RSC Sustainability

CRITICAL REVIEW

Cite this: *RSC Sustainability*, 2025, **3**, 2455

Design for recycle of devices to ensure efficient recovery of technology critical metals

Molly E. Keal,[a] Sean Scott,[a] Bashayer N. N. Alsulami,[b] Jeff Kettle,[b] Andrew Feeney,[b] Jacqueline S. Edge,[c] Paul A. Anderson,[c] Gavin D. J. Harper,[c] Allan Walton,[c] Guillaume Zante[d] and Andrew P. Abbott*[a]

Received 24th February 2025
Accepted 25th April 2025

DOI: 10.1039/d5su00128e

rsc.li/rscsus

Many of the issues associated with recycling devices containing small but significant amounts of technology critical metals, arise from the choice of materials and, most importantly, the joining methods for different materials. In many cases, recycling could be simplified and made more efficient by employing design for recycle principles which consider the requirements for separation. This study highlights recent innovative recycling tools which can impart greater selectivity during material separation and shows how often small changes in device architecture can greatly simplify critical metal recovery and promote circularity. It also discusses how design can be used to enable these tools to be assembled into the recycling flowsheet, to decrease energy and chemical input and maximise the recovery of technology critical metals. It also promotes how digital product passports could be used in combination with AI to develop algorithms to develop smart recycling flowsheets.

Sustainability spotlight

The movement from a carbon-based economy to one based on renewable energy puts increased demands on a group of technology critical metals. It is essential that new devices for creating and storing electrical energy are designed to recover these critical elements. A toolbox of recycling techniques has recently been developed that make metal recovery more efficient and this article shows how product design can be modified to enable efficient use of these tools. This review is the first to show how design can be used to improve recycling efficiency for a variety of devices. It also includes a variety of case studies. The work aligns directly with UN SDGs 7, 9 and 11.

1 Introduction

The ongoing deployment of mobile technologies and sustainable energy production has accelerated demand for so-called technology critical metals (TCMs) across the globe. Subsequently, it is predicted that for most TCMs, including iron and aluminium, global demand could increase by up to 215% by 2050.[1] While TCMs like aluminium are relatively efficiently recovered and recycled, with 75% of all aluminium ever produced still being used today,[2] many TCMs used in high-tech applications are spread more disparately throughout complex architectures and are accordingly more difficult to recover and purify. For instance, recovery rates of TCMs from waste electrical and electronic equipment (WEEE) range from 20% for gold, down to less than 1% for some rare earth elements (REEs) and most semiconductor elements.[3] (REF) WEEE is one of the fastest growing sources of waste, due to clean energy generators and energy storage devices having finite lifetimes. In 2022, Europe had an annual production of 17.6 kg of WEEE per capita, while is one of the highest collection and recycling rates, those rates still remain lower than 43%.[4]

Therefore, implementation of a circular economy is necessary to deal with such a complex, worldwide problem, ideally with a strong focus on strategies aimed at reducing the volume of waste generated through reuse, repair, or lifetime extension. However, recycling is unavoidable in the long-term, at the eventual end-of-life (EOL) of devices. Improved recycling rates are vital, but hindered by the complex architecture of WEEE, which are often made of interpenetrated layers of metal, organic (polymers) and inorganic (ceramics, glass) materials.[5] The metals themselves can be found in various components, such as silver bus bars on solar cells, soldered tin–silver–copper on composite materials at the surface of printed circuit boards (PCBs) or powder-like mixed oxides attached to current collectors in the cathodes of lithium-ion batteries (LIBs).

Various methodologies have been explored to efficiently recover key materials from these complex devices and these have recently been highlighted in a critical review.[6] It demonstrated techniques such as; removal of organic layers by

[a]School of Chemistry, University of Leicester, Leicester, LE1 7RH, UK. E-mail: apa1@leicester.ac.uk
[b]James Watt School of Engineering, University of Glasgow, G12 8QQ, UK
[c]Birmingham Centre for Strategic Elements & Critical Materials, University of Birmingham, UK
[d]Université Paris-Saclay, CEA, CNRS, NIMBE, LICSEN, Gif-sur-Yvette, France

dissolution with appropriate organic solvents, shredding, or high temperatures. Inorganic materials are generally brittle and can be shredded or broken up with techniques such as ultrasound. A range of established physical techniques are available to concentrate the metals and, to some extent, separate materials, such as froth flotation, or separation by density, magnetic or electrostatic means, among others.[6] However, the effectiveness of such strategies is determined by the separation of each waste stream. Often multiple separation steps are required and, due to the interconnected nature of the devices, these steps often consume a lot of energy, driving up costs and also tend to have low recovery rates. For example, small WEEE products such as smartphones are difficult to recycle because they consist of multiple materials, with elements and components glued and soldered together, making their liberation harder and leading to recycling rates close to or lower than 20% for most products.

This should be addressed with appropriate design of the product aimed at facilitating repair, remanufacture, reuse and, ultimately, recycling. Ultimately the goal is to create products more akin to lead–acid batteries, where a combination of simplicity of design and appropriate regulations has led to a 99% recycling rate in Europe and USA.[7] The present study aims to highlight how simples changes in materials and their connectivity could enhance the effectiveness of recovery methodologies, enabling TCMs to be recovered more easily. This will be done by highlighting how designing devices for recyclability can change the key impact factors for life cycle assessment (LCA) and techno-economic assessment (TEA). Additionally, case studies relating to design for recyclability of LIBs, photovoltaic cells, high-powered magnets and WEEE will be discussed, to explore the current state-of-the-art and future perspectives for each technology.

Product design is just one aspect to achieving a circular economy. Numerous stakeholders including manufacturers, retailers, economists, recyclers, consumers, legislators and designers control the fate of a product during its lifetime. Device circularity has component integration and separation at its heart and design architecture is the cornerstone to simplifying segregation and recovery of TCMs. Identifying end of life fate as a design criterion is the first step to achieving circularity and AI, labelling, legislation, discussion and education are all tools that can be used to promote design for recycle and this article aims to start the discussion for the important area of technology critical metals. It will focus on the TCMs in devices used to make and store renewable energy as these will experience the largest biggest changes in their use. The approaches to achieve circularity are, however, transferrable to other devices not specifically discussed in the cases studies.

2 Techno-economic and life cycle analysis perspective

2.1 Retro-economic analysis

When assessing the economics of recycling it is useful to consider the value of the recovered materials. This could be based on the value of the elemental components – for example, Fig. 1 shows the relative value of components in WEEE and silicon solar cells, when returned to their elemental state. Where the majority of the value lies in minor components: gold in the former and silver in the latter.[8,9] However, in some cases, the metal salts possess more value as they can be used directly as precursors in remanufacturing processes. The purity and form of the recycled product needs to be of a similar grade to that required in remanufacture. It should also be appreciated that all recycling processes currently carry a processing "gate" fee which is usually applied per tonne of material processed and it is expected that this approach will continue. This removes the risk associated with metal content variability, while also raising the issue of where the cost for this is born in a circular economy.

When developing an entire recycling methodology, as a first approximation, the chemical and energy costs for the whole recycling process should be less than half the cost of the virgin raw materials. To ensure profitability, the cost of labour and overheads should be a fraction of the cost. For instance, it was estimated that the cost of recycling LIBs should be in the range US$2–6 per kg of battery waste.[10] While profitability is expected to improve with increased scale, labour in the US, UK and EU is a significant component of the processing costs therefore, process automation would need to be implemented to ensure recycling costs are kept within this range. Additionally, artificial intelligence is already starting to play a role in material identification and separation, being particularly advanced for plastic separation, where the composition is dominated by one component.[11] Similar machine learning algorithms are also being developed to sort through WEEE waste and remove certain components from different waste streams.[12] Overall, when designing products for circularity, it is useful to consider the potential value of the recycled components once they have been recovered – *i.e.* how much of the remanufacturing costs can be offset from recoverable starting materials.

2.2 Environmental impacts of recycling processes

Recycling secondary resources, particularly for metal-containing waste, significantly improves Life Cycle Assessment (LCA) outcomes, compared to extracting and processing primary resources. This is primarily due to the higher metal content within the waste and lower consumption of both energy and chemicals, which should generally reduce the global warming potential (GWP) and the amount of waste generated. For example, aluminium obtained from recycled sources can save up to 95% of the energy required for primary production from Bauxite ore, circumventing energy-intensive mining, crushing, grinding, and refining steps.[13] Additionally, during recycling schemes it is vital to use the minimum quantities of chemicals and energy on as highly concentrated a waste stream as possible, so as to minimise the chemical and energy inputs and their associated impacts. Also, it should be noted that while GWP, energy consumption and waste generation are the most commonly assessed LCA impacts due to the relatively ease of data acquisition, other impact factors, such as land usage,

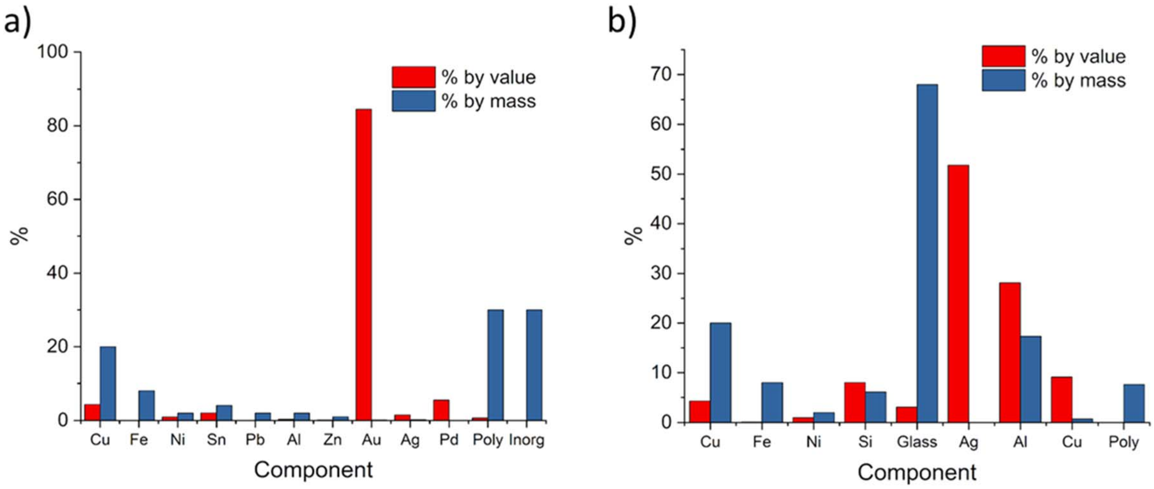

Fig. 1 Composition of typical (a) WEEE and (b) single crystal solar cell by mass and value. Value data calculated using London metal exchange prices 29/10/24, with the assumption that metals are returned to their elemental state.

acidification potential and ozone depletion potential should be considered.[14]

Generally, recycling processes can be classified into thermal (smelting and pyrolysis), mechanical, and chemical processes (in which a chemical reaction is involved), although there are processes within each category which require combinations of these. For example, smelting is usually required in most recycling methodologies, to process the metals into its required form for subsequent treatments. Fig. 2 shows the variable energy and chemical consumption of a range of recycling process categories.

Decisioning between recycling routes can also depend on aspects beyond environmental impact. For example, choosing between hydrometallurgical and pyrometallurgical routes often hinges on the feedstock characteristics, regulatory requirements, and economic trade-offs. LCA comparisons suggest that, while hydrometallurgy has a lower carbon footprint and energy demand, pyrometallurgy is more suitable for large-scale operations due to established infrastructure and lower waste management costs.[15]

2.2.1 Energy intensity in recycling. The significant energy consumption and the energy sources of some recycling processes need to be accounted for and addressed, to maximise the environmental benefits of recycling. Energy consumption of thermal processes, such as pyrolysis, is generally high, with smelting temperatures often exceeding 1000 °C. If the energy produced relies heavily on fossil fuels, the associated greenhouse gas emissions can reduce the potential environmental

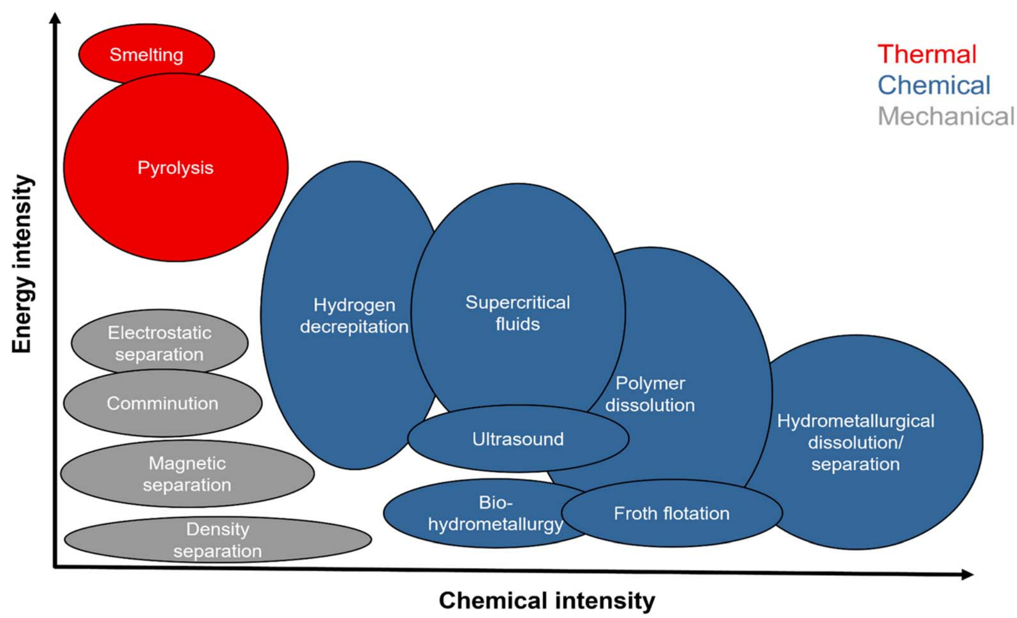

Fig. 2 Assessing the relative energy and chemical intensities of thermal, chemical and mechanical recycling processes.

advantages.[16] For instance, steel recycling, similarly to steel production, generates approximately 78% more CO_2 if coal-powered blast furnaces are used instead of electric arc furnaces.[17] Therefore, integrating appropriate renewable energy sources, such as wind, solar or hydropower, into energy-intensive processes presents an opportunity to further reduce emissions.[18] An example of this is the implementation of concentrated solar power (CSP) into calcination and roasting, where pilot projects have demonstrated the feasibility of using CSP, though scalability remains a challenge.[19,20]

Additionally, the usage of high temperatures for roasting, drying, solvent evaporation and pyrometallurgy also lead to emissions of SO_2, NO_x, fluorinated compounds and particulate matter. The air pollution, acid rain, and respiratory health issues are not accounted for in most LCAs. High energy inputs can also result from innocuous looking processes such as electrostatic separation as they require both drying and comminution steps as pre-treatments. Both processes require relatively large energy inputs and, in the case of comminution, can require more energy depending on the type of materials processed and the magnitude of the desired particle size reduction.[21]

2.2.2 Chemical inputs in recycling. In hydrometallurgical recycling schemes, chemical inputs, rather than energy intensity, are often the focal point of LCA, due to them operating at ambient or moderately elevated temperatures using aqueous solutions.[22] These inputs are essential to leach metals from waste streams as well as other key processes like purification and the removal of contaminants. While effective, proper management is required to prevent toxic emissions.[23] Therefore, LCA is vital to gain a complete understanding of the long-term effects to the local environment. Crucially, LCA can also be used in decision making *e.g.*, to identify process 'hotspots' and 'bottlenecks' as well as choosing compable methodologies. For instance, an LCA study on a hydrometallurgical process treating WEEE, identified the nitric acid leaching step as the primary contributor in several environmental impact categories such as global warming potential, ecotoxicity and acidification due to the high consumption of harmful chemicals and generation of liquid waste.[24]

Reuse of waste from another process can significantly reduce environmental and energy impact *e.g.* waste organic solvents can be used when removing plastics from a waste stream and these can be reused following redistillation.[25] Similarly, processes that use supercritical fluids can use pressure to regenerate the fluids, however, this does require elevated temperatures. Additionally, advancements in green chemistry offer alternatives to hazardous chemicals. For example, bioleaching uses microorganisms to recover metals from waste, reducing the need for strong acids and lowering environmental risks.[26] Furthermore, ionic liquids and deep eutectic solvents are being explored as alternative solvents for metal extraction due to their ability to control speciation in media of low water activity.[27] These approaches are, however, best used on concentrated metal streams with higher value metals, where smaller volumes of liquids are required.

In some cases, the concentrations of metals in a waste stream can be very low and may be uneconomic to extract using hydrometallurgy. There is also a growing research potential of various lithotrophic and organotrophic microorganisms for extracting metals by producing inorganic and organic acids (bioleaching), through selective excretion of metallic nodules or by making use of the metal-binding ability of various biomaterials, including algae, fungi, bacteria, and yeasts.[28]

2.2.3 Separation efficiency. Comminution (grinding, shredding *etc.*) can have a mixed effect on separation by both reducing particle size and liberating materials but it can also lead to the loss of valuable fine particles and increase energy consumption and disperse the valuable phase.[29,30] While shredding is commonly applied to reduce a material into a manageable form, it results in a high degree of mixing making the target materials more dilute and thus harder to recover. Structured disassembly enables pure major phases to be separated *e.g.* removal of aluminium frames from solar cells. This can simplify subsequent processing, such as hydrometallurgy, resulting in significant improvements in environmental performance through smaller volumes of solvents and lower energy inputs.

Physical separation uses differences in physical properties, such as density, size, magnetic, electrostatic charge, to separate components. The most commonly used is magnetic separation, as ferrous metals make up >90% of all metals used each year. Magnetism is particularly effective in separating steel and iron from waste streams.[31] For conductive non-ferrous metals, including aluminium and copper, there are processes such as eddy current separation, which induces eddy currents in these metals, repelling them from the waste stream.[32] Sensor-based sorting technologies have advanced significantly, enabling the identification and separation of materials based on colour, conductivity, atomic density, and other properties.[33] X-ray transmission (XRT) sorting can differentiate materials based on atomic density, which is useful for separating metals with similar physical properties.[34] The increasing complexity of products, especially WEEE, poses challenges for physical separation. Mixed materials and miniaturisation make it difficult to isolate metals without prior dismantling.[35]

Shredding can, in some cases significantly simplify separation *e.g.* in photovoltaic devices glass is *ca.* 70% by mass, but its value is relatively small, and it is glued to the most valuable components. This separation is complex, but the glass and substrate are brittle, so shredding rapidly reduces the size and separates the glass, plastic and silicon phases which can be separated electrostatically and reduces the mass by 80%. The remaining 20% has >60% of the value but the expensive leaching and recovery steps can be carried out on a smaller mass of material.

Knowledge of the spatial location of the TCMs within a device can significantly improve recycling efficiency through efficient separation. Walton *et al.* demonstrated that imaging of computer hard drives enabled the magnet segment to be guillotined from the rest of the structure, which significantly concentrated all of the REEs.[36,37] The magnet segments are then demagnetised and exposed to a hydrogen atmosphere, where they become brittle and break down into a demagnetised alloy powder. This enabled jet milling and recycling of the alloy powder into sintered magnets. This is an example of short loop recycling, where the components are not taken back to their starting materials before being remanufactured.

Smaller or embedded items will always be a complex stream, unless a mechanism for consumer segregation can be easily enabled. An example of this are mobile phone cameras, which are currently soldered onto the motherboard making replacement or facile recovery almost impossible. A small number of modular mobile phones are starting to appear on the market where major components can be easily exchanged for repair or upgrade. Furthermore, the movement to have hermetically sealed batteries in devices also means that the component most likely to limit the usability of the device cannot be easily replaced. This shortens the entire product's service lifetime and makes recycling more hazardous and less efficient.

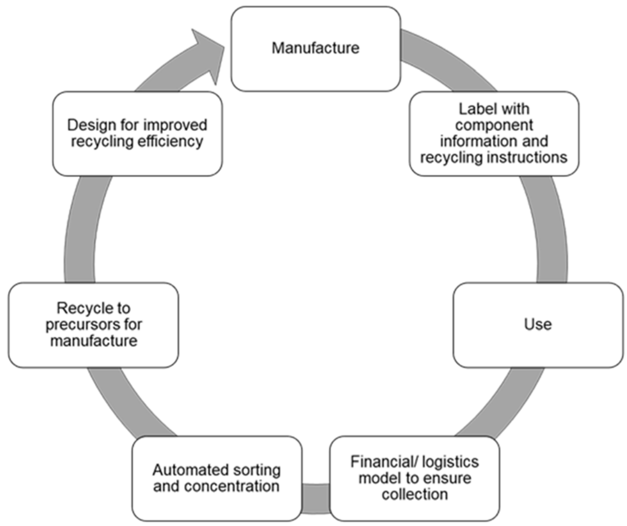

Fig. 3 Potential model for the life cycle of products in a circular economy when sustainable design criteria is implemented.

2.3 Recycling challenges

2.3.1 Compliance with legislation. Legislation and standards are proven methods for enabling circularity. Regulation is seen most noticeably in the management of WEEE.[38] The European WEEE directive is a prominent regulatory framework designed to address the EOL management of electronic devices and enhance the efficiency of recycling processes. Updated in 2019 (directive 2012/19/EC), it sets a goal of collecting 65% of WEEE, alongside specific targets for reuse, recovery, and recycling. However, few EU countries currently meet these targets, due to a mixture of economic, logistical, and technical challenges.[39] Furthermore, the existing recycling targets do not target the recovery of critical raw materials (CRMs), which are immensely important for circularity in electronics. New legislation from the EU "Ecodesign for Sustainable Product Regulation" will gradually improve product durability, reusability and repairability. But more significantly, it will address the substances that currently inhibit circularity, set targets for improving energy/resource efficiency and recycled content in electronics, as well as requiring information on environmental impacts of electronic goods such as carbon footprints.[39] While not currently enforced, manufacturers may adopt a scenario similar to that presented in Fig. 3, to meet these targets.

These targets seek to establish an Extended Producer Responsibility (EPR) that makes manufacturers responsible for monitoring their products and WEEE waste collection and treatment schemes.[40,41] Effective labelling would be a key enabler to inform recyclers about content and potential recycling protocols. Labelling is being explored for battery technology as a 'battery passport' and could consist of RFID/NFC tags or QR codes linked to cloud-based systems containing key information on the battery such as state of health, cell conformation and electrode chemistry.[42,43] "Battery passports" are expected to be implemented as early as 2026.[44] However, it is unclear how detailed the information will be as it is often sensitive to battery and/or car manufacturers. In addition, the Digital Product Passport for WEEE is expected to become mandatory around 2027 in alignment with the EU's Ecodesign for Sustainable Products Regulation (ESPR).

2.3.2 Generation and collection of WEEE. Fragmented regional or national regulation can hinder waste management and make it difficult to monitor WEEE flow.[45] Generally, the "production and trade" of WEEE are used for monitoring waste generation, as there is a strong link between trade statistics and national production statistics.[46] WEEE is collected through various channels, including retailers, communal collection points, or local government services. However, in the EU, less than 40% of WEEE is formally collected in this manner.[47] Collected waste may be recycled, sent to landfill or exported (mostly to West Africa or SE Asia). A significant portion of WEEE also ends up in mixed waste bins. In this instance WEEE is typically incinerated or sent to landfill sites without material recovery. The cumulative effect is that WEEE is often lost to landfill, which minimises how many TCMs can be recovered.

2.3.3 A hierarchy of assemblies. Any product which is an assembly of components is, in turn, composed of a collection of sub-assemblies or functional modules, which are themselves composed of components containing critical materials. For any technological product to be recycled, the first challenge is splitting the product into relevant sub-assemblies, to get to the component level, and then disassembling the components themselves in order to get to the point where materials can be recovered. Automated industrial disassembly will be a key enabler of recycling systems, to break products down to the component level, where their materials can be valorised.[48]

Automated disassembly is hindered by the vast array of connectors used to create these sub-assemblies. For instance, depending on the type of LIB cells (pouch, prismatic or cylindrical) battery modules can be assembled with nuts and bolts, structural adhesives, screws or using solder.[49] Standardisation of connectors would allow for automated disassembly and simplification of structure and manufacture. Fig. 4 shows the hierarchy of connectors for dis/re-assembly in terms of how easy they are to use. This outlines which connectors, or combination

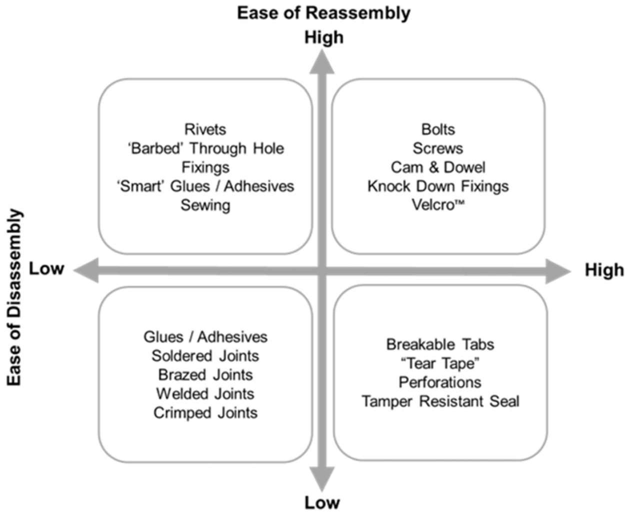

Fig. 4 A matrix of fixing methods ranked for 'Ease of Disassembly' on the x-axis and 'Ease of Reassembly' on the y-axis.

of connectors, can be utilised to ensure rapid disassembly and reassembly. For example, it was proposed that a combination of pressure sensitive adhesives in combination with Nylon zip cables could be used for small battery module assemblies. Where the time taken to dismantle the module was reduced from a few hours to under a minute, due to the removal of adhesive glues and welds.[50]

2.3.4 Shredding and separation. Full disassembly of WEEE is often not preferred due to the lack of full automation minimising the economic viability.[51] Instead, the standard method for recycling WEEE involves the manual removal of casings, such as plastics, glass, steel, aluminium, or copper wiring. Then, a series of sequential steps are applied to reduce the size of the material. This typically involves shredding, crushing, pulverising, and grinding. During shredding, the waste is broken down into smaller fragments, which are then further granulated into fine particles. Inline processing techniques can be commonly used to analyse the material composition and homogeneity, with methods such as X-ray fluorescence spectroscopy (XRF) or surface-enhanced Raman spectroscopy (SERS).[52] After pulverisation, various separation methods are applied to divide raw metals from non-metal components, such as gravity separation (to remove plastics, resins and glass from heavy metals), pyrometallurgy (to remove plastics), electrostatic or magnetic separation (to remove ferrous metals) and hydrometallurgy (to dissolve metals). However, there are few sites that deploy the full range of processes, so waste management facilities tend to specialise in one or two of these separation approaches and in collaboration with other separation companies.

2.4 Design for recycle approaches to deliver a circular economy

2.4.1 Current state of the art separation processes. Currently there are various techniques that can be employed, each targeting specific materials within the waste stream. Examples of separation processes and their specific targets are given in Table 1.

Many of these procedures originate from traditional mining and waste recycling industries, *e.g.*, magnetic separation has historically been used to remove impurities from non-metallic ores as well as treat solid waste streams, such as bottom ash from incinerators, and is now used to remove steel casings and other iron and nickel rich phases from WEEE waste.[63,64] Furthermore, density-based methods, such as air jets, are effective for sorting lightweight plastics from other, denser, materials. However, separation processes such as these become challenging when materials with different densities remain interlocked, *e.g.* the solar panel example discussed above. Many of techniques listed in Table 1 exhibit limited effectiveness

Table 1 Techniques available for the processing and separation of materials from TCM-containing wastes

Type of recycling process	Example techniques	Target	Example
Comminution and pre-treatments	Manual inspection	Separate WEEE that could be reused or is hazardous	Ensure fluids are removed Removal of batteries Confirm WEEE does not contain radioactive sources
	Disassembly or shredding	Brittle materials	Processing of battery cells and modules into 'black mass'[53]
	Sieving	Particle size	Mixed WEEE streams
Separation	Density separation	Heavy metals	Lead acid batteries
	Ultrasound	Brittle materials	Delamination of active material from current collectors in batteries[53]
	Electrostatic separation	Plastics	Separation of mixed polymer waste (*e.g.* polyethylene from polyvinyl chloride)[54]
	Magnetic separation	Ferrous metals	Separation of iron and nickel from WEEE.[55,56]
	Froth flotation	Hydrophobic materials	LIBs,[57] PCBs,[58] photovoltaics[59]
Leaching	Acid leaching (or use of selective redox catalysts)	Base metals	Dissolve Cu from PCBs,[60] recovery of Co and Ni from LIB cathodes.[61,62]

where target metals are present in low concentrations *e.g.* WEEE. Therefore, incorporating design for recycle principles, can significantly enhance the efficiency and effectiveness of these tools. The missing element is an open, cooperative conversation between manufacturers, recyclers and legislators on what these tools are and how they can be applied.

Recovery of TCMs can be improved by selective separation from plastic waste. Most recycling centres rely on optical sorting and infrared cameras to sort plastic waste as it moves along a conveyor belt. The speed of the conveyor belt is a critical factor for achieving high throughput and maximising efficiency. However, several important factors must be considered. Electronic substrates in WEEE often contain additives such as brominated flame retardants and use pigments that extend the lifetime of the plastics in the environment *via* their breakdown into microplastics.[65] Both of these lead to the classification of such waste as "hazardous", necessitating proper treatment and sorting. However, these darker plastics also complicate the application of infrared sorting techniques, so the success of these processes in sorting WEEE waste and avoiding the environmental impact lies in design changes and the use of alternative pigments.[65] Current regulatory trends are moving towards lowering the concentration thresholds for hazardous chemicals commonly found in WEEE (*e.g.* perfluoroalkylated substances, and brominated fire retardants).[66,67] This regulatory evolution has significant implications, as detecting lower concentrations of these substances may require a shift from infrared to X-ray fluorescence detection techniques, which could in turn reduce conveyor belt speeds and affect the magnitude of the waste treated and the subsequent profitability of a recycling scheme.

Other design criteria can have unintended consequences for recycling efficiency. Extensive use of structure adhesives, for example, may extend the product lifetime but may make recovery of TCMs impossible at EoL. The fast pace of change with product design may make smart recycling techniques outdated as product designs never reach a steady state. Regulations for control tend to be harsher on recyclers and easier on manufacturers. Principles of responsible innovation need to be applied to legislation, particularly in the area of WEEE.[68]

2.4.2 Identified "design for recycle" strategies. The need for design for recycle is obvious but for most manufacturers there is little incentive other than obtaining raw materials at reduced costs. This is why legislation, and standards also need to be implemented along with business and logistic models to ensure products become part of a circular economy and are economically viable to ensure market growth. Another essential mechanism to ensure the delivery of a sustainable recycling market is to establish forums for users, designers and recyclers where examples of improved design and generic tools can be showcased. Some examples for LIBs, photovoltaic devices and mobile phones already exist as case studies.[69–72]

Design for recycle tends to find favour is specific markets where the product is relatively uncomplicated, can be obtained in a standard form and is available in large volumes with higher intrinsic value. It is easier to apply to items such as solar cells than small devices such as mobile phones.[73,74] It is also clear that for mixed waste streams such as WEEE shredding followed by physical then chemical purification will always win as it is design and chemistry agnostic, and has a higher throughput compared to disassembly.

The choice of which materials are used in complex devices need to consider their performance, longevity and their recyclability in equal measure. An example of this is with the metals utilised in the casings, wiring and active materials. To enhance the product's durability, metals that are resistant to corrosion should be prioritised. However, these metals must also be amenable to dissolution during the recycling process. Base metals are generally easier to dissolve and should be considered for applications where recyclability is a key concern. The challenge lies in balancing the need for corrosion resistance with the requirement for efficient dissolution during recycling, making the careful selection of metals essential to both the product's longevity and its environmental impact. Additionally, choosing metals that can be efficiently recovered and purified with minimal energy input and waste production can further improve the sustainability of the recycling process.

Choice of polymers used in a device is almost as important as the types of metals as the formation of microplastics and fluorine-containing degradation products can have significant health and environmental consequences.[65,75] In many devices, structural adhesives are used to bind components together, due to the low cost, high strength and universal compatibility with different types of materials (metals, plastics, ceramics *etc.*). Many of these adhesives are thermoset materials, like epoxy resins, which do not dissolve or soften in solvents and do not become brittle at low enough temperatures to facilitate energy efficient removal steps.[76] This essentially irreversible bond necessitates shredding and pyrolysis steps when dealing with this kind of waste. However, debondable adhesives present an alternative to structural adhesives that possess either reversible linker units or structures with a fatal flaw.[76] In these adhesives the bonding can be reversed with a variety of stimuli such as heat, light, magnetism, electric fields or microwaves, as shown in Fig. 5. A simple example of this is the use of water-soluble binders to replace fluorinated binders within LIB electrode active materials.[77,78] The disadvantage of this approach is that exposure to natural stimuli can lead to unintended debonding. This could be circumvented by designing a need for two stimuli to be simultaneously applied, *e.g.* both a chemical and a thermal trigger.

The use of TCMs in short-lived products should be prohibited, where possible, for example lithium in single use batteries. The next tool is to enable reuse or repair where possible, extending the energy embodied in the device, from the extraction of its component materials and its fabrication. Design for recycle and design for repair do however have slightly different principles, which are sometimes in conflict.[79] For example, modularisation of a product would facilitate 'design for repair' by allowing failing modules to be taken out and repaired in isolation of the rest of the device. However, this principle will create barriers for recycling as modular designs hinder dismantling processes and could add additional material contaminants into shredded waste streams. The bonding of

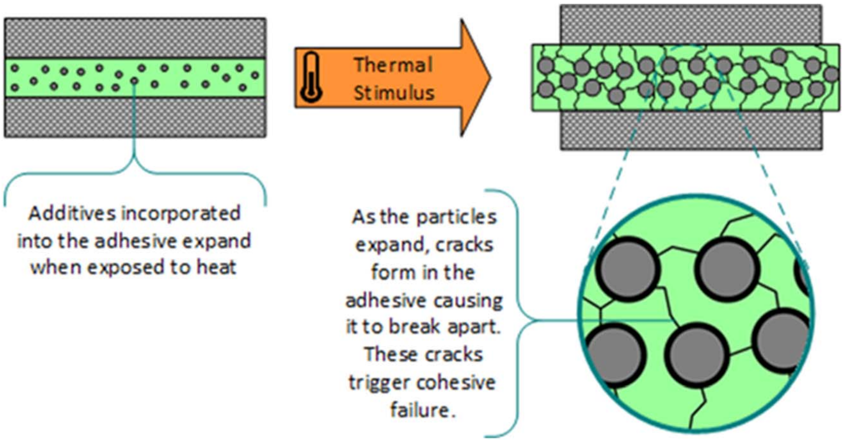

Fig. 5 Schematic representation of an example debonding process, triggered by a thermal stimulus. Adapted from Scheme 11 in ref. 76.

internal components and materials typically occur through welds, adhesive joints or physical connections, such as screws, clips or bolts. The first two of these are poorly reversible, whereas the latter, while reversible, can cost additional time to separate. The use of these physical connections in place of welds and adhesives enables repairs and replacement of modules, extending the life of the product. This concept has recently been discussed in terms of LIBs and the effect of design on disassembly time and costs has also been quantified (Table 2).[51]

3 Case studies

EOL fate is currently not a topic that commonly appears in design criteria lists. Extended producer responsibility is one method to ensure that this becomes an important criterion that designers and manufacturers consider. In order to illustrate how some of the aforementioned strategies can be implemented, this section will focus on five case studies: LIBs, photovoltaic cells, fuel cells, wind turbines and WEEE. These are some of the key technologies associated with the transition to sustainable energy creation and storage and are the key areas where the establishment of a circular economy is most critical, due to their high usage of critical materials. While the cases described are not exclusive, many of the technologies discussed are transferrable *e.g.*, the use of base metal substrates for catalytic metals is applicable across a wide range of applications.

3.1 Lithium-ion batteries

LIBs are often cited as being the most significant technology to power a net zero future and are currently the fastest growing energy technology on the market, due to falling costs and improving performance.[80] LIBs see widespread use across the transport and energy sectors, in electric vehicles (EVs) and stationary energy storage solutions for wind/solar farms and local communities. By 2023, the global number of EVs reached

Table 2 Potential design for recycle tools along with example applications and their respective advantages and disadvantages

Tool	Potential application	Advantages	Disadvantages
Etchable substrate under TCMs	Supported PGM catalysts or PCBs	Easier recovery of TCMs	Performance?
Additive manufacturing	Electronic circuitry	Easier assembly	May slow production
Fewer metals in alloys	Aerospace metals	More applications of recycled alloy	Performance?
More easily detachable tech	Speakers in cars or portable electronics, electric motors, replaceable batteries	Pre-concentration of TCMs	Trained disassembly required
Biodegradable composites	Electronic circuitry, wind turbine blades	Reduced EoL waste	Potential degradation in service
Water/solvent dispersible binders	LIBs	Easier to recycle, reduces use of harmful solvents during manufacturing	Need to ensure sufficient stability during use
Design for expected life	All technology	Get the most value out of components during initial use	Critical materials tied up in technology for longer. Can be at odds with design for recycle
Design out TCMs	LIBs, aerospace alloys	Reduces initial costs of manufacturing	Less economic incentive to recycle

45 million and the total volume of batteries used in the energy sector surpassed 2400 GW h globally.[80] In both instances, this is predicted to rise substantially, so that by 2035 there will be between 525 and 590 million EVs on the roads and a six-fold increase to stationary energy storage capacity supplied almost completely by LIBs.[80,81] Conversely, the capacity for battery recycling is in its relative infancy, where recycling capacity reached 300 GW h in 2023, pushed mainly by China. In a 'best case' scenario, where all proposed recycling projects are fulfilled, this is predicted to increase to 1500 GW h in 2030, but is highly dependent on future supply chains, overall effectiveness of recycling strategies on different waste streams and the environmental impacts.[81]

In their present form, LIBs are primarily designed with regards to optimising energy or power density, as well as their durability/longevity and cost. This usually involves the use of a large amount of packs or cells joined together with structural adhesives, such as thermoset resins, and an array of permanent, physical fastenings, such as screws and welds.[49,51,76] However, these design choices lead to severely limited economically viable disassembly routes for LIBs sent for recycling. Disassembly is preferable to shredding from an LCA perspective but the number of connectors and the complexity of the pack design hinders automation of disassembly.[10] Furthermore, the inherent stability of the polymeric binders present in the electrodes limits the effectiveness of delamination.[77] Quick and relatively cheap electrode delamination processes are essential, in order to separate cell components such as the metal oxides, current collector foils and graphite into distinct waste streams. However, current binders require high energy input or the use of expensive and toxic solvents, in most cases, to achieve this. Both of these factors contribute to shredding being the industry's preferred pre-treatment step for LIB waste.[73]

Recent reviews and articles have highlighted key factors, which could result in making LIBs easier to recycle. The main recommendations are:

- Reduce, where possible, the number of interfaces by replacing the conventional pack-module-cell conformations with battery packs that have no modules. *i.e.* replace modular Nissan Leaf pack designs with conformations like that of the BYD 'blade' cells (Fig. 6)
- Standardise physical connectors (clips, screws, bolts) within the battery to simplify tooling during disassembly.
- Replace structural adhesives, such as thermoset resins, with debondable adhesives in pack designs. These in-built failure modes can be made stable during use but can be exploited at EOL to simplify disassembly.[50,76]
- Place or anchor the anodes and cathode tabs on opposite ends of the pack, to simplify dismantling and collection of the electrodes into separate waste streams for subsequent recycling processes.[69]
- Use alternative water miscible binders other than carboxymethyl cellulose/styrene butadiene rubber to facilitate efficient delamination in environmentally friendly solvents.
- Implement battery passports, allowing recyclers to determine the state of health of an entire battery (on a cell, module and pack level), the pack design used, and the cell chemistries involved. They require this information to determine the most effective recycling processes to use, for safe material recovery at high yields.[83,84]

Adoption of these design principles are necessary to ensure that battery recycling can feed into a circular economy, whilst keeping costs and disassembly times low. It is also important to note that these concepts need to be accounted for in all future battery technologies beyond lithium-ion, in order to establish and maintain a sustainable supply chain and circular economy for these emerging technologies.

3.2 Photovoltaics

Photovoltaic (PV) systems are predicted to be a significant driver in the growth of renewable energy capacity, where new solar

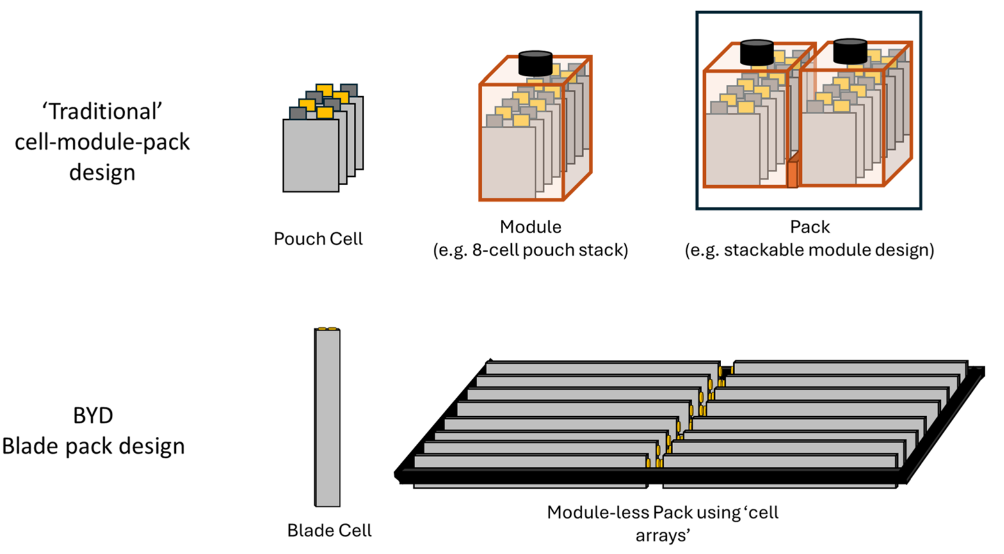

Fig. 6 Schematic diagrams showing an example of a 'traditional' battery pack design, using cells and modules, with an example of an 8-pouch cell configuration, and the BYD blade battery pack design, exhibiting a module-less system. Adapted from Fig. 3 in ref. 82.

systems will make up 80% of the growth between now and 2030.[85] However, the growth in waste PVs is projected to grow at a similarly rapid rate, so that by 2030 1.7–8 million tonnes will be generated, growing to 78 million tonnes by 2050.[86] This surge in waste will stem from older PV systems reaching the end of their life cycle, whilst the deployment of new PV installations rapidly expands. Additionally, any large-scale recycling processes for PV modules remain rudimentary, relying on mechanical shredding and crushing techniques, followed by chemical processing to recover the more valuable materials.[87] The preliminary mechanical treatments are necessary due to the sealed, sandwich-like structure of the PV modules, which are manufactured to prioritise durability over recyclability due to the large warranty periods (20–30 years) expected of PV devices.[88] Subsequently, due to the mixed waste streams produced from these processes and the poor quality of the recovered material, primarily due to antimony contamination, current PV recycling methodologies generally fail to turn a profit, as the products often end up downcycled into less valuable forms.[89,90] Alternative recycling pre-treatment steps, focusing on the disassembly of PV modules and the separation of each component, would lead to the creation of higher value products, however, the difficulty of separating encapsulation materials, *i.e.* EVA, from module components such as cells, glass, and backsheets, hinders PV recycling.

When considering how PVs can be designed with recyclability in mind, similar aspects to LIBs are quickly identified, such as minimising the use of non-reversible adhesives as encapsulation materials, replacing them with alternatives with exploitable failure modes, which can facilitate disassembly and material liberation. While EVA is the most widely used encapsulation material, primarily due to its good performance and low cost, alternatives have been investigated in an attempt to improve PV performance and lifetime, while reducing toxicity and manufacturing time.[91–93]

In some cases, these proposed encapsulation materials also present improved recyclability benefits, due to exploitable failure modes and a lack of additives like UV stabilisers. For instance, polyvinyl butyral (PVB) already sees use in other products which are recycled, such as safety glass and windscreens, where the recycling rate of PVB is 98%.[94] It is thought that, to some extent, the use of PVB would allow for similar principles to be exploited in PV recycling, however, the greater capacity for contamination within PVs may limit the effectiveness of such recovery routes.[94–96]

Another approach investigated recently focuses on the production of modules without any encapsulant material. Apollon Solar introduced NICE, utilizing technology derived from the insulating glass industry.[97] Here, nitrogen gas replaces the encapsulation material and is sealed into the modules using polyisobutylene (PIB).[97,98] This approach ensures long-term resistance to air and humidity, while maintaining mechanical contact between the module's components and facilitates a simplified manual disassembly, allowing intact component recovery. Fig. 7 shows the key differences between a traditional PV module and a module utilising the NICE technology. The separation and subsequent purity of the recovered PV components are especially important when considering reintroduction of recycled materials into PV manufacturing. This is because some of the components are particularly susceptible to contamination, such as silicon ingots and wafers. For silicon materials to enter secondary material markets, they must meet stringent purity specifications, where some specific combinations of materials make them incompatible for certain markets. Furthermore, recovery of PV-grade silicon is a priority, as production of ingots and wafers, which constitute only 3% of the mass of a PV panel, contribute the highest impacts to the life cycle of a PV panel, in terms of energy consumption and greenhouse gas emissions.

As with battery technology, the implementation of some of these design strategies and interfacial chemistries could allow for easier disassembly and subsequent recovery of purer PV components. However, it should be noted that a passport system would also need to be implemented, in order to allow recyclers to account for possible contaminants, such as antimony, lead, iron and fluorinated waste, all of which can cause additional environmental and safety risks during recycling procedures.[100–102]

3.3 Proton exchange membrane fuel cells (PEMFCs)

Proton Exchange Membrane Fuel Cells (PEMFCs) are considered as a promising clean energy source with the majority of applications in transportation, as well as stationary and portable energy storage. Within the transportation sector, the adoption of fuel cell electric vehicles (FCEVs) remains in its early stages. In 2021, there were 51 600 fuel cell electric vehicles (FCEVs) worldwide, a relatively modest figure compared to EVs, which totalled approximately 16.5 million vehicles in the same year.[103] The global PEMFC market was valued at US$0.44 billion in 2022, with transportation applications accounting for 79% of the total. This market is forecasted to surge to US$22.6 billion by 2031, largely driven by increased demand in the automotive industry.[104] Despite FCEV vehicles being commercially available for over a decade, the growth of the market when compared to EVs is constrained by a number of barriers. Arguably the largest barrier to the widespread adoption of this technology is the limited and expensive hydrogen refuelling infrastructure; it was reported that there were only 730 refuelling stations worldwide as of 2021.[103] Other factors include the lack of commercial models on the market and the high fuel and purchase costs. Geographically, the FCEV market is predominantly concentrated in Korea and the US with over 60% of the stock; the remainder being in Japan and China.[103]

In some respects, PEMFCs are similar to LIBs in terms of their structural architecture. Their architecture comprises thin layers of platinum group metals or metal oxides forming the cathode and anode, joined to opposite sides of an ion exchange polymeric membrane, using a fluorinated polymer binder which is commonly perfluorosulfonic acid (PFSA). The crucial aspect of cell design is the stability of the polymer membrane and binders. Given the chemistry of the anodic and cathodic processes, it is difficult to use non-fluorinated binders as they often lack the necessary durability and resistance to

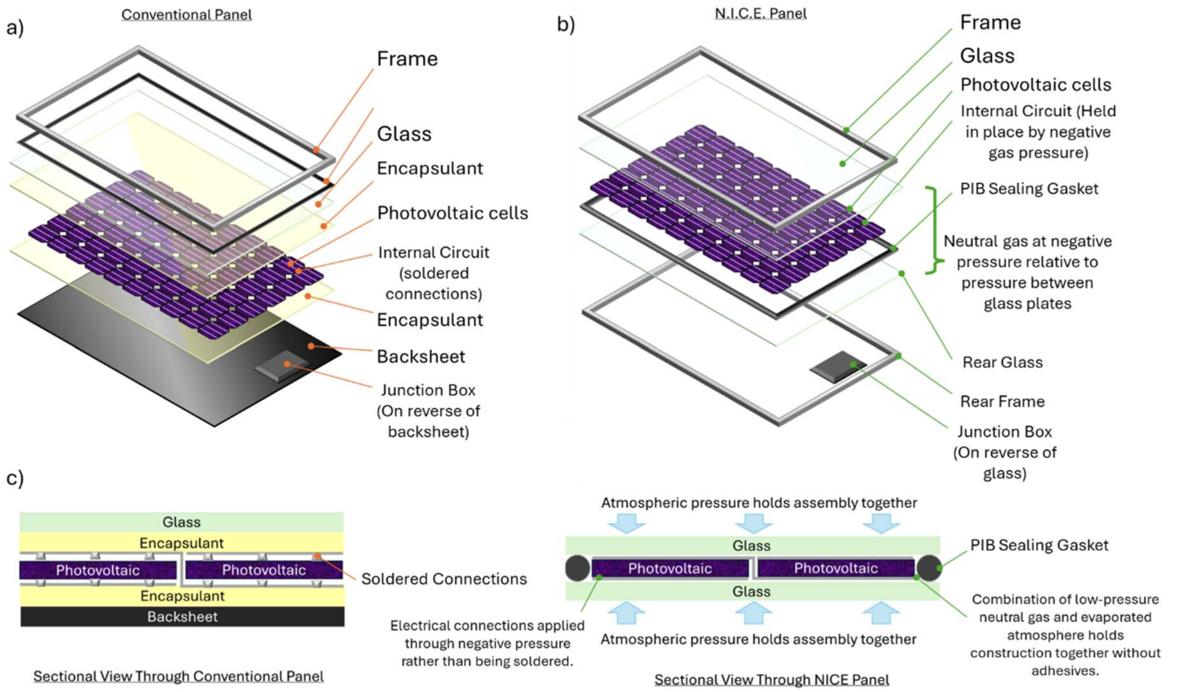

Fig. 7 (a) a schematic diagram depicting a traditional structure of a PV module utilising polymer encapsulation materials and (b) a schematic diagram showing a PV module with the NICE technology, removing the encapsulation material. (c) Sectional view through a traditional PV module and sectional view through a module with NICE technology.[99]

degradation under operating conditions. Some attempts have been made to investigate alternative hydrocarbon binders with limited success.[105]

Recently, it has been shown that a facile recycling technique can be applied, which briefly pre-soaks the membrane in ethanol and then removes both the anode and cathode layers using ultrasound in water. The overall removal takes less than 1 min and no detectable residue of PGM remains on the Nafion™ membrane.[106] This potentially allows most of the components to be recycled, however does not consider the separation of different metal catalyst nanoparticles. Tailoring the size of these nanoparticles could facilitate easier separation of different metals at EOL. However, achieving an optimal balance between particle size for catalytic activity and recovery efficiency is a critical trade-off, so could make this design change difficult to implement.

3.4 Magnets and motors

Rare earth elements are used across a wide variety of applications including catalysts, phosphors, display devices, batteries and optics. By far the largest sector is for their use in rare earth permanent magnets, based on an alloy of neodymium iron boron (NdFeB), particularly for electric motors in EVs and in generators in wind turbines. Wind turbines can contain 0.5 tonnes of NdFeB magnets per MW of power generation. The magnets are very large, and they are hazardous due to their high field strength. Therefore, it is important to be able to rapidly and safely separate the individual modules for repair or recycle. While magnet recycling has been carried out using hydrometallurgy, pyrometallurgy and hydrogen decrepitation there are factors such as carbon inclusion which can affect the properties of the recycled materials.[107,108] The magnets are often laser welded into hermetically sealed cans and potted in epoxy. As the magnets are large then surface contamination is less of the problem due to the epoxy resins. Automation is likely to play a role in the disassembly of turbines due to the significant safety considerations of handling these large magnets. The design issues are similar to those for LIBs *i.e.* large structures, hazardous to handle and breaking down to manageable units is the complex part.

EVs have many components which contain rare earth magnets including for example drive motors, regen braking, fans, speakers, power steering units and starter motors. Electric drives are likely to have a longer lifespan than electric batteries as they do not suffer from the same degradation issues. However, the designs for electric motors have evolved through time as have the strength of the magnets so reuse of these components in new cars is unlikely. However, there is a growing market for second hand parts for older vehicles. Standardisation of motor geometries could further facilitate reuse, but this should not be at the expense of efficiency in use. Magnet-containing applications are often heavily embedded in the vehicles which often makes them uneconomic to extract for recycling or reuse. Once the component (eg – drive motor) is extracted from the vehicle then the rotor needs separating from the stator which is often time consuming due to the fixations holding the motor together. The magnets are sometimes coated in epoxy resin and glued onto the outside of the rotor, or they are internally mounted and potted in resin. This means that to recycle the magnets they need to be removed from the epoxy binders, coatings or potting materials and they need to be separated from the electrical steels in the motor.

The inclusion of epoxy residues in recycled NdFeB magnets can increase the carbon content of the extracted materials. If the carbon content is too high in the alloy powders, then it can significantly reduce the magnetic properties of the directly re-sintered magnets. To lower the carbon content of the alloy powders this requires more complicated mineral processing and if this does not remove the epoxy then chemical treatment is required to separate the elements and to then pass the material back through metal winning, casting and then magnet manufacture. This significantly increases the environmental impact and cost of the recycling processes. Debondable adhesives could significantly improve the purity of the extracted NdFeB alloys and make direct short loop recycling easier, it would increase the product quality, reduce cost and reduce the complexity of the recycling processes.[109]

An additional complication when recycling NdFeB magnets is when the magnets have an anti-corrosion coating used to protect the magnet during service. A variety of metallic, organic and inorganic coatings have been used, which needs to be removed prior to many recycling processes. Some coatings have been shown to be easier to remove than others and therefore any design for recycle approach should take this into account.

The magnets used in different applications contain a variety of compositions and this has changed through time. For example, heavy rare earths (Dy and Tb) are typically added to the bulk of the magnet or to the surface to improve the coercivity of the magnets which stops them demagnetising in high-speed motors. This results in multiple grades of magnets. Any labelling of magnet containing components should state the composition and coating type in order to make automated sorting easier into different compositional streams.

3.5 Other WEEE

With an annual global production of >60 Mt WEEE is a major topic for design for recycle.[4] The mostly fibreglass substrate used gives strength which can aid longevity but it is also the main difficulty when it comes to recycling. A debondable thermoset would be a major advantage in the disassembly of printed circuit boards. An alternative which has been proposed for products of a shorter life is cellulose based electronics.[110,111] Use of a debonding agent or a biodegradable substrate would enable the metallic web of wires to be separated. Typically, between 20–40% by weight of electronic circuitry is metallic. Printed electronics achieved by additive manufacturing not only increase sustainability and recovery of materials at EOL but also decreases the carbon footprint during primary manufacture.[112,113] The use of graphene, printed in ink form, on a variety of surfaces has received significant interest for both ease of manufacture and disassembly.

Strong links have been shown to exist between design and recyclability of WEEE.[74] The ability of metals to be liberated from their matrices have been studied for ores but these are different from those observed for secondary waste. Extensive studies have correlated WEEE design recovery rates and found that in general recovery efficiency depends on the mechanical properties of the components and their joining methods, the type of joints between interfaces, the relative sizes and properties of neighbouring components and the complexity of spatially close phases and their interconnectivity.[114,115] This affects the degree of liberation and the extent of randomness of the shredded material and is connected to the type of

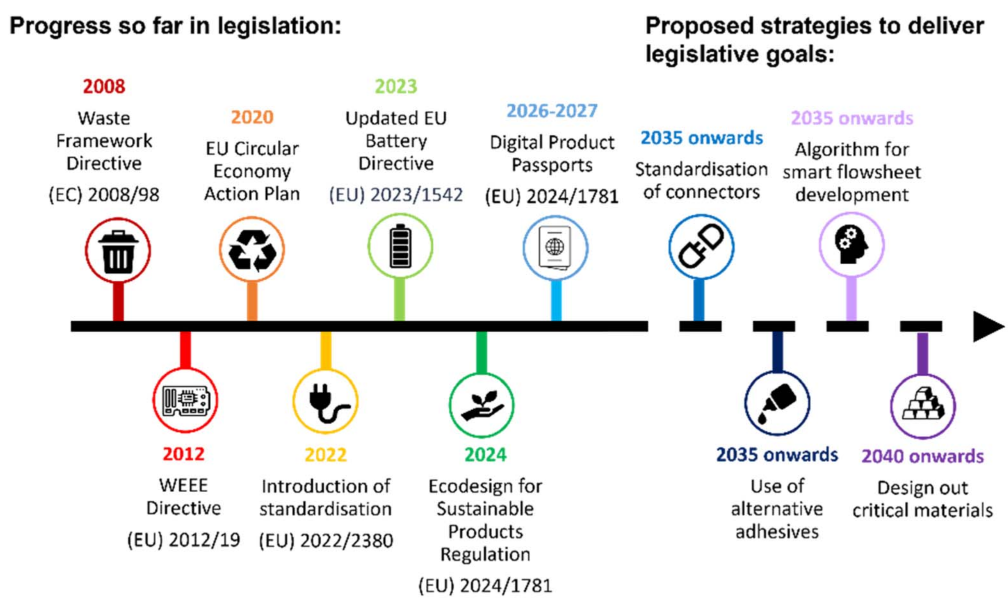

Fig. 8 Timeline of key "design for recycle" legislation influencing the recovery of technology-critical metals and proposed strategies to deliver legislative goals.

comminution method used together with the size of the particulate produced.

4 Conclusions

The concept of "design for recycle" is only just beginning to be implemented, particularly for complex materials containing technology critical metals. Fig. 8 presents a timeline of key legislative milestones, highlighting the progress to date and suggesting the most effective strategies we propose to integrate circularity into product design.

Early successes have been noted for plastic items by simplifying design and including fewer components. Products with fewer components are easier to sort into purer streams and lead to higher value source materials for recycled products. Technology critical metals are harder to separate and concentrate due to their dilute presence in most waste streams There are several recommendations which can significantly improve the recyclability of TCMs:

• Tagging and labelling items with digital product passports for cases where there is higher intrinsic value and creating reversible bonding mechanisms to promote their facile separation are key to efficient separation.

• Promote novel recycling tools to ensure that circularity becomes a criterion in product design. Forums where product designers can discuss manufacture with recyclers would start to remove some of the irreversible bonding techniques which make recycling so complex. Extended producer responsibility would make the concept of design for recycle more important for the manufacturer. Extending product lifetimes with different ownership models can simplify circularity and product logistics when combined with robust product labelling.

• Legislation and standards should be selective drivers to bring about circularity.

• Develop an algorithm for smart flowsheet development which will assess the value of critical metals within a device or product and suggest a suitable economically viable recycling flowsheet that minimises energy and chemical inputs.

• Simplify components *e.g.*, fewer plastic types, substitute TCMs where possible.

• Use more reversible connections *i.e.*, more physical connectors in place of structural adhesives.

• Use adhesives which are debondable or have green solvents which can be used to detach them.

• Electronic or block-chain labelling of devices containing TCMs to enable more efficient separation, concentration and recycling. This is set to be implemented by the EU in the form of digital product passports for energy-related products between 2026–2027.

Circularity for the critical metals used in sustainable energy creation and storage *e.g.* PV and wind turbines should be comparatively easier due to the unit sizes of devices and the limited number of owners. EVs need legislation to help create circularity as the volumes in use gets larger. Miscellaneous small electronic devices cause difficulties due to the large volume, difficulty of collection and variable value of its components. The frequent use of non-detachable batteries limits reuse and can cause safety issues. Labelling, design and standards can make TCM recovery more efficient.

Data availability

No primary research results, software or code have been included and no new data were generated or analysed as part of this review.

Conflicts of interest

There are no conflicts to declare.

Acknowledgements

The authors would like to thank the UKRI Interdisciplinary Circular Economy Centre for Technology Metals, Met4Tech project (EP/V011855/1) and the Faraday Institution (Faraday Institution grant code FIRG057, project website https://relib.org.uk), Engineering and Physical Sciences Research Council (EPSRC) Sonocat project (EP/W018632/1), REcycling CRitical Elements in Advanced Technologies for the Environment, RECREATE project (EP/Y53058X/1), 'GEOPIC' (EP/W019248/1) and REACT (UKRI240).

References

1 T. Watari, K. Nansai and K. Nakaijima, Major metals demand, supply, and environmental impacts to 2100: A critical review, *Conserv. Recycl.*, 2021, **164**, 105107.
2 EuRIC AISBL, Metal Recycling Factsheet, https://circulareconomy.europa.eu/platform/sites/default/files/euric_metal_recycling_factsheet.pdf#:~:text=Metalsare broadlypresentinavarietyof, accessed 5 February 2025.
3 S. Zhang, Y. Ding, B. Liu and C. Chi Chang, Supply and demand of some critical metals and present status of their recycling in WEEE, *Waste Manag.*, 2017, **65**, 113–127.
4 C. P. Baldé, R. Kuehr, T. Yamamoto, R. McDonald, E. D'Angelo, S. Althaf, G. Bel, O. Deubzer, E. Fernandez-Cubillo, V. Forti, V. Gray, S. Herat, S. Honda, G. Iattoni, D. S. Khetriwal, V. L. di Cortemiglia, Y. Lobuntsova, I. Nnorom, N. Pralat and M. Wagner, *Global E-Waste Monitor 2024, International Telecommunication Union (ITU) and United Nations Institute for Training and Research (UNITAR)*, Global E-waste Monitor 2024, Geneva/Bonn, 2024.
5 X. Zeng and J. Li, Measuring the recyclability of e-waste: An innovative method and its implications, *J. Clean. Prod.*, 2016, **131**, 156–162.
6 G. Zante, C. E. Elgar, J. M. Hartley, R. Mukherjee, J. Kettle, L. E. Horsfall, A. Walton, G. D. J. Harper and A. P. Abbott, A toolbox for improved recycling of critical metals and materials in low-carbon technologies, *RSC Sustain.*, 2024, **2**, 320–347.
7 M. Seban and O. Nowak, An Analysis of EU Collection and Recycling of Lead-based Automotive Batteries During the Period 2015-2017, https://www.eurobat.org/wp-content/

uploads/2021/09/RECYCLING-Dec20.pdf, accessed 19 February 2025.
8 F. Cucchiella, I. D'Adamo, S. C. Lenny Koh and P. Rosa, Recycling of WEEEs: An economic assessment of present and future e-waste streams, *Renew. Sustain. Energy Rev.*, 2015, **51**, 263–272.
9 A. Kuczyńska-Łażewska, E. Klugmann-Radziemska, Z. Sobczak and T. Klimczuk, Recovery of silver metallization from damaged silicon cells, Solar Energy Materials and Solar Cells, *Sol. Energy Mater. Sol. Cells*, 2018, **176**, 190–195.
10 D. Thompson, C. Hyde, J. M. Hartley, A. P. Abbott, P. A. Anderson and G. D. J. Harper, To shred or not to shred: A comparative techno-economic assessment of lithium ion battery hydrometallurgical recycling retaining value and improving circularity in LIB supply chains, *Resour. Conserv. Recycl.*, 2021, **175**, 105741.
11 Y. Fang, Y. Wen, L. Dai, C.-H. Wang, S. You and W. Li, Artificial intelligence in plastic recycling and conversion: A review, *Resour. Conserv. Recycl.*, 2025, **215**, 108090.
12 H. Zhang, J. Peeters, E. Demeester, J. R. Duflou and K. Kellens, *Proced. CIRP*, 2022, **106**, 264–269.
13 C. Athènes and EP intergroup, Achieving the full circular potential of aluminium, https://ebcd.org/wp-content/uploads/2022/01/European-Aluminuim-Catherine-Athenes-PDF-Presentation.pdf, accessed 19 February 2025.
14 N. Santero and J. Hendry, Harmonization of LCA methodologies for the metal and mining industry, *Int. J. Life Cycle Assess.*, 2016, **21**, 1543–1553.
15 G. Van Hoof, B. Robertz and B. Verrecht, Towards Sustainable Battery Recycling: A Carbon Footprint Comparison between Pyrometallurgical and Hydrometallurgical Battery Recycling Flowsheets, *Metals*, 2023, **13**, 1915.
16 D. Burchart-Korol, Life cycle assessment of steel production in Poland: A case study, *J. Clean. Prod.*, 2013, **54**, 235–243.
17 Steel Manufacturers Association, Steelmaking Emissions Report, https://steelnet.org/steelmaking-emissions-report-2022/, accessed 4 February 2025.
18 A. Cardarelli and M. Barbanera, Substitution of Fossil Coal with Hydrochar from Agricultural Waste in the Electric Arc Furnace Steel Industry: A Comprehensive Life Cycle Analysis, *Energies*, 2023, **16**, 5686.
19 T. Eglinton, J. Hinkley, A. Beath and M. Dell'Amico, Potential applications of concentrated solar thermal technologies in the Australian minerals processing and extractive metallurgical industry, *JOM*, 2013, **65**, 1710–1720.
20 B. A. A. Yousef, A. A. Hachicha, I. Rodriguez, M. A. Abdelkareem and A. Inyaat, Perspective on integration of concentrated solar power plants, *Int. J. Low Carbon Technol.*, 2021, **16**, 1098–1125.
21 O. D. Neikov, in *Handbook of Non-ferrous Metal Powders*, Elsevier, Amsterdam, 2nd edn, 2019, pp. 65–90.
22 P. M. Bell, Recent trends in extractive metallurgy, *EOS Trans. Am. Geophys. Union*, 1982, **63**, 220.
23 J. Cui and L. Zhang, Metallurgical recovery of metals from electronic waste: A review, *J. Hazard. Mater.*, 2008, **158**, 228–256.
24 E. M. Iannicelli-Zubiani, M. I. Giani, F. Recanati, G. Dotelli, S. Puricelli and C. Cristiani, Environmental impacts of a hydrometallurgical process for electronic waste treatment: A life cycle assessment case study, *J. Clean. Prod.*, 2017, **140**, 1204–1216.
25 T. W. Walker, N. Frelka, Z. Shen, A. K. Chew, J. Banick, S. Grey, M. S. Kim, J. A. Dumesic, R. C. Van Lehn and G. W. Huber, Recycling of multilayer plastic packaging materials by solvent-targeted recovery and precipitation, *Sci. Adv.*, 2020, **6**, eaba7599.
26 P. Kinnunen and S. Hedrich, Biotechnological strategies to recover value from waste, *Hydrometallurgy*, 2023, **222**, 106182.
27 A. P. Abbott, G. Frisch, J. Hartley and K. S. Ryder, Processing of metals and metal oxides using ionic liquids, *Green Chem.*, 2011, **13**, 471–481.
28 S. Ilyas and J. chun Lee, Biometallurgical Recovery of Metals from Waste Electrical and Electronic Equipment: a Review, *ChemBioEng Rev.*, 2014, **1**, 148–169.
29 J. M. Yoo, J. Jeong, K. Yoo, J. chun Lee and W. Kim, Enrichment of the metallic components from waste printed circuit boards by a mechanical separation process using a stamp mill, *Waste Manag.*, 2009, **29**, 1132–1137.
30 E. Ventura, A. Futuro, S. C. Pinho, M. F. Almeida and J. M. Dias, Physical and thermal processing of Waste Printed Circuit Boards aiming for the recovery of gold and copper, *J. Environ. Manage.*, 2018, **223**, 297–305.
31 M. Brojbiou, L. Mandache and V. Ivanov, in *2013 4th International Symposium on Electrical and Electronics Engineering, ISEEE 2013 - Proceedings*, 2013, pp. 1–5.
32 J. H. Zhang, T. D. Y. Chung and K. R. Oldenburg, A simple statistical parameter for use in evaluation and validation of high throughput screening assays, *J. Biomol. Screen*, 1999, **4**, 67–73.
33 F. Pfaff and U. D. Hanebeck, Sensor-based sorting, *Automatisierungstechnik*, 2020, **68**, 229–230.
34 M. M. Veras, A. S. Young, C. R. Born, A. Szewczuk, A. C. Bastos Neto, C. O. Petter and C. H. Sampaio, Affinity of dual energy X-ray transmission sensors on minerals bearing heavy rare earth elements, *Miner. Eng.*, 2020, **147**, 106151.
35 M. Kaya, Recovery of metals and nonmetals from electronic waste by physical and chemical recycling processes, *Waste Manag.*, 2016, **57**, 64–90.
36 C. Burkhardt, F. Ortiz, K. Daoud, T. Björnfot, F. Ahrentorp, J. Blomgren and A. Walton, Automated High-Speed Approaches for the Extraction of Permanent Magnets from Hard-Disk Drive Components for the Circular Economy, *Magnetism*, 2024, **4**, 295–304.
37 A. Walton, H. Yi, N. A. Rowson, J. D. Speight, V. S. J. Mann, R. S. Sheridan, A. Bradshaw, I. R. Harris and A. J. Williams, The use of hydrogen to separate and recycle neodymium-iron-boron-type magnets from electronic waste, *J. Clean. Prod.*, 2015, **104**, 236–241.

38 R. A. Patil and S. Ramakrishna, A comprehensive analysis of e-waste legislation worldwide, *Environ. Sci. Pollut. Res.*, 2020, **27**, 14412–14431.
39 C. P. Baldé, M. Wagner, G. Iattoni and R. Kuehr, *In-depth Review of the WEEE Collection Rates and Targets in the EU-28*, United Nations University (UNU)/United Nations Institute for Training and Research (UNITAR) – co-hosting the SCYCLE Programme, Norway, Switzerland and Iceland, Bonn, Germany, 2020.
40 UK Environment Agency, Electrical and electronic equipment (EEE): producer responsibilities, **https://www.gov.uk/guidance/electrical-and-electronic-equipment-eee-producer-responsibility#eee-producers-what-you-must-do**, accessed 6 February 2025.
41 European Union, European Union, Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE), **https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32012L0019#d1e32-53-1**, accessed 6 February 2025.
42 L. Wang, X. V. Wang, L. Gao and J. Váncza, A cloud-based approach for WEEE remanufacturing, *CIRP Ann.–Manuf. Technol.*, 2014, **63**, 409–412.
43 P. Nowakowski, A novel, cost efficient identification method for disassembly planning of waste electrical and electronic equipment, *J. Clean. Prod.*, 2018, **172**, 2695–2707.
44 V. Rizos and P. Urban, Implementing the EU Digital Battery Passport, **https://circulareconomy.europa.eu/platform/sites/default/files/2024-03/1qp5rxiZ-CEPS-InDepthAnalysis-2024-05_Implementing-the-EU-digital-battery-passport.pdf**, accessed 5 February 2025.
45 A. Kumar, M. Holuszko and D. C. R. Espinosa, E-waste: An overview on generation, collection, legislation and recycling practices, *Resour. Conserv. Recycl.*, 2017, **122**, 32–42.
46 V. Forti, C. P. Baldé, R. Kuehr and G. Bel, *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, 2020.
47 S. Salhofer, B. Steuer, R. Ramusch and P. Beigl, WEEE management in Europe and China – A comparison, *Waste Manag.*, 2016, **57**, 27–35.
48 S. Glöser-Chahoud, S. Huster, S. Rosenberg, S. Baazouzi, S. Kiemel, S. Singh, C. Schneider, M. Weeber, R. Miehe and F. Schultmann, Industrial disassembling as a key enabler of circular economy solutions for obsolete electric vehicle battery systems, *Resour. Conserv. Recycl.*, 2021, **174**, 105735.
49 K. Wegener, S. Andrew, A. Raatz, K. Dröder and C. Herrmann, Disassembly of electric vehicle batteries using the example of the Audi Q5 hybrid system, *Proced. CIRP*, 2014, **23**, 155–160.
50 S. Scott, Z. Islam, J. Allen, T. Yingnakorn, A. Alflakian, J. Hathaway, A. Rastegarpanah, G. D. J. Harper, E. Kendrick, P. A. Anderson, J. Edge, L. Lander and A. P. Abbott, Designing lithium-ion batteries for recycle : The role of adhesives, *Next Energy*, 2023, **1**, 100023.
51 L. Lander, C. Tagnon, V. Nguyen-Tien, E. Kendrick, R. J. R. Elliott, A. P. Abbott, J. S. Edge and G. J. Offer, Breaking it down: A techno-economic assessment of the impact of battery pack design on disassembly costs, *Appl. Energy*, 2023, **331**, 120437.
52 M. Chakraborty, J. Kettle and R. Dahiya, Electronic Waste Reduction Through Devices and Printed Circuit Boards Designed for Circularity, *IEEE J. Flex. Electron.*, 2022, **1**, 4–23.
53 C. Lei, I. Aldous, J. M. Hartley, D. L. Thompson, S. Scott, R. Hanson, P. A. Anderson, E. Kendrick, R. Sommerville, K. S. Ryder and A. P. Abbott, Lithium ion battery recycling using high-intensity ultrasonication, *Green Chem.*, 2021, **23**, 4710–4715.
54 D. K. Yanar and B. A. Kwetkus, Electrostatic separation of polymer powders, *J. Electrost.*, 1995, **35**, 257–266.
55 D. Marinos and B. Mishra, An Approach to Processing of Lithium-Ion Batteries for the Zero-Waste Recovery of Materials, *J. Sustain. Metall.*, 2015, **1**, 263–274.
56 A. Marra, A. Cesaro and V. Belgiorno, Separation efficiency of valuable and critical metals in WEEE mechanical treatments, *J. Clean. Prod.*, 2018, **186**, 490–498.
57 R. Zhan, Z. Oldenburg and L. Pan, Recovery of active cathode materials from lithium-ion batteries using froth flotation, *Sustain. Mater. Technol.*, 2018, **17**, e00062.
58 N. İ. Dinç, A. U. Tosun, E. Baştürkcü, M. Özer and F. Burat, Recovery of valuable metals from WPCB fines by centrifugal gravity separation and froth flotation, *J. Mater. Cycles Waste Manag.*, 2022, **24**, 224–236.
59 W. Berger, F. G. Simon, K. Weimann and E. A. Alsema, A novel approach for the recycling of thin film photovoltaic modules, *Resour. Conserv. Recycl.*, 2010, **54**, 711–718.
60 S. Gulliani, M. Volpe, A. Messineo and R. Volpe, Recovery of metals and valuable chemicals from waste electric and electronic materials: a critical review of existing technologies, *RSC Sustain.*, 2023, **1**, 1085.
61 R. Golmohammadzadeh, F. Faraji and F. Rashchi, Recovery of lithium and cobalt from spent lithium ion batteries (LIBs) using organic acids as leaching reagents: A review, *Resour. Conserv. Recycl.*, 2018, **136**, 418–435.
62 L. P. He, S. Y. Sun, Y. Y. Mu, X. F. Song and J. G. Yu, Recovery of Lithium, Nickel, Cobalt, and Manganese from Spent Lithium-Ion Batteries Using l -Tartaric Acid as a Leachant, *ACS Sustain. Chem. Eng.*, 2017, **5**, 714–721.
63 J. Ku, K. Wang, Q. Wang and Z. Lei, Application of Magnetic Separation Technology in Resource Utilization and Environmental Treatment, *Separations*, 2024, **11**, 130.
64 S. Back, K. Ueda and H. Sakanakura, Determination of metal-abundant high-density particles in municipal solid waste incineration bottom ash by a series of processes: Sieving, magnetic separation, air table sorting, and milling, *Waste Manag.*, 2020, **112**, 11–19.
65 S. Key, P. G. Ryan, S. E. Gabbott, J. Allen and A. P. Abbott, Influence of colourants on environmental degradation of plastic litter, *Environ. Pollut.*, 2024, **347**, 123701.

66 M. Rani, M. Keshu, M. Sillanpää and U. Shanker, An updated review on environmental occurrence, scientific assessment and removal of brominated flame retardants by engineered nanomaterials, *J. Environ. Manage.*, 2022, **321**, 115998.
67 UK Environment Agency, Poly- and perfluoroalkyl substances (PFAS): sources, pathways and environmental data, https://assets.publishing.service.gov.uk/media/611e31fbd3bf7f63b19cea2d/Poly-_and_perfluoroalkyl_substances_-sources_pathways_and_environmental_data_-_report.pdf, accessed 19 February 2025.
68 G. Biggi, Steering responsible innovation: the impact of EU REACH legislation on the novelty and toxicity of chemical inventions, *Int. J. Innov.*, 2024, **31**, 922–943.
69 D. L. Thompson, J. M. Hartley, S. M. Lambert, M. Shiref, G. D. J. Harper, E. Kendrick, P. Anderson, K. S. Ryder, L. Gaines and A. P. Abbott, The importance of design in lithium ion battery recycling-a critical review, *Green Chem.*, 2020, **22**, 7585–7603.
70 A. Norgren, A. Carpenter and G. Heath, Design for Recycling Principles Applicable to Selected Clean Energy Technologies: Crystalline-Silicon Photovoltaic Modules, Electric Vehicle Batteries, and Wind Turbine Blades, *J. Sustain. Metall.*, 2020, **6**, 761–774.
71 K. Schischke, A. Berwald, G. Dimitrova, J. Rückschloss, N. F. Nissen and M. Schneider-Ramelow, *Proced. CIRP*, 2022, **105**, 619–624.
72 F. Poppelaars, C. Bakker and J. van Engelen, Design for divestment in a circular economy: Stimulating voluntary return of smartphones through design, *Sustainability*, 2020, **12**, 1488.
73 G. Harper, R. Sommerville, E. Kendrick, L. Driscoll, P. Slater, R. Stolkin, A. Walton, P. Christensen, O. Heidrich, S. Lambert, A. Abbott, K. Ryder, L. Gaines and P. Anderson, Recycling lithium-ion batteries from electric vehicles, *Nature*, 2019, **575**, 75–86.
74 A. Van schaik and M. A. Reuter, in *Waste Electrical and Electronic Equipment (WEEE) Handbook*, 2012, pp. 163–211.
75 W. J. Hall and P. T. Williams, Analysis of products from the pyrolysis of plastics recovered from the commercial scale recycling of waste electrical and electronic equipment, *J. Anal. Appl. Pyrolysis*, 2007, **79**, 375–386.
76 K. R. Mulcahy, A. F. R. Kilpatrick, G. D. J. Harper, A. Walton and A. P. Abbott, Debondable adhesives and their use in recycling, *Green Chem.*, 2022, **24**, 36–61.
77 S. Scott, J. Terreblanche, D. L. Thompson, C. Lei, J. M. Hartley, A. P. Abbott and K. S. Ryder, Gelatin and Alginate Binders for Simplified Battery Recycling, *J. Phys. Chem. C*, 2022, **126**, 8489–8498.
78 D. Bresser, D. Buchholz, A. Moretti, A. Varzi and S. Passerini, Alternative binders for sustainable electrochemical energy storage-the transition to aqueous electrode processing and bio-derived polymers, *Energy Environ. Sci.*, 2018, **11**, 3096–3127.
79 M. Gharfalkar, Z. Ali and G. Hillier, Clarifying the disagreements on various reuse options: Repair, recondition, refurbish and remanufacture, *Waste Manag. Res.*, 2016, **34**, 995–1005.
80 International Energy Agency (IEA), Batteries and Secure Energy Transitions, https://iea.blob.core.windows.net/assets/cb39c1bf-d2b3-446d-8c35-aae6b1f3a4a0/BatteriesandSecureEnergyTransitions.pdf, accessed 10 January 2025.
81 International Energy Agency (IEA), Global EV Outlook 2024, https://iea.blob.core.windows.net/assets/a9e3544b-0b12-4e15-b407-65f5c8ce1b5f/GlobalEVOutlook2024.pdf%0A, accessed 19 February 2024.
82 X. Meng, The Next-Generation Battery Pack Design: from the BYD Blade Cell to Module-Free Battery Pack, https://medium.com/batterybits/the-next-generation-battery-pack-design-from-the-byd-blade-cell-to-module-free-battery-pack-2b507d4746d1, accessed 15 January 2025.
83 K. Berger, J. P. Schöggl and R. J. Baumgartner, Digital battery passports to enable circular and sustainable value chains: Conceptualization and use cases, *J. Clean. Prod.*, 2022, **353**, 131492.
84 Y. Bai, N. Muralidharan, Y. K. Sun, S. Passerini, M. Stanley Whittingham and I. Belharouak, Energy and environmental aspects in recycling lithium-ion batteries: Concept of Battery Identity Global Passport, *Mater. Today*, 2020, **41**, 304–315.
85 International Energy Agency (IEA), Renewables 2024, https://www.iea.org/reports/renewables-2024, accessed 15 January 2025.
86 S. Weckend, A. Wade and G. Heath, END-OF-LIFE management: solar photovoltaic panels, https://www.irena.org/publications/2016/Jun/End-of-life-management-Solar-Photovoltaic-Panels, accessed 15 January 2025.
87 EPRI, Alliance for Sustainable Energy, Operator of the National Renewable Energy Laboratory and Wambuch-Consulting, *Program on Technology Innovation: Insights on Photovoltaic Recycling Processes in Europe*, Palo Alto, CA, A Survey-Based Approach., 3002008846.
88 M. Aghaei, A. Fairbrother, A. Gok, S. Ahmad, S. Kazim, K. Lobato, G. Oreski, A. Reinders, J. Schmitz, M. Theelen, P. Yilmaz and J. Kettle, Review of degradation and failure phenomena in photovoltaic modules, *Renew. Sustain. Energy Rev.*, 2022, **159**, 112160.
89 C. C. Farrell, A. I. Osman, R. Doherty, M. Saad, X. Zhang, A. Murphy, J. Harrison, A. S. M. Vennard, V. Kumaravel, A. H. Al-Muhtaseb and D. W. Rooney, Technical challenges and opportunities in realising a circular economy for waste photovoltaic modules, *Renew. Sustain. Energy Rev.*, 2020, **128**, 109911.
90 G. Granata, P. Altimari, F. Pagnanelli and J. De Greef, Recycling of solar photovoltaic panels: Techno-economic assessment in waste management perspective, *J. Clean. Prod.*, 2022, **363**, 132384.
91 B. Adothu, F. R. Costa and S. Mallick, in *Proceedings of the 38th European Photovoltaic Solar Energy Conference and Exhibition*, Lisbon, Portugal, 2021, pp. 631–639.

92 M. Kempe, in *2011 37th IEEE Photovoltaic Specialists Conference (PVSC)*, 2011, pp. 85–90.
93 G. Oreski, J. Stein, G. Eder, K. Berger, L. S. Bruckman, J. Vedde, K.-A. Weiss, T. Tanahashi, R. H. French and S. Ranta, *Designing New Materials for Photovoltaics: Opportunities for Lowering Cost and Increasing Performance through Advanced Material Innovations*, 2021, DOI: **10.2172/1779380**.
94 S. Khouri, M. Behun, L. Knapcikova, A. Behunova, M. Sofranko and A. Rosova, Characterization of customized encapsulant polyvinyl butyral used in the solar industry and its impact on the environment, *Energies*, 2020, **13**, 5391.
95 M. Tupy, P. Mokrejs, D. Merinska, P. Svoboda and J. Zvonicek, Windshield recycling focused on effective separation of PVB sheet, *J. Appl. Polym. Sci.*, 2014, **131**, 39879.
96 L. Šooš, M. Matúš, M. Pokusová, V. Čačko and J. Bábics, The recycling of waste laminated glass through decomposition technologies, *Recycling*, 2021, **6**, 26.
97 E. Saint-Sernin, R. Einhaus, K. Bamberg and P. Panno, in *23rd European Photovoltaic Solar Energy Conference and Exhibition*, Valencia, Spain, 2008, pp. 2825–2828.
98 A. K. Schnatmann, F. Schoden and E. Schwenzfeier-Hellkamp, Sustainable PV Module Design—Review of State-of-the-Art Encapsulation Methods, *Sustainability*, 2022, **14**, 9971.
99 R. Einhaus, NICE Technology for Bifacial PV Modules 'New Industrial Cell Encapsulation', **https://serchile.cl/wp-content/uploads/2015/03/2.7-ROLAND-EINHAUS-bifi-PSDA-Antofagasta-Chile-2015.pdf**, accessed 19 February 2025.
100 C. M. Motta, R. Cerciello, S. De Bonis, V. Mazzella, P. Cirino, R. Panzuto, M. Ciaravolo, P. Simoniello, M. Toscanesi, M. Trifuoggi and B. Avallone, Potential toxicity of improperly discarded exhausted photovoltaic cells, *Environ. Pollut.*, 2016, **216**, 786–792.
101 V. Aryan, M. Font-Brucart and D. Maga, A comparative life cycle assessment of end-of-life treatment pathways for photovoltaic backsheets, *Prog. Photovoltaics Res. Appl.*, 2018, **26**, 443–459.
102 M. Thuis, N. M. Al Hasan, R. L. Arnold, B. King, A. Maes, D. C. Miller, J. M. Newkirk, L. T. Schelhas, A. Sinha, K. Terwilliger, S. Ulicna and K. Van Durme, A Comparison of Emerging Nonfluoropolymer-Based Coextruded PV Backsheets to Industry-Benchmark Technologies, *IEEE J. Photovolt.*, 2022, **12**, 88–96.
103 International Energy Agency (IEA), Global EV Outlook 2022, **https://iea.blob.core.windows.net/assets/ad8fb04c-4f75-42fc-973a-6e54c8a4449a/GlobalElectricVehicleOutlook2022.pdf**, accessed 19 February 2025.
104 Transparency Market Research, PEM Fuel Cell Materials Market Outlook 2031, **https://www.transparencymarketresearch.com/pem-fuel-cells-materials.html%0A**, accessed 19 February 2025.
105 Y. S. Kim, Hydrocarbon Ionomeric Binders for Fuel Cells and Electrolyzers, *Adv. Sci.*, 2023, **10**, 2302914.
106 T. Yingnakorn, J. Hartley, M. E. Keal, R. Gordon, D. M. Florido, A. P. Abbott and J. M. Yang, Catalyst Coated Membranes for Fuel Cell and Water Electrolyser Delamination Induced by Organic Solution Soaking and Water Ultrasonication, *RSC Sustain.*, 2025, **3**, 1900–1908.
107 T. Elwert, D. Goldmann, F. Roemer and S. Schwarz, Recycling of NdFeB Magnets from Electric Drive Motors of (Hybrid) Electric Vehicles, *J. Sustain. Metall.*, 2017, **3**, 108–121.
108 M. Kaya, An overview of NdFeB magnets recycling technologies, *Curr. Opin. Green Sustain. Chem.*, 2024, **46**, 100884.
109 C. Jönsson, M. Awais, L. Pickering, M. Degri, W. Zhou, A. Bradshaw, R. Sheridan, V. Mann and A. Walton, The extraction of NdFeB magnets from automotive scrap rotors using hydrogen, *J. Clean. Prod.*, 2020, **277**, 124058.
110 M. Reuter and A. Van Schaik, in *Electronics Goes Green 2012+, ECG 2012 - Joint International Conference and Exhibition, Proceedings*, 2012, pp. 1–8.
111 R. R. Nair, J. Wolansky, K. Uhlig, A. Solgi, L. Teuerle, T. Zhang, T. A. Jonas Schröder, J. Benduhn, H. Kleemann and K. Leo, Leaftronics: Natural lignocellulose scaffolds for sustainable electronics, *Sci. Adv.*, 2024, **10**, eadq3276.
112 A. Sudheshwar, N. Malinverno, R. Hischier, B. Nowack and C. Som, The need for design-for-recycling of paper-based printed electronics – a prospective comparison with printed circuit boards, *Resour. Conserv. Recycl.*, 2023, **189**, 106757.
113 T. Zhang, A. Bainbridge, J. Harwell, S. Zhang, M. Wagih and J. Kettle, Life cycle assessment (LCA) of circular consumer electronics based on IC recycling and emerging PCB assembly materials, *Sci. Rep.*, 2024, **14**, 29183.
114 M. Kaya, in *Waste Electrical and Electronic Equipment Recycling*, Woodhead Publishing, Cambridge, 2018, pp. 33–93.
115 R. Cherrington and K. Makenji, in *Waste Electrical and Electronic Equipment (WEEE) Handbook*, Woodhead Publishing, Cambridge, 2nd edn, 2019, pp. 283–310.