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# Material efficiency strategies to reducing greenhouse gas emissions associated with buildings, vehicles, and electronics—a review

**Edgar G Hertwich[1], Saleem Ali[2], Luca Ciacci[3], Tomer Fishman[1,4], Niko Heeren[1], Eric Masanet[5], Farnaz Nojavan Asghari[1], Elsa Olivetti[6], Stefan Pauliuk[7], Qingshi Tu[1] and Paul Wolfram[1]**

[1] School of Forestry & Environmental Studies, Yale University, New Haven, CT 06511, United States of America
[2] College of Earth, Ocean, and the Environment, University of Delaware, Newark, DE 19716-3501, United States of America
[3] Department of Industrial Chemistry, University of Bologna, Viale Risorgimento, 4, I-40136 Bologna, Italy
[4] Interdisciplinary Center, Herzliya, Israel
[5] Department of Mechanical Engineering, Northwestern University, Evanston, IL 60208-3109, United States of America
[6] Department of Material Science and Engineering, Massachusetts Institute of Technology, Cambridge, MA 02139, United States of America
[7] University of Freiburg, Faculty of Environment and Natural Resources, D-79106 Freiburg, Germany

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## Abstract

As one quarter of global energy use serves the production of materials, the more efficient use of these materials presents a significant opportunity for the mitigation of greenhouse gas (GHG) emissions. With the renewed interest of policy makers in the circular economy, material efficiency (ME) strategies such as light-weighting and downsizing of and lifetime extension for products, reuse and recycling of materials, and appropriate material choice are being promoted. Yet, the emissions savings from ME remain poorly understood, owing in part to the multitude of material uses and diversity of circumstances and in part to a lack of analytical effort. We have reviewed emissions reductions from ME strategies applied to buildings, cars, and electronics. We find that there can be a systematic trade-off between material use in the production of buildings, vehicles, and appliances and energy use in their operation, requiring a careful life cycle assessment of ME strategies. We find that the largest potential emission reductions quantified in the literature result from more intensive use of and lifetime extension for buildings and the light-weighting and reduced size of vehicles. Replacing metals and concrete with timber in construction can result in significant GHG benefits, but trade-offs and limitations to the potential supply of timber need to be recognized. Repair and remanufacturing of products can also result in emission reductions, which have been quantified only on a case-by-case basis and are difficult to generalize. The recovery of steel, aluminum, and copper from building demolition waste and the end-of-life vehicles and appliances already results in the recycling of base metals, which achieves significant emission reductions. Higher collection rates, sorting efficiencies, and the alloy-specific sorting of metals to preserve the function of alloying elements while avoiding the contamination of base metals are important steps to further reduce emissions.

## Introduction

The production of major materials (iron and steel, aluminum, cement, chemical products, and pulp and paper) accounted for 26% of global final energy use and 18% of $CO_2$ emissions from fossil fuels and industrial processes in 2014 [1]. *Material-* or *resource efficiency* [2–5] measures the quantity of physical services provided per unit of material. For climate change mitigation, material efficiency (ME) strategies seek to achieve similar outcomes with the use of less materials or less emissions-intensive materials [6]. ME strategies such as light-weighting of and lifetime extension for products, reuse, remanufacturing, recycling of materials, and appropriate material choice, have recently been recognized as an important yet hereto largely untapped opportunity for emissions abatement [7, 8].

Among policy makers, a recent surge in the interest in ME was triggered by the popularity of the Circular Economy and concerns about plastic pollution of oceans. Only recently policy makers have started to

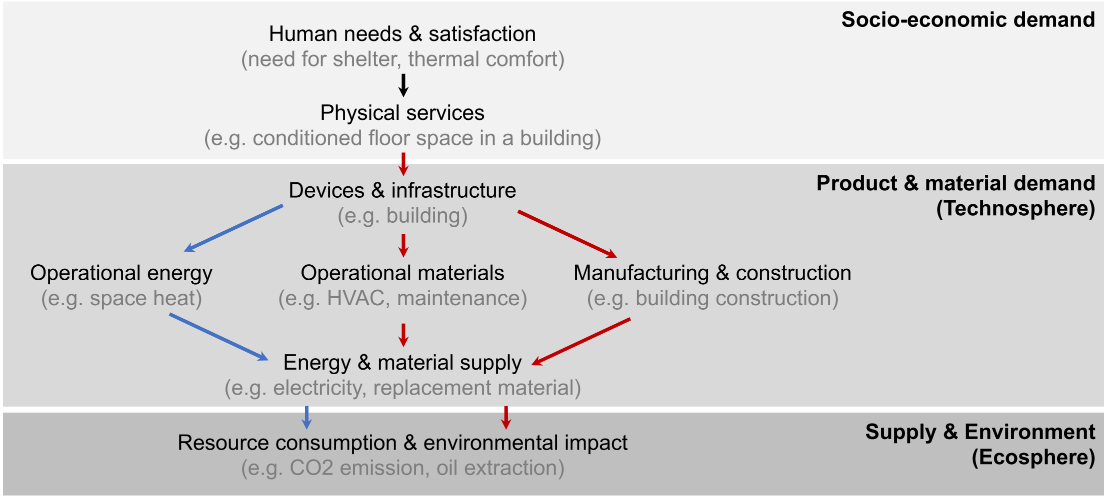


**Figure 1.** Human need fulfillment depends on material consumption, including basic needs like nutrition and shelter and more advanced needs such as connectivity or self-realization. These services are satisfied by manufactured products whose production, delivery, and operation requires both energy (blue arrows) and resources (red arrows) and causes emissions. Each step in the chain between needs and resource use presents an opportunity for decoupling and reducing resource use and associated emissions.

focus on potential synergies and trade-offs between ME and greenhouse gas (GHG) mitigation, for example through the G7 Alliance on Resource Efficiency [9] and the Resource Efficiency Dialogue of the G20 [10, 11]. In these policy circles, the term *resource efficiency* is used in a manner that is synonymous with the use ME in the scientific literature [3], and we use the more precise scientific term in this review.

This review addresses the current state of knowledge regarding GHG abatement through ME, focusing on products groups for which ME strategies are particularly relevant: buildings, vehicles, and electrical and electronic equipment (EEE) [2, 3, 12]. The focus on the product perspective was chosen because consumers, producers, and policy directly relate to them. Demand projections for products can be linked to sustainable development scenarios. We review research and policy analyses to answer the following questions: what strategies have been identified for each product group? What are the potential GHG emission reductions of different strategies? What are important gaps that encumber our understanding?

In the past decade, Allwood [13], Gutowski [14], Worrell [2], and colleagues have taken the lead in the investigation of a wide range of ME opportunities. National and European assessments of waste management policies have sometimes quantified emission reductions connected to waste management and recycling [5, 15]. In a first model-based assessment conducted by the International Energy Agency (IEA), ME makes a small but not insignificant contribution of 0.6 Gt to emission reductions in the industry sector of 8 Gt by 2060 [1]. Addressing a more comprehensive range of measures in a bottom-up approach, a European think tank recently estimated a much more substantial mitigation potential of 56% in emissions from the steel, aluminum, plastics and cement production sectors [12]. Both efforts drew on existing, bottom-up assessments of specific products and strategies, e.g. for steel [16], and were hampered by a lack of established data, as well as agreed methods and models to estimate emissions reductions.

## Defining ME strategies

The goods and services to satisfy human needs typically consist of, or require, materials for their production and delivery (figure 1). Materials are as fundamental to economic activity as energy and labor. However, there are great differences in the amounts and types of material or product that are required to fulfil a service. ME has been defined both as an indicator—i.e. the amount of physical service provision per unit material—and as a strategy for climate change mitigation. A meeting convened by the Royal Academy [6] offers following definition: '[ME] entails the pursuit of technical strategies, business models, consumer preferences and policy instruments that would lead to a substantial reduction in the production of high-volume energy-intensive materials required to deliver human well-being.' The following strategies are described in the literature (figure 2) [3, 5].

1. More intensive use [4]: less product to provide the same service, e.g. through a more space-efficient design of buildings or multifunctionality of gadgets [17], or use of a product at a higher utilization rate, e.g. through sharing.
2. Lifetime extension (including through repair, resale, remanufacturing) [18, 19]: more service provided by an existing product.

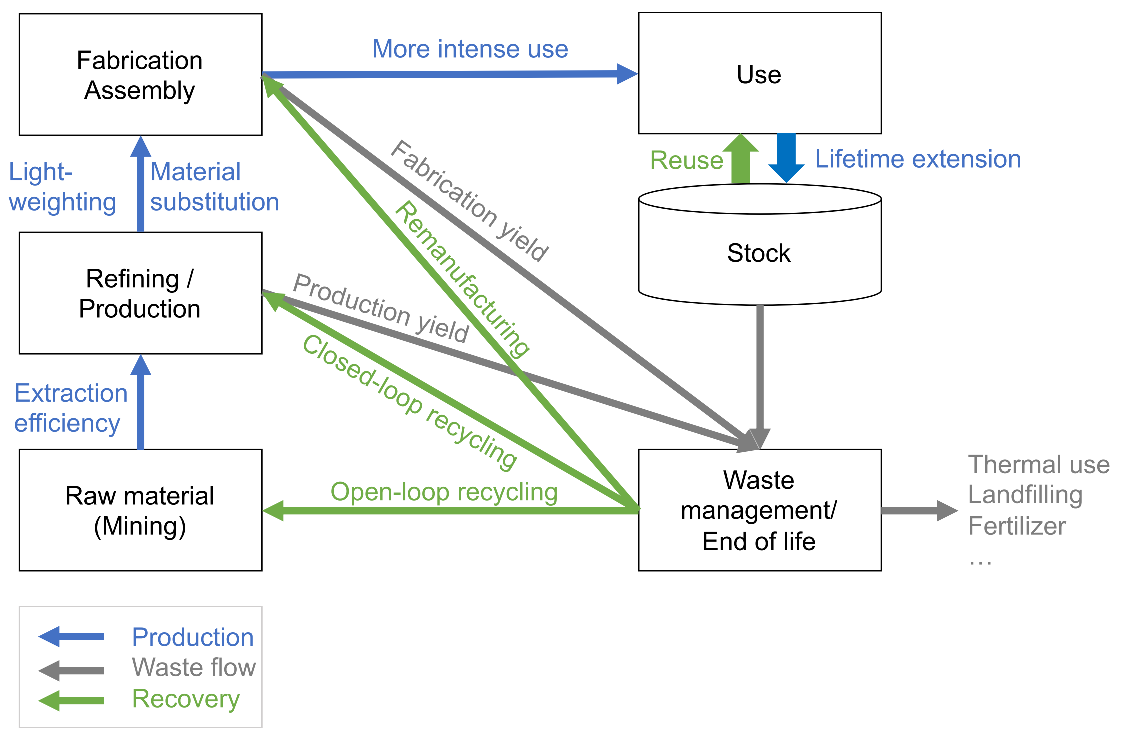


**Figure 2.** Material cycles and the identification of material efficiency strategies.

3. Light-weight design [20] and materials choice [21]: less material and/or lower GHG emissions in the production of a product.
4. Reuse of components [22], including through remanufacturing [18] and modularity [23].
5. Recycling, upcycling [24], cascading [25].
6. Improved yield in production, fabrication, waste processing [26].

To evaluate whether a strategy provides a way to deliver a similar or the same service with reduced GHG emissions, one needs to compare the life cycle GHG emissions of service provision with and without the strategy implemented. These comparisons can be based on modeling and then rely on a set of assumptions, or based on actual implementation, where technological performance and behavioral response are considered simultaneously.

Figure 2 illustrates the life cycle of materials products and indicates where different resource efficiency strategies apply.

## GHG emissions of material production and use

In 2015, cradle-to-gate GHG emissions from the production of materials was 11.4 Gt of $CO_2$-equivalent (figure 3). Direct emissions from material producing sectors constituted more than half of the cradle-to-gate emissions, energy production contributed 35%, mining 3%, and other economic sectors 8% (figure 3(a)), according to an analysis of EXIOBASE [27]. Iron and steel contributed most to the total cradle-to-gate impact of materials in society, with 32%. Aluminum contributed 5%, other metals summed to 4%. Rubber and plastics contributed 13%, cement, lime, and plaster 26%, other non-metallic minerals 13%, and paper and wood products 8% when ignoring land-use related emissions (figure 3(b)).

The most important uses of materials in terms of embodied GHG emissions are those of cement, lime and plaster in the construction sector (2.9 Gt $CO_2$eq), and of steel in the manufacturing sector (2.8 Gt $CO_2$eq). Materials contribute 50% or more to the carbon footprint of buildings and infrastructure, machinery, vehicles, and other transport equipment. In terms of the industries using material, 40% of emissions related to material production were for materials used in construction, 18% machinery and equipment, 8% transport equipment, and 3% electronics (figure 3(c)).

The IEA foresees that by 2060, the economy will add another 220 billion square meters of building floor area and another billion of light-duty vehicles, doubling current numbers [1]. Growth of EEE is even more rapid, with interconnected devices projected to grow from 8.4 billion in 2017 to 20 billion in 2020 [28]. The Organisation for Economic Co-operation and Development [29] and the International Resource Panel [30] foresee a doubling of global material use from 2015–2060.

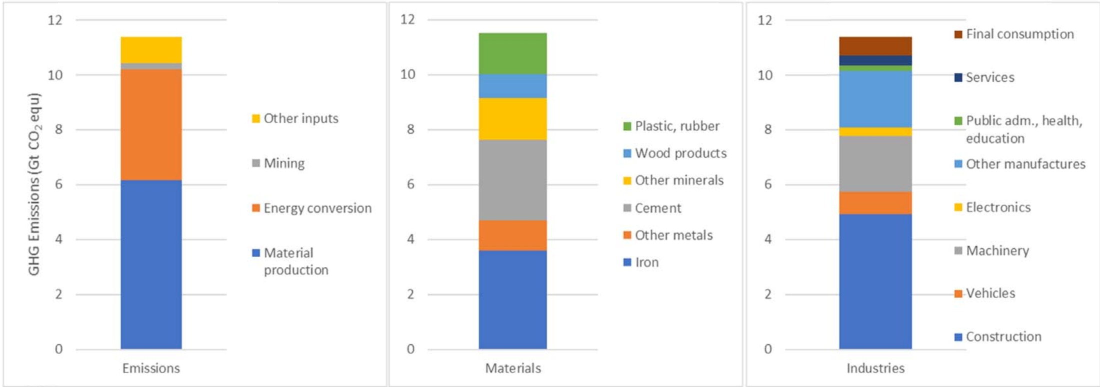


**Figure 3.** (a) Source of GHG emissions, i.e. material production itself (scope 1), energy inputs (scope 2), mining or other purchases (scope 3). (b) Cradle-to-gate greenhouse gas emissions from the production of key materials in 2015, identified by material. (c) Material-related GHG emissions by industries using materials [27].

## ME in buildings

In 2010, about 30 Gt of nonmetallic minerals were extracted globally, of which over 95% are construction minerals [31–33]. Modern construction is dominated by the use of concrete, constituted of nonmetallic minerals cement, aggregate, and sand [31, 33] mixed with water [34]. As for other construction materials like wood, bricks, glass, and tiles, their availability is of increasing concern in some regions and longer-distance transport of construction materials will be necessary in the future to satisfy increasing demand, also when accounting for secondary materials [35–37]. For structural purposes concrete and steel are used together as reinforced concrete. Steel is also used as beams and other structural elements, and as cladding. Estimates from the EU, Japan, and Vietnam indicate that about half (+/−50%) of construction minerals end up in buildings and the rest in civil infrastructure like roads, ports, and dams [32, 38–40]. In the US in 2016, 31% of cement was used for highways and streets, 27% for residential buildings, and 15% for commercial buildings [41]. Of the 1 Gt of steel produced annually, over 40% is for buildings and about 15% for infrastructure. According to EXIOBASE, production of materials (figure 3) accounted for 56% of the carbon footprint of the construction sector, or 3.3 Gt $CO_2$ [42].

Buildings and infrastructure have lifespans of decades to centuries and require ongoing materials and energy for their operation and maintenance. These long lifespans *may* lead to lock-ins of specific use patterns which no longer meet current needs or reflect the current state of energy efficiency [43–45].

Future building materials demand and related emissions can be reduced through more intensive use of buildings (reducing per capita floor area), building lifetime extension, the use of lighter constructions and less carbon-intensive building materials (e.g. wood-based construction instead of steel and cement), reduction of construction waste (e.g. through prefabrication) [46, 47], the reuse of structural elements, and the recycling of building materials [13]. The potential of various strategies depends on a region's stage of development and its local building material resources, as well as its existing building stock, with measures targeting new buildings being more important in developing countries and measures related to lifetime extensions, reuse and recycling being more pertinent to countries with a large existing stock.

### More intensive use

Per capita floor area trends upwards with time and increasing GDP [48], but average floor area range from 30–70 $m^2$ per person in countries with a GDP of \$50 000 per capita and year, indicating that different conditions and policies result in very different material requirements [49]. Although urban dwellers have less floor space per person than rural dwellers [50, 51], the ongoing transition of humanity to cities is not enough to counterbalance the overall trend of increasing per capita floor area. Scenarios of future residential buildings often assume that buildings will become more spacious [44, 52–55] which is detrimental to ME. In Switzerland, a continued growth of floor area by 20% until 2050 would lead to an increase in cumulated material-related GHG emissions of 8% compared to a baseline scenario [44]. Swilling *et al* [56] anticipate an increase in global urban land area by a factor of three between 2010–2050 to accommodate housing for 2.4 billion more people, following a trend of decreasing urban densities [57].

Therefore, bucking the trend of increasing floor area through better designed and furnished residences with less residential space per capita has a large potential to reduce emissions. A 'more intense use' scenario for future residential buildings in Norway shows that the climate impacts of buildings could be reduced by 50% compared to baseline as a result of reduced material demand and reduced energy demand to heat a smaller area [58]. Milford *et al* [16] identify more intensive use as the most effective ME strategy for steel. Grübler *et al* [59] also assume floor space limits in a 1.5 degree scenario focusing on consumption-oriented solutions rather than relying on negative emissions. While most scenarios assume that more intensive use just implies smaller residences, other options include larger household sizes, fewer second homes, dual-use spaces, and shared or multi-purpose office spaces.

**Lifetime extension**

In the US, the average lifetime of residential buildings is 50–60 years [43, 48, 60], in Europe it exceeds 100 years [61–63], while recent historical building lifetimes have been 30–40 years in Japan [64, 65] and just 25 years in China [66–68]. While short historical building lifetimes in emerging Asian countries can be explained by the inadequacy and inflexibility of buildings built during rapid early urbanization and industrialization, the question arises whether and how the rapid obsolescence of currently constructed buildings can be avoided and how new buildings can be more flexibly designed and easily modified to meet evolving demands.

Numerous studies explored the potential reductions in resource demands by extending building lifespans [52, 62, 67, 69, 70], which directly reduce upstream energy demands. Cai *et al* [66] estimated that extending Chinese building lifespans to 50 years could dramatically reduce $CO_2$ emissions by over 400 Mt per year (one fifth of current construction-related emissions) and save 3 EJ of energy per year.

**Lightweight design and material choice**

The GHG emissions of new buildings can be reduced either through using less materials, such as lighter structures, or using less carbon intensive materials, such as replacing steel and concrete with wood where such solutions are appropriate.

Carruth *et al* [20] analyzed the material use associated with different load-bearing structures and found that a variable cross-section steel beam could save one third of the material compared to a universal standard beam, while a truss-structure could offer additional savings at cost of needing more volume. Moynihan and Allwood [71] investigated the design of 23 steel-structured buildings and found that for over 100 00 beams, on average less than half of the load-bearing capacity was being utilized, indicating a substantial scope of savings in steel due to closer specifications and different load-bearing elements. Milford *et al* [16] conservatively assume a reduction of the mass of steel to provide the same function by 19% in their global ME scenarios for future steel demand.

The climate benefit of using wood over steel and concrete in construction is well established [21, 72–77], even considering trade-offs in energy storage in the building shell (figure 4) [78]. Cross-laminated timber can even be used in tall structures [79, 80]. The benefit is a result of two effects: first, the storage of carbon in wooden biomass in buildings, which delays its oxidation [81]. The storage benefit increases with the storage period and with forest regrowth speed [72]. Second, displaced materials such as cement and steel have high emissions during production [76]. The quantification of this effect needs to carefully consider system boundary choices (inclusion of waste mgt. stage or not, use phase (thermal insulation) included or not) and overcome a lack of transparency of many studies. Petersen *et al* [82] compiled literature findings for Norway and Sweden and report that avoided emissions from using timber typically lie between 100 and 400 kg $CO_2$-eq/$m^3$ timber, although the entire range spans minus 310 to plus 1060 kg $CO_2$-eq/$m^3$. Kayo *et al* [83] estimate that increasing wood construction in Japan could lead to a net GHG emission reduction of 1.23 $tCO_2$-eq/$m^3$ Sathre and O'Connor [84] compiled displacement factors of wood product substitution, measured in tons of C emissions reduced per additional ton of C used in construction, for 21 case studies. They find positive replacement factors in most but not all cases. Oliver *et al* [76] find it feasible to replace 10% of construction materials, resulting in substantial $CO_2$ emission reductions. The potential for additional wood harvests is, however, controversial, given already unsustainably high harvest rates in some regions. Three quarters of the world's forests are currently used for timber production, yielding 2 Gt dry matter [77], of which 1/4th is currently used as construction material. Given the limited availability of timber, it is hence important to focus on structures where carbon benefits are largest.

**Reuse**

The reuse of energy-intensive building components could result in substantial savings of energy [85, 86]. Most investigations have focused on the reuse of metal elements. Ideas for reusing concrete panels from the walls of pre-fabricated buildings have been proposed [87], but potentials and issues are not yet well understood. A case study of reusing steel components, Pongigilione and Calderini [88] describe the construction of a railway station in Genoa, where the reuse of steel components was an explicit design objective. 30% of the steel in the new station came in the form of components from the demolished station. Reuse saves 0.36 kg $CO_2$/kg of steel compared to recycling given the energy requirements of remelting in an electric arc furnace, which is much less than replacing virgin steel (1.78 kg $CO_2$/kg) but still appreciable [89]. In the UK, 8%–11% of steel from demolition is reused, with a downward trend [89, 90]. Cooper and Allwood estimate a total reuse potential of 27% for metal products, with structural steel and cladding from buildings being the largest two sources. By contrast, concrete reinforcement bars have a low potential for

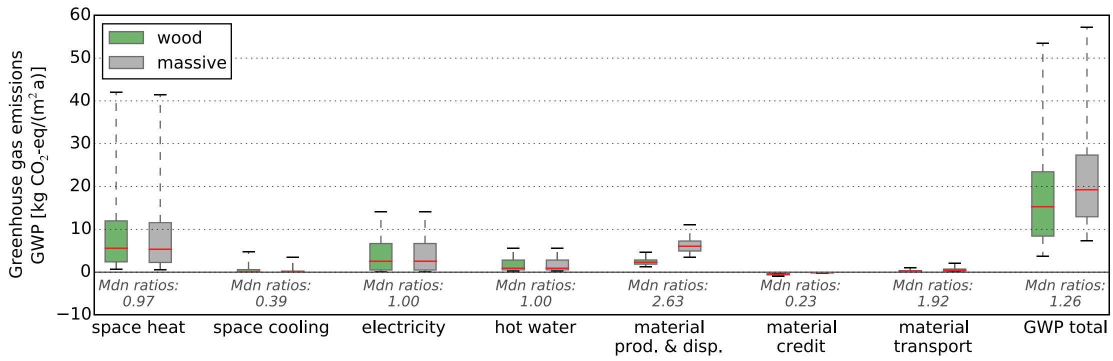


**Figure 4.** GHG emissions associated with wood and massive residential buildings under Swiss conditions show a median life cycle benefit of 25% for wood constructions, exclusively due to material production, with increased energy demand for heating and cooling the building due to loss of thermal mass [78].

reuse [86], but the use of modular constructions opens new opportunities [91]. Important barriers are the (perceived) availability of correctly specified components to be reused, issues associated with quality assurance and risk, and (perceived) costs [89, 92, 93]. Proposals to overcome these barriers have been made. Ness *et al* [94] suggest the use of radio frequency tagging of components and the use of building information modeling to track components and assemblies and import them into building design software at the design stage. Dunant *et al* [95] suggest the introduction of new market actors that would identify, quality-control, stock, and market disused components.

## Recycling

Construction and demolition wastes constitute about a third of all solid waste in Europe, and twice as much as municipal solid waste in the United States [96]. It is common practice to recycle metal elements. The recycling of metals has higher environmental benefit when measured in terms of GHG emissions avoided than the recycling of other materials [97]. For wood as construction material, energy recovery brings significant benefits [73]. Concrete and other mineral building materials are most often downcycled to coarse aggregates. Investigating a case of aggregate production near Rome, Simion *et al* [98] indicate that secondary materials have only 40% of the impact of aggregates from natural resources, but not all uses result in such environmental benefits [99]. Some studies indicate that when using low-grade recycled aggregates in concrete production, more cement is required to obtain the same quality of concrete [100, 101]. The environmental benefit of recycling of minerals depends in part on the comparative transportation distances of virgin and secondary resources [102, 103]. For fine particle size construction and demolition waste, recycling is technologically more challenging. Methods to recycle hydrated cement waste into new cement have been developed [104]. An assessment suggests a reduction of $CO_2$ emissions by up to 30% [105]. Some promote the recovery of unhydrated cement from concrete [12]. Technologies to recycle all components of cement are under development and unreviewed life cycle assessments suggest substantial reductions in GHG emissions [106], which have yet to be verified.

Similar issues with the quality of the secondary feedstock exist also for metals, but their impact is less severe. Haupt *et al* [107] estimated that 'sweetening' low quality steel scrap requires about 1.4 times more energy than high quality steel scrap. For aluminum, the energy penalty was estimated up to 20% [108]. Issues of alloy-specific recycling are further discussed in the section on vehicle recycling.

There is a renewed interest in the enhanced carbonation of concrete, a process by which $CO_2$ is absorbed from the atmosphere [73, 102, 109]. In an investigation focused on the US, it was estimated that the enhanced $CO_2$ absorption from crushing concrete waste could offset 2%–3% of the emissions of the construction sector [109]. However, enhanced weathering results in the increased release of toxic compounds, so precautions have to be undertaken [102].

The 'lost stock' of construction materials mostly in sub-surface layers, including foundations, and the 'hibernating stock' in delipidated and abandoned construction provide additional potential for reuse and recycling of building material, but the limited value of the materials may constitute a major barrier [64, 110, 111].

## Trade-offs between material and energy efficiency

A heat recovery ventilation system, extra window panes, a ground-source heat pump, and insulation all increase building energy efficiency, but also influence the materials footprint of a building (table 1). Chastas [112] harmonized 90 building case studies and found that the embodied emissions increase with the energy efficiency of a building while the total life cycle emissions decrease, echoing earlier findings [113, 114]. Koezjakov *et al* [115] performed a prospective assessment of the Dutch residential building stock

**Table 1.** Trade-off between material use and energy use of selected material and energy efficiency strategies for buildings.

| | | Material Use and related GHGs | | |
|---|---|---|---|---|
| | | Decreasing | Neutral | Increasing |
| *Operational or Construction Energy Use* | Increasing | Reuse and recycling of cement and aggregates<br>Lifetime extension<br>Wood structures | Higher indoor temperature | Larger units |
| | Neutral | Recycling of steel<br>Reuse<br>Light-weighting | | High-rise buildings |
| | Decreasing | More intensive use, smaller units | Lower inner temperature (heated buildings) or higher temperature (cooled buildings), reduction of heated/cooled area | Building stock renewal<br>Heat exchange ventilation systems<br>Extra insulation<br>Passive solar design and heat storage |

and anticipate that as energy efficiency improves and energy supply decarbonizes, construction-related emissions will become dominant by the year 2050. There are, however, few papers that investigate the energy costs of ME. Heeren *et al* [78] show that there is a slightly higher energy consumption in the shoulder season related to the loss of thermal mass when using wood instead of concrete or stone masonry buildings. Grant and Ries [116] show that longer building lifetimes increase operational energy use when older buildings are designed to poorer standards. Individual case studies indicate that refurbishments can have lower life cycle impacts than replacements if and only if refurbished to ambitious energy standards [58, 117]. This section indicates that some ME strategies such as more intensive use and light-weighting reduce material use and related emissions without increasing energy consumption, while other strategies such as lifetime extension or the use of wood instead of massive and steel structures may face trade-offs that require more systematic evaluation (table 1).

## ME in vehicles

Similar to buildings, road transport is characterized by substantial direct $CO_2$ emission of 5.5 Gt in 2012 [118], while the production of gasoline caused 0.6 Gt [119]. The materials delivered directly to motor vehicles and other transport equipment manufacturing caused emissions equal to 0.7 Gt $CO_2$ (see figure 3: 440 Mt for iron and steel, 200 Mt for rubber and plastics, 50 Mt for aluminum, 20 Mt for glass). Materials constituted 55% of the carbon footprint of vehicle and transport equipment manufacturing of 1.6 Gt [27, 42, 119]. For battery electric vehicles, which are considered important mitigation technologies within the transport sector [1], studies have found that battery production is an energy-consuming process that offsets some of the efficiency gains of electric motors over internal combustion engines [120]. Similarly, the production and operation of information and communication technology (ICT) systems may substantially offset the benefits from automated driving, platooning and other energy-saving operations that are enabled by these ICT systems [121, 122]. Car ownership is often seen as a hallmark of the middle class [123] and has been rising quickly in emerging economies. As larger populations join the middle class, car ownership is forecasted to increase, adding another billion of vehicles by 2060 [1].

The future materials demand for vehicle manufacturing depends on future transport demand, the number of vehicles required to satisfy a given transport demand, the mass of material per vehicle, and the emissions intensity of those materials. Demand for materials can be reduced through measures that reduce transport demand, car ownership, and vehicle mass, which is also a function of vehicle size. Apart from affluence, access to public transport, car and ride sharing opportunities, urban design, and costs of car ownership including parking influence the rate of car ownership, while culture, urban lay-out and costs influence car size.

Emissions associated with vehicle manufacturing are also influenced by material choice, where there is often a trade-off, with lighter materials desired to reduce fuel consumption often being more energy-intensive to produce [124]. Further, the increasing penetration of electric vehicles increases the importance of decarbonizing the electricity supply [120, 125]. Understanding life cycle impacts is of critical importance, given that electric vehicle shares of up to 90% of the global passenger vehicle fleet are foreseen in many climate-mitigation scenarios [1].

Fuel combustion is often assumed to cause 80%–88% of the life cycle emissions of internal combustion engine vehicles [124], resulting in a predominant focus on improving on-board energy efficiency over other improvements. In reality, direct emissions of vehicles account only for two thirds of road transport related emissions in the US, the rest are mainly associated with fuel production, vehicle manufacturing and maintenance, and construction, operation and maintenance of road infrastructure [126]. Trade-offs between operational and upstream emissions arise even under current conditions, and their importance increases with increasing energy efficiency and electrification. In a scenario of high electric vehicle and renewable electricity penetration in Australia, upstream GHG emissions exceed direct tailpipe GHG emissions of the passenger vehicle fleet already before 2040 [127].

### Vehicle fleet size, more intensive use, and the potential impact of self-driving vehicles

Personal vehicles, while important symbols of affluence and convenience, are utilized on average only 5% of the time and for 1/3 of their capacity [128–130], indicating that there is a significant potential to reduce the amount of materials tied up in a largely stationary vehicle stock. The average utilization rate of vehicles decreases further with vehicle age [131]. Measures that shift transport demand away from privately owned vehicles have the potential to reduce emissions. In regions with a higher population density, public transport, biking and walking provide convenient alternatives that reduce GHG emissions, but this is not always the case in areas with lower population density [132]. Car-pooling has long been a focus of efforts to reduce congestion and air pollution; in recent years, car-sharing and ride-sharing have emerged as alternatives that may increase the rate of vehicle utilization [133–135]. Through trip-chaining, autonomous taxis (ATs) could radically reduce the number of vehicles required, potentially at the cost of increased vehicle turn-over and longer distances. Other environmental effects arise from the easier electrification of the fleet, the higher initial energy and material requirements of ATs, the issue of empty trips, and benefits through eco-driving and platooning [121, 122, 136]. The impact of ATs, carsharing and ride-hailing on overall travel demand seems to be inconclusive and may depend on many local factors. On the one hand, these options can support multi-modal traffic in urban areas and thereby reduce the number of vehicle-km [137, 138]. On the other hand, they may favor urban sprawl and compete with public transport, leading to increased travel demand [139–141]. Currently, the main barrier to a large-scale adoption of autonomous vehicles is the high costs, which are expected to reduce significantly [142]. Given the increasing importance of materials for electric and autonomous vehicles, a scenario-based life cycle assessment of this trade-off will likely underline the importance of recycling for attaining emissions reductions from more intensive use.

### Lifetime extension

With vehicle utilization rates of 5%, the effect of lifetime extension is ambiguous as reduced material and energy requirements for manufacturing new vehicles is offset by performance differentials between new and used vehicles if fuel efficiency increases, although estimates of this increase range between 1.8–3% per year [14, 143]. Use scenarios can be constructed which lead to modest emission reductions both for lifetime extension [144] and early retirement [145–147]. As fuel efficiencies plateau and vehicle manufacturing comprises a larger share of life cycle emissions, the benefit of lifetime extension will rise.

### Light-weighting and right-sizing

Different factors have affected vehicle mass in the past. On the one hand, the desire to decrease fuel consumption has prompted a shift to light-weight designs and materials, which has been facilitated by steady improvements through computer-aided design and in material properties [124]. On the other hand, the collision-advantage of relatively larger vehicles and the introduction of more ancillary, computing, and safety components, such as airbags, anti-intrusion bars, air conditioning, electric windows, entertainment units, and electronics have increased vehicle mass [124]. Shifting the vehicle fleet to smaller cars would reduce fuel consumption and material requirements at the same time. One option to attain such goals is car sharing, which may give participants access to trip-appropriate car sizes [148]. For AT, such a right-sizing effect of deploying vehicle sizes to match occupancy requirements of each trip has also been hypothesized [149].

Light-weighting is often but not always [150] based on shifting the composition of vehicles from steel to lighter materials such as fiber composites, aluminum, and magnesium, which require more energy in their production. Reduction of component mass allows design changes such as the reduction of structural material and engine size, which result in further savings [150–152]. For gasoline-driven vehicles, this type of light-weighting results in a reduction of life cycle emissions due to the reduction in operational energy use and despite the increased energy requirement for material production [151–153]. In a scenario to 2050, developed by Modaresi *et al* [153], steel-intensive light-weighting can reduce mass by 11% compared to business-as-usual, reducing life cycle emissions by 5%, while an aluminum-extreme scenario reduces mass by 26% and results in life cycle emission reductions of 8%. Through alloy-specific recycling of the aluminum components, the additional energy use for producing aluminum components can be more than offset [154].

Additive manufacturing (AM) of vehicle components may offer additional resource-saving benefits in select applications. AM can produce optimized lighter weight part geometries not achievable using conventional manufacturing methods—thereby delivering greater vehicle fuel economies [155]. It is for these reasons that some aircraft manufacturers have already begun adopting AM parts in non-critical applications to save both operating fuel and raw materials costs [156], as manufacturing yields from liquid aluminum to machined aircraft component can be below 10% [13]. If deployed in technically feasible aircraft applications, AM may have the potential to reduce the fuel use of the US aircraft fleet by around 6% by 2050, with raw material reductions as high as 85% for feasible titanium, nickel, aluminum, and steel aircraft components [157]. With current technology, economic use of AM components is limited to those with complex geometries, low production quantities, expensive raw materials, and significant redesign optimization potential, combinations of which may be limited in the transport sector. AM processes currently show high production costs, low throughput rates, surface roughness, and part fatigue life limitations [157–159], factors that limit their near-term application. The extent and pace of AM market uptake will depend on continued technical progress to improve its competitiveness compared to conventional methods, and its overall benefits must be established from a life cycle perspective.

### Remanufacturing and reuse

It has always been common practice to reuse car parts, sometimes requiring repair, refurbishment or remanufacturing [160]. According to Liu *et al* [161], remanufacturing a diesel engine can save 69% of embodied GHG emissions compared to producing a new diesel engine. Similarly, Sutherland *et al* [162] find a 90% energy use reduction for remanufacturing a diesel engine, supported by findings in other countries [163]. Remanufacturing of components such as tires can result in energy savings on the order of 80% compared to new parts, but the question arises whether the performance of a remanufactured product is on par with a new product. Remanufacturing can often restore performance to like-new [160], reversing performance loss through aging, but it is not always equal to a newly manufactured part which will have benefitted from technological progress [14]. For an energy-using product, one needs to weigh operational and manufacturing energy use to find the optimal replacement strategy [145, 146].

### Recycling

End-of-life vehicles are commonly recycled, which results in the recovery or thermal utilization of 85% of materials [164–167]. Scrap metals often undergo downcycling because vehicles are complex products that contain many alloys and metals, resulting in the mixing of incompatible elements [168, 169]. For example, the assortment of high-quality steel in a car becomes construction steel. In the process, the functionality of alloying elements is lost. Such downcycling constitutes itself an energy loss: pig iron production causes emissions of 1.5 kg $CO_2$ equivalent per kg iron, while alloying elements range from similar (1.9 kg $CO_2$/kg metal for ferrochromium) to much higher (11 kg $CO_2$/kg nickel from sulfide ores) [170], so that the emissions associated with highly alloyed steel can be significantly higher than those of construction steel. Further, alloying elements and other metals mixed in as part of the shredding process become contaminants that compromise the quality of the material in question even for bottom applications, potentially leading to a future where secondary material needs to be discarded [154, 171]. Copper and tin contamination limits the usefulness of secondary steel and scenarios foresee a possible saturation of the steel stock with copper within material tolerances, impeding further recycling [171]. Similarly, secondary aluminum will need to be discarded unless alloy-specific recycling is introduced, in particular when internal combustion engine blocks, which currently absorb much of the low-quality supply, are no longer needed [154]. A national level material flow analysis of alloying elements in steel for Japan indicates that a better dismantling and sorting of iron and steel products provides a route to preserve the function of alloying elements even over a 100-year time scale [172]. Focusing on the recycling of Japanese cars, Ohno *et al* [173] show that dismantling and sorting can reduce the need for adding alloying elements to electric arc furnaces by 10% and as a result reduce the GHG emissions of the alloying elements required in the recycling process by up to 28%.

Only a fraction of the increasing amount of electronics is recycled as electronic parts are distributed throughout the car, as these parts are not easily collected [121, 174]. Plastic, fabrics and other materials usually end up in automotive shredder fluff which is landfilled or combusted. Combustion, favored by an international expert panel [166] delivers energy that can replace fossil fuels but emits more carbon than deriving the same energy from natural gas [22]. In a zero-emissions scenario, such a strategy is only acceptable in a facility with energy valorization and/or $CO_2$ capture or if plastics are made from renewable sources, all of which are feasible in the medium term. Seventeen of 25 identified specialty metals used in vehicles for their particular properties are currently not functionally recycled [175, 176].

With the expected electrification of fleets, the demand for lithium (Li) batteries [177] and charging infrastructure [178] will increase material-related energy requirements. A Li battery can contribute 31% of cradle-to-gate GHG emissions of a medium BEV [127], while the charging infrastructure may account for ca. 10% of life cycle energy use [178]. Two major strategies to reducing GHG emissions from Li battery manufacturing have been identified: (1) reducing the energy use and/or using renewable energy during cell manufacture, and (2) battery recycling/use of recycled metals during battery production [179, 180].

### Trade-offs between material and energy efficiency

Strategies such as product down-sizing and more intensive use often achieve synergies between material and energy efficiency (table 2). Other strategies, such as light-weighting, lifetime extension, and electrification have trade-offs, which indicates that wider system boundaries need to be considered and the savings may not be as great as anticipated. We also see that there can be interactions between strategies; e.g. with optimal recycling strategies enhancing the attractiveness of light-weighing through a shift to more energy-intensive specialty materials. The effect of different strategies may depend on both geographical factors and policy design. An integration with public transport may be required for ride-sharing and AT to lead to a decrease in vehicle travel and congestion. Overall, strategies of ME for vehicles can contribute to a substantial reduction of emissions in vehicle production. Synergies and trade-offs with energy efficiency are notable (table 2) and should be considered in the selection of strategies and the design of policies.

## ME in EEE

Due to rapid technological development in the consumer electronics industry, there has been a significant attention to the obsolescence of EEE. Estimates of total volumes of waste electrical and electronic equipment (WEEE) range from 20–70 million tons per year [181, 182]. Household appliances constitute about half of the mass, consumer equipment around 20%, and ICT equipment around 15% [181]. Research and

**Table 2.** Trade-off between material use and energy use of selected material and energy efficiency strategies for vehicles.

| | | Material-related GHG emissions | | |
|---|---|---|---|---|
| | | Decreasing | Neutral | Increasing |
| *Operational energy use* | Increasing | Lifetime extension<br>Remanufacturing | | Larger vehicles |
| | Neutral | More intensive use (ride sharing, car-pooling)<br>Recycling (esp. closed loop) | | |
| | Decreasing | Down-sizing (smaller vehicles)<br>Additive manufacturing<br>Light-weighting | Improved engine control<br>Driving style | Electrification of vehicles<br>Driving assistants and autonomous vehicles |

policy making efforts have focused on consumer electronics and ICT, for two primary reasons: the environmental burden associated with WEEE management and the economic loss from incomplete recovery of materials within these devices [183]. Lead in solder and some flame retardants used in plastics can cause environmental contamination and detrimental effects to human health. Materials contained in EEE usually include base metals, such as aluminum and copper; precious metals, such as silver and gold; critical raw materials, such as rare earths, gallium, indium; and plastics [184], most of which are very valuable [185–188]. Concentrations of gold and silver within printed circuit boards can reach ten times those seen in their respective ores [189]. However, a significant portion of these materials are not recovered. In the European Union, 3.3 million tons of WEEE were collected in 2012, while over 6 million tons were not accounted for [190].

Based on the results of several studies, the embodied or upstream GHG-impact of EEE (i.e. outside of the use phase) includes impact from high volume constituents such as steel and aluminum for industrial equipment and appliances to higher value constituents such as integrated circuits and other active components for electronic devices, such as ICT [191–193]. For these higher value constituents, the impact is, therefore, not just around the extraction and processing of materials, such as silicon, but also the emissions intensive processes of manufacturing these devices. For EEE, ME strategies include reuse, remanufacturing, recycling to recover valuable materials, and functional integration potentially leading to consumption reduction. In general, resulting benefits of ME strategies depend on study assumptions around the volume of devices recovered, the fate of the recovered materials or components, and the resulting rebound implications. Few studies have demonstrated that strategies to reduce EEE resource consumption lead to a reduction in life cycle GHG-emissions.

**More intensive use**

Given the rapid expansion of the ownership of EEE, little attention has been paid to sharing or other more intensive use strategies. It has, however, been observed that the integration of functions into smart-phones and other multi-use devices can contribute to reducing the number of devices owned by an individual and thus reduce the material and energy demand caused by the production (and operation) of EEE [17, 194]. Given these recent trends towards smaller, more integrated products, there has been a shift in the demand for material classes from reduced use of bulk material quantities, but increased quantities of active components such as integrated circuits.

**Lifetime extension**

Whether lifetime extension leads to a net GHG benefit depends on whether their resale offsets new product acquisition. For the case of reuse of small consumer products, components may be downcycled (cascaded use of mobile phone chips, for example) while whole products may be reused if cycled to a less affluent user. These reuse options tend to mean that the product would be relocated to another geographic market. The labor-intensive processes associated with enabling lifetime extension mean that this ME strategy is typically restricted to the refurbishment of high-value subassemblies, such as mobile phones [195], photocopier modules [196, 197], and industrial equipment components. Cooper and colleagues found evidence that remanufacturing of industrial equipment could lead to a lifespan doubling [19, 86].

Estimates for remanufacturing savings of EEE range from 50%–80% when the use phase is excluded [19]. Gutowski *et al* [198] argue when use phase is included the claimed energy savings for remanufacturing might be dampened based on increases in energy efficiency of new items, whereas King *et al* [196] identify both socio-economic and environmental benefits for remanufacturing over other waste reduction strategies. Quariguasi-Frota-Neto and Bloemhof [199] explore remanufacturing of personal computers and mobile phones. They argue remanufacturing reduces the total energy used during the life cycle of personal computers and mobile phones, except when the second life span of the product is substantially shorter than the first lifespan. Truttmann and Rechberger [200] compare two scenarios of normal product life and an intensive extended product life by reuse with the latter reducing total resource consumption (materials and energy) of a highly developed industrial economy by less than 1%. Geyer and Blass [201] investigated mobile phone reuse and recycling from an economic point of view concluding that reuse is the largest driver of end-of-use handset collection and recycling is a by-product. Further examples of repurposing, or adaptive reuse, include using liquid crystal display and screens as televisions, notebook computers as thin clients, Advanced Technology eXtended (ATX) power supplies for battery charging applications, and smart phones in parking meters [202].

The main barriers to reuse are costs (due to scarcity of parts and labor), technology obsolescence, consumer perception, lack of reverse supply chain infrastructure, as well as data privacy and security issues [203]. Although data privacy concerns have been observed primarily for ICT, this issue may become more and more relevant with the growing relevance of the 'internet of things'. We underscore that based on a few limited studies it appears unlikely that, without specific regulatory attention, an increase in the reuse of products will translate to an equal decrease in the sale of new products. Recently, Makov and Vivanco [204] estimated that one third, and potentially the entirety, of emission savings resulting from smartphone reuse could be lost based in part on this imperfect substitution [204].

In addition, products stored unused (i.e. 'hibernating stock') influence the total time a product remains with the consumer. For instance, Thièbaud *et al* [205] found that the hibernating stock accounted for about 25% in mass of the total in-use stock of electronic devices in Switzerland in 2014. The same authors estimated that hibernation extends the apparent lifetime of mobile phones and smartphones from 3 to 7 years, and for desktops and laptops from 5 to 8 and 9 years respectively. However, even though this hibernating stock delays recycling and waste treatment, it does not reduce the demand for new products.

### Recycling

In the case of recycling, the fate of the recovered materials will influence whether the GHG savings are borne to the EEE sector itself. Rapid advance of technologies and increasing product complexity may discourage closed-loop recycling, as the secondary material may not fit into the new generation of products. In addition, the composition of electronic products evolves rapidly, so complete compositional characterization of these products is challenging. This lack of information hinders recycling. Therefore, in most cases recovered material replaces primary inputs to anther sector. Quantified GHG benefit from recycling ranges from 1% to 10% of life cycle emissions. However, recycling is often motivated by preserving access to functionally important metals and preventing toxic emissions from waste incineration and landfills [181].

Rapid technical improvements shorten the lifetime of electronics, but they also increase energy efficiency and reduce material use through miniaturization. This tends to involve the components themselves, rather than whole products, but the reduced materials use in some cases has been 50% [206]. Within EEE, while studies do generally find some GHG-benefit for ME strategies, reductions in other environmental impacts tend to be higher.

Overall, we find that there is minimal scope to reduce GHG emissions through additional ME strategies applied to EEE, given that lifetime extension may increase operational energy use, secondary markets may fail to off-set new purchases, and recycling beyond existing levels yields only modest reductions of GHGs.

## The state of evidence

### Evaluation of ME strategies

The literature indicates a significant potential for individual ME strategies to provide shelter and automotive transport with less materials and lower overall GHG emissions. The evidence regarding potential emission reductions from ME in EEE is limited.

Table 3 provides an overview of the potential, synergies and trade-offs, barriers and drivers for different strategies, as identified in the literature. The level of support for claimed reductions is evaluated according to the amount of evidence available and the unanimity of support, following the scoring used by the Intergovernmental Panel on Climate Change.

There is a limited to intermediate level of support in the literature for the potential of an intensified use of buildings and vehicles and its ability to reduce the demand for materials and associated emissions (table 3). The number of studies identified is not very high, but there is agreement across studies and a strong logic supporting this strategy. There is a potential co-benefit of reduced operational energy use, especially for buildings, and savings concern primarily new products and are available immediately. Empirical studies of realized cases and programs could substantially strengthen the evidence base.

For light-weighting of buildings, there is also limited evidence but a strong agreement about a substantial potential for emission reductions in the construction phase with few trade-offs. There is a stronger level of support for significant operational energy use reductions from the light-weighting of cars through material substitution, which results in increased material-related GHG emissions. There is medium evidence and strong agreement that a downsizing of vehicles could achieve significant material- and energy-related emission reductions.

**Table 3.** State of evidence for the contribution of material efficiency to total climate change mitigation. ↓ indicates a reduction, ↑ an increase, and—a neutral effect. ◊ denotes a barrier and → a driver.

| Product | Strategy | Material-related GHG savings potential | Operational energy use | Net GHG effect[a] | Level of support[b] | Barriers ◊ Drivers → |
|---|---|---|---|---|---|---|
| Buildings | More intensive use | ↓40% [16] | ↓ | ↓ | LM | ◊ ↑GDP, ↓family size |
| | | | | | | → urbanization, ↑prices |
| | Lifetime extension | ↓47% [16] | –↑ | ↓– | LM | ◊ ↑GDP |
| | | ↓40% [66] | | | | →aging |
| | Light-weight design | ↓19 [16]–50% [71] | –↑ | ↓– | MH | ◊ Conventions, labor costs |
| | | | | | | → materials price |
| | Reuse | | –↑ | ↓– | LM | ◊ Logistics, labor cost |
| | Metals | ↓15% [16] | | | | |
| | Minerals | ↓0%–5% | | | | → materials price |
| | Remanufacturing | | — | ↓– | LM | |
| | Recycling | ↓10%–20% | | | | → Materials price (for metals) |
| | Metals | above baseline | — | ↓ | RH | |
| | Minerals | ↓0%–20% | ↑– | ↕– | LM | ◊ transport cost, low value (for minerals) |
| Light duty vehicles | More intensive use | ↓39% steel fleet [16] | – | ↓ | MM | ◊ ↑GDP, ↓family size |
| | | ↓93%–96% vehicle [149] | | | | → urbanization, technology development |
| | Lifetime extension | ↓13% steel fleet [16] | –↑ | ↓– | LM | ◊ ↑Model variety |
| | | | | | | →standardization of platforms |
| | Light-weight design | ↓5%–45% steel [16, 207] | ↓ | ↓ | MH | ◊ Costs |
| | | ↑50% metals fleet (Al replacing steel) [153, 154] | | | | → ↑fuel efficiency standards |
| | Reuse | ↓30% steel fleet [16] | –↑ | ↕– | LM | ◊ Logistics, labor cost |
| | | ↓2.8%–5.1% fleet [163] | | | | → materials price |
| | Remanufacturing | ↓69%–90% for a diesel engine [161, 162] | ↓– | ↕– | LM | → Materials price |
| | | No fleet evidence | | | | |
| | Recycling | ↓10%–38% vehicle [152, 208] | — | –↓ | MH | ◊ Sorting and separation |
| | | ↓50% Al in fleet [153, 154] | | | | → materials price (for metals) |

[a] Assessment of the author team based on reviewed case studies.

[b] Availability of evidence: L limited, M medium, R robust; Level of agreement: L low, M medium, H high. Studies with limited evidence cannot have a high level of agreement. Limited evidence: 2–3 studies, medium evidence: 4–6 studies, robust evidence: >7 studies. Agreement reflects an expert judgment based on the quality of evidence, the degree of potential disagreement and the size of the literature.

There is a limited evidence and medium agreement on the contribution of lifetime extension to emission reductions in buildings, when refurbishment to reduce operational energy use is undertaken. More studies investigate emission reductions from lifetime extension for private vehicles, but they show little agreement; there is a trade-off that is the larger the quicker operational energy use declines for new age-cohorts. As operational emissions stabilize at low levels or car use intensifies, the strategy may become more important.

The reuse of building elements and car parts can result in substantial emission reductions for the production of the parts in question, but the scope of application is limited by practical considerations.

Remanufacturing can be mostly seen as a reuse/lifetime extension strategy. There is a limited number of studies, but these support the ability of the strategy to reduce emissions in cases with a limited scope, but the wider applicability of the strategy within the product groups reviewed here is not well understood.

There is a medium level of evidence and a high level of agreement that the recycling of metals from buildings and vehicles already contributes to substantial emission reductions, while the recycling of EEE addresses other environmental concerns but contributes little to overall GHG mitigation. There is a limited level of evidence but agreement that further emission reductions can be achieved by sorting metals according to alloys to avoid the contamination of metal flows and allow for recycling even when metal stocks are no longer increasing. There is a medium level of evidence and agreement on the benefit of recycling of construction minerals, with high agreement that existing recycling as aggregates reduces the energy demand associated with aggregate production, but limited evidence for the benefit of recycling cement or concrete to anything but aggregate. There is insufficient evidence to evaluate the suitability of recycling of construction minerals and plastics under future conditions of a more stringent emissions control policy.

Overall, strategies to reduce the demand for materials or the products themselves, through more intensive use, down-sizing, light-weighting, and lifetime extension offer the largest emission reductions. Many of these would be available in the short run. More intensive use and lifetime extension apply to the existing stock as well. Further research and development are needed to improve these strategies, the policies to support them, and to avoid adverse trade-offs and rebounds. There are specific applications in which reuse, remanufacturing, and recycling can also achieve worth-while further emission reductions, which are likely to become more important in the long run.

*Achieving measurable emissions reductions from ME*

Where reviewed studies have indicated emission reductions from ME, it has usually been with respect to a referenced service. Change in attributes and costs of the service may affect either the acceptance of ME or the consumption level of the service. Where ME changes attributes of the service, such as driving a smaller vehicle or living in a refurbished rather than a new flat, the question is whether ME service is as attractive as a more conventional one. Where ME reduces costs, such as with light-weighted or shared vehicles, the question is whether it will result in an increased demand. Both modeling and empirical evidence point to a sizable rebound effect to energy efficiency [209] and a similar effect applies to materials [210, 211]. We have highlighted some fundamental behavioral questions, such as whether ATs will be used to complement public transportation (last mile) or whether they will multiply the trips taken and reduce urban densities. For other strategies, such responses are less likely, such as lighter buildings, which cost as much as conventional ones. The behavioral response to ME is an open question that deserves research attention. The question of whether a technology-push strategy for resource efficiency will contribute to GHG mitigation depends on the outcome of such research.

Within the context of climate mitigation scenarios, ME offers another technological solution which reduces the cost of achieving a desired level of mitigation and can be hence seen as desirable. In a modeling exercise, the carbon price employed to reach such a target would be lower than without these options available, and it may still guard against a rebound.

*ME in integrated policy studies*

While the preceding sections suggest that significant emissions reductions may be achieved from a technical perspective [3, 7, 13], more integrated policy modeling is necessary to assess the broader economic, social, and environmental dimensions of ME strategies [212]. However, existing integrated assessment models (IAMs) necessary for such multi-dimensional assessment are generally poorly equipped to analyze ME options due to pervasive structural and data limitations [213]. Key barriers include lack of data on ME technology performance and costs, application markets and barriers, and intersectoral (i.e. life cycle) effects as well as lacking representation of material-containing product stocks (buildings and structure, vehicles, machinery) in the models. As a result, few studies have taken integrated analysis approaches, and their results are generally limited to macro-level insights that are insufficient for the detailed policy design necessary to accelerate ME as a mitigation strategy. For example, the IEA has represented selected ME strategies in its two main integrated energy systems models—the World Energy Model and ETP-TIMES—to provide global estimates of achievable GHG emissions savings in its WEO 2015 and ETP 2017 scenarios, respectively [1, 214]. However, savings estimates were not inclusive of upstream (e.g. reduced freight) or downstream (e.g. lighter-weight vehicles using less fuel) effects due to a lack of life cycle systems data, nor were cost implications considered. More recently, Materials Economics estimated EU-level GHG emissions reductions associated with ME policies, but did so using independent models for each industrial sector, thereby lacking important economy-wide perspectives [12]. As a recent review found that current policies are insufficient to tap the significant mitigation potential of ME [215], improved IAM capabilities for robust, policy-relevant assessment of ME strategies should be a critical priority. Emerging work on a country level may offer indications for how the effect of ME can be modeled [216, 217].

## Conclusions

The literature supports a strong role for ME as an avenue for reducing GHG emissions connected to material-intensive systems, including buildings and light-duty vehicles, while evidence for emission reductions within EEE is more limited. There is a significant potential to reduce the substantial emissions connected to producing materials used in buildings and vehicles. The contribution of ME to climate change mitigation is supported by a wide number of case studies and by a very limited number of studies attempting an up-scaling and scenario development, as well as very few ex-post studies. These studies offer a strong support for emission reductions, which can be substantial for more intensive use, light-weighting of buildings, lifetime extension of buildings in countries with short building lifetimes, and right-sizing of vehicles in countries with large default vehicles. There are situations in which trade-offs with operational energy use and rebound effects are important, so that determining an optimal strategy requires a proper analysis, e.g. for lifetime extension related strategies including reuse and remanufacturing. Studies have often focused on highly developed countries or China and there is a lack of information from other regions, even though gains are likely to be larger in developing countries. The global potential emission reductions from material are still poorly characterized.

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## ORCID iDs

Edgar G Hertwich https://orcid.org/0000-0002-4934-3421
Luca Ciacci https://orcid.org/0000-0002-5151-5384
Tomer Fishman https://orcid.org/0000-0003-4405-2382
Niko Heeren https://orcid.org/0000-0003-4967-6557
Stefan Pauliuk https://orcid.org/0000-0002-6869-1405
Paul Wolfram https://orcid.org/0000-0002-8470-427X

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