can serve to evaluate the sustainability performance of current LIBs technologies and new developments, for example, the replacement of the cell with chemical composition NMC111 for NMC811, the replacement of cobalt by another type of raw material, or supply from another country.

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4.6 Indicators framework for sustainability impact categories

After defining the thematic axes (final objectives - endpoint categories), and the operational subcategories (midpoints) for the LIBs, a complete set of indicators was proposed for each operational subcategory. Table 4 presents a part of the indicators built for the Management for Sustainability axis. The complete table of indicators for all operational axes and categories is available in Supplementary Material S10.

Adapting the indicators to the life cycle of LIBs helps in measuring impacts, as it makes visible how the indicator can be evaluated at the product level. For example, the indicator for the Financing of forest management category (“Amount (in dollars) invested to finance forest management by stage of the life cycle, per year”); the indicator for the Access to financial services category (“Proportion of workers with access to financial services by life cycle stage, per year”). These are examples of how a macro-level indicator can be adapted to the product level. Some indicators need assessment both at the country and life cycle levels, as for the R&D workers category (“Researchers by skill level per million inhabitants per country/by life cycle stage.year”) and (“Researchers by occupation per million inhabitants by country/by life cycle stage.year”). This is necessary due to data constraints, as sometimes R&D data are found at national levels and can express the importance of the industry within a country (OECD, 2022).

Other indicators are relatively easier to adapt to the product level, such as resource use in Efficiency and Productivity level category (“Efficiency rate in the LIBs life cycle/year”), waste management in Chemical Waste Generation and Management category (“Proportion of waste treated by life cycle stage/year”). More challenging is adapting social data to the life cycle context, as those on poverty and inclusion, for example, are aggregated at the country level, making it difficult to directly link to industrial activity. Particularly in multifaceted issues such as poverty,

defining how much a product system contributes to reducing or increasing poverty in a given location is not easy – it involves investigating many interrelated factors (Hall, 2021).

Table 4 - Indicators for the Management for Sustainability axis in the LIBs life cycle.

SUSTAINABLE BATTERY					Indicators
			Legend	Operational subcategories (midpoint)	
AXES	CATEGORIES	SUBCATEGORIES LEVEL 1	SUBCATEGORIES LEVEL 2	SUBCATEGORIES LEVEL 3	
MANAGEMENT FOR SUSTAINABILITY	CORRUPTION	CORRUPTION INCIDENCE			-Number of corruption cases reported and convicted by life cycle stage in the last year.
	EFFICIENCY AND PRODUCTIVITY	EFFICIENCY AND PRODUCTIVITY LEVEL			-Efficiency rate in the life cycle of the battery in the last year.
					-Productivity rate in the life cycle of the batteries in the last year;
	DEVELOPMENT POLICY	ACTIONS TO COMBAT CLIMATE CHANGE	CAPACITY AND PLANNING FOR CLIMATE CHANGE		-Number of projects aimed at increasing resilience to climate change in the life cycle in the last year.
					-Investment in dollars in actions to reduce climate change by country in the last year.
		FOREST MANAGEMENT	FINANCING FOREST MANAGEMENT		-Amount (in dollars invested) to finance the forest management by stage in the life cycle, in the last year.
					-Forest area by life cycle stage in the last year.
		STRENGTHENING THE INSTITUTIONS	SUPPORT FOR SMALL AND MEDIUM-SIZED ENTERPRISES		-Proportion of Small and Medium Enterprises with loans or credit lines by life cycle stage, by country, in the last year.
			ACCESS TO FINANCIAL SERVICES	ACCESS TO FINANCIAL SERVICES FOR INDUSTRIES	-Proportion of industries with access to financial services by life cycle stage, in the last year.

				ACCESS TO FINANCIAL SERVICES FOR SMALL ENTERPRISES	-Proportion of small enterprises with access to financial services by country in the last year.
				ACCESS TO FINANCIAL SERVICES WORKERS	-Proportion of workers with access to financial services by life cycle stage, in the last year.
	MODERN, RESILIENT, AND SUSTAINABLE INFRASTRUCTURE	CLEAN TECHNOLOGIES AND PROCESSES			-Amount of emissions released to air, water and soil by process and technology by life cycle stage in the last year.
		UNIVERSAL ACCESS TO INFORMATION AND COMMUNICATION TECHNOLOGY (ICT)			-Proportion of workers by sex, age, race, with access to ICT (and internet) in the battery life cycle, in the last year.
	EFFECTIVE AND TRANSPARENT INSTITUTIONS	SUSTAINABILITY REPORTS			-Number of companies that prepare transparency reports (e.g. GRI, Compliance) by stage of the battery life cycle in the last year.
	SCIENTIFIC AND TECHNOLOGICAL CAPACITY	INNOVATION			-Number of patents generated by battery life cycle stage in the last year.
					-Number of patents sold by stage of the battery life cycle in the last year.
		PUBLIC AND PRIVATE R&D SPENDING			-Total expenditure on R&D in proportion to Gross Domestic Product (GDP) by country in the last year.
					-Total expenditure on R&D in proportion to gross profit per life cycle stage, in the last year.
		R&D WORKERS			-Researchers by skill level per million inhabitants per country/by life cycle stage in the last year.
					-Researchers by occupation per million inhabitants by country/by life cycle stage in the last year.

	CHEMICAL WASTE GENERATION AND MANAGEMENT	CHEMICALS PRODUCT MANAGEMENT			-Amount of chemical products used by life cycle stage in the last year.
					-Proportion of waste treated by life cycle stage in the last year.

4.7 Revision and validation of the proposed framework by experts

The sustainability assessment framework for LIBs were assessed by a group of experts of different knowledge areas related to the life cycle of LIBs, LCSA and LCA. In total, 17 experts replied to the online survey. The experts are from nine different countries: Brazil (7 experts; 41%), Italy (1; 6%), India (1; 6%), Germany (1; 6%), Austria (1; 6%), France (2; 12%), Canada (1; 6%), Spain (2; 12%) and Turkey (1; 6%). 41% of the specialists are professors or researchers at universities, 17% are sustainability and management consultants, 12% are actors in some of the stages of the LIBs supply chain, 6% are professionals in the development of LCA software, 6% are of the environmental area in the productive sector, 6% are in the transport sector and 12% are specialists in the third sector and public administration.

In evaluating the sets of criteria, the experts judged the adequacy of the framework using a 5-point Likert scale evaluating each of the six criteria defined in section 2. The experts could also contribute with open comments. For the first five criteria, the mode of the experts' evaluation was 4, that is, the set of impact categories represents essential sustainability objectives for the life cycle of LIBs, is understandable, operational, has minimal redundancy and is as reduced as possible. However, the last criterion, which assesses the preferential absence of cause-and-effect relationships amongst categories, had the worst evaluation (mode = 2), indicating some cause-and-effect relationships. Although the experts did not provide in-depth justification of such cause-effect

links, two experts noted that due to the lack of complete knowledge about the interrelationships in the mechanisms in the LCA and the broad themes covered by the SDGs, can occur relationships of the cause and effect between the categories. For example, see these quotes from two experts:

Expert 1: *“In relation to my topic (LCA) the effects are still partially identified/correlated in some LCA impact categories”.*

Expert 2: *“I did not do a deep analysis on this, but, considering the breadth of some aspects, such as ‘Implementation of sustainable production and consumption patterns’ and ‘Strengthening of institutions (Small and Medium Enterprises and Institutions of Financial Services)’ , I imagine that there will be cause and effect relationships between the listed aspects”.*

The fact that the categories in the structure present some relationship dependence is neither a surprise nor an impediment to the use of the structure, as this is a phenomenon already recognized in LCA. Environmental impact categories do not show absolute independence, as the same environmental mechanism can impact different midpoint categories, and the same midpoint category contributes to damages in several endpoint categories (Huijbregts et al., 2016). Environmental, social and economic impacts are characterized by uncertainties regarding possible effects, as they are multidimensional phenomena that are almost always interrelated (Weber & Weber, 2020).

Regarding the non-redundancy criterion, one of the experts made the following comment:
Expert 7: *“There may be the possibility of complementation. In Clean energy x Climate change, the Fossil Carbon indicator can classify Clean Energy level”.*

The experts too suggested the inclusion of categories or a change in terminology for the categories. For example, for Expert 3: *“I would call “energy consumption”=“Primary Energy Demand”, because I think it is not about electricity consumption, but about the consumption of*

energy carriers to produce the manufacturing energy...". For Expert 4: "Access to transport is missing- renewable/low Carbon energy as well". Expert 5: "I would add: 1) Categories /Development Policy/ Sublevel 1 (Regulation for the protection of biodiversity and for ecosystem services) 2) Categories / Modern, resilient and sustainable infrastructure / Sublevel 2 (gradual removal of subsidies for sources of fossil resources and implementation of carbon taxation). 3) Categories / Scientific and Technology capacity / sublevel 2 (structuring of intellectual property rights and incentives for biotechnology-based companies)". Expert 6: "Examples of what I missed: Traditional peoples - impact mineral extraction; %, method, disposal impact in whole or in parts throughout the cycle (in case of no opportunity for the 100% recyclable cycle)".

The results of the categories evaluation and the comments by experts were analyzed carefully, seeking to refine and enrich the proposed sustainability assessment framework. To view statistical results in each criteria see supplementary material S11. Based on the experts feedback, the following changes have been included in the assessment framework:

- The energy consumption category has been changed to Primary Energy Demand;
- Regulation for protection for biodiversity and for ecosystem service: included in the framework as a subcategory level 1 of the Development Policy category;
- "Fossil fuel subsidies" subcategory level 1 has been changed to "Fossil fuel subsidies and taxation" in the Energy matrix category;
- Biotechnology innovation: included in the framework as a subcategory level 2 of the Scientific and Technological Capacity category.

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4.8 Refinement of categories based on literature reviews

A further refinement was made based on the categories mapped in the two literature reviews. Four additional categories were identified that were not included in the initial map derived from the SDGs and were not suggested by the consulted experts, namely:

- Sustainable Transport: included in the framework as a subcategory level 1 of the Resilient, Modern and Sustainable Infrastructure category (based Onat et al., 2021). In this case, Sustainable Transport involves several issues, such as access to different types of transport, type of energy source, safety, emissions and energy efficiency (Haghshenas & Vaziri, 2012). These were included in the framework as subcategories level 2;
- Social acceptance of energy technology: included as a subcategory level 1 of the Energy matrix category (based Haase et al., 2022);
- Social Tensions: included as a subcategory level 1 of the Equal Rights category. In this case, as Social Tensions involve several issues, such as tensions generated by limited access to resources, conflicts with illegal actors, aggression against political opponents and inequality in the sharing of costs and benefits (Mancini & Sala, 2018), these were included in the framework as subcategories level 2;
- Thefts: included in the framework as a subcategory level 1 of the Violence category (based on Mancini & Sala, 2018).

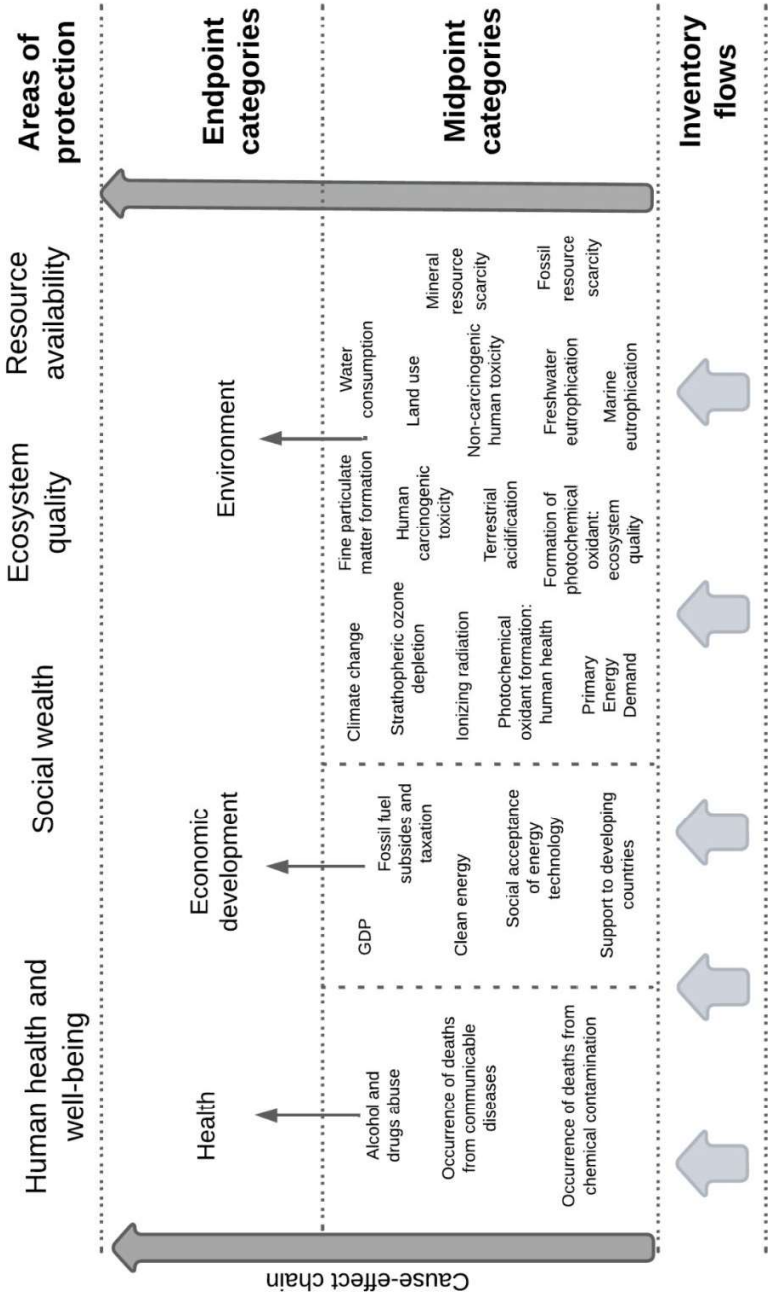
To see the evaluation framework and indicators after this refinement, see the supplementary material S9 e S10.

4.9 LCSA structure for the SDGs categories and cause-effect relationship

Based on the previous steps, this study proposes an illustrative map of the cause-effect relationships from the SDG targets to the final Areas of Protection (AoP), representing an LCSA impacts pathway for LIBs. The LCSA framework was developed similarly to that done by Souza et al. (2015). The central axes of the structure shown in Figure 3 are endpoint categories, whilst the operational aspects are considered midpoint categories. When compared with previously proposed LCSA structures (Kloepffer, 2008; Traverso et al., 2012; UNEP, 2011), it is observed that the previous categories are covered, and there is an expansion in the categories to include the questions considered central to SD. The impact pathways of LCSA originating from the SDGs are shown in Figures 4 and 5.

The proposed cause and effect structure adopts at the midpoint level single problem categories and at the endpoint categories aggregate damages that address the endpoint indicators already consolidated in the LCA literature (Human Health; Biodiversity loss; Resources Scarcity), but also other indicators aggregates that include Economic Development, Sustainability Management and Citizenship and Inclusion.

Figure 4 - LCSA impact categories for LIBs based on the SDGs (Part I).



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648 To validate the proposed impact pathways, an extensive literature review, covering
649 different areas. It is important to point out that these causal relationships are still defined in a
650 qualitative way, aiming at a sufficiently detailed structure to support SD. No characterization
651 model is presented now, which would allow for assessing the level of importance of each inventory
652 flow within an impact category. The present analysis is not definitive, as there are many challenges
653 associated with defining cause-effect relationships in social phenomena, that are heavily dependent
654 on perception and interpretation (Jorgensen et al., 2010; Iofrida et al., 2019).

655 The adopted Areas of Protection (AoPs) include those established in environmental LCA:
656 Human Health, Ecosystem Quality, and Resource Availability (Huijbregtes et al., 2016). In
657 addition, two additional AoPs were included, namely Well-being (Neugebauer et al., 2016;
658 Valdivia et al., 2021) and Social Wealth, which can be understood as prosperity (Nansai et al.,
659 2019; Neugebauer et al., 2016). Human health and well-being were intertwined, as evidence shows
660 that these aspects are intrinsically related (Feschet et al., 2013).

661 The SDGs endpoint categories represent the axes of the proposed structure in Figure 3 and
662 Table 3, expressing broad sustainability objectives related to each AoP. For example,
663 improvements in aspects of health, citizenship and inclusion, management for sustainability,
664 economic development, and the environment can improve human and ecosystem health and well-
665 being (Neugebauer et al., 2017). The increase in GDP, measured by the increase in economic
666 activity and income growth, leads to improvements in the health of a country's population, and
667 consequently in the perception of well-being (Feschet et al., 2013). On the other hand, increasing
668 inequality has negative consequences for health (Bocoum et al., 2015).

669 Using the same rationale, midpoint categories are qualitatively linked to those SDGs
670 endpoints, supported by the literature. For the Health endpoint, midpoint categories relate to issues

that affect the ability of individuals to fully perform their tasks or deaths (Ju-Chang et al., 2018). The Inclusion and Citizenship endpoint category encompasses 29 midpoints, involving multidimensional themes such as education, access to basic services and human rights (Hardadi & Pizzol, 2017; Neugebauer et al., 2014; Souza et al., 2015). Education is highly relevant for human development and can determine the future living conditions of individuals, where the lack of education reduces access to prosperity and wealth (Neugebauer et al., 2014). The salary level is directly related to the living condition and nutritional status, which can be related to life expectancy and, therefore, to human health and social well-being (Neugebauer et al., 2017). The complete visualization of the analysis of the impact pathways is presented in supplementary material S7.

The AoP aims to safeguard key dimensions of sustainable development (Schaubroeck & Rugani, 2017). This broad view of the role of the LIBs life cycle in protecting these key areas can lead to improvements in the development of the product system. The environmental, social and economic issues in the life cycle of LIBs have the potential to cause damage to the AoP and consequently hamper the achievement of the SDGs. Therefore, making the life cycle of LIBs more sustainable is essential. However, this requires the development of analytical and systematic tools that allow a holistic assessment of the contribution, as well as the advancement of the LIBs life cycle toward SD. LIBs have been identified as key elements in the low-carbon energy transition, which will lead to the mitigation of climate change and, consequently, the reduction of social and economic challenges, protecting health, ecosystems, resources and the generation of wealth (Harper et al., 2019).

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5 CONCLUSIONS

This study presents an LCSA framework for the LIBs life cycle, using a methodological approach that uses the SDG framework as a basis to elicit the impact categories. By using the SDGs as the basis for developing the LCSA, a comprehensive number of sustainability themes are incorporated into the assessment framework, going beyond the categories already proposed in sustainability assessments for the LIBs life cycle, and in general LCSA frameworks. New impact assessment categories were identified as relevant to the LIBs life cycle, such as: Housing, Modern and Resilient Infrastructure, Scientific and Technological Capacity, Energy Matrix, Support for Developing Countries, Productive Inclusion and Income, Access to Basic Services, Sustainable transport, Disease transmission, Alcohol and drug abuse, Discrimination, Food Security and Violence.

The results indicate that LIBs life cycle has the potential to contribute to the achievement of the SDGs in all countries where extraction, production, use, and disposal activities take place. The materiality analysis revealed that the goals related to SDGs 1, 3, 6, 7, 8, 9, 13, 15 and 16 were more correlated to LIBs life cycle, as they express social, environmental, and economic issues widely affected in the locations where the stages of the life cycle occur. The stages that have the most potential for impacting the SDG targets are the extraction and processing of raw materials, production of materials and cells, and end-of-life. This is because the countries involved in the upstream stages are underdeveloped or developing, which still have serious social problems, such as poverty, strong dependence on the extraction of natural resources for economic development, and weak environmental and labor legislation, which leads to serious risks to ecosystems and socio-economics. The end-of-life stage can happen in any country, but it is also associated with serious social problems, as there is still little regulation in all countries on the management of the

end-of-life of LIBs, with many LIBs being managed through informal channels, which are associated with serious environmental and socioeconomic problems. The countries that most present sustainability risks associated with LIBs are the DRC, China, and Chile.

Regarding the methodology applied to develop LCSA based on the SDGs, the problem structuring methods facilitate the mapping of impact pathways and the elucidation of cause-and-effect mechanisms to identify and define impact categories. To develop impact pathway in an LCSA, it may be better to break sustainability down into strategic axes than into the three dimensions of the triple bottom line, as the division on the sustainability tripod can limit the understanding of cause-and-effect relationships. The in-depth detailing of the cause-and-effect pathways allows the relationships between the themes and their contribution to each AoP to be better understood. The application of PSMs in conjunction with the principles of life cycle thinking makes it possible to integrate relationships within a sustainability assessment framework into a general framework.

The developed framework provides the means for assessing the sustainability of the life cycle of LIBs, making it possible to assess the present situation, as well as changes in materials, processes and supply locations and their possible effects on achieving the SDGs (adopting the CLCA). In addition, the framework can be applied to assess the sustainability of other types of products that share the same supply chain. The results presented also contribute to the literature by presenting a systematic method for the development of LCSA structures based on SDGs.

Finally, despite the support in the literature for the development of the framework and the external consultation of experts for validation, some limitations must be considered. The main limitation is related to the qualitative analysis applied to the definition of potential impact categories and end-point and mid-point levels. In addition, the framework was developed

considering the context of the life cycle of LIBs and cannot be easily used by other products that do not share the same supply chain. Future developments may enhance the framework through practical assessment applications to validate proposed impact pathways. In addition, future work can also validate the proposed method by applying it to other types of products.

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REFERENCES

Ackermann, F. (2012). Problem structuring methods 'in the Dock': Arguing the case for Soft OR. *European Journal of Operational Research*, 219(3), 652-658. <https://doi.org/10.1016/j.ejor.2011.11.014>

Almanza, A. M. H., & Corona, B. (2020). Using Social Life Cycle Assessment to analyze the contribution of products to the Sustainable Development Goals: a case study in the textile sector. *The International Journal of Life Cycle Assessment*, 25(9), 1833-1845. <https://doi.org/10.1007/s11367-020-01789-7>

Amnesty International. (2017). Time to recharge: Corporate action and inaction to tackle abuses in the cobalt supply chain.

Asongu, S. A., Orim, S. M. I., & Nting, R. T. (2019). Inequality, information technology and inclusive education in sub-Saharan Africa. *Technological Forecasting and Social Change*, 146, 380-389. <https://doi.org/10.1016/j.techfore.2019.06.006>

Baars, J., Domenech, T., Bleischwitz, R., Melin, H. E., & Heidrich, O. (2021). Circular economy strategies for electric vehicle batteries reduce reliance on raw materials. *Nature Sustainability*, 4(1), 71-79. <https://doi.org/10.1038/s41893-020-00607-0>

Backes, J. G., & Traverso, M. (2022). Life Cycle Sustainability Assessment as a metrics towards SDGs Agenda 2030. *Current Opinion in Green and Sustainable Chemistry*, 100683. <https://doi.org/10.1016/j.cogsc.2022.100683>

- Barke, A., Thies, C., Popien, J. L., Melo, S. P., Cerdas, F., Herrmann, C., & Spengler, T. S. (2021). Life cycle sustainability assessment of potential battery systems for electric aircraft. *Procedia CIRP*, 98, 660-665.
- Beske, F., Haustein, E., & Lorson, P. C. (2019). Materiality analysis in sustainability and integrated reports. *Sustainability Accounting, Management and Policy Journal*. <https://doi.org/10.1108/SAMPJ-12-2018-0343>
- Bielewski, M., Blagoeva, D., Cordella, M., Di Persio, F., Gaudillat, P., Hildebrand, S., ... & Zampori, L. (2021). Analysis of sustainability criteria for lithium-ion batteries including related standards and regulations. doi:10.2760/811476
- Bocoum, I., Macombe, C., & Revéret, J. P. (2015). Anticipating impacts on health based on changes in income inequality caused by life cycles. *The International Journal of Life Cycle Assessment*, 20(3), 405-417. <https://doi.org/10.1007/s11367-014-0835-x>
- Buchmayr, A., Verhofstadt, E., Van Ootegem, L., Delmás, D. S., Thomassen, G., & Dewulf, J. (2021). The path to sustainable energy supply systems: Proposal of an integrative sustainability assessment framework. *Renewable and Sustainable Energy Reviews*, 138, 110666. <https://doi.org/10.1016/j.rser.2020.110666>
- Calabrese, A., Costa, R., Levialdi, N., & Menichini, T. (2016). A fuzzy analytic hierarchy process method to support materiality assessment in sustainability reporting. *Journal of Cleaner Production*, 121, 248-264. <https://doi.org/10.1016/j.jclepro.2015.12.005>
- Caiado, R. G. G., Leal Filho, W., Quelhas, O. L. G., de Mattos Nascimento, D. L., & Ávila, L. V. (2018). A literature-based review on potentials and constraints in the implementation of the sustainable development goals. *Journal of cleaner production*, 198, 1276-1288. <https://doi.org/10.1016/j.jclepro.2018.07.102>
- Centobelli, P., Cerchione, R., Del Vecchio, P., Oropallo, E., & Secundo, G. (2022). Blockchain technology for bridging trust, traceability and transparency in circular supply chain. *Information & Management*, 59(7), 103508. <https://doi.org/10.1016/j.im.2021.103508>
- Chandrakumar, C., & McLaren, S. J. (2018). Towards a comprehensive absolute sustainability assessment method for effective Earth system governance: Defining key environmental indicators using an enhanced-DPSIR framework. *Ecological Indicators*, 90, 577-583. <https://doi.org/10.1016/j.ecolind.2018.03.063>
- Chen, Z., Zhang, L., & Xu, Z. (2020). Analysis of cobalt flows in mainland China: Exploring the potential opportunities for improving resource efficiency and supply security. *Journal of Cleaner Production*, 275, 122841. <https://doi.org/10.1016/j.jclepro.2020.122841>
- Christensen, P. A., Anderson, P. A., Harper, G. D., Lambert, S. M., Mrozik, W., Rajaeifar, M. A., ... & Heidrich, O. (2021). Risk management over the life cycle of lithium-ion batteries in electric

1
2
3 827 vehicles. *Renewable and Sustainable Energy Reviews*, 148, 111240.
4 828 <https://doi.org/10.1016/j.rser.2021.111240>
5 829
6
7 830 Colangelo, F., Petrillo, A., & Farina, I. (2021). Comparative environmental evaluation of recycled
8 831 aggregates from construction and demolition wastes in Italy. *Science of The Total Environment*,
9 832 798, 149250. <https://doi.org/10.1016/j.scitotenv.2021.149250>
10 833
11 834 Crenna, E., Gauch, M., Widmer, R., Wäger, P., & Hischier, R. (2021). Towards more flexibility
12 835 and transparency in life cycle inventories for Lithium-ion batteries. *Resources, Conservation and*
13 836 *Recycling*, 170, 105619. <https://doi.org/10.1016/j.resconrec.2021.105619>
14 837
15 838 Cusenza, M. A., Bobba, S., Ardente, F., Cellura, M., & Di Persio, F. (2019). Energy and
16 839 environmental assessment of a traction lithium-ion battery pack for plug-in hybrid electric
17 840 vehicles. *Journal of cleaner production*, 215, 634-649.
18 841 <https://doi.org/10.1016/j.jclepro.2019.01.056>
19 842
20 843 Dunn, J. B., Gaines, L., Kelly, J. C., James, C., & Gallagher, K. G. (2015). The significance of Li-
21 844 ion batteries in electric vehicle life-cycle energy and emissions and recycling's role in its reduction.
22 845 *Energy & Environmental Science*, 8(1), 158-168. DOI: 10.1039/C4EE03029J
23 846
24 847 Eden, C., & Ackermann, F. (2018). Theory into practice, practice to theory: Action research in
25 848 method development. *European Journal of Operational Research*, 271(3), 1145-1155.
26 849 <https://doi.org/10.1016/j.ejor.2018.05.061>
27 850
28 851 Ekvall, T., Azapagic, A., Finnveden, G., Rydberg, T., Weidema, B. P., & Zamagni, A. (2016).
29 852 Attributional and consequential LCA in the ILCD handbook. *The International Journal of Life*
30 853 *Cycle Assessment*, 21(3), 293-296. <https://doi.org/10.1007/s11367-015-1026-0>
31 854
32 855 Fahimi, A., Ducoli, S., Federici, S., Ye, G., Mousa, E., Frontera, P., & Bontempi, E. (2022).
33 856 Evaluation of the sustainability of technologies to recycle spent lithium-ion batteries, based on
34 857 embodied energy and carbon footprint. *Journal of Cleaner Production*, 338,
35 858 130493. <https://doi.org/10.1016/j.jclepro.2022.130493>
36 859
37 860 Farooq, M. B., Zaman, R., Sarraj, D., & Khalid, F. (2021). Examining the extent of and drivers for
38 861 materiality assessment disclosures in sustainability reports. *Sustainability Accounting,*
39 862 *Management and Policy Journal*. <https://doi.org/10.1108/SAMPJ-04-2020-0113>
40 863
41 864 Feschet, P., Macombe, C., Garrabé, M., Loeillet, D., Saez, A. R., & Benhmadi, F. (2013). Social
42 865 impact assessment in LCA using the Preston pathway. *The International Journal of Life Cycle*
43 866 *Assessment*, 18(2), 490-503. <https://doi.org/10.1007/s11367-012-0490-z>
44 867
45 868 Franco, L. A., & Montibeller, G. (2010). Problem structuring for multicriteria decision analysis
46 869 interventions. *Wiley encyclopedia of operations research and management science*.
47 870
48
49
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52
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55
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57
58
59
60

- Françoço, R. V., & Belderrain, M. C. N. (2022). A problem structuring method framework for value-focused thinking. *EURO Journal on Decision Processes*, 10, 100014. <https://doi.org/10.1016/j.ejdp.2022.100014>
- Goodenough, J. B. (2018). How we made the Li-ion rechargeable battery. *Nature Electronics*, 1(3), 204-204. <https://doi.org/10.1038/s41928-018-0048-6>
- Gebhardt, M., Beck, J., Kopyto, M., & Spieske, A. (2022). Determining requirements and challenges for a sustainable and circular electric vehicle battery supply chain: A mixed-methods approach. *Sustainable Production and Consumption*, 33, 203-217. <https://doi.org/10.1016/j.spc.2022.06.024>
- Haase, M., Wulf, C., Baumann, M., Rösch, C., Weil, M., Zapp, P., & Naegler, T. (2022). Prospective assessment of energy technologies: a comprehensive approach for sustainability assessment. *Energy, Sustainability and Society*, 12(1), 1-41. <https://doi.org/10.1186/s13705-022-00344-6>
- Hall, M. R. (2021). The living wage gap—a quantitative measure of poverty in global supply chains. *The International Journal of Life Cycle Assessment*, 26(9), 1867-1877. <https://doi.org/10.1007/s11367-021-01945-7>
- Haghshenas, H., & Vaziri, M. (2012). Urban sustainable transportation indicators for global comparison. *Ecological Indicators*, 15(1), 115-121. <https://doi.org/10.1016/j.ecolind.2011.09.010>
- Hannouf, M. B., Padilla-Rivera, A., Assefa, G., & Gates, I. (2022). Methodological framework to find links between life cycle sustainability assessment categories and the UN Sustainable Development Goals based on literature. *Journal of Industrial Ecology*. <https://doi.org/10.1111/jiec.13283>
- Hardadi, G., & Pizzol, M. (2017). Extending the multiregional input-output framework to labor-related impacts: A proof of concept. *Journal of Industrial Ecology*, 21(6), 1536-1546. <https://doi.org/10.1111/jiec.12588>
- Harper, G., Sommerville, R., Kendrick, E., Driscoll, L., Slater, P., Stolkin, R., ... & Anderson, P. (2019). Recycling lithium-ion batteries from electric vehicles. *nature*, 575(7781), 75-86. <https://doi.org/10.1038/s41586-019-1682-5>
- Havardi-Burger, N., Mempel, H., & Bitsch, V. (2021). Framework for sustainability assessment of the value chain of flowering potted plants for the German market. *Journal of Cleaner Production*, 329, 129684. <https://doi.org/10.1016/j.jclepro.2021.129684>
- Henzler, K., Maier, S. D., Jäger, M., & Horn, R. (2020). SDG-based sustainability assessment methodology for innovations in the field of urban surfaces. *Sustainability*, 12(11), 4466. <https://doi.org/10.3390/su12114466>
- Huijbregts, M. A., Steinmann, Z. J., Elshout, P. M., Stam, G., Verones, F., Vieira, M., ... & van Zelm, R. (2017). ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and

1
2
3 917 endpoint level. *The International Journal of Life Cycle Assessment*, 22(2), 138-
4 918 147.<https://doi.org/10.1007/s11367-016-1246-y>
5 919
6 920 INTERNATIONAL LABOUR ORGANIZATION (ILO). (2019). Child labour in mining and
7 921 global supply chains. Child Labor Platform.
8 922
9 923 INTERNATIONAL INSTITUTE FOR SUSTAINABLE DEVELOPMENT. (2018). Green
10 924 Conflict Minerals: The Fuels of Conflict in the Transition to a Low-carbon Economy.
11 925
12 926 Iofrida, N., De Luca, A. I., Silveri, F., Falcone, G., Stillitano, T., Gulisano, G., & Strano, A. (2019).
13 927 Psychosocial risk factors' impact pathway for social life cycle assessment: an application to citrus
14 928 life cycles in South Italy. *The international journal of life cycle assessment*, 24(4), 767-
15 929 780.<https://doi.org/10.1007/s11367-018-1482-4>
16 930
17 931 Jørgensen, A., Lai, L.C.H. & Hauschild, M.Z. (2010). Assessing the validity of impact pathways
18 932 for child labour and well-being in social life cycle assessment. *Int J Life Cycle Assess* 15, 5.
19 933 <https://doi.org/10.1007/s11367-009-0131-3>
20 934
21 935 Kang, D. H. P., Chen, M., & Ogunseitan, O. A. (2013). Potential environmental and human health
22 936 impacts of rechargeable lithium batteries in electronic waste. *Environmental science & technology*,
23 937 47(10), 5495-5503.<https://doi.org/10.1021/es400614y>
24 938
25 939 Klöpffer, W. (2008). Life cycle sustainability assessment of products. *The International Journal*
26 940 *of Life Cycle Assessment*, 13(2), 89-95. <https://doi.org/10.1065/lca2008.02.376>
27 941
28 942 Kühnen, M., Silva, S., Beckmann, J., Eberle, U., Hahn, R., Hermann, C., ... & Schmid, M. (2019,
29 943 March). Contributions to the sustainable development goals in life cycle sustainability assessment:
30 944 Insights from the Handprint research project. In *NachhaltigkeitsManagementForum|*
31 945 *Sustainability Management Forum* (Vol. 27, No. 1, pp. 65-82). Springer Berlin Heidelberg.
32 946 <https://doi.org/10.1007/s00550-019-00484-y>
33 947
34 948 Laurent, A., Molin, C., Owsianiak, M., Fantke, P., Dewulf, W., Herrmann, C., ... & Hauschild, M.
35 949 (2019). The role of life cycle engineering (LCE) in meeting the sustainable development goals—
36 950 report from a consultation of LCE experts. *Journal of Cleaner Production*, 230, 378-382.
37 951 <https://doi.org/10.1016/j.jclepro.2019.05.129>
38 952
39 953 Lèbre, É., Stringer, M., Svobodova, K., Owen, J. R., Kemp, D., Côte, C., ... & Valenta, R. K.
40 954 (2020). The social and environmental complexities of extracting energy transition metals. *Nature*
41 955 *communications*, 11(1), 1-8.<https://doi.org/10.1038/s41467-020-18661-9>
42 956
43 957 Liu, B., Zhang, Q., Liu, J., Hao, Y., Tang, Y., & Li, Y. (2022). The impacts of critical metal
44 958 shortage on China's electric vehicle industry development and countermeasure policies. *Energy*,
45 959 248, 123646. <https://doi.org/10.1016/j.energy.2022.123646>
46 960
47 961 Maier, S. D., Beck, T., Francisco Vallejo, J., Horn, R., Söhlemann, J. H., & Nguyen, T. T. (2016).
48 962 Methodological approach for the sustainability assessment of development cooperation projects

- for built innovations based on the SDGs and life cycle thinking. *Sustainability*, 8(10), 1006. <https://doi.org/10.3390/su8101006>
- Mancini, L., Eslava, N. A., Traverso, M., & Mathieux, F. (2021). Assessing impacts of responsible sourcing initiatives for cobalt: Insights from a case study. *Resources Policy*, 71, 102015. <https://doi.org/10.1016/j.resourpol.2021.102015>
- Mancini, L., & Sala, S. (2018). Social impact assessment in the mining sector: Review and comparison of indicators frameworks. *Resources Policy*, 57, 98-111. <https://doi.org/10.1016/j.resourpol.2018.02.002>
- Mio, C., Panfilo, S., & Blundo, B. (2020). Sustainable development goals and the strategic role of business: A systematic literature review. *Business Strategy and the Environment*, 29(8), 3220-3245. <https://doi.org/10.1002/bse.2568>
- Mohr, M., Peters, J. F., Baumann, M., & Weil, M. (2020). Toward a cell-chemistry specific life cycle assessment of lithium-ion battery recycling processes. *Journal of Industrial Ecology*, 24(6), 1310-1322. <https://doi.org/10.1111/jiec.13021>
- Montiel, I., Cuervo-Cazurra, A., Park, J., Antolín-López, R., & Husted, B. W. (2021). Implementing the United Nations' Sustainable Development Goals in international business. *Journal of International Business Studies*, 1-32. <https://doi.org/10.1057/s41267-021-00445-y>
- Morley, J., Widdicks, K., & Hazas, M. (2018). Digitalisation, energy and data demand: The impact of Internet traffic on overall and peak electricity consumption. *Energy Research & Social Science*, 38, 128-137. <https://doi.org/10.1016/j.erss.2018.01.018>
- Mossali, E., Picone, N., Gentilini, L., Rodriguez, O., Pérez, J. M., & Colledani, M. (2020). Lithium-ion batteries towards circular economy: A literature review of opportunities and issues of recycling treatments. *Journal of environmental management*, 264, 110500. <https://doi.org/10.1016/j.jenvman.2020.110500>
- Nansai, K., Kondo, Y., Giurco, D., Sussman, D., Nakajima, K., Kagawa, S., ... & Tohno, S. (2019). Nexus between economy-wide metal inputs and the deterioration of sustainable development goals. *Resources, Conservation and Recycling*, 149, 12-19. <https://doi.org/10.1016/j.resconrec.2019.05.017>
- Nature Materials. (2022). Batteries show the difficulties of being greener. *Nat. Mater.* 21, 131 (2022). <https://doi.org/10.1038/s41563-022-01198-1>
- Nkulu, B. L. C., Casas, L., Haufroid, V., De Putter, T., Saenen, N. D., Kayembe-Kitenge, T., ... & Nemery, B. (2018). Sustainability of artisanal mining of cobalt in DR Congo. *Nature sustainability*, 1(9), 495-504. <https://doi.org/10.1038/s41893-018-0139-4>
- Neugebauer, S., Traverso, M., Scheumann, R., Chang, Y. J., Wolf, K., & Finkbeiner, M. (2014). Impact pathways to address social well-being and social justice in SLCA—fair wage and level of education. *Sustainability*, 6(8), 4839-4857. <https://doi.org/10.3390/su6084839>

1
2
3 1009
4 1010 Neugebauer, S., Forin, S., & Finkbeiner, M. (2016). From life cycle costing to economic life cycle
5 1011 assessment—introducing an economic impact pathway. *Sustainability*, 8(5), 428.
6 1012 <https://doi.org/10.3390/su8050428>
7 1013
8 1014 Neugebauer, S., Emara, Y., Hellerström, C., & Finkbeiner, M. (2017). Calculation of fair wage
9 1015 potentials along products' life cycle—Introduction of a new midpoint impact category for social life
10 1016 cycle assessment. *Journal of Cleaner Production*, 143, 1221-
11 1017 1232.<https://doi.org/10.1016/j.jclepro.2016.11.172>
12 1018
13 1019 OECD. (2022). Main science and technology indicators. Available:
14 1020 <https://www.oecd.org/sti/msti.htm>.
15 1021
16 1022 Olawumi, T. O., & Chan, D. W. (2018). A scientometric review of global research on sustainability
17 1023 and sustainable development. *Journal of cleaner production*, 183, 231-
18 1024 250.<https://doi.org/10.1016/j.jclepro.2018.02.162>
19 1025
20 1026 Olivetti, E. A., Ceder, G., Gaustad, G. G., & Fu, X. (2017). Lithium-ion battery supply chain
21 1027 considerations: analysis of potential bottlenecks in critical metals. *Joule*, 1(2), 229-
22 1028 243.<https://doi.org/10.1016/j.joule.2017.08.019>
23 1029
24 1030 Omisore, A. G. (2018). Attaining Sustainable Development Goals in sub-Saharan Africa; The need
25 1031 to address environmental challenges. *Environmental development*, 25, 138-
26 1032 145.<https://doi.org/10.1016/j.envdev.2017.09.002>
27 1033
28 1034 Onat, N. C., Abdella, G. M., Kucukvar, M., Kutty, A. A., Al-Nuaimi, M., Kumbaroğlu, G., &
29 1035 Bulu, M. (2021). How eco-efficient are electric vehicles across Europe? A regionalized life cycle
30 1036 assessment-based eco-efficiency analysis. *Sustainable Development*, 29(5), 941-956.
31 1037 <https://doi.org/10.1002/sd.2186>
32 1038
33 1039 Pollok, L., Spierling, S., Endres, H. J., & Grote, U. (2021). Social Life Cycle Assessments: a
34 1040 review on past development, advances and methodological challenges. *Sustainability*, 13(18),
35 1041 10286.<https://doi.org/10.3390/su131810286>
36 1042
37 1043 Rajaeifar, M. A., Ghadimi, P., Raugei, M., Wu, Y., & Heidrich, O. (2022). Challenges and recent
38 1044 developments in supply and value chains of electric vehicle batteries: A sustainability perspective.
39 1045 *Resources, Conservation and Recycling*, 180, 106144.
40 1046
41 1047 Redman, A. (2018). Harnessing the Sustainable Development Goals for businesses: A progressive
42 1048 framework for action. *Business Strategy & Development*, 1(4), 230-243.
43 1049 <https://doi.org/10.1002/bsd2.33>
44 1050
45 1051 Rocchi, M., & Pescatore, G. (2022). Modeling narrative features in TV series: coding and
46 1052 clustering analysis. *Humanities and Social Sciences Communications*, 9(1), 1-11.
47 1053
48
49
50
51
52
53
54
55
56
57
58
59
60

- Rosenhead J. (2013). Problem Structuring Methods. In: Gass S.I., Fu M.C. (eds) Encyclopedia of Operations Research and Management Science. Springer, Boston, MA.
- Sachs, J. D. (2012). From millennium development goals to sustainable development goals. *The lancet*, 379(9832), 2206-2211. [https://doi.org/10.1016/S0140-6736\(12\)60685-0](https://doi.org/10.1016/S0140-6736(12)60685-0)
- Sachs, J. D., Schmidt-Traub, G., Mazzucato, M., Messner, D., Nakicenovic, N., & Rockström, J. (2019). Six transformations to achieve the sustainable development goals. *Nature Sustainability*, 2(9), 805-814. <https://doi.org/10.1038/s41893-019-0352-9>
- Saraiva, A. B., Souza, R. G., Mahler, C. F., & Valle, R. A. B. (2018). Consequential lifecycle modelling of solid waste management systems—Reviewing choices and exploring their consequences. *Journal of Cleaner Production*, 202, 488-496. <https://doi.org/10.1016/j.jclepro.2018.08.038>
- Schaubroeck, T., & Rugani, B. (2017). A revision of what life cycle sustainability assessment should entail: Towards modeling the net impact on human well-being. *Journal of Industrial Ecology*, 21(6), 1464-1477. <https://doi.org/10.1111/jiec.12653>
- Singh, G. G., Cisneros-Montemayor, A. M., Swartz, W., Cheung, W., Guy, J. A., Kenny, T. A., ... & Ota, Y. (2018). A rapid assessment of co-benefits and trade-offs among Sustainable Development Goals. *Marine Policy*, 93, 223-231. <https://doi.org/10.1016/j.marpol.2017.05.030>
- Sommerville, R., Zhu, P., Rajaeifar, M. A., Heidrich, O., Goodship, V., & Kendrick, E. (2021). A qualitative assessment of lithium ion battery recycling processes. *Resources, Conservation and Recycling*, 165, 105219. <https://doi.org/10.1016/j.resconrec.2020.105219>
- Souza, R. G., Rosenhead, J., Salhofer, S. P., Valle, R. A. B., & Lins, M. P. E. (2015). Definition of sustainability impact categories based on stakeholder perspectives. *Journal of Cleaner Production*, 105, 41-51. <https://doi.org/10.1016/j.jclepro.2014.09.051>
- Spaiser, V., Ranganathan, S., Swain, R. B., & Sumpter, D. J. (2017). The sustainable development oxymoron: quantifying and modelling the incompatibility of sustainable development goals. *International Journal of Sustainable Development & World Ecology*, 24(6), 457-470. <https://doi.org/10.1080/13504509.2016.1235624>
- Subramaniam, N., Junior, R. M., Akbar, S., Ji, H., & Situ, H. (2020). *SDG Measurement and Disclosure 2.0 A study of ASX150 companies*. RMIT University.
- Sulis, F., Agostinho, F., Almeida, C. M., & Giannetti, B. F. (2021). Recognizing the wealth of non-marketable food in distribution centres: The environmental benefits of donation. *Journal of Cleaner Production*, 318, 128482. <https://doi.org/10.1016/j.jclepro.2021.128482>
- Sullivan, K., Thomas, S., & Rosano, M. (2018). Using industrial ecology and strategic management concepts to pursue the Sustainable Development Goals. *Journal of Cleaner Production*, 174, 237-246. <https://doi.org/10.1016/j.jclepro.2017.10.201>

3.11 Artigo 11 – Leachate analysis of Waste electrical and electronic equipment disposal in landfills and open dumps – context of a Brazilian city *

* Nascimento, G.A., Souza, R.G., Leachate analysis of Waste electrical and electronic equipment disposal in landfills and open dumps – context of a Brazilian city. Submetido ao J Environ Pollut, em dezembro/2022.

ABSTRACT

Brazil is the fifth largest generator of waste electrical and electronic equipment (WEEE) worldwide, but the estimated proportion of WEEE that undergoes the correct disposal is very low. The WEEE is commonly discarded mixed with municipal solid waste (MSW), the most common destinations for which are sanitary landfills and open dumpsites. The objective of this study was to evaluate the characteristics of leachate that results from disposal of WEEE in soil. Different forms of WEEE and MSW landfilling and open dumping were simulated in column reactors to analyse the resultant leachates. Simulation environments included a blank reactor (soil without waste), two open dumps (one with mixed MSW and WEEE and the other with only WEEE), and WEEE landfilling (alternated waste with soil cover layers). Precipitation was simulated considering the six rainiest months in the city of São José dos Campos, Brazil, and the leachate was sampled monthly. Analyses of heavy metals in the leachates were performed in two ways: via inductively coupled plasma-mass-spectrometry with acid digestion of samples, following American Standard 3015A, and via inductively coupled plasma-optical emission spectrometry without acid digestion (USEPA, 2007). For both the methodologies, the greatest number of contaminants was observed in the tanks that contained WEEE, regardless of the final disposal method. Metal concentrations found in the landfill and dump simulations were similar, but a higher concentration of metals was found in samples with acid digestion than those without acid digestion. Additionally, it was evident that the timescale of observation influenced the results.

Keywords: Landfill; Open dumps; Heavy metals; Leachate; WEEE.

Leachate analysis of Waste electrical and electronic equipment disposal in landfills and open dumps – context of a Brazilian city

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ABSTRACT

Brazil is the fifth largest generator of waste electrical and electronic equipment (WEEE) worldwide, but the estimated proportion of WEEE that undergoes the correct disposal is very low. The WEEE is commonly discarded mixed with municipal solid waste (MSW), the most common destinations for which are sanitary landfills and open dumpsites. The objective of this study was to evaluate the characteristics of leachate that results from disposal of WEEE in soil. Different forms of WEEE and MSW landfilling and open dumping were simulated in column reactors to analyse the resultant leachates. Simulation environments included a blank reactor (soil without waste), two open dumps (one with mixed MSW and WEEE and the other with only WEEE), and WEEE landfilling (alternated waste with soil cover layers). Precipitation was simulated considering the six rainiest months in the city of São José dos Campos, Brazil, and the leachate was sampled monthly. Analyses of heavy metals in the leachates were performed in two ways: via inductively coupled plasma-mass-spectrometry with acid digestion of samples, following American Standard 3015A, and via inductively coupled plasma-optical emission spectrometry without acid digestion (USEPA, 2007). For both the methodologies, the greatest number of contaminants was observed in the tanks that contained WEEE, regardless of the final disposal method. Metal concentrations found in the landfill and dump simulations were similar, but a higher concentration of metals was found in samples with acid digestion than those without acid digestion. Additionally, it was evident that the timescale of observation influenced the results.

Keywords: landfill, open dumps, heavy metals, leachate, WEEE.

1. Introduction

Electrical and electronic equipment (EEE) contains toxic, precious, and base metals, such as Ba, Cd, Cr, Pb, Sb, Al, Cu, Fe, Sn, Zn, Ag, and Au (Oguchi et al., 2013). The degree of human toxicity or ecotoxicity depends on the environmental setting of the metals (Woo et al., 2016).

Waste electrical and electronic equipment (WEEE) refers to electrical products discarded after use (Green Eletron, 2021). WEEE recycling can be risky because of high concentrations of toxic metals, which can harm waste workers through dust inhalation, oral ingestion, or skin contact (Lepawsky and McNabb, 2010). Adverse health effects in adults and children include coughing, choking, tremors, convulsions, and even death (Yu et al., 2006). Because children are still growing and developing, contact with toxic substances from WEEE can impact their neurological system, increasing the risk of chronic diseases, such as cancer (WHO, 2012).

WEEE leads to negative environmental effects in water, soil, and air. Chemical analyses carried out in areas of inadequate WEEE processing showed the presence of As, Cd, Cr, Hg, and Pb in the soil, water, and sediments (Houessionon et al., 2021). Material recovery practices at informal WEEE recycling systems include open-air burning and acid leaching (Sepúlveda et al., 2010; Wang and Xu, 2014). If uncontrolled, these recovery practices can emit toxic compounds to the air, harming workers and residents in surrounding areas (Cesaro et al., 2019).

The Brazilian National Solid Waste Policy (PNRS) requires that manufacturers, importers, distributors, and traders implement reverse logistics systems for WEEE and their components upon return of end-of-life products by consumers (BRASIL, 2010). However, the estimated adequate disposal of WEEE in Brazil is only 2.7% (Wagner et al., 2022) and less than 1% of this waste is recycled via formal recycling systems. The majority is processed by informal workers (CEMPRE, 2018).

Brazil is the fifth largest generator of WEEE worldwide and the second largest generator in the American continent (Forti et al., 2020). In 2019, 2.1 t of WEEE was generated in Brazil, equivalent to 10.2 kg per inhabitant (ABRELPE, 2020). The average annual generation of WEEE per capita is estimated as 3.1 kg in São Paulo (Rodrigues et al., 2015) and 4.0 kg in São José dos Campos (Abbondanza and Souza, 2019).

In 2019, the final destination of approximately 83% of the WEEE generated worldwide was uncertain. For middle- and low-income countries, such as Brazil, the majority of this waste is recycled via the informal sector through cooperatives and waste pickers (Forti et al., 2020). In 2018, an estimated 145 t of WEEE was collected by informal collectors in Brazil. WEEE is more commonly collected by informal collectors than other waste types (ANCAT, 2018).

A large part of the Brazilian population discards electronic waste as recyclable or common waste (Green Eletron, 2021). In 2021, there was an estimated 76.1 million tons of municipal solid waste (MSW), of which 60.2% underwent adequate disposal (i.e., sanitary landfill) and 39.8% underwent inadequate disposal (i.e., dumps and controlled landfills) (ABRELPE, 2021). In 2018, the São José dos Campos sanitary landfill received 173,664 t of MSW and 29 t of WEEE that was incorrectly disposed of and had not undergone selective collection (URBAM, 2018).

An environmentally adequate final disposal describes distribution of waste to sanitary landfills to avoid damage or risk to public health and to minimise adverse environmental impacts (BRASIL, 2010). Most of the MSW collected in Brazil goes to landfills for non-hazardous waste. In 2020, São Paulo had 346 sanitary landfills and 585 cities in the state that adequately disposed of waste, while 53 inadequately disposed of waste in open dumps or open-air deposits. Waste disposal in São José dos Campos is considered adequate (CETESB, 2020), and the leachate generated is stored in specific centres and transported by trucks to ETE Lavapés – SABESP, a wastewater treatment plant (URBAM, 2022.).

Final disposal areas cannot be considered as the final destination for substances contained in the waste because water can percolate through the waste, leading to contaminant leaching (Celere et al., 2007). Landfill leachates include high concentrations of heavy metals, suspended solids, and organic compounds, and the composition depends on rainfall, contact time, and waste characteristics. The highly soluble substances present in WEEE can reach the surface and infiltrate the soil, compromising the quality and potential uses of groundwater (Jiang et al., 2014).

The main impacts associated with dumps are contamination of soil and groundwater by leachate, in addition to increasing the number of vector-borne diseases (Mahmood et al., 2022). Leachate from landfills and open dumps contains non-biodegradable substances that accumulate in soil, water, plants, and animals. It includes heavy metals, such as Cd, As, Cu, Hg, Co, and Pb (Afonso, 2018).

In 2019, the Ministry of the Environment introduced the Zero Dump Programme, an initiative that aims to comply with the federal directive to eliminate existing dumps. The programme objectives include adequate environmental disposal of MSW and strengthening of reverse logistics processes. By 2020, there was a 17% reduction in the number of dumps compared with 2019. The new Legal Framework for Basic Sanitation also encouraged municipalities to seek appropriate forms of waste disposal and avoid use of dumps (ABETRE, 2020).

2. Material and Methods

A laboratory-scale experiment was conducted to simulate landfills and open dumps. The experiment was divided into three main phases: pre-experiment, operation, and leachate analysis (Figure 1).

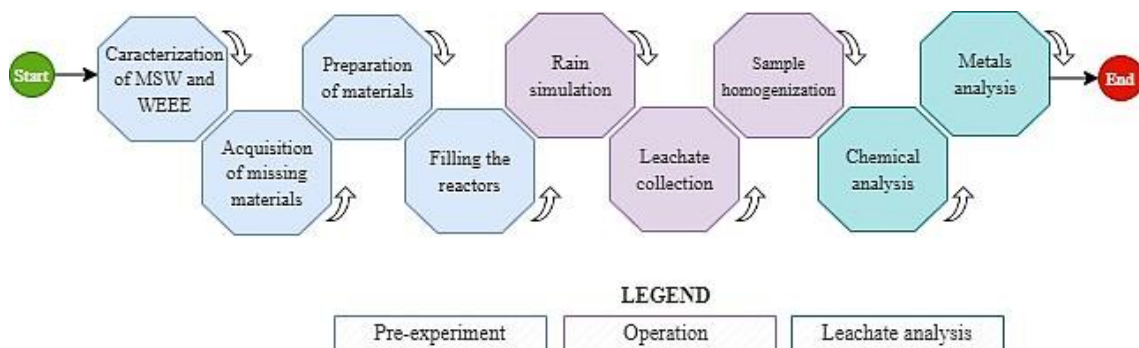


Figure 1. Summary of laboratory-scale experimental process

Landfill and open dump disposal conditions were simulated in five column reactors made of 200 mm diameter polyvinyl chloride (PVC) tubes. Three reactors were 100 cm tall, while two reactors were 150 cm tall to accommodate alternating waste and soil layers. The reactors were supported by a base of solid wood and fixed to a steel structure. Amber glass bottles were placed at the base of the reactors to collect leachate, which was transported by rubber hoses connected to the bottom of the reactors. The following scenarios were simulated (Figure 2): (1) soil only (white); (2) open dump with MSW + WEEE; (3) open dump with only WEEE; (4) landfill with MSW + WEEE; and (5) landfill with only WEEE.

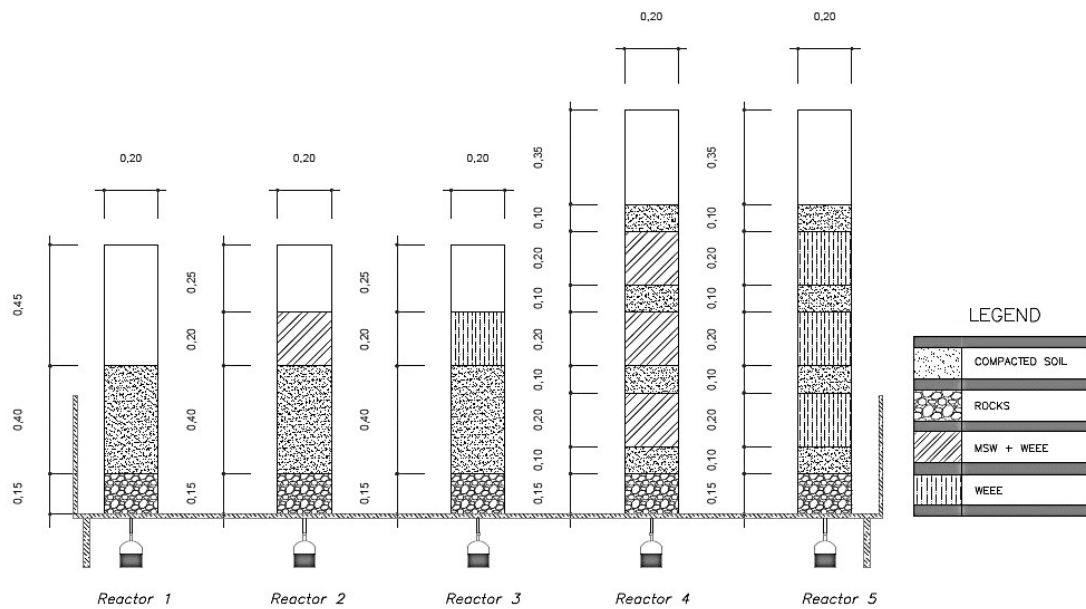


Figure 2. Schematic of research reactors (dimensions in metres).

The height of the waste layers was determined and scaled to the height of the tallest reactors to take advantage of the entire length, maintaining 15 cm of gravel in the bottom layer and >25 cm of soil in the top layer for safety reasons. The soil used in the reactors is natural from São Jose dos Campos and was characterised according to Brazilian standards NBR 7182 and NBR 7180 (ABNT, 2016a; ABNT, 2016b), NBR 6459, and NBR 7181 (ABNT, 1984a; ABNT, 1984b).

To estimate water infiltration, this study used the water balance method (Subramanian and Abraham, 2019) and rainfall data from a 30-year historical series of the city published by Climatempo (2021). The amount of water needed to simulate precipitation in each reactor was determined (Table 1) considering the six rainiest months of the year (October to March) and the reactor surface area. Owing to national restrictions during the COVID-19 pandemic, the experiment was carried out from July 2021 to January 2022, with the first leachate collected in August 2021.

Month	Water quantity (litres)	Weekly quantity (litres)
October	3.30	0.83
November	3.32	0.83
December	5.50	1.38
January	5.26	1.32
February	4.11	1.03
March	3.62	0.91

Table 1. Quantity of water needed to simulate precipitation in each reactor considering rainfall data from October to March 2021.

To determine the amount of each waste type needed in each reactor, gravimetry data from São José dos Campos were used (URBAM, 2018), along with the density of each type of waste obtained by CEMPRE (2014) or calculated in the laboratory. Using these values and considering the specified reactor heights, the mass of each waste type was determined. Table 2 lists the total material weights used in the reactors.

Category	Quantity required (kg)	% total	Category	Quantity required (kg)	% total
Aluminium	0.10	0.36	Mixed plastic	0.89	3.19
Ferrous metals	0.32	1.15	Radiological film	0.00	0.01
Paper	1.01	3.60	Polystyrene	0.13	0.47
Cardboard	0.73	2.61	Glass	0.59	2.10
Newspaper	0.07	0.23	Wood	0.38	1.33
Tetra Pak	0.20	0.72	Rubber	0.03	0.11
High-density polyethylene	0.13	0.45	Tissue	1.24	4.42
PET	0.23	0.82	Others	3.53	12.62
Plastic film	2.35	8.41	Organic material	15.08	53.86

Table 2. Amount of waste required by type

Bulky materials were manually fragmented into particles with an area of approximately 8 cm² using industrial scissors. Less bulky materials were kept at their original size. The materials were then homogenised and quartered before being transferred to the reactors. The soil layers in the landfill simulations had similar heights to the operational coverage in sanitary landfills (ABNT, 2010), and the amount of soil was equal in all reactors. To avoid clogging, all reactors were equipped with protection grids in the leachate outlet region. For reactors 4 and 5, which simulated landfills, the waste fraction was divided into layers, compacted using a 2 kg compactor, and covered by soil layers. In reactors that simulated open-air conditions, the waste fraction was placed in the top layer and compacted.

In each of the reactors, 840 g of mixed electronic waste was introduced, including batteries, cathode-ray tube monitor glass, liquid-crystal displays, wires, green printed circuit boards (PCBs), and brown PCBs. This mixture was prepared by manually dismantling and fragmenting the materials into particles with a maximum surface area of 10 cm². The distribution of the materials varied according to the type of simulated

disposal. For sanitary landfills, WEEE + MSW were jointly distributed between soil layers (base and cover). For open dump conditions, WEEE was distributed over a single layer of soil.

The generated leachate was collected weekly, and samples were homogenised and measured monthly. Measurements included the temperature, the pH (using a pH-meter mPA-210), and conductivity (using a conductivity meter mCA-1501). After pH adjustment, samples were refrigerated and sent for chemical analyses. At the LAREX laboratory at USP, analysis of the metals from undigested samples was performed according to the American Standard Method 3015A (USEPA, 2007) using the Optima™ 8300 ICP-OES spectrometer, which has detection limits of 1–100 ppb. The other samples were digested in acid with the CEM-MARS 6 from ICT-UNESP using two solutions: HNO₃+HCl and HNO₃ (USEPA, 2007). The acid-digested samples were later analysed by ICP-MS at the FEB-UNESP laboratory, which has detection limits of the order of parts per trillion (ppt). The analyses focused on quantifying Ag, Al, Ba, Be, Ca, Cd, Cr, Co, Cu, Fe, Hg, K, Mg, Mn, Ni, Pb, Sb, Se, Sn, and Zn.

The results were compared between the two methodologies and with results obtained in other landfill simulation studies. The results were also compared with Brazilian legislation and leachate from Brazilian landfills.

3. Results and Discussion

At the beginning of the experiment, the operation of reactor 4 was interrupted due to an internal blockage of the leachate outlet. Thus, the study continued using the other four operating reactors. Average values, pH values, and conductivity results for the leachate generated are listed in Table 3.

Average quantity of leachate generation (l)			
T1	T2	T3	T5
1.90 ± 0.87	1.97 ± 0.76	2.23 ± 1.04	1.72 ± 0.77
Average pH values			
T1	T2	T3	T5
7.40 ± 0.26	6.81 ± 0.22	7.03 ± 0.26	6.60 ± 0.54
Average conductivity values (uS/cm)			
T1	T2	T3	T5
274.47 ± 90.76	376.05 ± 69.09	203.42 ± 52.21	170.84 ± 66.64

Table 3. Average quantity, pH, and conductivity for the experimentally generated leachate.

Following the method proposed by [Casagrande \(1948\)](#), it was determined that the soil was predominantly silt. The quantity of leachate generated during the experiments was approximately 48% of the total quantity of added water. The remaining water was retained in the soil or lost through evaporation, as observed by [Li et al. \(2009\)](#). The measured pH values were close to neutral and similar in all reactors. The measured conductivity values were lower than those observed in the leachate from Brazilian landfills, likely due to an increased decomposition by organic matter in the physical landfills, since these landfills have been in operation for years ([Morais et al., 2019](#)).

Due to cost constraint, some strategies were not used to reduce the preferential water flow path. Therefore, voids accumulated between the aggregates, creating a preferential path for infiltrating water to reach the surface ([Li et al., 2021](#)), which may have influenced the leaching results.

For leachate samples with acid digestion (Figure 3) and without digestion (Figure 4), the maximum observed metal concentrations were greater than the value observed in the 'blank' tank (T1). None of the reactors recorded a metal concentration higher than the limit established for the release of effluents in Brazil ([CONAMA, 2011](#)). Although the experimental values were lower than the official limit, it is important that producers take responsibility for their disposal (Extended Producer Responsibility) in view of the daily quantities of generated WEEE ([Lundgren, 2012](#)). Furthermore, shared responsibility for the lifecycle of products is a constitutive principle of the PNRS ([BRASIL, 2010](#)).

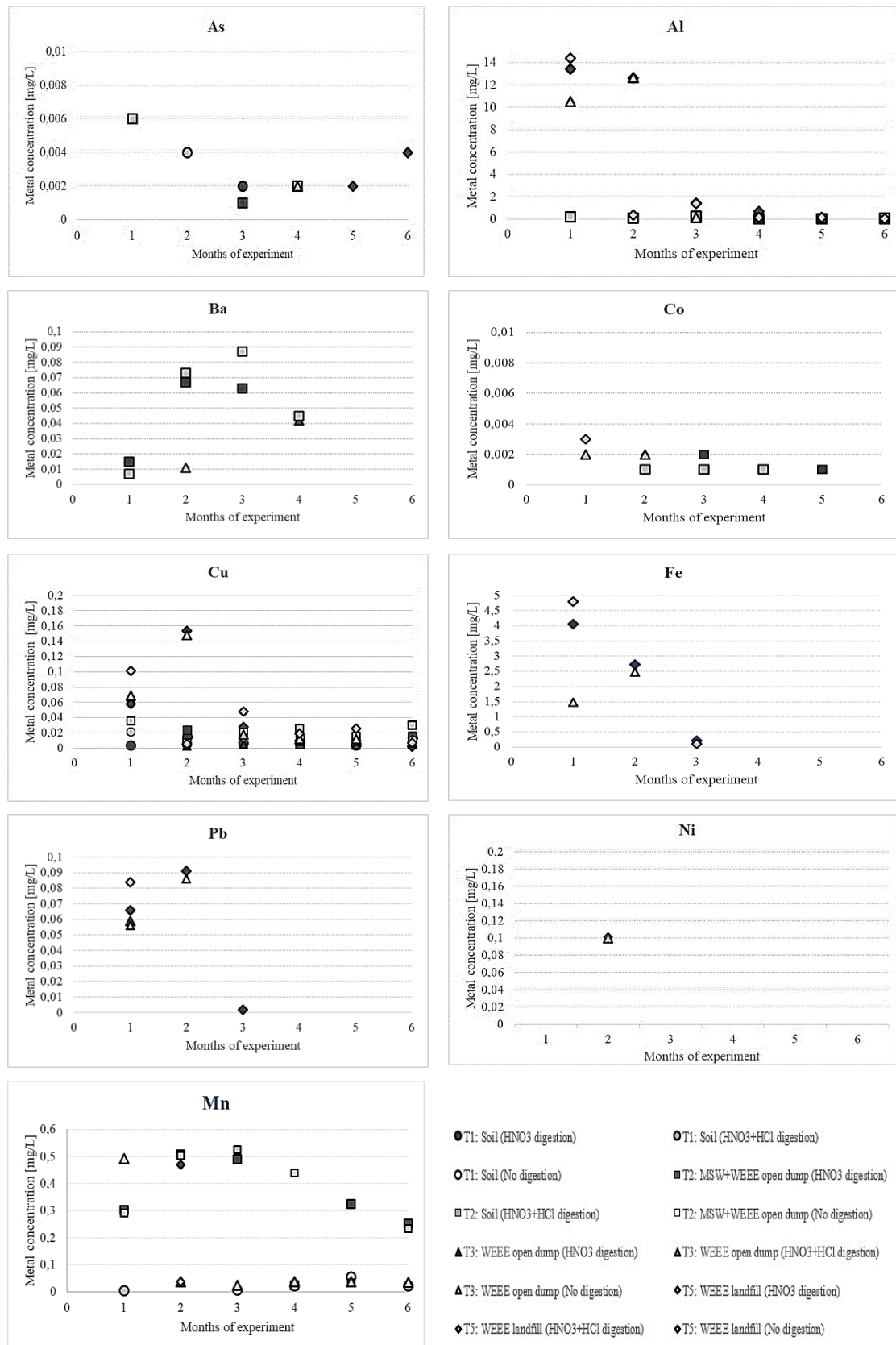


Figure 3. Results of chemical analyses of leachate samples with acid-digestion

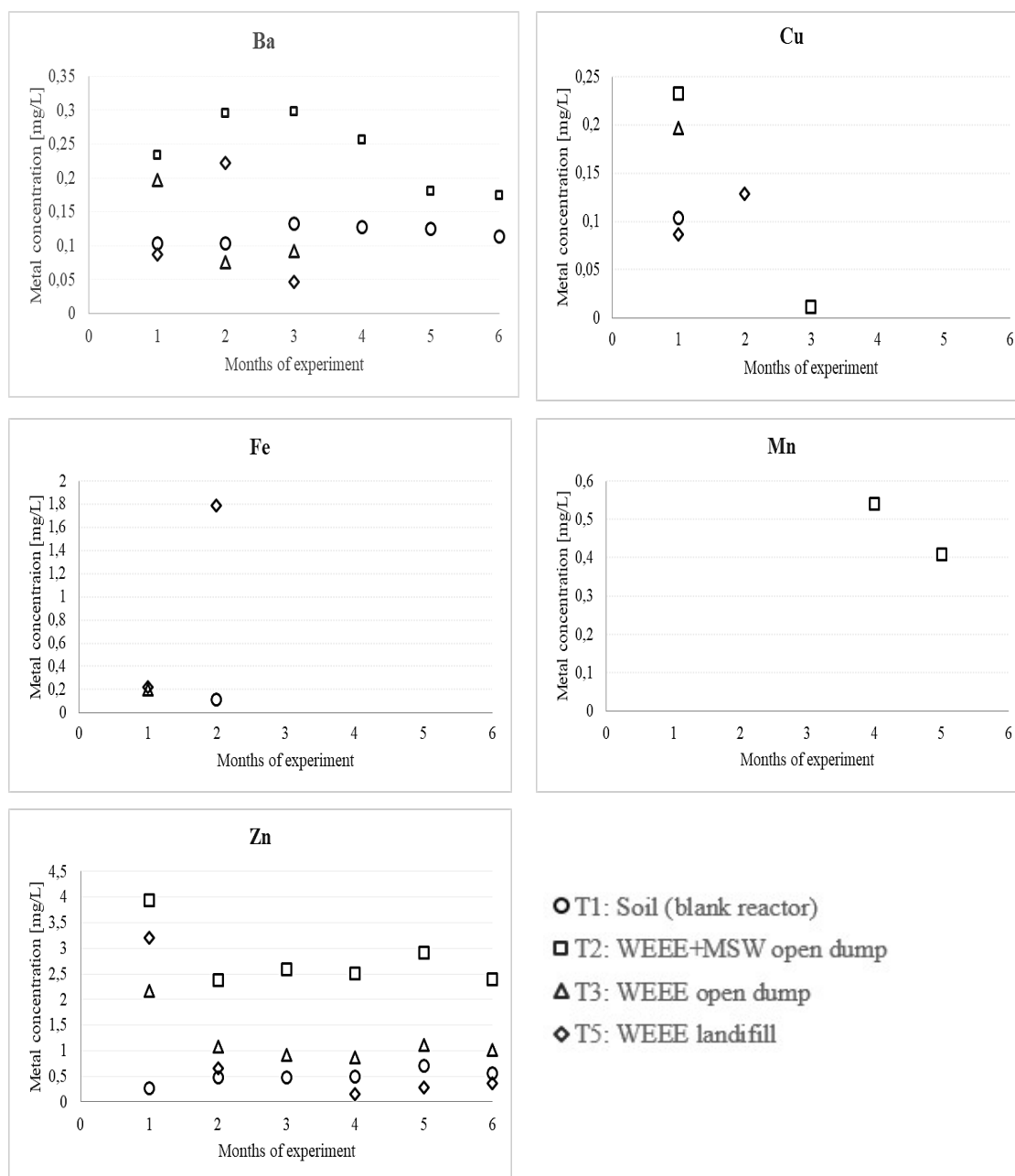


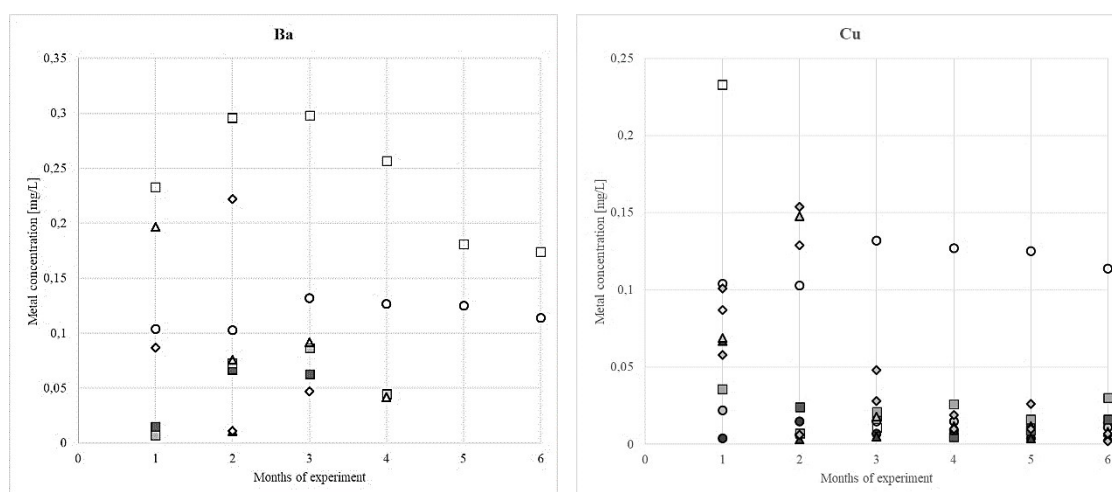
Figure 4. Results of chemical analyses of leachate samples without acid digestion

Pb and Cu were highly concentrated in reactors with only WEEE (T3 and T5), indicating possible interference of MSW in the leaching of these metals (as observed by Li et al., 2009). The highest concentration of Mn was observed in the scenario where MSW was present, which may indicate an influence of MSW on Mn availability, as observed in other studies (e.g. Karim et al., 2017), where the highest bioavailability of Mn, Pb, and Cu was related to open dumps or landfills with a high amount of organic matter and more than ten years of operation.

For the proportions of waste and the timescales tested in this study, all metals were found in lower concentrations than those found in leachates from Brazilian landfills (e.g. [Morais et al., 2019](#)). However, the evaluated Brazilian leachates came from landfills in operation for more than 7 years, which influences the availability of nutrients and metals in the leachate ([Gomes, 2009](#); [Gomes et al., 2007](#)).

Al, Ba, Cu, Fe, and Pb showed peak concentrations and then decreased, as in other simulations of landfills containing WEEE. However, about a year later in these studies, the metals reappeared at similar concentrations (or higher) than those at the beginning of the experiment, indicating that the timescale of observation and sample collection interfered with the results ([Kiddee et al., 2013](#); [Li et al., 2009](#)). In addition, some metals are more or less soluble depending on their fraction size ([Jiang et al., 2014](#)).

Metals identified in both analyses with and without acid digestion are shown in Figure 5. The concentration of Cu in T1 without digestion was higher than that in the other reactors. A possible explanation is that the soil was previously contaminated with metals. Silty soils have a high capacity to retain metals among aggregates ([Cesar et al., 2012](#); [Malara and Oleszczuk, 2013](#)) and the soil had not been previously analysed. In addition, the Cu^{+1} cation is insoluble depending on its combination with other anions ([Forte et al., 2019](#)).



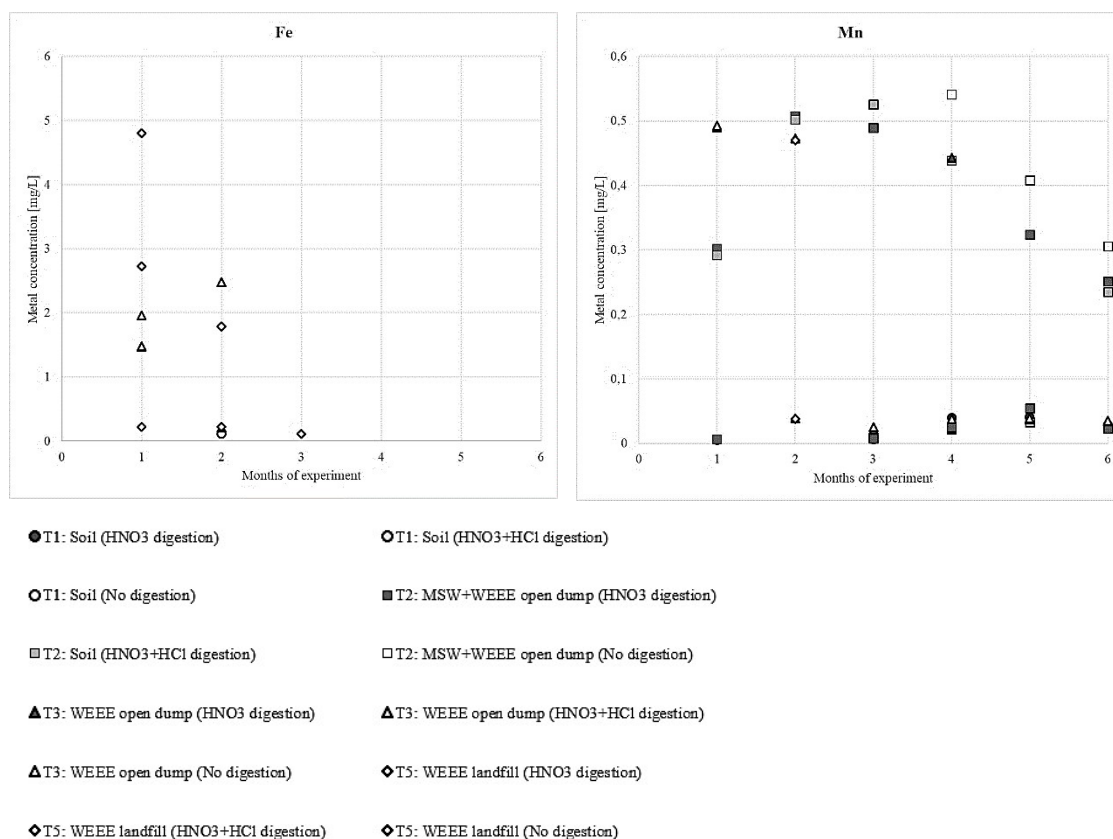


Figure 5. Metals found in both analyses

These results agree with previous studies that simulated landfills containing WEEE and MSW and considered acid digestion in leachate samples. However, Zn was only detected in undigested samples, despite the high concentration of this metal in digested samples from previous studies (Visvanthan et al., 2010; Li et al., 2009). This may indicate an influence of observation timescale on metal availability in the leachate. In previous studies, Zn had only appeared after 6 months of analysis.

The chelate effect, defined as the ability of ligands to form more stable complexes, needs to be considered (Sengupta et al., 2018). Some metal ions, such as Zn, Cu, Cd, Fe, Co, Ni, Al, Fe, and Pb, are known as chelating agents and the resulting complex is termed a chelate (Norwell, 2018). When this effect occurs, the extraction rate of Co, Zn, Pb, and Cd in leachates decreases considerably (Xu and Chen, 2015).

4. Conclusions

Based on the results presented here, the simulations were adequate for the proposed objective. Toxic metals were mostly found in leachates from reactors that

contained WEEE, indicating leaching of the metals in electrical devices. The highest metal concentrations were observed in sanitary landfill conditions, but the variation was small compared with simulations of open-air deposits, demonstrating that the method of final disposal did not affect the metal concentration in leachate. However, it was observed that most of the metals were not present in scenarios containing MSW, indicating that MSW can absorb metals or their decomposition may facilitate absorption by the soil.

Finally, most of the metals identified were found in digested samples. The correlation between metal concentration and acid digestion was dependent on the specific element. For example, Ba was found in higher concentrations in samples without acid digestion, whereas Zn was only found in acid-digested samples.

Some cost-associated limitations affected aspects of the experimental design, including the experimental duration (which limited observation of peak metal concentrations and discussion of the effect of material aging), the transition between layers (soil and liquid), and the absence of hydrodynamic profile analysis.

In future studies, a second leachate outlet is recommended in case of clogging of the main hole. Analysis of the simulated solids (MSW, WEEE, and soils) would enable verification of metal absorption in these materials and evaluation of the two methods of leachate sample preparation (i.e., with and without acid digestion).

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References

- Abbondanza, M.N.M., Souza, R.G., 2019. Estimating the generation of household e-waste in municipalities using primary data from surveys: A case study of Sao Jose dos Campos, Brazil. *Waste Manag.* 85, 374–384. <https://doi.org/10.1016/j.wasman.2018.12.040>.

- 310 ABRELPE, 2021. Panorama dos resíduos sólidos no Brasil,
311 <https://abrelpe.org.br/panorama-2021/> Acess in. 02, 2022.
- 312 Afonso, J.C., de Equipamentos Eletroeletrônicos, R., 2018. O antropocentro bate à nossa
313 porta. *Rev. Virtual Quim.*, sixth ed., ISSN: 1984-6835. 10.
314
- 315 Associação Brasileira De Normas Técnicas – ABNT. NBR 6459 – Determinação do
316 Limite de Liquidez, 1984a. NBR 7181 – Granulometria, 1984b. NBR 15849 – Resíduos
317 Sólidos Urbanos – Aterros sanitários de pequeno porte - diretrizes NBR 7182 –
318 Compactação dos Solos, 2016a e NBR 7180 – Determinação do Limite de Plasticidade,
319 2016b.
- 320 Associação Brasileira De Tratamento De Resíduos E Afluentes – ABETRE, 2019. Um
321 retrato da realidade brasileira dos resíduos sólidos. [https://abetre.org.br/atlas-da-](https://abetre.org.br/atlas-da-destinacao-final-de-residuos-brasil-2020/)
322 [destinacao-final-de-residuos-brasil-2020/](https://abetre.org.br/atlas-da-destinacao-final-de-residuos-brasil-2020/) Acess in. 05, 2022
- 323 Associação Nacional dos catadores e catadoras de Materiais Recicláveis – ANCAT,
324 2018. Anuário de Reciclagem 2017-2018, [https://cempre.org.br/wp-](https://cempre.org.br/wp-content/uploads/2020/11/2-Anu%C3%A1rio-daReciclagem.pdf)
325 [content/uploads/2020/11/2-Anu%C3%A1rio-daReciclagem.pdf](https://cempre.org.br/wp-content/uploads/2020/11/2-Anu%C3%A1rio-daReciclagem.pdf) Access in. 07, 2021.
326
- 327 BRASIL - Ministério do Meio Ambiente, Brasil. Brasília, DF, 2010. Política Nacional
328 dos Resíduos Sólidos - Lei nº 12.305/2010.
329 http://www.planalto.gov.br/ccivil_03/_ato2007-2010/2010/lei/112305.htm. Acess in. 07,
330 2021.
331
- 332 Casagrande, A., 1948. Classification and identification of soils. *T. Am. Soc. Civ. Eng.*
333 *ASCE (Trans.)*. 113, 901–930. <https://doi.org/10.1061/TACEAT.0006109>.
- 334 Celere, M.S., Oliveira, A.S., Trevilato, T.M.B., Segura-Muñoz, S.I., 2007. Metals in
335 landfill leachate in Ribeirão Preto, São Paulo, Brasil, and its relevance for public health
336 *Cad. Saúde Publ.* 23, 939–947.
337
- 338 CEMPRE, 2014. Guia da Coleta Seletiva. 2. Ed.: 12. [https://cempre.org.br/wp-](https://cempre.org.br/wp-content/uploads/2020/11/4-GuiadaColetaSeletiva2014.pdf)
339 [content/uploads/2020/11/4-GuiadaColetaSeletiva2014.pdf](https://cempre.org.br/wp-content/uploads/2020/11/4-GuiadaColetaSeletiva2014.pdf) Acess in. 04, 2020.
- 340 CEMPRE, 2018. Lixo Municipal: Manual de Gerenciamento Integrado: 4. ed.
341 http://cempre.org.br/upload/Lixo_Municipal_2018.pdf. Acess in. 04, 2022.
- 342 Cesar, R., Silva, M., Colonese, J., Bidone, E., Egler, S., Castilhos, Z., Polivanov, H., 2012.
343 Influence of the properties of tropical soils in the toxicity and bioavailability of heavy
344 metals in sewage sludge-amended lands. *Environ. Earth Sci.* 66, 2281–2292.
345
- 346 Cesaro, A., Belgiorno, V., Gorrasi, G., Viscusi, G., Vaccari, M., Vinti, G., Jandric, A.,
347 Dias, M.I., Hursthouse, A., Salhofer, S., 2019. A relative risk assessment of the open
348 burning of WEEE. *Environ. Sci. Pollut. Res. Int.* 26, 11042–11052.
349 <https://doi.org/10.1007/s11356-019-04282-3>.
350
- 351 Characterization and substance flow analysis in waste treatment processes. *Sci. Total*
352 *Environ.*, v. 463, 1124–1132, <http://dx.doi.org/10.1016/j.scitotenv.2012.07.078>

- 353 Climatempo, 2021. Climatologia e histórico de previsão do tempo em São José dos
354 Campos – BR. <https://www.climatempo.com.br/climatologia/553/saojosedosc campos-sp>
355 Acess in. 02, 2020.
- 356
- 357 Companhia Ambiental do Estado de São Paulo - CETESB, 2020. Inventário Estadual de
358 Resíduos Sólidos – 2020. [https://cetesb.sp.gov.br/residuossolidos/wp-](https://cetesb.sp.gov.br/residuossolidos/wp-content/uploads/sites/26/2021/07/Inventario-Estadual-de-Residuos-Solidos-Urbanos2020.pdf)
359 [content/uploads/sites/26/2021/07/Inventario-Estadual-de-Residuos-Solidos-](https://cetesb.sp.gov.br/residuossolidos/wp-content/uploads/sites/26/2021/07/Inventario-Estadual-de-Residuos-Solidos-Urbanos2020.pdf)
360 [Urbanos2020.pdf](https://cetesb.sp.gov.br/residuossolidos/wp-content/uploads/sites/26/2021/07/Inventario-Estadual-de-Residuos-Solidos-Urbanos2020.pdf). Acess in. 08, 2021.
- 361
- 362 Conselho Nacional do Meio Ambiente, - CONAMA. Resolução nº 430, de 13 de Maio de
363 2011. <http://conama.mma.gov.br/> Acess in: 05, 2022.
- 364 Forte, C.M.S., Pacheco, L.C.M., Queiroz, Z.F., 2019. Química Analítica: I. 2. Ed. Editora
365 da Universidade Estadual do Ceará – EDUECE.
- 366 Forti, V., Baldé, C.P., Kuehr, R.Bel G., 2020. The Global E-Waste Monitor 2020:
367 Quantities, Flows and the Circular Economy Potential. *United Nations University Press*
368 *(United Nations University)/United Nations Institute for Training and Research*
369 *(UNITAR)*, Bonn/Geneva/Rotterdam.
- 370 Gomes, L.P., Sólidos, R., 2009. Estudos de caracterização e tratabilidade de lixiviados de
371 aterros sanitários para as condições brasileiras. PROSAB. 1. Ed.
372 [http://www.finep.gov.br/images/apoio-e-financiamento/historico-](http://www.finep.gov.br/images/apoio-e-financiamento/historico-deprogramas/prosab/prosab5_tema_3.pdf)
373 [deprogramas/prosab/prosab5_tema_3.pdf](http://www.finep.gov.br/images/apoio-e-financiamento/historico-deprogramas/prosab/prosab5_tema_3.pdf) Acess in. 11, 2002.
- 374
- 375 Gomes, N.A., Almeida, M.V.A., Melo, M.A., Monteiro, V.E.D., OLIVEIRA, Influência,
376 R. de parâmetros físicos-químicos na composição de constituintes tóxicos em lixiviado
377 de aterro sanitário, 2017. Rev. Mater. 29. [https://doi.org/10.1590/S1517-](https://doi.org/10.1590/S1517-707620180003.0489)
378 [707620180003.0489](https://doi.org/10.1590/S1517-707620180003.0489).
- 379
- 380 Green Eletron. Resíduos eletrônicos no Brasil, 2021.
381 https://greeneletron.org.br/download/RELATORIO_DE_DADOS.pdf. Acess in. 02, 2021
382
- 383 Houessionon, M.G.K., Ouendo, E.D., Bouland, C., Takyi, S.A., Kedote, N.M., Fayomi,
384 B., Fobil, J.N., Basu, N., 2021. Environmental Heavy Metal Contamination from
385 Electronic Waste (E-Waste) recycling activities worldwide: A systematic review from
386 2005 to 2017. Int. J. Environ. Res. Public Health. 18, 3517.
387 <https://doi.org/10.3390/ijerph18073517>.
- 388
- 389 Jiang, S.Y.N., Yang, F., Chan, K.L., Ning, Z., 2014. Water solubility of metals in coarse
390 PM and PM 2.5 in typical urban environment in Hong Kong. Atmos. Pollut. Res., second
391 ed. 5, 236–244. <https://doi:10.5094/apr.2014.029>.
- 392 Karim, Md.R., Kuraoka, M., Higuchi, T., Sekine, M., Imai, T., 2017. Assessment of heavy
393 metal contamination from municipal solid waste open dumping sites in Bangladesh. Int.
394 J. Environ. Waste Manag. 19. <https://doi.org/10.1504/IJEW.2017.084297>.

- 395 Kiddee, P., Naidu, R., Wong, M.H., 2013. Metals and polybrominated diphenyl ethers
396 leaching from electronic waste in simulated landfills. *J. Hazard. Mater.* 252–253, 243–
397 249. <https://doi.org/10.1016/j.jhazmat.2013.03.015/>.
- 398 Lepawsky, J., McNabb, C., 2010. Mapping international flows of electronic waste. *Can.*
399 *Geogr.* 54, 177–195. <https://doi.org/10.1111/j.1541-0064.2009.00279.x>.
400
- 401 Li, M., Yao, J., Yan, R., Cheng, J., 2021. Effects Of Infiltration Amounts On Preferential
402 Flow Characteristics And Solute Transport In The Protection Forest Soil Of Southwestern
403 China. *Water*. 13, 1301. <https://doi.org/10.3390/w13091301>.
- 404 Li, Y., Richardson, J.B., Mark Bricka, R.M., Niu, X., Yang, H., Li, L., Jimenez, A., 2009.
405 Leaching of heavy metals from E-waste in simulated landfill columns. *Waste Manag.* 29,
406 2147–2150. <https://doi.org/10.1016/j.wasman.2009.02.005>.
- 407 Lundgren, K., 2012. The global impact of e-waste: Addressing the challenge. *International*
408 *Labour Office*, Programme on Safety and Health at Work and the Environment
409 (SafeWork), Sectoral Activities Department (SECTOR). International Labour
410 Organization, Geneva.
- 411 Mahmood, K., Faizi, F., Mushtaq, F., 2022. Satellite based bio-thermal impact insights
412 into MSW open dumps: A pair-unified proximity scenario. *Geomatics Natural Hazards*
413 *and Risk*, first ed. 13, 667–685. <https://doi.org/10.1080/19475705.2022.2038695>.
- 414 Malara, A., Oleszczuk, P., 2013. Application of a battery of biotests for the determination
415 of leachate toxicity to bacteria and invertebrates from sewage sludge-amended soil.
416 *Environ. Sci. Pollut. Res. Int.* 20, 3435–3446. [https://doi.org/10.1007/s11356-012-1268-](https://doi.org/10.1007/s11356-012-1268-3)
417 [3](https://doi.org/10.1007/s11356-012-1268-3).
- 418 Morais, M.M., Morais, G.A., Zoby Junior, L.C., Pereira Neto, L.M., Motta Sobrinho,
419 M.A. Análise de componentes principais para avaliar as características de lixiviados de
420 aterros de resíduos sólidos urbanos, 2019. *30º Congresso brasileiro de Eng. Sanit. E*
421 *Ambient*.
- 422 Norwell, W.A., 2018. Reactions of metal chelates in soils and nutrient solutions.
423 *Micronutrients Agric.*, 127–187, ISBN: 978-089118878-0, 978-089118797-4. *Sci.*, v.66,
424 2281–2292. <https://doi.org/10.136/sssabookser4.2ed.c7>.
425
- 426 Oguchi, M., Sakanakura, H., Terazono, A., 2013. Toxic Metals in WEEE.
427 Rodrigues, A.C., Gunther, W.M.R., Boscov, M.E.G., 2015. Estimativa da geração de
428 resíduos de equipamentos elétricos e eletrônicos de origem domiciliar: Proposição de
429 método e aplicação ao município de São Paulo, São Paulo, Brasil. *Eng. Sanit. Ambient.*
430 20, 437–447. <https://doi.org/10.1590/S1413-41522015020000133701>.
- 431 Sengupta, A., Seitz, A., Merz, K.M., 2018. *J. Am. Chem. Soc.* Simulating the chelate
432 effect. *J. Am. Chem. Soc.* 140, 15166–15169, <https://doi.org/10.1021/jacs.8b09371>.
- 433 Sepúlveda, A., Schluep, M., Renaud, F.G., Streicher, M., Kuehr, R., Hagelüken, C.,
434 Gerecke, A.C., 2010. A review of the environmental fate and effects of hazardous
435 substances released from electrical and electronic equipments during recycling:

- 436 Examples from China and India. *Environ. Impact Assess. Rev.* 30, 28–41,
 437 <https://doi.org/10.1016/j.eiar.2009.04.001>
 438
- 439 Subramanian, T.S., Abraham, M., 2019. Assessment of natural groundwater recharge: A
 440 case study of North Chennai Aquifer. *Environ. Geosci.*, second ed. 26, 41–50.
 441 <https://doi.org/10.1306/eg.01091918005>.
- 442 Urbanizadora Municipal – URBAM, 2020. Estudo de Caracterização Gravimétrica,
 443 2018. [https://www.sjc.sp.gov.br/media/34209/apresentacaourbam-caracterizacao-](https://www.sjc.sp.gov.br/media/34209/apresentacaourbam-caracterizacao-gravimetrica-sjc-2018.pdf)
 444 [gravimetrica-sjc-2018.pdf](https://www.sjc.sp.gov.br/media/34209/apresentacaourbam-caracterizacao-gravimetrica-sjc-2018.pdf). Access in: 10, 2021.
- 445 Urbanizadora Municipal – URBAM, 2022. Aterro Sanitário.
 446 https://www.urbam.com.br/aterro_sanitario. Access in. 07, 2021
- 447 US Environmental Protection Agency - USEPA. Method 3015a – Microwave Assisted
 448 Acid Digestion of Aqueous Samples and Extracts, 2007. *Office of Solid Waste and*
 449 *Emergency Response*, first revision.
- 450 Visvanathan, C., Yin, N.H., Karthikeyan, O.P., 2010. Co-disposal of electronic waste with
 451 municipal solid waste in bioreactor landfills. *Waste Manag.* 30, 2608–2614.
 452 <https://doi.org/10.1016/j.wasman.2010.08.006>.
- 453 Wagner, M.C.P., Baldé, V.L., Nnorom, I.C., Kuehr, R., Iattoni, G., 2022. Regional E-
 454 Waste Monitor for Latin America: Results for the 13 Countries Participating in Project
 455 UNIDO-GEF 5554, Bonn (Germany): 05. [https://ewastemonitor.info/regional-e-waste-](https://ewastemonitor.info/regional-e-waste-monitors/)
 456 [monitors/](https://ewastemonitor.info/regional-e-waste-monitors/) Access in. 07,2022.
 457
- 458 Wang, R., Xu, Z., 2014. Recycling of non-metallic fractions from waste electrical and
 459 electronic equipment (WEEE): A review. *Waste Manag.* 34, 1455–1469.
 460 <https://doi.org/10.1016/j.wasman.2014.03.004>.
- 461 Woo, S.H., Lee, D.S., Lim, S.R., 2016. Potential resource and toxicity impacts from
 462 metals in waste electronic devices. *Integr. Environ. Assess. Manag.* 12, 364–370, [http://](http://dx.doi.org/10.1002/ieam.1710)
 463 dx.doi.org/10.1002/ieam.1710.
 464
- 465 World Health Organization – WHO. Children and digital dumpsites. 2021.
 466 <https://www.who.int/publications/i/item/9789240023901>. Access in. 02, 2022.
- 467 Xu, Y., Chen, Y., 2015. Leaching heavy metals in municipal solid waste incinerator fly
 468 ash with chelator/biosurfactant mixed solution. *Waste Manag. Res.* 33, 652–661.
 469 <https://doi.org/10.1177/0734242X15592276>.
- 470 Yu, J., Welford, R., Hills, P., 2006. Industry responses to EU WEEE and ROHS
 471 Directives: Perspectives from China. *Corp. Soc. Responsib. Environ. Manag.*, v. 13 n. 5,
 472 286–299, <https://doi.org/10.1002/csr.131>

3.12 Artigo 12 – Indium recovery from end-of-life e-waste: chemical determination and recycling viability *

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ABSTRACT

The interest in In recycling has been increasing in the last 5 years, mainly due to its strategic, economic importance and scarcity in the Earth's crust. This element can be found in flat panel displays (FDPs), especially in those devices that employ technology based on liquid crystal displays (LCD). The aim of this study was to investigate a simple analytical method to extract the In element from LCD screen. It was possible to solve some In determination problems by ICP OES and, propose an optimized method using design of experiment (DoE). The samples were ground with the polymer adhered to the substrate and, it was possible to mix three types of samples (smartphone, monitors and TV screens). Leaching was performed with 1 mol/L H₂SO₄ and 25% of sample using magnetic plate with stirring and temperature (~ 80°C). With this procedure, almost 100% of In was extracted and the DoE was used successfully. In addition, interviews were conducted with key operators for urban mining and other segments in Brazil, where it was observed that there is no consolidated market for recycling LCD screens with a focus on In recovery in the country.

Keywords: Chemical recycling; Hydrometallurgical recycling; Liquid Crystal Display; material recovery; E-waste.

Indium recovery from end-of-life e-waste: chemical determination and recycling viability

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Abstract

The interest in In recycling has been increasing in the last 5 years, mainly due to its strategic, economic importance and scarcity in the Earth's crust. This element can be found in flat panel displays (FDPs), especially in those devices that employ technology based on liquid crystal displays (LCD). The aim of this study was to investigate a simple analytical method to extract the In element from LCD screen. It was possible to solve some In determination problems by ICP OES and, propose an optimized method using design of experiment (DoE). The samples were ground with the polymer adhered to the substrate and, it was possible to mix three types of samples (smartphone, monitors and TV screens). Leaching was performed with 1 mol/L H_2SO_4 and 25% of sample using magnetic plate with stirring and temperature ($\sim 80^\circ\text{C}$). With this procedure, almost 100% of In was extracted and the DoE was used successfully. In addition, interviews were conducted with key operators for urban mining and other segments in Brazil, where it was observed that there is no consolidated market for recycling LCD screens with a focus on In recovery in the country.

1. Introduction

The industries that produce electrical and electronic equipment (EEE) operated for a long time in an economy system based on the linear model (extraction-production-use-disposal of material), transforming natural resources into products that end up being discarded in the environment. This observation in combination with the massive use of natural resources, has led to increasing impacts on the environment.¹ Due to these problems, the circular economy (CE) concept emerges as a new model based on rethinking and redesigning products and services, implementing the recirculation of natural resources within production value chains, in order to regenerate natural systems and reduce the generation of pollution and waste. Even with the growing interest in CE, the concept has gained relatively little attention in supply chain management, as indicated in the literature.²

In this way, CE can be a practical solution for the reuse of geologically scarce elements used in new technological products, such as smartphones, wind turbines and solar panels.^{3,4} These chemical elements, such as gallium (Ga), germanium (Ge), indium (In), tellurium (Te), tantalum (Ta) and platinum group metals (PGMs) are of particular concern in terms of geological scarcity and their difficult recovery in finished products.⁵ The element In presents special economic interest for several reasons, being classified as a “critical metal” due to scarcity and importance for the manufacturing industries of solar panels, LCD screens (Liquid Crystal Display) and other applications of emerging technologies.^{6,7}

LCD screens are one of the main constituents of cell phones, as well as other electronic products as HDTVs, monitors, laptops, digital watches, among other products. In a cell phone, 4.4% by weight corresponds to the LCD screen, 1.2% by weight is organic material (films adhered to the screens) and the rest of the mass corresponds to the other components. A typical LCD screen contains conductive electrodes made of indium tin oxide (ITO, or tin-doped indium oxide), which are located between two glass plates. ITO is a mixture of indium (III) oxide (In_2O_3) and tin (IV) oxide (SnO_2), with 80-90% by weight of In_2O_3 and 10-20% by weight of SnO_2 . In addition to ITO, LCD screens include glass and polymers in their composition, which can also be recovered.⁶⁻⁸

As mentioned by Ueberschaar et al.⁹, recycling of In from LCD screens is not yet carried out on an industrial scale, however, the authors comment that

many scientific papers have been carried out recently on the topic of In recovery from LCD screens. One of the most used approaches to In recovery is the transfer of ITO substrate to a liquid state via acidic dissolution.¹⁰ The main acids used are sulfuric (H_2SO_4), nitric (HNO_3) and hydrochloric (HCl). Most of the approaches tested use a high temperature for a better and more efficient reaction. Other concentration steps through hydrometallurgical procedures, such as solvent extraction, purify the In output and separate substances that are not of interest to the process.^{9,11-13}

Akcil *et al.*⁶ reported that in the period 2013-2035, the demand for In could be much higher than the recovery rate of end-of-life LCD screen technologies. The authors also highlight the need to improve existing In recovery technologies.⁶ In this way, and considering the disparity of supply and demand, global competitiveness, In supply security, stable price and to maintain the technologically advanced lifestyle, new recycling technologies should be a reliable solution. An economically viable, eco-efficient and sustainable recycling/recovery process for In can be viable from the concepts of CE, low carbon, clean energy and ecological perspective.^{6,8}

The aim of this study was to obtain more information about the procedure for sample preparation and determination of In *via* ICP OES in order to solve some analytical problems and propose a simple analytical method to extract this element from LCD screens. In addition, this research shed some light the potential In recycling market in Brazil.

2. Experimental

2.1. Overview of preliminary experiments

Two strategies to remove the polymer adhered to the substrate were performed: (1) thermal shock using an oven or hot air blower and, (2) pyrolysis at a temperature of 580-600°C for 30 minutes.

The first test for the In extraction was accomplished using a full factorial design 2^3 , with variables tested in two levels: (1) sample type (monitor and smartphone), (2) sample mass (300 and 500 mg) and (3) extractor type (aqua regia and inverted aqua regia). The process was performed in a digester block (Marconi, Brazil) with PFA closed vessels (Saville, USA) with a temperature of

90°C for 4 hours. The elements were determined by ICP OES (Thermo, iCAP 7000 model).

For the second test, experiments using a microwave (CEM, model: MarsXpress) system with more oxidizing concentrated acids (HNO_3) and complexing (HF) and subsequent neutralization using boric acid (H_3BO_3) were performed with the samples from pyrolysis. The elements were also determined by ICP OES.

Another studied strategy was (third test): extraction using diluted acids (HNO_3 , HCl and H_2SO_4) with samples from pyrolysis and milling, and samples from thermal shock. The extraction method was ultrasonic bath (Delta Ultrasons).

The next step was conducted with new samples from other forms of LCD pre-processing (fourth test): (1) with polymer and milling; (2) with polymer and without milling (samples in small fragments); (3) without polymer (thermal shock) and with milling; (4) without polymer and samples in small fragments. In this case, H_2SO_4 in three concentrations was tested (1, 4 and 7 mol/L). The extractions were made in digester block and magnetic plate (Fisatom, model 752A, Brazil) using temperature of 90°C and 2 hours of leaching.

2.2. Sample preparation

LCD screens from smartphones, monitors and TVs were acquired from local market in the São Paulo state, Brazil. The availability of screens of different sizes, manufacturers and models in electronic repair stores is remarkable, whereas in just one store, more than 500 smartphone screens were collected.

The disassembly of the collected material was performed manually. Table 1 presents some details of the dismantling and milling process, which was performed using a knife mill (TE-650 mill type Willye, Tecnal, Brazil).

Table 1. Quantity, mass balance (g) for TV, monitor and smartphone during the disassembly and milling process, and its sample size fractions description.

	TV	monitor	smartphone
Quantity of screens	1	7	212
ITO substrate (g)	1864	1807	1161
Polymers (g)	332	1147	854
Frames and other small parts (g)	254	1375	6100
Total mass (g)	2450	4329	8115
% ITO substrate in screen	76	42	14
% ITO substrate: > 600µm	50	45	29
% ITO substrate: between 212 and 600µm	31	33	50
% ITO substrate: <212µm	19	22	21

It is important to mention that the TV screen sample contained only the frontal part (without the ferrous from the rear), so the % ITO in screen was high (76%). Only the ITO substrate was milled, and Table 1 presents the particle size distribution. For TV and Monitor samples, 50% and 45% of the ITO substrate mass, respectively, presented particle size higher than 600µm and, for smartphone, 50% of the ITO substrate mass presented particle size between 212 µm and 600 µm. There is a polymer adhered to the ITO substrate, which was not removed during milling process (items 3.1 and 3.2 show more details).

In order to investigate each particle size of the sample, experiments were performed using the smartphone LCD sample (1) cut in small fragments; (2) with particle size higher than 600 µm; (3) between 212 µm and 600 µm and, (4) less than 212 µm. The experimental procedure was performed with 0.5g of the sample, 10 mL of H₂SO₄ 1 mol/L and digester block where the analytical frequency is higher than when microwave oven is considered. The temperature and time of leaching was 90°C and 2 hours, respectively. The samples were filtered and diluted for further ICP OES elements determination, where the analytical curve was prepared in the same medium as the samples using standards with high degree of purity. A standard addition procedure was also performed using In and Sn standards in order to calculate recovery.

2.3. Optimization of leaching process

Some variables were studied through Design of experiments – DoE (Doehlert approach¹⁴) in order to optimize the solid liquid ratio and extraction time and observe its interaction. Both variables are important in an industrial process: the higher the % of solids and the shorter the extraction time, the faster the process. Table 2 shows the Doehlert experimental design performed.

Table 2. Doehlert experimental design with two variables.

Experiments	Variable 1 (time, minutes)		Variable 2 (% of solid)	
	Coded	Real	Coded	Real
1	0	150	0	15
2	1	270	0	15
3	0.5	210	0.866	25
4	-1	30	0	15
5	-0.5	90	-0.866	5
6	0.5	210	-0.866	5
7	-0.5	90	0.866	25
8	0	150	0.433	20
9	0	150	-0.433	10

With the help of Doehlert design it was possible to evaluate both variables (time and % of solid) in different number of levels. The time (variable 1) for instance was studied from -1 (30 min) up to 1 (270 min) due to its clear importance in the recycling process, playing an important role in the final cost of the process. The % of solids was verified in 3 levels ranging from -0.866 (5%) up to +0.866 (25%).

The experiments were performed using H₂SO₄ 1 mol/L and magnetic plate with and without stirring, and temperature of 80°C; these conditions were optimized beforehand (more details in 3.1 and 3.2). The sample was the milled ITO substrate from smartphone. After extraction, the leachate solutions were filtrated and diluted for subsequent determination by ICP OES. In this case, just In was determined. The concentration from ICP OES was evaluated using a

regression model (Anova) and contour plot were prepared to visualize the best optimal conditions.

After these experiments, more were performed in order to verify the previous design. Fig. 1 shows the performed experimental domain of the design. The % of solids was tested in more levels: 25%, 30%, 40% and 50% and, the extraction time was tested in 30, 90 and 150 minutes. The experimental conditions were the same (H_2SO_4 1 mol/L and magnetic plate with stirring and temperature). Then, all the expected chemical elements were determined (Al, Ba, Ca, Cu, Cr, Fe, In, Mg, Ni, Sn, Sr, Si, Zn). The results were also evaluated using regression models (Anova) and contour plots were prepared to visualize the most favorable condition.

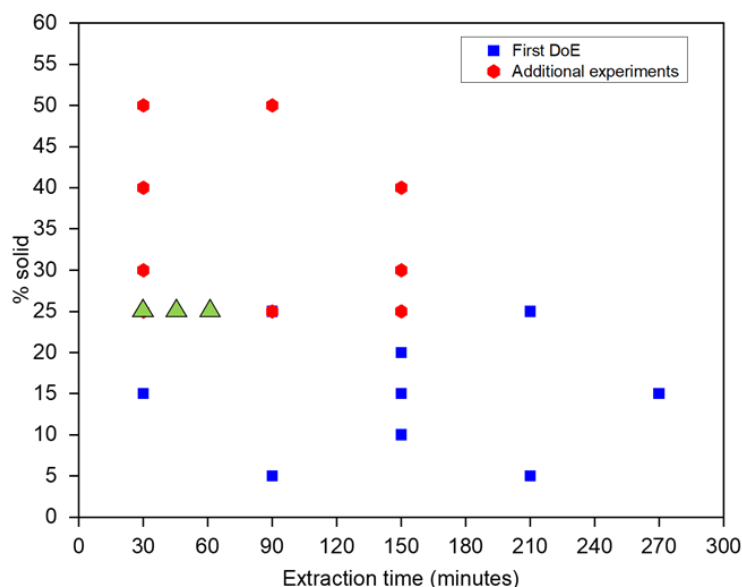


Fig. 1 Experimental domain of the first DoE (blue squares, see Table 2) and additional experiments (red and green lozenges and triangles, respectively).

After visualizing the optimal conditions according to DoE, it was necessary to perform additional three new experiments (see green triangles in Figure 1) to identify the most appropriate extraction time and to apply for the other samples:

- H_2SO_4 1 mol/L;
- Ratio of solid/liquid: 1g:3mL (25% of solid);
- Magnetic plate with stirring and temperature ($\sim 80^\circ\text{C}$);
- Sample: smartphone, monitor, TV, mix (smartphone, monitor and TV mixed). All the samples were ground and, to mix the three types, a homogenization was performed.

The extraction time levels tested were 30, 45 and 60 minutes. These additional experiments are depicted as green triangles in Fig. 1. After leaching, the samples were filtrated and diluted to further determination by ICP OES. The combination of extraction time of 30 min and % of solids of 25 was tested twice.

3. Results and discussion

3.1. Difficulties in the In determination

During several preliminary experiments, the In concentrations determined by ICP OES were very low when compared to those results observed in the scientific literature. In these experiments, the ITO substrate was adhered to the polymer and, our main hypothesis was that this could jeopardize the In determination. Therefore, some alternatives to remove the polymer adhered to the substrate were tested and more experiments were proposed (more details at item 2.1).

With thermal shock, while heated, the polymer can be easily removed. On the other hand, it cools down quickly, adhering on the substrate again. Furthermore, the substrate is a very brittle glass, which fragments in small parts, being an inviable process. Besides, during the process of pyrolysis, the polymer was totally removed, and the substrate was grinded for further mineralization.

For the first test (item 2.1), the In concentration ranged from 9mg/kg to 12mg/kg, and no difference in the concentration values among the experiments was observed. Even using a more energetic extraction method (second test, item 2.1), the results were not favorable with In concentration of 9mg/kg for monitor LCD samples and 13 mg/kg for smartphones.

In the third test (item 2.1), it was possible to vary the acids, acid concentration and extraction method, but there was no significant improving in the In concentration, ranging from 6mg/kg to 14mg/kg.

What could be happening to In determination? Could it be the sample pre-processing or the determination via ICP OES? After these experiments, it was noticed that the removal of the polymer from substrate was not interfering in the In determination. Therefore, the next steps were focused in preparing new samples by other pre-processing forms.

Even varying the pre-processing of the samples (fourth test, item 2.1), the In concentration remained low (from 3 to 8 mg/kg). But it was possible to visualize

that the samples with polymer and milling showed the best results, and there is no difference between leaching by digester block and magnetic plate. In addition, using H_2SO_4 1 mol/L as extractor, the best results for In determination were obtained.

In order to verify if the analytical problem was with the samples or with the determination, it was performed an extraction with a commercial (Aldrich, nº CAS 50926-11-9) indium and tin oxide, but even so the results were below expectations (recovery around 4%). Other test was a standard addition using an aqueous standard of In on a smartphone LCD sample using H_2SO_4 1 mol/L for leaching. This test also did not show good results. Therefore, it was concluded that the problem was in the determination by ICP OES.

3.2. How to solve the In determination problem?

A detailed study using standard addition was performed in order to investigate each experimental procedure steps. The experimental procedure employed a digester block with 90°C and 2 hours using 10 mL of H_2SO_4 1 mol/L. In this way, standard additions of 1mg/L of In and Sn were added in different steps from the procedure: (1) before and (2) after leaching (for the samples) and (3) in solutions containing only standard (without sample).

Other point was the calibration curve, which was previously prepared with the same acid used in the leaching, but its concentration was 1% v/v. In this case, the calibration curve was prepared now with the same concentration and acid. For example, after the leaching and further dilutions, the final solution is 0.1 mol/L or 0.55% v/v. Therefore, solutions for the ICP OES standards were also prepared at this concentration.

After all experiments, the recovery values ranged from 101 to 122% for In and, from 112 to 121% for Sn. The In concentrations observed for the samples were 188 mg/kg (samples with polymer and milling), 257 mg/kg (samples with polymer and small fragments), 142 mg/kg (samples without polymer and with milling) and 117 mg/kg (samples without polymer and small fragments).

With this study, it was possible to conclude that In determination is very sensitive to variations in acid concentration in the analytical curve, suffering a high transport effect in ICP OES determinations. From that point on, all calibration curve solutions were prepared exactly the same as the sample solutions.

3.3. Optimization of leaching process

3.3.1. Particle size of the samples

A study about the particle size was performed to identify the In concentration in each fraction of the sample (Item 2.2). Fig. 2 shows the In and Sn concentrations. With these results, was concluded that the samples with particle size less than 212 μm present In and Sn concentrations higher than the samples with larger particle size. When performed the grinding, it is noticed that the polymer is more concentrated in the particle size higher than 600 μm , but there is also In in this process.

To grind the entire mass of the sample to particle size smaller than 212 μm , more than one grinding step may be required. Therefore, it is better to operate with the sample containing all particle sizes as long as it is homogenized, avoiding loss of In in the other fragments. The sieve of the knife mill was 20 mesh (800 μm), being the particle size less than 800 μm .

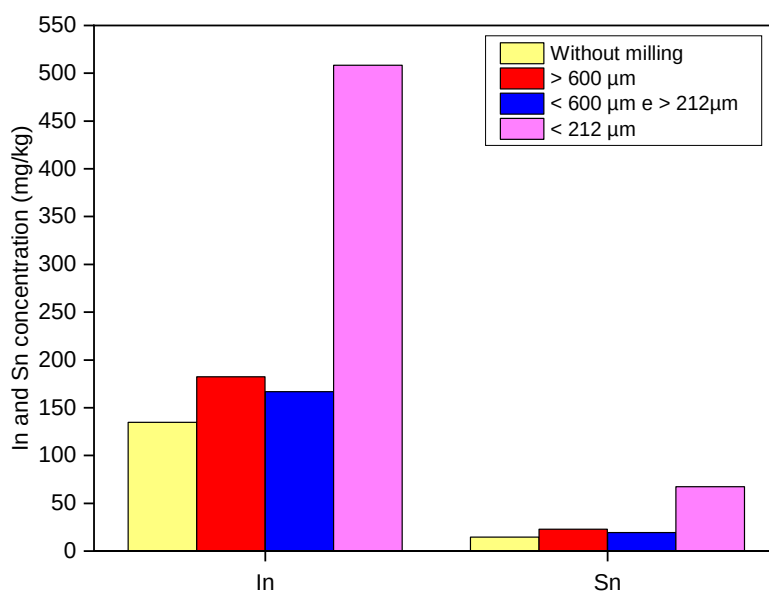


Fig. 2 In and Sn concentrations (mg/kg) in the smartphone LCD sample with different particle size.

3.3.2. Design of experiments

The extractor type and its concentration were well defined with the experiments mentioned at item 3.1: H_2SO_4 1 mol/L. Therefore, the first design of experiment studied the extraction time (v_1) and % of solid (v_2) (Table 2). The In

concentration obtained from ICP OES was used to calculate a regression model ($[In] = 258.11 + 25.75v_1 + 47.63v_2$), and Fig. 3 shows its contour plot.

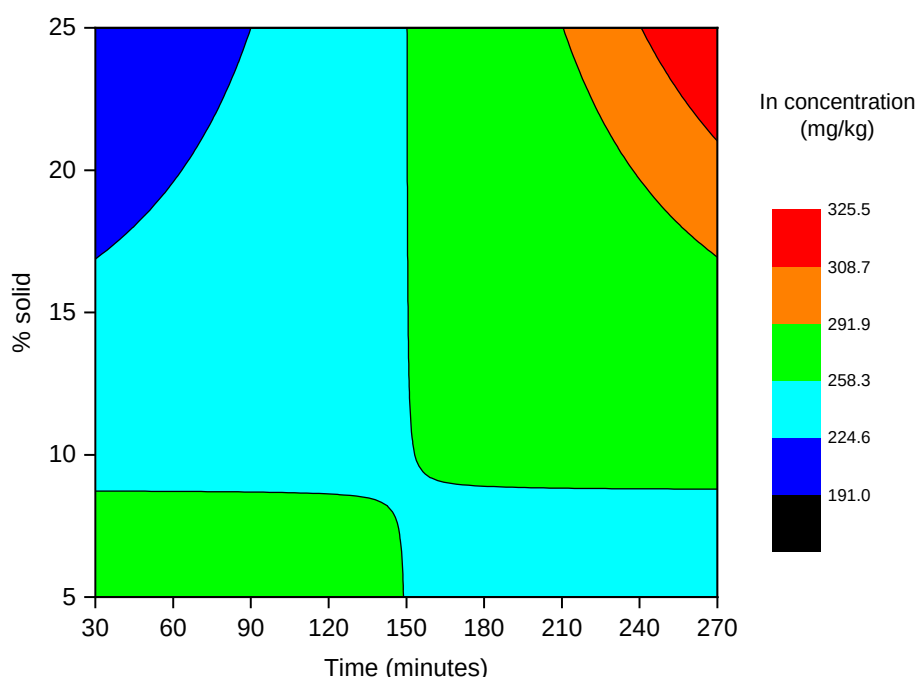


Fig. 3 Contour plot for the model calculated using the In concentration.

To obtain the highest In concentration, a high % of solid (positive aspect) and a long time of extraction (aspect that reflects a high cost) are required (red part from the graph). The red part presented at Figure 3 shows that the predicted In concentration can reach values around 300 mg/kg. On the other hand, there was not a huge variation among the different experiments. The In concentration obtained experimentally ranged from 228 mg/kg to 297 mg/kg.

These results are from the experiments using stirring, which were very similar to those without stirring. Even introducing an additional cost for the process, the stirring can be important to avoid saturation of the region of the solution close to the solid, requiring a movement of the solute in the medium, which is known as diffusion. Therefore, the next experiments were performed using the magnetic plate with stirring.

In order to investigate more about the % of solid (v_2) and extraction time (v_1), more experiments were made (Fig. 1), and Fig. 4 shows the contour plots using the regression models calculated. In this case, all chemical elements concentrations were considered and two regression models were calculated: one

for In concentration ($[In] = 300.98 + 33.76 v_1$) and other for all chemical elements using the desirability calculate ($D = 0.26 - 0.22 v_1$).¹⁵

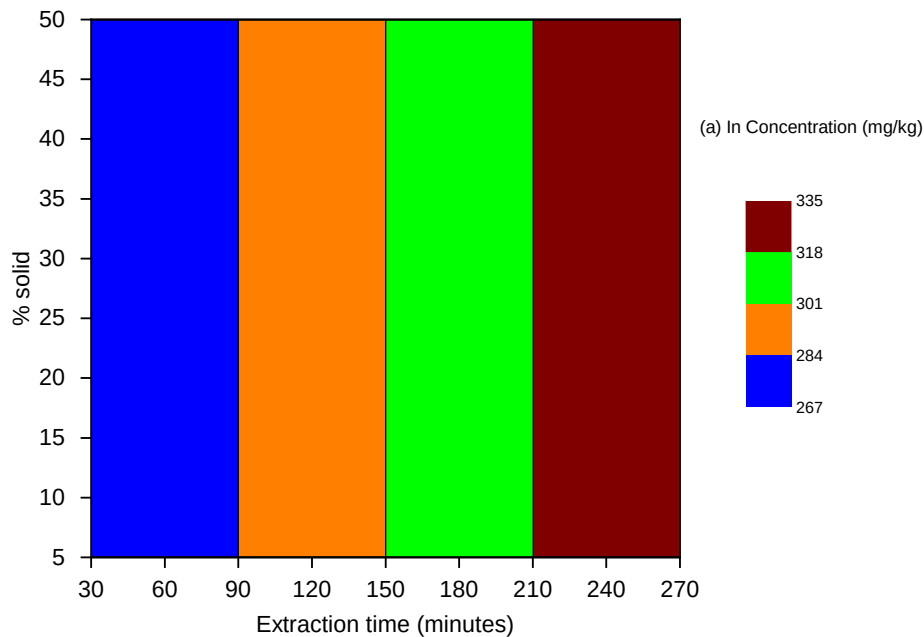
The desirability is calculated to code the responses obtained between 0 and 1. These parameters (0 and 1) describe the worse and the best responses, respectively. The goal was to maximize for In extraction and minimize for the other elements, i.e., the best condition is that leach more In and less the other elements, aiming at the further recycling steps. The equations used to desirability are:

$$d = \left(\frac{y-L}{T-L} \right)^s \text{ (maximize),}$$

where y is the monitored response, L is the lower acceptable working condition, T is the target value and, s is the weight.

$$\text{To minimize, } d = \left(\frac{U-y}{U-T} \right)^s,$$

where U is the acceptable response. For the model with all chemical elements (14 responses), it is necessary to calculate the individual desirability of each one, and calculate the geometric mean generating the global desirability (D_{global}), which is used in the regression model calculation.



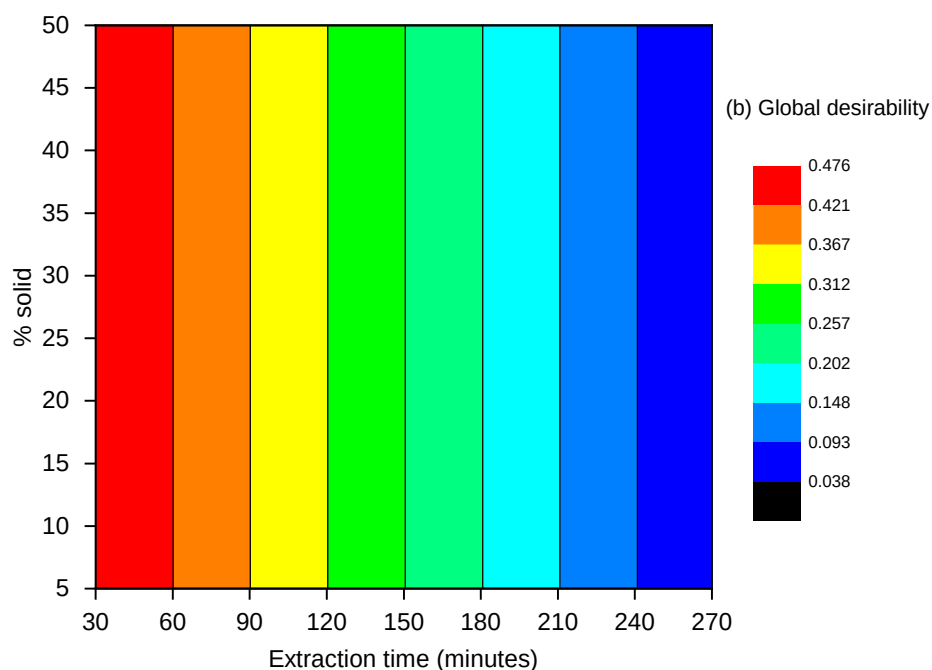


Fig. 4 Contour plots for (a) In concentration, (b) all chemical elements using the global desirability.

In conclusion, the % of solid does not statistically interfere in the result. When the % of solid is greater than 25%, it makes the leaching difficult, therefore, it was chosen work with 25% of solids. Regarding time, it is noted that the longer the extraction time, the greater the amount of chemical elements extracted from the substrate (including In). On the other hand, when looking at the contour plot using the global desirability (Figure 4b), the best option is to work in shorter times, where it is possible to obtain an average desirability. Fig. 5 shows a comparison of four elements in relation to time. For the case of Al (Fig. 5a), Ca (Fig. 5b) and Fe (Fig. 5c), the concentrations presented a strong linear correlation with the time (mainly for Al and Ca). When In was evaluated (Fig. 5d), there is not such a significant increase from shortest to longest extraction times. Therefore, the main observation derived from the results described up to this point is that it is more appropriate to carry out extractions with shorter leaching times, a more profitable process. Additionally, smaller amounts of the other chemical elements are extracted, that are not of commercial interest.

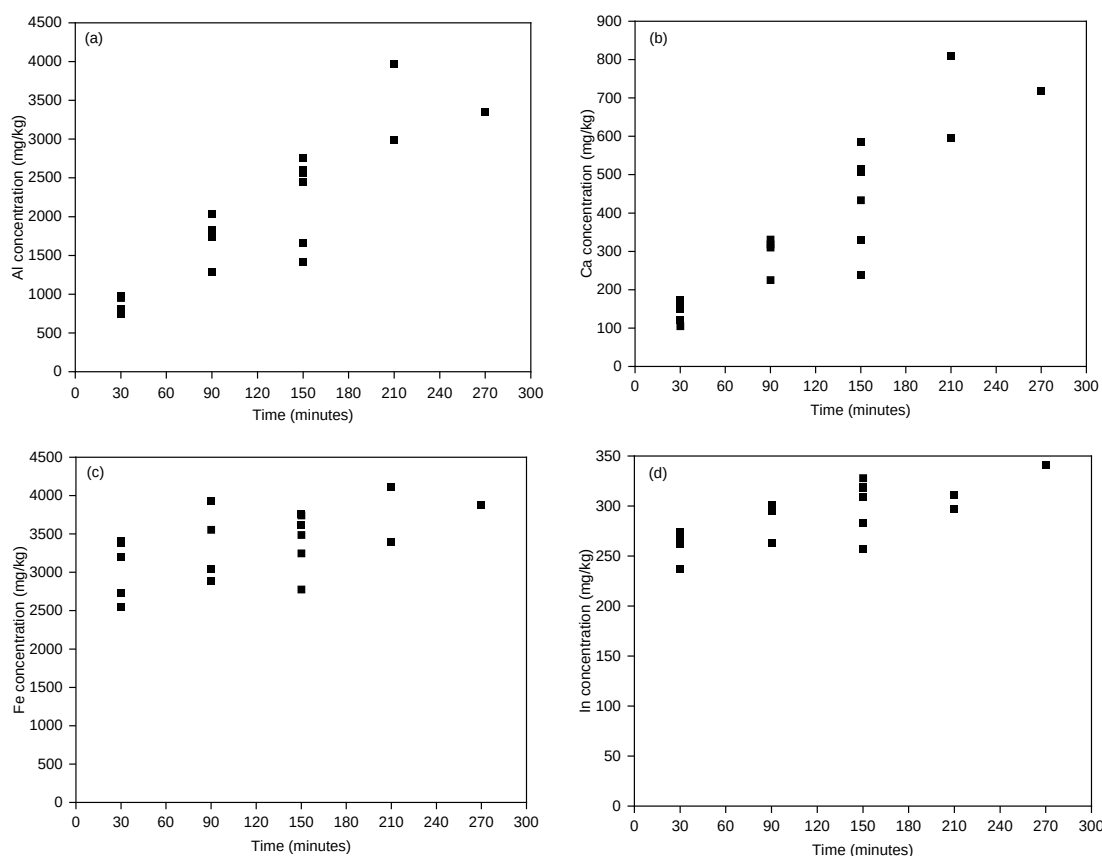


Fig. 5 Concentrations for (a) Al, (b) Ca, (c) Fe and (d) In in relation to extraction time.

Fig. 6 shows the In concentration results (mg/kg) for smartphone, monitor, TV and the mix (prepared from the three sample types) for 30, 45 and 60 minutes of extraction. There is no remarkable difference among the studied samples. The monitor and TV samples have the lower and the highest In concentration, respectively. With the mix sample (inverted triangle in Fig. 6), the In concentration is in an intermediary range, so, it is better to carry out the experiments with a mix, mainly thinking in an industrial process, where the samples can be prepared together. In relation to time, there was not a huge difference among the tested intervals, therefore the 45 minutes of extraction was chosen. Table 3 shows the concentrations for all chemical elements present in mix sample with 45 minutes of leaching and temperature of $\sim 80^{\circ}\text{C}$. Fig. 7 shows an overview of the optimized method.

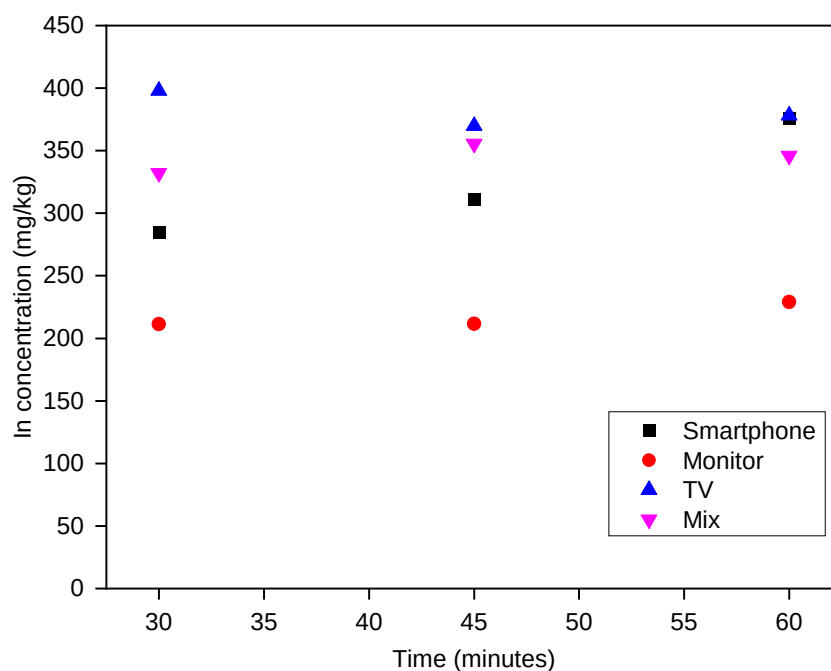


Fig. 6 In concentration (mg/kg) in relation to extraction time for the different samples analyzed.

Table 3. Chemical elements concentration (mg/kg) present in mix sample.

Chemical elements	Emission line (nm) monitored using ICP OES	Concentration (mg/kg) $\pm$ standard deviation
Al	393.152	927 $\pm$ 23
Ba	455.403	5.0 $\pm$ 0.4
Ca	393.366	388 $\pm$ 20
Cr	283.563	51 $\pm$ 7
Cu	327.396	46 $\pm$ 2
Fe	259.940	2192 $\pm$ 15
In	325.609	355 $\pm$ 18
Mg	279.553	60 $\pm$ 3
Mn	257.610	13 $\pm$ 1
Ni	231.604	6 $\pm$ 1
Si	251.611	162 $\pm$ 3
Sn	189.989	28 $\pm$ 3
Sr	407.771	76 $\pm$ 3
Zn	213.856	5.0 $\pm$ 0.3

1ª step: pre-processing of the sample

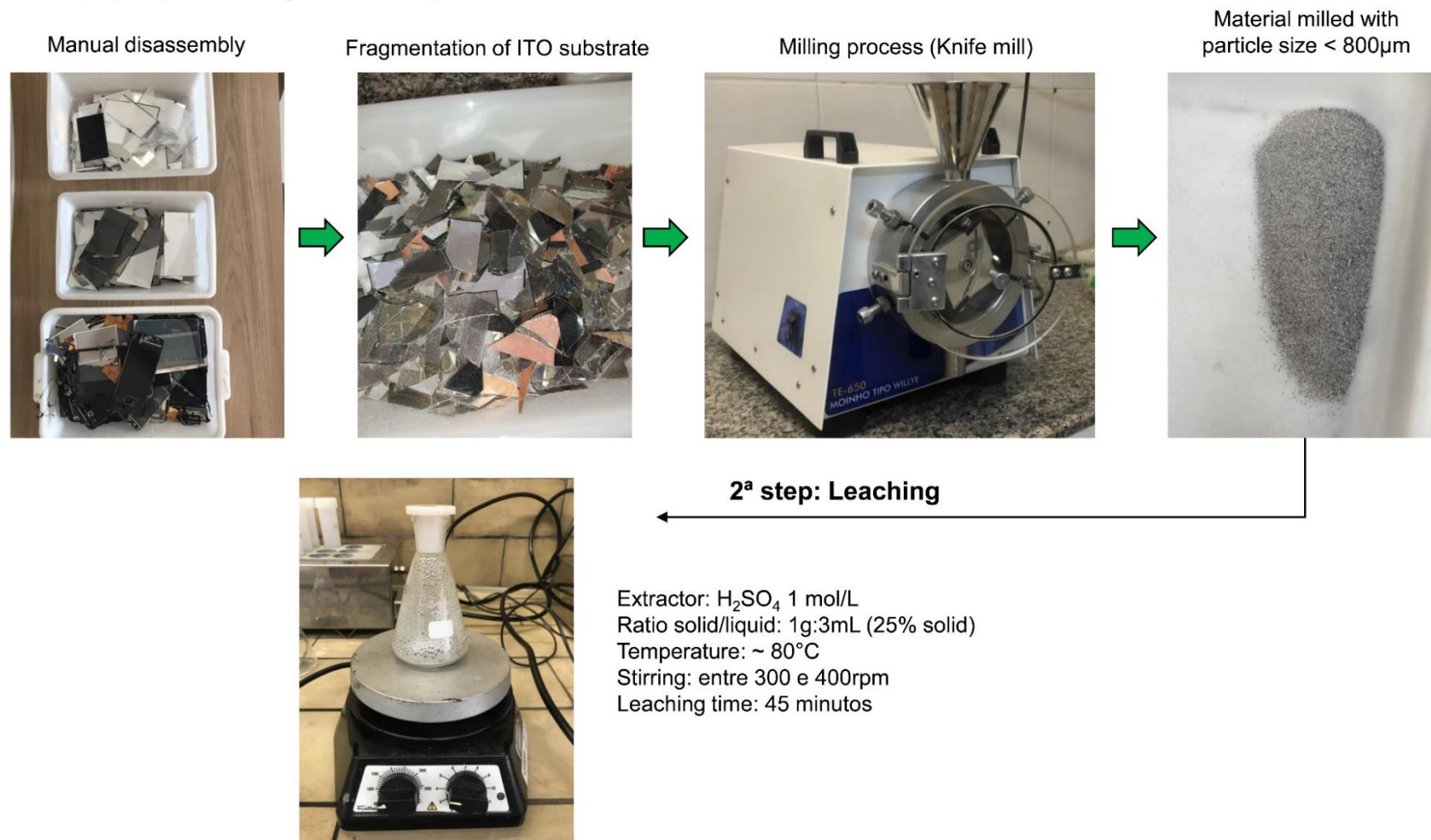


Fig. 7 Overview of optimized method.

3.3.3. Method validation

Besides the standard addition of In during all the experiments performed, it was also made an ITO addition in the optimized method in order to observe its recovery as In. Besides the mix sample, an analysis of leaching residues (solid residue obtained) was performed to visualize if there is remaining In in the residue. Additions of 15 mg from oxide were made before the leaching for the mix sample and residue.

For the mix, the In concentration was 321 ± 58 mg/kg and for the residue it was 8.0 ± 0.4 mg/kg, this shows that practically all the In (98%) is being extracted from LCD screens in the first leaching process. The recovery of the additions was 105% for the mix sample and 86% for the residue, being within the acceptable range and validating the proposed method. The validation was performed in this way due to the lack of certified reference material.

3.4. Prospection of In recycling market in Brazil

In 2019, estimated 2.14 Mt of e-waste were generated in Brazil¹⁶, and it is estimated that only ~4% are formally recycled¹⁷. A fraction of e-waste, including TVs and LCD monitors, is destined for selective collection in cities, that are most focused on separating common recyclables (plastics, metals, papers, glasses, packaging in general). However, recyclables selective collection only attends to 35% of the Brazilian population¹⁸ and, approximately 50% of the population reported having discarded e-waste mixed with Municipal Solid Waste (MSW).¹⁹ When there is not a specific process for e-waste separation, or when the population do not deliver them at specific collection points, this residue is disposed as mixed MSW to landfills and dumps; or are retrieved informally by waste pickers, thus following the informality chain. In some cities, there are particular processes to e-waste processing integrated to the selective recyclables collection. In this case, the e-waste is separated and usually commercialized as mixed electronic scrap. Pre-sorting is carried out by device types and, in some cases, there is also the disassembly and commercialization of separate components.²⁰

The formal e-waste reverse logistics chains are enforced by the Brazilian National Solid Waste Policy (Law 12.305/2010). This system was formalized through a sectorial agreement (SA) in 2019, where the system mechanisms and

their implementation goals were established. The SA is a contract between the Federal Government and producers, importers, distributors and retailers of electrical and electronic equipment in Brazil. Currently, formal e-waste reverse logistics is managed by two entities responsible for the process operation and resources – Green Eletron and ABREE (Brazilian Association of EEE recyclers). In these systems, the e-waste is collected and sent to licensed operators, who carry out the stages of sorting, disassembly, mechanical processing, sale of recyclables and disposal of hazardous waste and tailings.

The present study carried out a total of 20 interviews with key actors in the Brazilian e-waste chain, including representatives of: (1) the e-waste reverse logistics management entities / key operators / recyclers (urban mining), (2) EEE producers/distributors and (3) In processing (Fig. 8). The study also monitored the e-waste recycling market through groups in social networks, where scrap dealers negotiate e-waste components.

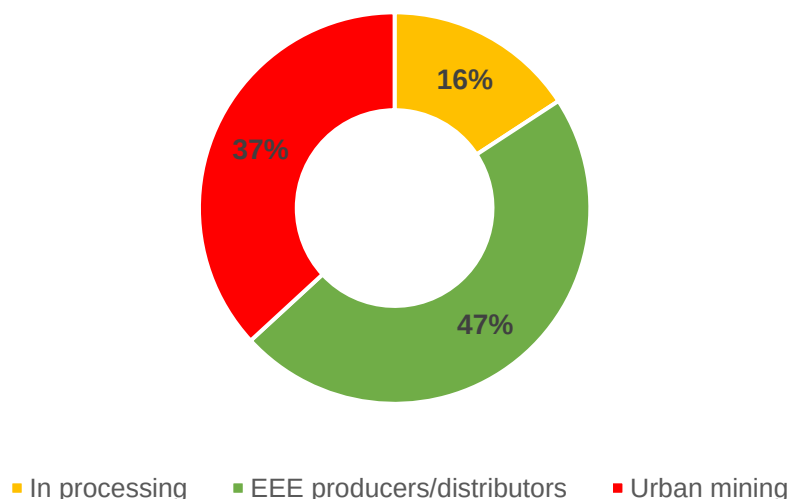


Fig. 8 Interviewers representation.

In this analysis, it was observed that there is no consolidated market for recycling LCD screens with a focus on In recovery in the country. The processing of this type of e-waste is concentrated in reverse logistic operators and in international companies with branches in Brazil. The main destinations after device disassembly are: (1) foundry in metallurgical and steel furnaces, targeting base metals, where probably In behaves like impurity (there is no much information about In); (2) pyrometallurgical processes to extract other elements in oxide form (with no information about In) or, (3) the In is processed together

with other precious metals and sent to companies abroad for further refining. Usually, LCD screens are not commercialized for recycling, but are destined associated with metal scrap to base metals recyclers. The only identified value for end-of-life LCD screens was in the repairing business for second-hand equipment, in which the identified price ranged from 5-10 BRL/kg (~1-2 USD).

The only recyclers who recover In from LCD are located in other countries, especially Belgium. However, as LCD waste has not been commercialized separately from scrap metals, it is possible to assume that the share of LCD destined for In recycling abroad is inexpressive. At formal e-waste reverse logistics operators, LCD have been usually destined as hazardous waste, with expenses rather than revenue from this waste.

The key potential clients for high purity recycled In are ITO manufacturers who are suppliers of the international EEE industry. In Brazil, there are no such manufacturers, as the LCD TVs and monitors are only assembled in the country, after importing LCD displays from international producers. Therefore, an In recycling business in Brazil would need to compete with the international In supply chain and reach for large ITO and EEE producers, mainly in China.

Another possibility is to focus on producing an In concentrate. For this product, the target clients could be large e-waste recyclers in Brazil, who are used to negotiate with e-waste and special metals refiners in Europe and Asia (ex. UMICORE, Aurubis). These refiners usually process a wide range of input material, not just e-waste, and they determine the purchase price of these inputs based on a laboratory analysis of special elements. In this case, an In concentrate could be an interesting product to export, and some of the interviewed reverse logistics operators in Brazil declared that they would be potentially interested in producing and exporting such concentrate. Another positive point with this product is that it does not require an advanced refining of In to the highest purity, so the recycling process can be more feasible.

4. Conclusion

It was possible to solve the problems on In determination by ICP OES and to extract the In from LCD screens using a simpler experimental procedure. For the pre-treatment of the sample, it was not necessary to remove the polymer adhered to the ITO substrate, reducing one step in the process. In addition, there

was a mix of several types of displays: TV, smartphone and monitors, being a positive aspect to process the materials together. In the leaching procedure, the use of diluted acid and magnetic plate with stirring and heating was effective to extract almost all the In with an extraction greater than 90%.

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5. References

- 1 J. Korhonen, A. Honkasalo, J. Seppälä, *Ecol Econ.*, 2018, **143**, 37-46.
- 2 M. V. H. Marquina, M. A. L. Dain, P. Zwolinski, I. Joly, *Procedia CIRP*, 2022, **105**, 607-612.
- 3 V. C. Costa, J. P. Castro, D. F. Andrade, D. V. Babos, J. A. Garcia, M. A. Sperança, T. A. Catelani, E. R. Pereira-Filho, *Trends Anal. Chem.*, 2018, **108**, 65-73.
- 4 D. F. Andrade, J. P. Castro, J. A. Garcia, R. C. Machado, E. R. Pereira-Filho, D. Amarasiriwardena, *Chemosphere*, 2022, **286**, 131739.
- 5 E. Worrel, M. A. Reuter, *"Handbook of Recycling State-Of-The-Art For Practitioners, Analysts, And Scientists"*, ISBN: 978-0-12-396459-5.
- 6 A. Akcil, I. Agcasulu, B. Swain, *Resour. Conserv. Recycl.*, 2019, **149**, 622-637.
- 7 A. V. M. Silveira, M. S. Fuchs, D. K. Pinheiro, E. H. Tanabe, D. A. Bertuol, *Waste Manag.*, 2015, **45**, 334-342.

- 8 B. Swain, C. G. Lee, *Waste Manag.*, 2019, **87**, 597-611.
- 9 M. Ueberschaar, M. Schlummer, D. Jalalpoor, N. Kaup, V. S. Rotter, *Recycling*, 2017, **2**, 7.
- 10 D. F. Andrade, F. M. Fortunato, E. R. Pereira-Filho, *Anal. Chim. Acta*, 2019, **1061**, 42-49.
- 11 E. Ma, R. Lu, Z. Xu, *Green Chem.*, 2012, **14**, 3395.
- 12 H-N. Kang, K-Y. Kim, J-Y. Kim, *Green Chem.*, 2013, **15**, 2200.
- 13 C. Deferm, M. V. de Voorde, J. Luyten, H. Oosterhof, J. Fransaer, K. Binnemans, *Green Chem.*, 2016, **18**, 4116.
- 14 S. L. C. Ferreira, W. N. L. Santos, C. M. Quintella, B. Barros Neto and J. M. Bosque-Sendra, *Talanta*, 2004, **63**, 1061–1067.
- 15 G. Derringer, R. Suich, *J. Qual. Technol.*, 1980, **12**, 214.
- 16 The global e-waste statistics partnership, 2022. Country sheets: Brazil. Available at: <https://globalewaste.org/statistics/country/brazil/2019/>. Accessed in: 25/10/2022.
- 17 P. Dias, J. Palomero, M. P. Cenci, T. Scarazzato, A. M. Bernardes, *Cleaner Waste Systems*, 2022, **3**, 100022.
- 18 CEMPRE (2020). Pesquisa Ciclosoft 2020. Available at: <https://ciclosoft.cempre.org.br>. Accessed in: 25/10/2022.
- 19 Green Eletron (2021). *Resíduos Eletrônicos no Brasil – 2021*. São Paulo: Green Eletron.
- 20 R.G. Souza, *In: Handb. Elect. Waste Manag.*, 2020, pp. 377-396.

4 CONSIDERAÇÕES FINAIS

Esta Tese sistematizou as principais publicações científicas produzidas ao longo da minha carreira acadêmica, dentro do assunto de avaliação da sustentabilidade do ciclo de vida (LCSA) de resíduos sólidos. A LCSA é uma metodologia em pleno desenvolvimento pela comunidade científica, pois deve permitir a estruturação e modelagem de sistemas complexos, integrando os aspectos sociais, ambientais e econômicos, e uma perspectiva de ciclo de vida. Por sua vez, a LCSA pode ser considerada um dos elementos necessários para a Gestão do Ciclo de Vida (GCV) de produtos e resíduos, uma vez que esta Gestão deve contemplar não somente a avaliação de desempenho de sustentabilidade, como processos de governança e engajamento de *stakeholders*, *design* de produtos, gestão estratégica, compras sustentáveis e outros (UNEP/SETAC, 2007). Apesar de a minha carreira ter se dedicado centralmente ao problema da avaliação da sustentabilidade usando a LCSA e suas componentes metodológicas, também expandi minha contribuição para as intersecções entre este assunto e a GCV de resíduos e produtos (especialmente observando os Artigos 1 e 9).

O Artigo 1 foi a minha primeira produção acadêmica neste assunto. Sua principal contribuição foi apresentar uma adaptação do Balanced Scorecard (BSC), respeitado método de gestão da estratégia, para o problema da gestão da sustentabilidade de RSU. Uma das principais vantagens do BSC é a integração dos objetivos gerais de uma organização em um mapa estratégico, que é um diagrama de causa-efeito que conecta os objetivos, permitindo detectar aqueles que são de curto, médio e longo-prazos, bem como caminhos críticos para o alcance da visão corporativa. Ao simular um potencial consórcio entre dois municípios, consultando os principais decisores envolvidos na gestão de RSU nestes locais, e com base nos princípios da sustentabilidade de resíduos identificados a literatura, foi possível definir potenciais objetivos estratégicos deste consórcio, e estruturar seu mapa estratégico. Também foi possível determinar objetivos específicos de cada setor que comporia o consórcio (e.g. gerência operacional, jurídico etc.), pela derivação dos objetivos gerais em consulta com especialistas envolvidos. Seguindo a metodologia do BSC, os objetivos estratégicos foram desdobrados em possíveis ações e indicadores de desempenho. Assim, este artigo trouxe elementos metodológicos que

foram aprofundados nos próximos estudos, como a consulta a *stakeholders*, estruturação de mapas de causa-efeito (um PSM), e definição de critérios e indicadores para avaliar a sustentabilidade em gestão de resíduos.

O Artigo 2 expressa um aprofundamento e maturidade metodológica neste processo de consulta a stakeholders e estruturação de mapas causais (mapas cognitivos), com vistas a identificar potenciais objetivos estratégicos, e particularmente, categorias de impacto da sustentabilidade do ciclo de vida de REEE. Neste estudo, foram consultados diversos especialistas no problema dos REEE no Brasil, representando a visão de diferentes setores, como: indústria de eletroeletrônicos, logística reversa, serviço municipal de gerenciamento de resíduos, catadores, dentre outros. A visão destes atores foi descrita nos mapas cognitivos, os quais foram integrados em um mapa geral unificado, de onde, observando a topologia das relações de causa-efeito, foi possível identificar potenciais categorias *midpoint* e *endpoint* para avaliação da sustentabilidade da logística reversa de REEE no Brasil. Este estudo foi inovador, não somente pela sua abordagem metodológica, como ao relevar categorias de impacto não destacadas pela literatura disponível até então, como a competição entre setor formal e informal, condições psicossociais de catadores, inovação para reciclagem, dentre outros. O artigo concentrou-se na identificação das categorias de impactos sociais e econômicos, pois a dimensão ambiental da LCSA já é contemplada de forma robusta pela ACV e suas categorias já consolidadas (e.g. aquecimento global, acidificação, toxicidade humana etc.).

O Artigo 3 é uma continuação do Artigo anterior, concentrado em dois problemas: a falta de dados para avaliação dos indicadores sociais e econômicos, o que é muito comum para os problemas de gerenciamento de resíduos, especialmente em países em desenvolvimento; e a necessidade de métodos multicritério para a integração entre as categorias sociais, ambientais e econômicas quali-quantitativas. Neste Artigo, foram simulados cenários de implementação da logística reversa de REEE no Brasil, que variavam segundo a localização dos pontos de entrega dos REEE, e das unidades de reciclagem de suas partes (exterior ou Brasil). A dimensão ambiental foi avaliada por um modelo de ACV, com base em dados da literatura e bases de ACV. As categorias sociais e econômicas foram medidas por consulta a especialistas, usando uma escala visual em que eles faziam uma comparação do desempenho relativo entre os cenários, para cada categoria. A

partir desta escala, foram medidos os desempenhos relativos dos cenários nas categorias, e aplicou-se um método multicritério chamado Composição Probabilística de Preferências, de forma que a resposta da integração das dimensões não fosse um índice adimensional de sustentabilidade, mas a probabilidade de um cenário ser preferível ao outro. Desta forma, buscou-se evitar a agregação das diferentes categorias na abordagem metodologicamente fraca de soma ponderada, o que permite a compensação entre as categorias. A análise probabilística permite uma melhor consideração sobre as incertezas na avaliação das categorias e na sua integração na abordagem multicritério para sustentabilidade.

O Artigo 4 concentra-se na dimensão ambiental da gestão de RSU, avaliando e discutindo a diferença entre os resultados da modelagem atribucional e consequential de ACV (em que a atribucional, mais difundida, limita os fluxos de entrada e saída aos processos dentro de um delimitado sistema de produtos). O estudo comparou três cenários para gerenciamento da fração orgânica dos RSU do Rio de Janeiro, que expressam: 1) sua completa destinação a aterros, misturados aos RSU (cenário de base); 2) separação de 50% da fração orgânica na fonte, com digestão anaeróbia e recuperação energética pelo biogás; e 3) separação mecânica posterior da fração orgânica mistura aos RSU, com digestão anaeróbia e recuperação energética do biogás. Em concordância com estudos anteriores, o Artigo detectou que a abordagem adotada para modelagem da ACV pode interferir na ordem de prioridade dos cenários analisados. A CLCA pode apresentar resultados melhores ou piores que a ACV atribucional, dependendo do método adotado para identificar os processos marginais afetados pela destinação dos resíduos orgânicos. Considerando que a recuperação energética do biogás pode substituir parcialmente o fornecimento de energia a partir da matriz energética brasileira, a magnitude do impacto evitado por este tratamento depende da magnitude dos impactos gerados pelo processo que é substituído. Quando são incluídos como processos marginais a expansão da vida útil de aterros, e a produção de fertilizantes minerais (substituído pelo orgânico), por exemplo, a magnitude dos impactos evitados pelo tratamento biológico com recuperação energética é maximizada.

O Artigo 5 aprofunda-se no problema de identificação de processos marginais em estudos de CLCA aplicados ao gerenciamento de resíduos sólidos. A

partir de uma revisão sistemática da literatura, identificaram-se as abordagens adotadas para identificação de tecnologias (de energia e materiais) afetadas pela reciclagem e tratamento de resíduos, e os critérios adotados para delimitação da fronteira de sistema. Do ponto de vista de tecnologias de produção de energia que são afetadas pela recuperação energética a partir de resíduos, a maioria dos estudos adotavam uma única tecnologia, mas também é comum que se adote a matriz energética nacional (mix de tecnologias), e até mesmo as redes fornecedoras de calefação, comuns no hemisfério norte. A substituição de materiais pela reciclagem pode enviesar os resultados da CLCA, quando se adotam dados médios da indústria nacional ou internacional, em estudos que comparam reciclagem com recuperação energética de resíduos. É necessário dedicar atenção para as incertezas relacionadas à identificação dos fluxos de materiais que são afetados pelas decisões estudadas. Com relação à delimitação da fronteira de sistema em CLCA, foram identificadas quatro abordagens, cuja escolha pode afetar os resultados da avaliação de impactos: i) consideração de alternativas perdidas, ou seja, processos que são evitados pela implementação da decisão em questão (ex. mais reciclagem, menos incineração); ii) efeitos indiretos da reciclagem, como a menor demanda por produtos de origem primária e de nutrientes minerais; iii) efeitos cascata do excesso de capacidade tecnológica: quando cria-se uma nova indústria de reciclagem, a capacidade anteriormente disponível (ex. incineradores) pode ser usada para processar outros materiais além de resíduos; e iv) efeitos rebote, como por exemplo a mudança nos padrões de consumo (processo marginal) em função da mudança nos preços entre produtos primários e reciclados. O Artigo destaca que os futuros estudos de CLCA de resíduos devem ter maior atenção ao nível de modelagem de sistemas (não somente processos unitários), e devem contemplar a consulta a especialistas, como economistas, para compreender melhor os processos afetados por mudanças nas cadeias de reciclagem e recuperação energética.

O Artigo 6 concentra-se em um outro problema da gestão de REEE, que pode afetar sua posterior avaliação da sustentabilidade: a estimativa do perfil qualitativo da geração destes resíduos, expressando o total de REEE que potencialmente segue pelas cadeias de processamento e destinação de resíduos. Muitos estudos científicos e técnicos adotam estimativas de vida útil de equipamentos que são defasadas, representam outros países, e quase sempre são

médias discretas que não contemplam variações nos padrões de consumo dos aparelhos. A partir de estudos anteriores, este Artigo considerou como referência o método de estimativa de REEE “*market supply*”, que adota um perfil probabilístico para a vida útil dos equipamentos. Este Artigo estimou o perfil probabilístico da vida útil de equipamentos eletroeletrônicos (EEE) a partir de uma survey com uma amostra representativa da população de São José dos Campos-SP. O estudo observou que, neste município, os valores médios de vida útil tendem a ser menores que os anteriormente praticados na literatura, o que pode representar um subdimensionamento da logística reversa de REEE quando tomando em conta estes estudos anteriores. O Artigo também notou que os padrões de vida útil variam de acordo com as Zonas do município, podendo expressar uma correlação com os diferentes padrões socioeconômicos da população. Embora haja incertezas associadas às respostas dadas nas entrevistas, essa abordagem por *surveys* é recomendada para obter dados locais, quando não houver um rastreamento dos REEE gerados, o que é o cenário comum no Brasil e na maioria dos países.

O Artigo 7 é uma revisão teórica sobre os desafios e oportunidades da contribuição do gerenciamento de resíduos sólidos para a economia circular no Brasil. O Artigo discute a forte relação entre estes dois assuntos, demonstrando que as cadeias de resíduos sólidos podem contribuir para a economia circular tanto nos ciclos biológicos como técnicos e tecnológicos. Por outro lado, destaca os desafios do Brasil enquanto um país continental, evidenciando a diferença entre as regiões do país com relação ao gerenciamento de resíduos e capacidade industrial, bem como as limitações logísticas para melhorar estas cadeias. Ademais, o artigo discute os problemas nacionais com relação a resíduos de setores específicos, como a mineração, agricultura e indústria tecnológica.

O Artigo 8 apresenta uma revisão sistemática da literatura sobre as soluções atualmente adotadas para estudos de ACV Social (SLCA) aplicados ao gerenciamento de resíduos sólidos. Observou-se que estes estudos são mais frequentes em países em desenvolvimentos, evidenciando os problemas sociais relacionados aos seus sistemas de resíduos, como a informalidade e riscos de contaminações, especialmente com foco na disposição final, que em muitos casos são lixões a céu aberto. Por este motivo, os trabalhadores e comunidades locais foram os stakeholders mais enfocados, sendo avaliadas principalmente as

categorias de impacto relacionadas às condições de trabalho, saúde e segurança. O estudo constatou que ainda há grande necessidade de evolução da metodologia de SLCA, que apresenta desafios para se alinhar ao método já bem estabelecido e normatizado para a ACV (ambiental), que deve servir de referência para as demais componentes da LCSA.

Já o Artigo 9 concentra-se em outra etapa do ciclo de vida dos produtos, antes de se tornarem resíduos: a compra sustentável. A partir de uma revisão sistemática da literatura, complementada pela consulta a especialistas, e adotando o mapeamento cognitivo (de forma semelhante ao Artigo 2), este Artigo estabeleceu um conjunto de critérios de sustentabilidade para a compra de produtos sustentáveis. A partir deste mapeamento, observou-se, além das três dimensões clássicas da sustentabilidade, uma quarta dimensão relevante para a decisão sobre as compras, que consiste em critérios de Governança e *Compliance* do Fornecedor. Esta dimensão compreende categorias de impactos associadas aos *endpoints* “maturidade do sistema de gestão”, “auditoria externa” e “legitimidade (legalidade)”. A contribuição dos especialistas permitiu não somente validar categorias bem estabelecidas na literatura, como revelou outras categorias de sustentabilidade inovadoras, como: “estímulo a movimentos e questões sociais”, “custos durante o uso” e “custos pós-uso do produto”.

O Artigo 10, na mesma linha dos Artigos 2 e 9, adota PSM em combinação com a revisão de literatura e consulta a stakeholders, para determinar categorias de impacto de LCSA – neste caso, específicas para as baterias de íons de lítio (LIBs), e tendo como eixo teórico os Objetivos de Desenvolvimento Sustentável (ODS) e suas respectivas metas. O estudo observou que as etapas do ciclo de vida de LIBs com maior potencial para afetar os ODS são a extração de matéria prima, produção de materiais e células, e o fim-de-vida (resíduo). Estes problemas são maximizados quando estas etapas ocorrem em países em desenvolvimento, o que é mais crítico na extração de matéria prima (intensificada no Congo e América Latina).

Os Artigos 11 e 12 concentram-se no problema de geração de dados primários, que podem subsidiar estudos de ACV mais robustos e regionalizados no Brasil. O Artigo 11 simulou cenários de disposição final de REEE em aterros e lixões no Brasil, misturados aos RSU. Ao coletar e analisar quimicamente os lixiviados gerados, observou-se maior concentração de metais pesados nos tanques que

continham REEE. O método de preparação das amostras de lixiviados para análise também interferiu nos resultados. O Artigo 12, por sua vez, apresenta o desempenho e a caracterização química de uma tecnologia nacional para reciclagem do índio de resíduos de telas de cristal líquido (LCD). O estudo também detalha o fluxo de insumos (substâncias, energia) necessários para a aplicação desta reciclagem e obtenção de seu produto final (concentrado de In).

Este histórico de publicações demonstra uma atuação científica consistente deste candidato, desenvolvendo estudos e conhecimento em assuntos relacionados à sustentabilidade do ciclo de vida de produtos. Alguns destes estudos contemplam a modelagem de ACV (ambiental) para avaliar sistemas de resíduos sólidos no Brasil (Artigo 3 e Artigo 4). Ainda sobre a ACV ambiental, os Artigos 4 e 5 aprofundam-se na importância da abordagem Consequencial (CLCA), de forma a identificar e avaliar impactos em processos marginais afetados por mudanças nos sistemas de gerenciamento de resíduos sólidos. Os Artigos 2, 9 e 10 concentram-se na adoção de PSM combinada com revisão sistemática da literatura e consulta a especialistas, visando determinar categorias de impacto social e ambiental do ciclo de vida de produtos e resíduos. O Artigo 8 aprofunda-se na forma como a ACV social (SLCA) tem sido aplicada a problemas de resíduos sólidos, apresentando o estado-da-arte e diretrizes metodológicas para futuros estudos, da mesma forma que o Artigo 5 o faz para estudos de CLCA aplicados a resíduos. O Artigo 6 procura solucionar um dos problemas associados à geração de dados primários e regionalizados em ACV/LCSA, que são um dos principais gargalos para estudos de ACV e LCSA: neste caso, o perfil quantitativo e qualitativo da geração de REEE pela população Brasileira, que seguem diferentes cadeias de processamento de resíduos no país. A geração de dados primários para potenciais estudos de ACV e LCSA também foi o objetivo dos Artigos 11 e 12, que visam melhorar a confiabilidade e reduzir a incerteza de futuros estudos de ACV: o Artigo 11 analisou o teor de metais em lixiviados de simulações de aterros e lixões contendo REEE e RSU; e o Artigo 12 apresenta a caracterização química dos fluxos de insumos necessários e produtos gerados (concentrado de In) em um processo de reciclagem de Índio de telas de cristal líquido (LCD). Dados primários como estes permitem modelos de ACV muito confiáveis, detalhados em nível de processos unitários, e representando a situação nacional, seja na disposição final de resíduos, como potenciais tecnologias de

reciclagem atualmente desenvolvidas no Brasil.

Expandindo-se para a Gestão do Ciclo de Vida, o Artigo 1 abordou a gestão estratégica de resíduos sólidos, adotando uma metodologia que foi aprofundada e aperfeiçoada nos Artigos 2 e 9. O Artigo 9 expandiu a preocupação com a sustentabilidade do ciclo de vida para a fase pré-uso dos produtos (compra), enquanto os resíduos se caracterizam como tal no pós-uso.

Em uma perspectiva maior, observa-se que não há como diferenciar o ciclo de vida do produto e do resíduo, pois as etapas de seus ciclos de vida se sobrepõem. Nestes casos, a única limitação é a delimitação do escopo para o estudo de LCSA, que conforme as normas de ACV, deve ser definida em função do objetivo do estudo. Observando a complexidade envolvida na sustentabilidade do ciclo de vida de produtos e resíduos, e considerando o aprofundamento teórico permitido por estes estudos, considera-se que as fronteiras de sistemas de ciclo de vida devem ser o mais amplas possíveis. Assim como na CLCA, isto permite ter uma visão mais abrangente das cadeias de causa e efeito associadas a cada determinada etapa do gerenciamento de resíduos ou do ciclo de vida, e todos os processos e mercados que são significativamente afetados por alterações nestas etapas. Assim, compreende-se que o próximo ciclo de estudos deste candidato deve não somente aprofundar-se na modelagem de ACV e SLCA, mas contemplar temas associados como: LCSA consequencial, desenvolvimento de produtos sustentáveis, função das cadeias de resíduos para a economia circular e ODS, e análises químicas para geração de dados primários relevantes. Esta tendência tem se demonstrado pelo tema dos Artigos 10 a 12, em revisão por periódicos indexados internacionais, representando o próximo ciclo de possíveis publicações do candidato.

Por fim, destaca-se que esta Tese não é exaustiva da produção acadêmica deste candidato, pois não contempla outras produções como: livros e capítulos de livro, orientações de TCC que não geraram artigos indexados, artigos em desenvolvimento, artigos publicados em eventos científicos, e orientações de Mestrado e Doutorado em andamento. Esta atuação complementar tem explorado temas associados aos assuntos destacados acima, reforçando os potenciais eixos condutores para futuras publicações científicas.

REFERÊNCIAS

- ABNT NBR ISO 14040: avaliação do ciclo de vida: princípios e estrutura. Rio de Janeiro: ABNT, 2009, 21 p
- Amnesty International. Democratic Republic of the Congo: Time to recharge: Corporate action and inaction to tackle abuses in the cobalt supply chain. Amnesty International, 2017. Available at: <https://www.amnesty.org/en/documents/afr62/7395/2017/en/>
- Angelo A.C.M., Marujo L.G. Chapter 12 - Life cycle sustainability assessment and decision-making under uncertainties, in: Life Cycle Sustainability Assessment for Decision-Making. Editor(s): Ren, J., Toniolo, S. Elsevier, Pages 253-268, 2020. ISBN 9780128183557. <https://doi.org/10.1016/B978-0-12-818355-7.00012-9>.
- Berardocco C, Delawter H, Putzu T, Wolfe LC, Zhang H. Life Cycle Sustainability Assessment of Single Stream and Multi-Stream Waste Recycling Systems. *Sustain.*, 14(24):16747, 2022. <https://doi.org/10.3390/su142416747>.
- Bernstad Saraiva, A., Souza, R. G., Mahler, C. F., Valle, R. A. B. Consequential lifecycle modelling of solid waste management systems – Reviewing choices and exploring their consequences. *J. Cleaner Prod.* 2018; 202: 488–496. doi: n°. 10.1016/j.jclepro.2018.08.038
- Beula D., Sureshkumar M. A review on the toxic E-waste killing health and environment – Today's global scenario. *Mater. Today: Proceedings*, 2021.
- Ducoli S., Fahimi A., Mousa, E., Ye, G., Federici, S., Frontera, P., & Bontempi, E. ESCAPE approach for the sustainability evaluation of spent lithium-ion batteries recovery: Dataset of 33 available technologies. *Data in Brief*, 42, 2022. <https://doi.org/10.1016/J.DIB.2022.108018>
- Erakca, M.; Bautista, S.P.; Moghaddas, S.; Baumann, M.; Bauer, W.; Leuthner, L.; Weil, M. Closing gaps in LCA of lithium-ion batteries: LCA of lab-scale cell production with new primary data. *J. Clean. Prod.*, 384, 135510, 2022. <https://doi.org/10.1016/j.jclepro.2022.135510>
- Fahimi A., Ducoli S., Federici S., Ye G., Mousa E., Frontera P., & Bontempi E. Evaluation of the sustainability of technologies to recycle spent lithium-ion batteries, based on embodied energy and carbon footprint. *J. Cleaner Prod.*, 338, 2022. <https://doi.org/10.1016/J.JCLEPRO.2022.130493>.
- Gómez-Sanabria A. et al. Potential for future reductions of global GHG and air pollutants from circular waste management systems. *Nature Communications* 2022 13:1, v. 13, n. 1, p. 1–12, 2022.
- Harper G., Sommerville R., Kendrick E., Driscoll L., Slater P., Stolkin R., Walton A., Christensen P., Heidrich O., Lambert S., Abbott A., Ryder K., Gaines L., & Anderson P. Recycling lithium-ion batteries from electric vehicles. *Nature* 575:7781, 575(7781), 75–86, 2019. <https://doi.org/10.1038/s41586-019-1682-5>.
- Herreras-Martínez L., Anta M., Bountis R. et al. (2021). Recommendations for tackling fires caused by lithium batteries in WEEE- A report of the Batteries Roundtable. Available at: <https://weee-forum.org/wp->

<content/uploads/2021/07/Tackling-fires-caused-by-batteries-in-e-waste.pdf>.

Accessed in: 05 December 2022.

Jiang S., Zhang L., Li F., Hua H., Liu X., Yuan Z., Wu H. Environmental impacts of lithium production showing the importance of primary data of upstream process in life-cycle assessment. *J. Environ. Manag.*, 262, Article 110253, 2020. Doi: 10.1016/j.jenvman.2020.110253

Mrozik W., Rajaeifar M. A., Heidrich O., & Christensen P. (2021). Environmental impacts, pollution sources and pathways of spent lithium-ion batteries. *Energy & Environ. Sci.*, 14(12), 6099–6121. <https://doi.org/10.1039/D1EE00691F>

Oliveira D.F., Souza R.G. Life cycle sustainability impact categories for sustainable procurement, *J. Cleaner Prod.*, V. 383, 2023. <https://doi.org/10.1016/j.jclepro.2022.135448>.

Ren J., Ren X., Dong L., Manzardo A., He C. & Pan M. Multiactor multicriteria decision making for life cycle sustainability assessment under uncertainties. *AIChE J.*, 64: 2103-2112, 2018. <https://doi.org/10.1002/aic.16149>

Ripa A., Fiorentino G., Vacca V., Ulgiati S. The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). *J. Clean. Prod.*, 142, pp. 445-460, 2017. Doi: 10.1016/j.jclepro.2016.09.149

Sala S., Farioli F. & Zamagni A. Life cycle sustainability assessment in the context of sustainability science progress (part 2). *Int J Life Cycle Assess* 18, 1686–1697, 2013. <https://doi.org/10.1007/s11367-012-0509-5>

Schüler D., Dolega P., & Degreif S. Social, economic and environmental challenges in primary lithium and cobalt sourcing for the rapidly increasing electric mobility sector. In *European Policy Brief. Strategic Dialogue on Sustainable Raw Materials for Europe (STRADE)*, 2018.

Souza R.G., Rosenhead J., Salhofer S.P., Valle R.A.B., Lins M.P.E. Definition of sustainability impact categories based on stakeholder perspectives. *J. Clean. Prod.*, 105, pp. 41-51, 2015. Doi: 10.1016/j.jclepro.2014.09.051

Souza, R.G., Clímaco, J.C.N., Sant'Anna, A.P., Rocha, T.B., Valle, R.A.B., Quelhas, O.L.G. Sustainability assessment and prioritisation of e-waste management options in Brazil. *Waste Manag.* 57, 46–56, 2016. <https://doi.org/10.1016/j.wasman.2016.01.034>.

UNEP/SETAC. Life Cycle Management: A Business Guide to Sustainability. UNEP. 2007. Available at: <https://www.unep.org/resources/report/life-cycle-management-business-guide-sustainability>. Accessed in: 17 February 2023.

UNEP/SETAC. Towards a Life Cycle Sustainability Assessment. UNEP, 2011. Available at: www.unep.org/resources/report/towards-life-cycle-sustainability-assessment-making-informed-choices-products. Accessed in: 17 February 2023.

Voss, R., Lee, R. P., & Fröhling, M. A consequential approach to life cycle sustainability assessment with an agent-based model to determine the potential contribution of chemical recycling to UN Sustainable Development Goals. *J Indust Ecol*, 00, 1– 20, 2022. <https://doi.org/10.1111/jiec.13303>