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# Assessing the life cycle costs and environmental performance of lightweight materials in automobile applications

Robert A. Witik [a], Jérôme Payet [a], Véronique Michaud [a], Christian Ludwig [b,c], Jan-Anders E. Månson [a,*]

[a] Laboratoire de Technologie des Composites et Polymères (LTC), Ecole Polytechnique Fédérale de Lausanne (EPFL), Lausanne CH-1015, Switzerland
[b] Paul Scherrer Institut (PSI), Forschungsbereich Allgemeine Energie, CH-5232 Villigen PSI, Switzerland
[c] École Polytechnique Fédérale de Lausanne (EPFL), Faculté de l'Environnement Naturel, Architectural et Construit (ENAC-IIE), CH-1015 Lausanne, Switzerland

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ABSTRACT

Life cycle assessment (LCA) and manufacturing focused life cycle costing are used to evaluate the potential advantages of composites in automotive applications. The life cycle costs and environmental performance of several suitable lightweight polymer composites are quantified and compared against magnesium and steel for a representative component. The results indicate that weight reduction will not always lead to improved environmental performance. Materials offering high weight savings such as carbon fibres and magnesium have been shown to give limited or negative environmental benefits over their life cycles due to increased environmental burdens associated with their production. Lower performance materials such as sheet moulding compounds were found to perform better from a life cycle perspective despite not being recycled. Lighter weight vehicle components were found to be always more costly; however their use did lead to reduced costs for the consumer through lower fuel consumption.


## 1. Introduction

Environmental protection is a growing concern for many industries today, with emphasis on the reduction of carbon dioxide ($CO_2$) emissions to mitigate climate change. This is of particular importance for the transportation industry, which is currently the second largest contributor of anthropogenic greenhouse gas emissions within the European Union (EU) [1]. Of these emissions, around 93% are generated by road transportation and as demand grows and vehicle fleet sizes increase, the industry faces the challenge of reducing emissions in order to meet targets set by the European commission, Non Governmental Organisations (NGOs) and to appease public pressure. Of all the modes of road transportation, passenger cars remain the dominant group in terms of greenhouse gas emissions. If targeted for efficiency improvements, they provide the opportunity to achieve a significant reduction of both sector and industry emissions.

The automotive industry has a number of approaches for improving vehicle efficiency and thus lowering emissions during the dominant use phase of a vehicle. These include drive train efficiency improvement [2], alternative fuel systems [3] and vehicle weight reduction [4]. Weight reduction is an approach offering advantages relatively quickly, through improved handling, braking and reducing propulsion power requirements. It is also seen as the first step towards the downsizing of other vehicle components, known as mass decompounding [5]. Lowering vehicle mass may be achieved with two key approaches: innovative design, where components are optimised to achieve higher performance, and materials substitution, where existing automotive materials such as steel are replaced with lighter weight alternatives. In terms of substitute materials, lighter metals such as aluminium and magnesium are increasing in use, as are high strength steels in some applications [6,7]. Polymer composite materials also hold potential for weight reduction in the automotive sector. Previous estimations suggest that the significant use of glass reinforced polymers could result in a 20–35% reduction in vehicle weights [8]. The automotive industry has to some extent already benefited from the use of composite materials in lower performance applications for several decades. Glass Mat Thermoplastics (GMT) have been in use for over 40 years in applications such as seatbacks, bumper beams and battery boxes [6,9–11]. Sheet moulding compounds (SMC) have also played a role in automotive manufacture since the 1970s with use further increasing in recent years. Despite the relative success of composites in these applications, their spread to higher performance areas with greater weight saving potential has been limited by high costs, and slow cycle times. These limitations have led to the development of automated manufacturing technologies and faster resin systems, moving the industry a step closer to providing solutions for a wider range of automotive applications.

As composites edge towards the higher performance applications, designers are faced with a wider range of materials from which

* Corresponding author.
*E-mail address:* jan-anders.manson@epfl.ch (J.-A.E. Månson).

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to choose. The materials selection process has typically been driven by cost [12] and previous work has moved some way to develop cost estimation models suitable for the composites industry [13,14], which have been applied to comparative cost studies determining the effects of materials and process change [15–17]. However, these models do not consider costs related to use, efficiency improvements through weight saving, or associated disposal costs. Use costs are of interest to the consumer, while disposal costs are becoming increasingly relevant to manufacturers as producer responsibility at the end of life grows and selecting materials on the basis of low cost is no longer appropriate. A lifecycle cost approach is therefore suggested to enable more holistic assessments.

Costs are an important consideration in the effort to produce affordable personal transportation. However, perhaps the primary driver for the use of lighter weight materials today is to lower emissions. It would, therefore, be prudent to also evaluate the environmental performance of a candidate material in addition to its cost. As with costs, the environmental performance of a material must be considered over the full life cycle of a product. Reductions in environmental burdens from one phase of a vehicle's life cycle may result in increases in other areas which could negate initial benefits. It is precisely these shifts of environmental burdens which need to be understood, thus ensuring materials are selected on their ability to improve complete environmental performance, rather than focusing on one life cycle phase.

The environmental assessment of materials, products and services is typically carried out using Life Cycle Assessment (LCA). LCA is an established methodology supported by the International Standards Organisation (ISO) [18,19] and is growing in use across many industries, and in particular the automotive industry. It has been used for standalone assessments of complete vehicles [20], materials [21] and alternative fuels [3]. Some composite specific studies have also been carried out. Joshi et al. [22] use LCA to compare natural fibre composites (NFC) with glass reinforced plastics (GRP), concluding that natural fibre composites are less burdening for the environment in automotive applications, provided that components are able to match the service life of GRP. More recently Duflou et al. [23] have used LCA to determine the environmental impact of carbon fibre reinforced composites in car manufacturing. Fuel savings during the vehicle's use phase were observed and an environmental breakeven point for the carbon fibre was found to be at 132,000 km into the vehicle's life.

Few studies exist which combine environmental and cost assessment of materials in an automotive context. Ungureanu et al. [24] developed a sustainability model to perform such an assessment. Composite materials were not considered, but aluminium was compared with steel for a Body in White (BIW) with costs and environmental impacts quantified over the life cycle of the structure. Cost modelling was used to calculate component manufacturing costs. Use phase costs considered fuel use only and were derived from the total quantity of fuel used during the vehicle's lifetime. The environmental assessment was carried out with $CO_2$ as a single environmental indicator, with the disadvantage that other important environmental effects upon ecosystem and resources are not identified.

Roes et al. [25] have used environmental and cost assessment to compare polypropylene nano-composites with steel for an automotive panel. Part weight equivalence was calculated with Ashby material indices [26], and the LCA was carried out according to the ISO format [17]. Manufacturing costs were quantified using what was described as "very rough estimates". Lloyd and Lave [27] and Song et al. [28] have also studied economic and environmental effects of materials substitution with composite materials. However, rather than using ISO LCA methodology, an economic input–output model developed at Carnegie Mellon [29] was applied deriving environmental impacts from the economic output of related industry sectors within the USA. Both studies again used Ashby material indices to determine weight of replacement components based upon stiffness. Lloyd and Lave [27] considered substituting nano-clay reinforced composites and aluminium for steel in light duty body panels, Song et al. [28] compare extruded carbon fibre components in trucks and buses.

The present study explores a wider range of materials with the aim to move towards establishing a more robust methodology for combined environmental and cost assessments of composite materials. A case study explores the economic and environmental effects of substituting steel for lighter weight alternatives with the focus on composite materials. Four composite scenarios have been chosen which include sheet moulding compound (SMC) and Glass Mat Thermoplastic (GMT) which represent more established high volume manufacturing processes. A more recent automated preforming technology, combined with reaction injection moulding is also included. Glass fibre reinforcement is considered as a lower cost option and carbon fibre a more costly alternative with higher weight saving potential. Finally, a magnesium variant is added to the study as a metallic lightweight alternative.

Manufacturing and life cycle costs are derived from a technical cost model which has been modified to include use and end of life costs, thus providing a more accurate basis for comparison than estimations used in previous studies. The environmental performance of each scenario is then quantified using LCA according to ISO guidelines. This approach is favoured over previously applied economic input output models, which are limited geographically to the USA. In addition, environmental impact assessment is not restricted to a single impact category but carried out using four, namely: human health, ecosystem quality, climate change and resources. This approach enables the overall environmental impact to be measured rather than focusing on a single specific impact such as climate change. Finally, previous studies have all considered weight savings based upon Ashby material indices whereas the component weights in this study have been derived from finite element analyses (FEA) performed by material suppliers. Weights are therefore more representative as the components have been specifically designed for the application incorporating geometrical design features to increase stiffness and other performance characteristics.

## 2. Methods

### 2.1. Life cycle cost modelling

The life cycle cost model is manufacturing focused, and based upon the parametric technical cost model previously described by Wakeman et al. [13] used previously for comparative studies of composites processing technologies [15–17]. For the purpose of this study the model has been extended to include use and End of Life (EoL) costs. The coefficient for the reduction of fuel consumption determines fuel saving as a function of weight reduction. The figure for petrol driven vehicles is 0.34–0.48 l/(100 kg $\times$ 100 km) and has been used here to quantify fuel use and associated cost over vehicle lifetime [30,31]. Consumption is first calculated at the vehicle level, and then allocated to the component as a proportion of total weight. EoL costs include transportation, shredding and final disposal, which either incur costs if incineration or landfill is chosen, or benefit if recycled. Key input parameters required for the cost model are shown in Fig. 1 with cost outputs. Data regarding materials quantities, power consumption, waste, etc., obtained from the cost model are subsequently used in the LCA.

### 2.2. Life cycle assessment

Life Cycle Assessment (LCA) can be described as an environmental accounting methodology, which enables the quantification and evaluation of environmental effects, associated with a specific

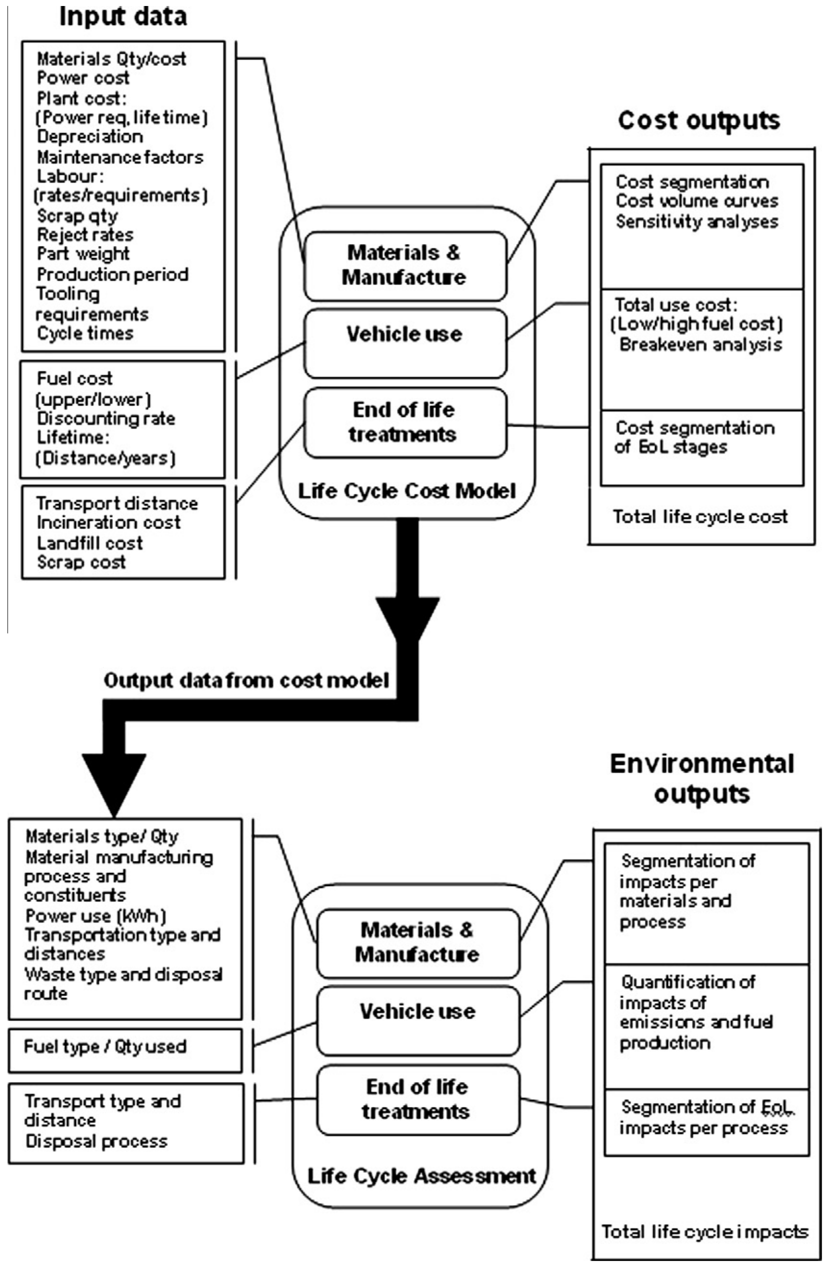


**Fig. 1.** Interaction between cost model and life cycle assessment study showing the key inputs and obtainable outputs.

service, manufacturing process or product. The assessment considers the complete life cycle, from the extraction of raw materials until the point at which all residuals are returned to the earth [18]. A typical product life cycle is deemed to be made up of four phases, which consider: raw materials acquisition, manufacture, use, and end of life treatment. Potential environmental impacts such as climate change, ozone depletion, tropospheric ozone creation (smog), eutrophication, acidification, toxicological stress on human health and ecosystems, depletion of resources, and land use are also considered [32]. These provide a more holistic indication of environmental effects than a single metric such as energy use or $CO_2$ emissions. The approach can be used to assist with the optimisation of environmental performance of a product, or to compare products to determine the most environmentally favourable solution. The methodology is supported by a set of standards from the ISO [18,19], and follows the four stages shown in Fig. 2 and summarised below.

*2.2.1. Goal and scope definition*

The goal and scope definition is the first stage of an LCA. It is where the purpose of the study is described and where the boundaries of the product system are defined according to factors such as time constraints, data available and depth of study required. At this point a "functional unit" is defined which forms the basis for comparison if two or more products are being considered, thus ensuring that the products are compared according to their ability to fulfil the specific function for which they were intended.

*2.2.2. Inventory analysis*

The second stage concerns the capturing of data related to the inputs and outputs of the system described in the "goal scope and definition". Quantities of raw materials, waste flows and emissions which are attributed to the products life cycle are quantified and allocated to the functional unit defined. Life cycle inventory (LCI) databases hold data on energy and materials supply,

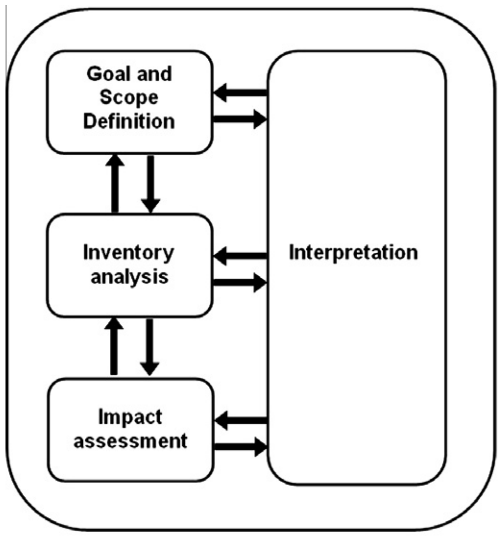


**Fig. 2.** Life cycle assessment framework showing interactions between stages [16].

chemicals, metals, resource extraction, transport and waste management. One such database is Ecoinvent [33,34], which is currently regarded as the as the world's leading database with around 4000 datasets accompanied by supporting documentation. Such databases may be linked to LCA specific software such as Simapro [35] which enable the user to build complex product systems. Data which is not available in such a database may be acquired from reliable industrial sources, experimentation or literature sources.

*2.2.3. Life cycle impact assessment*

The impact assessment stage of the study provides the means to determine the potential environmental impacts from the contributions of the emissions, waste and resources determined in the inventory analysis. ISO 14042 describes classification and characterisation as obligatory elements of this phase. Classification assigns the elements of the LCI data to relevant impact categories such as climate change, toxicological stress land use etc. For instance methane ($CH_4$) and $CO_2$ are both assigned to the global warming category. Once assigned to the appropriate impact category the relevant characterisation factors must be applied to determine the contribution of the specific LCI element to the impact category. Within the global warming category, results are given in kg of $CO_2$ equivalents (eqv), therefore 1 kg of $CO_2$ quantified in the LCI would be indicated by 1 kg of $CO_2$ eqv in the climate change impact category. $CH_4$ on the other hand contributes 25 times more to climate change than $CO_2$, therefore the characterisation factor would be 25 and 1 kg of $CH_4$ from the LCI would be communicated as 25 kg of $CO_2$ equivalents in this category. A number of Life Cycle Impact Assessment (LCIA) methods already exist such as Eco-indicator 99 [36], CML 2 [37], and Impact 2002+ [38]; the appropriate method is chosen with respect to the required outputs of the goals of the study.

**Table 1**
Materials and process scenarios considered with final part weight and percentage weight reduction achieved.

| Material | Processing | Component weight (kg) | Weight reduction |
|---|---|---|---|
| Steel | Stamping | 5.8 | Baseline |
| SMC | Press moulding | 2.5 | 57% |
| GMT | Press moulding | 2.4 | 59% |
| Glass fibres | Reaction injection moulding | 2.3 | 60% |
| Magnesium (AZ91) | Die-casting | 2.2 | 62% |
| Carbon fibres | Reactive injection moulding | 1.8 | 69% |

*2.2.4. Interpretation*

Here results are interpreted, summarised and discussed, conclusions are drawn and recommendations made against the initial goals. Fig. 2 shows that that there are interactions between interpretation and the other stages as the study is constantly measured against its initial goals and scope and refined during its duration.

## 3. Case study

### *3.1. Goal and scope definition*

A steel vehicle bulkhead component situated at the rear of a two seat vehicle separating the luggage space from the passenger compartment is considered for weight reduction through materials substitution. This part is a thin plate, rectangular in shape, with requirements of bending and torsion stiffness, up to 80 °C. The materials considered for the replacement are shown in Table 1, along with their processing routes, weights, and percentage weight reductions. The weight assumptions have been derived from finite element analyses performed during a previous study between the automotive industry and material suppliers. Weights are considered part specific, as designs reflect an optimum solution for each material to fulfil the required torsion, shear stiffness, and modal requirements of the application within the appropriate operating temperatures. Detailed results and performance criteria are not included for the purpose of respecting confidentiality. The goal of the study is to compare the light weight materials with the steel baseline in terms of cost and environmental performance, identifying the solution with the highest gains in both areas. The comparison is across the four main life cycle phases of the product: (i) raw materials acquisition, (ii) manufacture, (iii) use, and (iv) end of life. Shifts of costs and environmental burdens between these phases are of interest along with the most significantly contributing processes. The unit of comparison or functional unit (FU), to which all costs and impacts are allocated, is "one vehicle bulkhead over a vehicle lifetime of 200,000 km".

### *3.2. Scenario descriptions and inventories*

*3.2.1. General assumptions*

Prior to describing each material scenario in detail, some general assumptions are first defined. Manufacture, use and disposal of the bulkhead takes place within Western Europe where transportation is by 40t truck. Materials are sourced in mainland Europe with the exception of magnesium which is shipped by sea from China. Manufacturing plant layout diagrams and associated equipment costs are given in Figs. A1–A5 and Tables A1–A5 of the supporting information given in the appendix. Manufacturing cost input data assumptions are listed in Table 2. Ecoinvent 2.1 Life Cycle Inventory (LCI) data [34] is used for the LCA exclusively unless otherwise stated. The power mix assumed for manufacture is the European wide average from the Union for the coordination of the Transmission of Electricity (UCTE), the Chinese power mix was used for magnesium production [34]. Life cycle assessment models are defined in the commercially available LCA software Simapro [35] with Impact 2002+ applied as the impact assessment method [38].

**Table 2**
Case study manufacturing input data.

| Item | Qty | Description |
|---|---|---|
| Volume | 80,000 | Target production volume per year |
| Production period (years) | 7 | Period over which production occurs |
| Energy cost (€/kWh) | 0.1 | Industrial power cost |
| Plant operating cost (€/m2/year) | 90 | Area used by cell cost for heating, cleaning etc. |
| Working days | 250 | Plant operating days per year |
| Labour cost (€/h) | 33 | Salary cost |
| Depreciation time (years) | 7 | Estimated machine lifetime |
| Shifts | 3 | Number of shifts per day |
| Hour per shift | 7.5 | Number of hours per shift |
| Machine maintenance factor | 0.05 | Factor of original purchase cost |
| Efficiency factor | 0.85 | Unscheduled lost hours (breakdown etc.) |
| Consumables (€/h) | 0.5 | Consumables per person per hour |

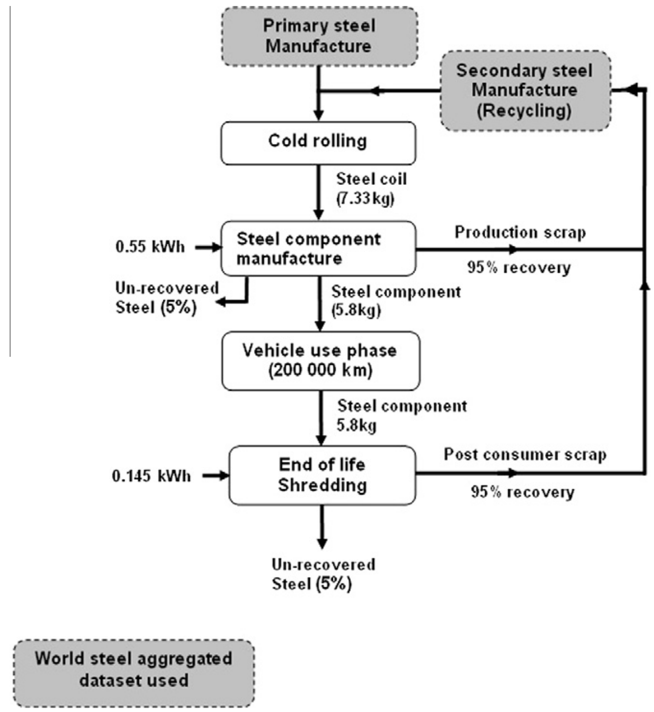


**Fig. 3.** System diagram showing material, energy use and waste flow for the steel bulkhead component.

*3.2.2. Scenario 1, steel*

Steel plays a dominant role in the production of structural automotive components and makes up 65–70% of body mass in today's passenger cars [39]. Its extensive use within automotive manufacture has been supported by a combination of factors such as low cost, high performance and suitability for high-speed manufacture. It is therefore considered to be the standard for BIW material and is the baseline for this study. The bulkhead component is manufactured from cold rolled steel supplied as coil. The LCI data for this material was obtained from the World Steel Association (WSA) and includes end of life recovery (95%) based upon the automotive sector within Europe [40]. The assumed manufacturing facility consists of 2 production cells; a coil handling/blanking cell and a stamping cell. The combined power requirement of the process per component produced is 0.55 kWh and the cycle time assumed is 2 s per cell. During the manufacturing process clean scrap is generated at 28% of the initial amount required, the value of which is deducted from the overall manufacturing costs within the cost model. For the end of life phase, power requirements for the shredding and sorting of the ferrous scrap has also been added as it is not included in the LCI data provided by WSA. The power consumption of a 3700 kW shredder is calculated per component and included in all scenarios in this study [41]. Material, energy and waste flows used in the LCA are summarised in the system diagram shown in Fig. 3.

*3.2.3. Scenario 2, magnesium*

The use of magnesium alloys within automotive applications has increased by around 15% over the last decade and is predicted to grow further [42]. The low density of magnesium together with recent advancements in processing techniques have increased its potential for use in a wider range of applications such as seat frames, cross beams and engine covering components [43]. In this scenario the bulkhead component is manufactured using AZ91 casting alloy in a high pressure die-casting operation. China currently produces over 70% of the world's magnesium [44], and so is considered to be the source with LCI data taken from literature [45]. The component manufacturing facility is an automated magnesium die-casting line situated in Europe and consisting of three cells; a die-casting cell, punching cell and machining/de-burring cell. The cycle time for the process has been modelled at 110 s per component for the die-casting cell and 65 s for the punching and trimming cells [46]. LCI die-casting data was used for cell 1 [34], and specific equipment power consumption used for cells 2 and 3 totalling 6.86 kWh per component manufactured [46].

Two types of magnesium waste are considered in this study, production, and post consumer waste. In a typical magnesium die-casting operation, only around 50% of the metal which enters the system ends up in the magnesium component, the remainder forms scrap consisting of circuit material, dross and chip from the machining process [47]. Different types of magnesium scrap are considered to have different qualities which subsequently affect the method of recovery and value. Scrap is classified by the magnesium industry into 10 different categories according to purity, cleanliness and size [48]. Each of the magnesium scrap categories require different treatments to either recycle the material as magnesium or to reuse within other applications. Circuit material from the casting process falls into the category of clean scrap (class 1A and 1B), and is the easiest to reprocess. In this study, clean production scrap is re-melted and reused within the manufacturing plant (closed loop recycling). Any additional waste from the casting process such as dross, chips and slurry falls into a higher magnesium scrap categories (2 or above) requiring more complex handling due to size or level of contamination. Recycling of this higher classification scrap is generally not considered economically viable due to the handling and extensive pre-treatment required to convert the magnesium waste into a useable product [48]. Waste not falling into the class 1A or 1B categories, which is around 8% of die-casting shot size, is considered lost [47,48]. Post consumer scrap is generated when a vehicle reaches the end of its life. When vehicles are shredded the magnesium remains within a non ferrous mix consisting of aluminium, zinc, copper, brass, stainless steel and magnesium [48]. Magnesium content is generally low in Europe [49]. Any, which is recovered, is deemed to fall into class 7 scrap category [47,48] and assumed lost in this study due to the levels of contamination present. Material, energy and waste flows used in the LCA are summarised in the system diagram shown in Fig. 4.

*3.2.4. Scenario 3, Glass Mat Thermoplastic (GMT)*

LCI data for the constituent materials of glass fibre and Polyamide 6 have been used and taken from Ecoinvent [34]. It is assumed that the Manufacturing of the GMT component occurs in a 2 cell facility consisting of compression and trimming cell. The cycle time

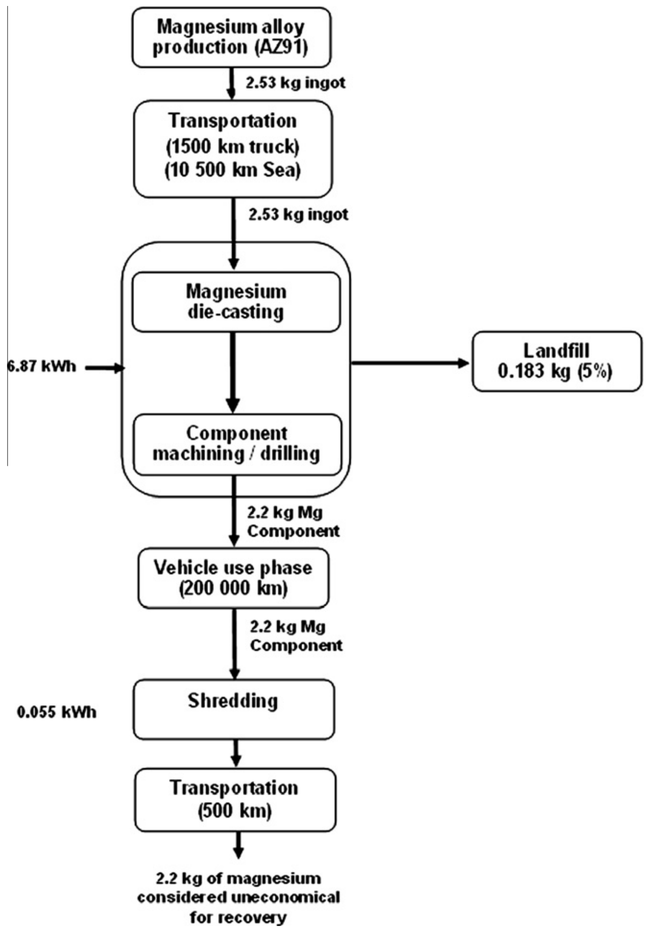


**Fig. 4.** System diagram showing material, energy use and waste flow for the magnesium bulkhead component.

for the process has been modelled at 60 s per component for the compression cell and 60 s for the trimming cell [46]. The total energy consumption is 9.5 kWh per component produced. A small amount of trim is generated during the manufacturing process (0.099 kg per component); this is disposed of through municipal waste incineration where energy recovery takes place. The Gross Calorific Value (GCV) of GMT is 25.2 MJ/tonne [50] and the incinerator energy recovery efficiency runs at 18%. [51]. At the end of life the GMT component remains within the vehicle and is shredded. GMT fragments are separated from the metallic fraction of shredder output, and classed as Automotive Shredder Residue (ASR) [52]. The ASR is transported to an incineration facility where energy is reclaimed and considered as an avoided product in the system shown in Fig. 5. This end of life treatment is considered for all of the composite scenarios within this study.

*3.2.5. Scenario 4, Sheet Moulding Compound (SMC)*

SMC is one of the most common composite materials used within the automotive industry today [11]. It is a low cost flow moulding compound comprised of unsaturated polyester resin, short glass fibres and fillers. When pressed it is capable of forming complex geometries with relatively short cycle times. SMC is also able to resist paint baking temperatures and produce high quality surface finishes making the material particularly attractive to the automotive industry. The facility required to produce the bulkhead component consists of 2 cells, a compression moulding cell and a trimming cell, which have a combined energy requirement 5 kWh per component produced [46]. During the process 0.212 kg of waste material is generated. Ecoinvent data [36] are used for the constituent materials of the SMC. Disposal of production and post consumer waste occurs through incineration where the GCV of 7.5 MJ/kg [50] is used to calculate recovered energy, again treated as an avoided product and benefiting the product system shown in Fig. 6.

*3.2.6. Scenarios 5 and 6, Structural Reaction Injection Moulding (SRIM)*

The SRIM process utilises both the Programmable Powdered Pre-forming Process (P4) developed by the Automotive Composite Consortium (ACC) and Reaction Injection Moulding (RIM) [53]. When combined these processes have been shown to be capable of producing cost competitive structural automotive components up to volumes of 50,000 per year with 25% reduction in component weight against steel [54]. Savings occur through taking advantage of the fast reactivity of the polyurethane resin chemistry employed with fast repeatable automated pre-forming. Two variants of this process have been considered. One component produced with glass fibre reinforcement and one with carbon fibre. LCI data for carbon fibres were obtained from the Japan Carbon Fibre Manufacturers Association (JCMA) [55]. The SRIM manufacturing facility consists of 3 cells; pre-forming, injection and trimming requiring 24.75 kWh per component produced. The carbon fibre process generates 0.091 kg of waste and the glass fibre component generates 0.113 kg of waste per component. Incineration is used to dispose of both production and end of life waste. The GCV values for the carbon and glass composites are 31.7 MJ/kg and 21 MJ/kg respectively [50]. Energy is recovered and treated as an avoided product within the system shown in Fig. 7.

*3.2.7. Vehicle use phase*

The environmental burdens from the vehicle use phase are calculated at the component level using LCI data for passenger car use (EURO 4) from Eco-invent [34]. The datasets used include impacts from fuel production as well as emissions from the vehicle itself. Only fuel costs were considered for the vehicle use phase which was 1.3 € per litre and was based upon the average unleaded petrol costs in the Euro 15 set of countries for 2008 [56].

## 4. Results

*4.1. Life cycle cost*

Fig. 8 shows the life cycle costs related to each material variant of the bulkhead component. The material variants are organised from left to right, starting from the base line material (steel) and decreasing in weight. The light weight material scenarios had weight reductions ranging between 62% and 69%. These subsequently generated use phase savings of between 57% and 69% compared to the steel component. The carbon fibre SRIM variant achieved the lowest use phase costs as it had the lowest weight.

All of the lightweight material scenarios showed increases in materials and manufacturing costs. These were between 37% and 324%, the two extremes being the SMC and carbon fibre SRIM respectively. Although steel achieved the lowest component manufacturing cost, it was the heaviest component with highest overall life cycle cost which was mostly related to the dominant use phase fuel costs. Component costs increased with weight saving potential as higher cost materials or processes were employed to obtain the weight savings. Of the lightweight materials examined, SMC achieved the lowest lifecycle cost despite being the heavier of the lightweight variants. The carbon fibre component achieved the greatest weight saving, but had highest lifecycle costs most of which was related to the materials and manufacture. EoL costs were very small for all scenarios, and represented 2% at most of the total life cycle cost of the composite components.

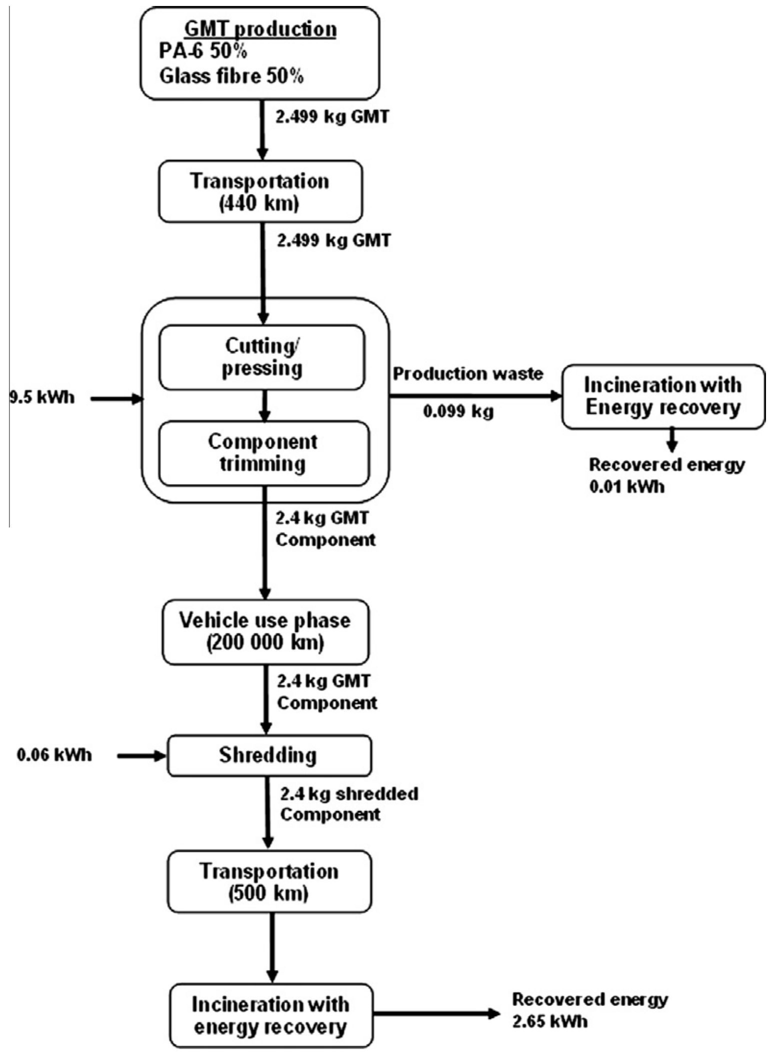


**Fig. 5.** System diagram showing material, energy use and waste flow for the GMT bulkhead component.

A breakeven analysis determined at which stage in the vehicle's life the lighter weight components would yield a cost benefit. The results of this analysis are shown in Fig. 9. The first of the lightweight scenarios to breakeven is SMC, which has the lowest materials and manufacturing costs after steel and benefits were achieved after only 17,200 km. The carbon fibre component had the lowest weight of all scenarios, but had the highest materials and manufacturing costs, resulting in benefits obtained relatively late in the vehicle's life at 121,000 km.

*4.2. Cost sensitivity*

Component cost (materials and manufacture) increases were observed for all lightweight materials. A cost sensitivity study was carried out to investigate the key parameters influencing component cost. Four material scenarios were considered: (i) steel, (ii) magnesium, (iii) SMC and (iv) carbon fibre. Five cost parameters were investigated: (i) labour, (ii) material, (iii) power, (iv) plant, and (v) tooling. Process cycle time was also included as an additional variable. Each of the parameters were varied by ±20%, their influence on component cost can be seen in Fig. 10a–d.

Fig. 10a shows the cost sensitivity analysis for the steel component. Material and tooling costs had the largest influence and contributed to 12.5% and 7% change in component cost respectively given the variation applied. Changes in the remaining parameters resulted in negligible variations of component cost (between 0.2% and 1.5%), these relatively small effects being due to the fast cycle times and the utilisation based amortisation scenario. For the lightweight scenarios, material cost was also the most dominant parameter affecting component cost, with the second being cycle time. The effect of cycle time variation on the carbon fibre SRIM component was slightly different than for the other material scenarios where a variation between +19% and −6% was observed when the cycle time was varied by ±20%. The faster cycle time led to expected cost reductions, however the increase in cycle time exceeded the production capacity of the plant; new manufacturing cells would have to be purchased in order to fulfil production volumes given the slower cycle times.

*4.2.1. Use phase cost*

The initial fuel price of 1.3 € per litre was used to calculate vehicle use cost and breakeven distances for each scenario. As fuel prices are subject to fluctuations, two additional scenarios were considered at ±50% of the original cost, in line with the fuel cost variation in the past 5 years. The cost breakeven points for each scenario are summarised in Table 3. The lower fuel cost scenario approximately doubled breakeven distance to the extent that it was no longer economically viable from a life cycle perspective to use carbon fibres. The 50% increase fuel cost scenario, reduced the breakeven distances to approximately 34% of that achieved

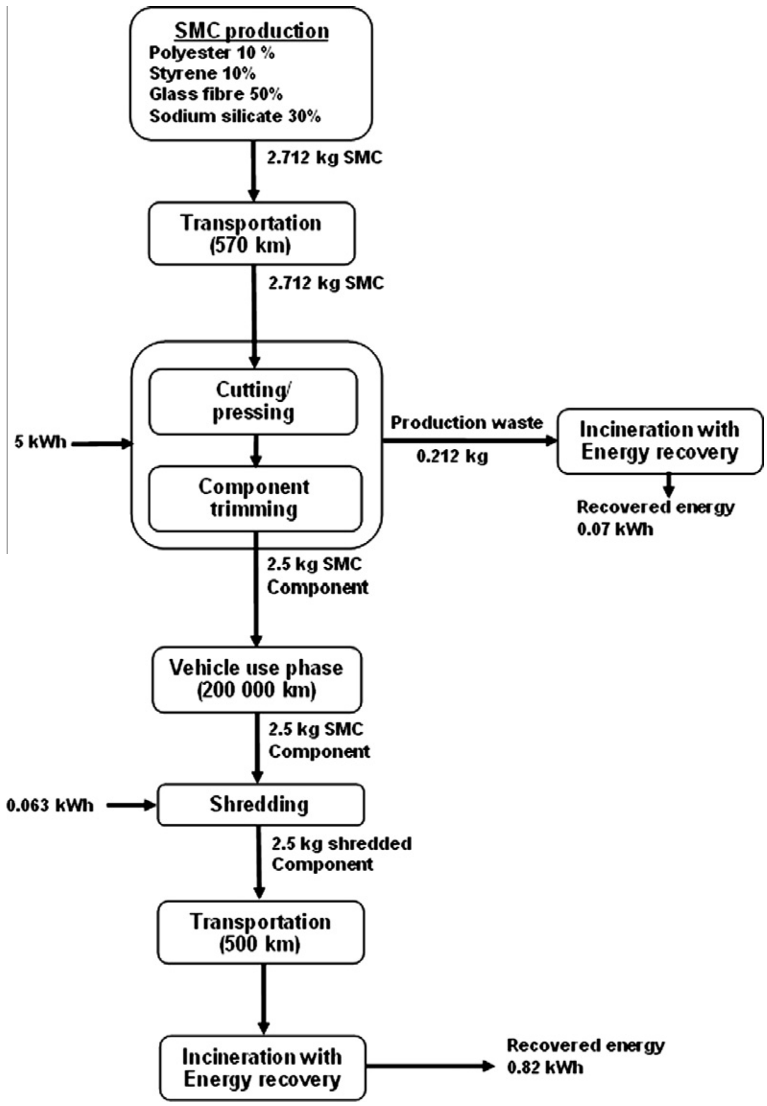


**Fig. 6.** System diagram showing material, energy use and waste flow for the SMC bulkhead component.

with the mid level cost. Life cycle costs were also calculated using the Inflation and discounting rates of 4% and 10% respectively with the total vehicle distance driven over 8 years. With this scenario breakeven for the SMC component would occur within the first year of vehicle operation at 16,800 km and during year six for the carbon fibre component at 137,000 km.

### 4.3. Life cycle assessment results

Results from the life cycle assessment component of this study are shown in Fig. 11a–d and are communicated via four endpoint damage categories (climate change, resources, ecosystem quality and human health) generated with the Impact 2002+ impact assessment method [38].

Climate change is considered first in Fig. 11a, units are in kg $CO_2$ equivalent emissions. The steel scenario is shown on the left of the figure and has the highest associated emissions of all the scenarios examined. The use phase is clearly dominant and accounts for 95% of the emissions for this component, with materials and manufacturing accounting for the remainder. Reduction of component weights led to reduced "use phase" $CO_2$ eqv emissions. These reductions ranged between 57% for SMC (2.5 kg) to 70% for the carbon fibre component (1.8 kg). All composite components had increased emissions from the materials and manufacturing phases with the exception of SMC which achieved a 45% reduction against the steel component. When considering the $CO_2$ eqv for the "complete" life cycle, SRIM, GMT and SMC showed reductions of 52%, 44%, and 56% respectively, while the CF scenario only achieved a relatively small 12% reduction despite having the largest weight saving. The magnesium component did not perform as well as the composites, despite achieving "use phase" emission reductions of 62%. $CO_2$ equivalent emissions from the complete lifecycle were increased by 4% with those emitted at materials and manufacturing phases being approximately 10 times higher than for the steel component. For this scenario materials and manufacturing emissions contributed to 66% of the life cycle emissions compared with just 5% for the steel. Fig. 12 shows a breakeven analysis for $CO_2$ equivalent emissions for all scenarios. The SMC component benefits environmentally from the outset, the carbon fibre component at 162,000 km and magnesium does not breakeven within the vehicle life time.

Fig. 11b shows the impact category "resources" which quantifies effects on resource extraction and non renewable energy (MJ). The effects of mineral extraction are calculated with the concept of surplus energy, assuming that the use of a specific mineral will deplete its available concentration, thus leading to increased

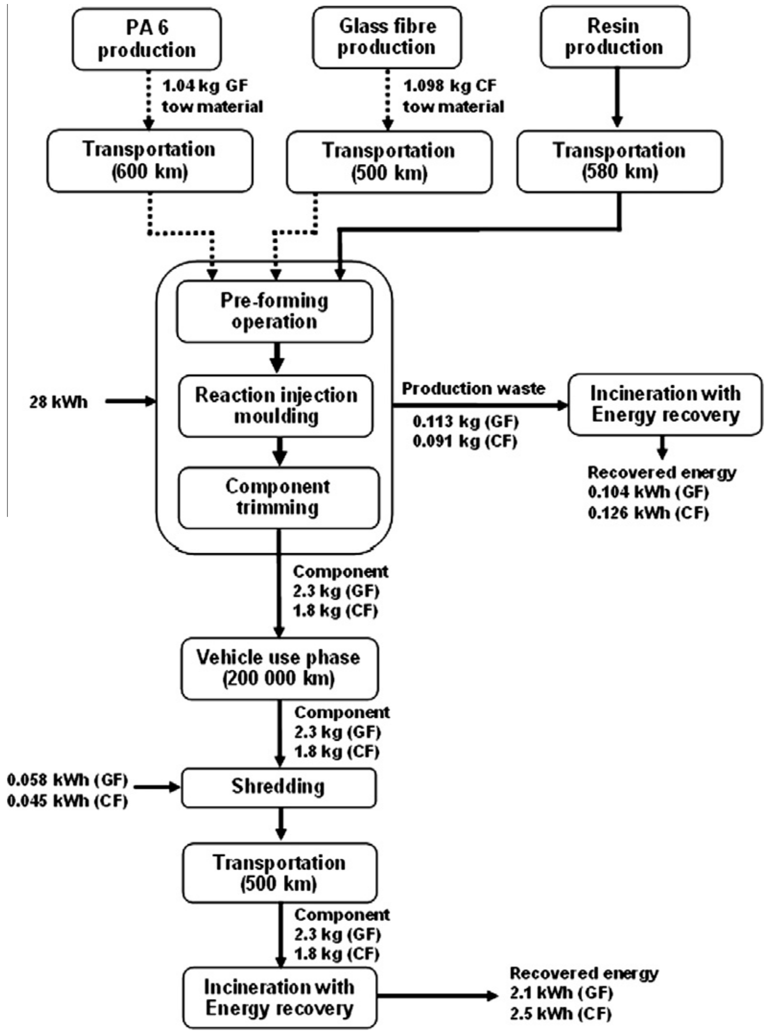


**Fig. 7.** System diagram showing material, energy use and waste flow for the carbon and glass SRIM components.

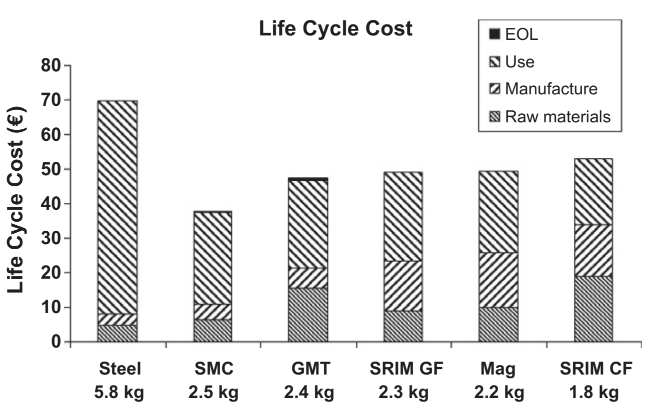


**Fig. 8.** Bar chart showing life cycle costs associated with each material variant considered for the bulkhead component. Cost in € is represented by the vertical scale, materials are identified on the horizontal axis with corresponding weights which decrease from left to right of the chart. Each material is represented by a divided bar showing the cost contribution of each lifecycle phase.

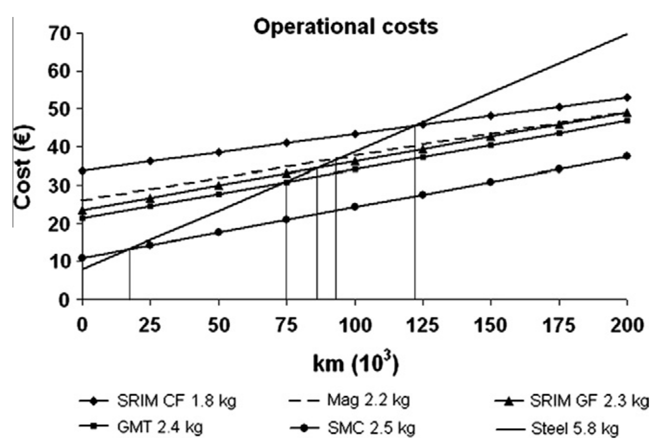


**Fig. 9.** Breakeven analysis results. Cost and vehicle distance are shown on the vertical and horizontal axes respectively. The line with the steepest gradient represents the steel component which is the heaviest of all the material scenarios at 5.8 kg. Each additional line represents one material scenario; the gradient of each line varies according to the component weight and the period at which they intersect the steel baseline corresponds to the breakeven distance.

energy requirements in the future to extract the equivalent quantity. The same concept is applied to that of non-renewable fuels together with quantification of total energy content lost as a resource [38]. Reductions in this impact category were observed for all the lightweight materials during the "use phase". Increases were again observed during the materials and manufacturing phases for all of

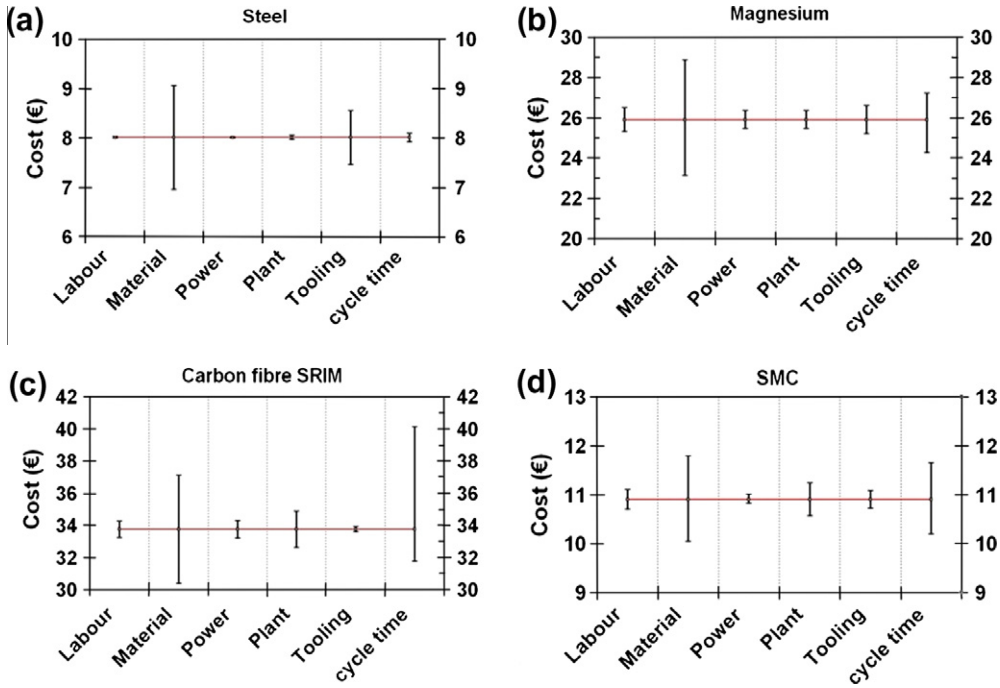


**Fig. 10.** (a–d) Manufacturing sensitivity analysis, for the steel, magnesium, carbon fibre SRIM and SMC materials. Showing the change in component cost given a 20% variation in key costs and parameters. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

**Table 3**
Break even distances for each material scenario using low, middle and high fuel costs.

| Material | Breakeven (km) fuel cost low (€0.65) | Breakeven (km) fuel cost mid (€1.3) | Breakeven (km) fuel cost high (€1.95) |
|---|---|---|---|
| SMC (2.5 kg) | 34,122 | 17,200 | 11,370 |
| GMT (2.4 kg) | 147,886 | 74,872 | 49,295 |
| GF SRIM (2.3 kg) | 167,247 | 84,626 | 55,749 |
| Magnesium (2.2 kg) | 187,617 | 94,916 | 62,539 |
| CF SRIM (2.1 kg) | Out of life | 122,807 | 80,988 |

the lightweight scenarios with the exception of SMC. The most notable increases during these phases were for the magnesium and carbon fibre SRIM components which were due to the energy intensive manufacture of carbon fibre [55] and fossil fuel use for magnesium production. Despite the increases in environmental burdens during the initial two phases of the life cycle, all lightweight components still achieved reductions over the complete life cycle against steel. The SMC variant achieved a 55% improvement, magnesium and carbon fibre SRIM saw relatively lower benefits at 15% and 7% respectively.

Impacts on human health are shown in Fig. 11c and are measured in Disability Adjusted Life Years (DALY) [38]. Human toxicity (carcinogenic and non carcinogenic effects) respiratory effects (inorganics and organics), ionising radiation and ozone layer depletion all contribute to health damage and are considered in this impact category. As with the other impact categories, contributions from the use phase of the vehicle were reduced with lighter weight components. Increases from the materials and manufacturing phases were observed for all lightweight scenarios with the exception of the SMC component. The materials and manufacturing phases of the magnesium and carbon fibre SRIM scenarios negated any benefits from light weighting resulting in overall increases of environmental burdens of 125% and 8% respectively. Glass fibre SRIM and GMT and SMC achieved 45% and 54% lower impacts respectively in this category.

Damage to ecosystem is expressed in Potentially Disappeared Fraction of ecosystem over an area over a specific time frame (PDF/$m^2$/yr). Fig. 11d shows that all of the lightweight material scenarios exhibit lower levels of damage to ecosystem over the full life cycle compared with the steel. Reductions were in the range of 45–63%, the extremes being magnesium and carbon fibre SRIM respectively, the SMC component yielded a 60% reduction of damage to ecosystem. In terms of materials and manufacturing phases, only the magnesium and carbon fibre SRIM scenarios showed notable increases in damage compared to the other scenarios.

### *4.4. Contribution analysis*

Achieving a lower weight automotive component has, with the exception of SMC, resulted in higher environmental burdens from the materials and manufacturing phases of the product life cycle. A crucial step towards improving environmental performance would be to understand these increases, and target changes more effectively. A contribution analysis has been performed focusing upon the initial two life cycle phases (materials and manufacture) to determine the materials and processes with the highest contributions to $CO_2$ eqv emissions and sensitivity to change. Fig. 13a–d show the dominant emissions for the steel, magnesium, SMC, and carbon fibre SRIM scenarios.

Of the lightweight scenarios the magnesium and carbon fibre components had the poorest performance in terms of life cycle $CO_2$ equivalent emissions, with the majority emitted during the materials and manufacturing phases. Process contributions for the magnesium component are shown in Fig. 13b. Magnesium alloy production dominates, contributing to 94.5% of emissions with the remaining processes and transportation accounting for 5.6%. Coal burning operations related to electrical power and heat

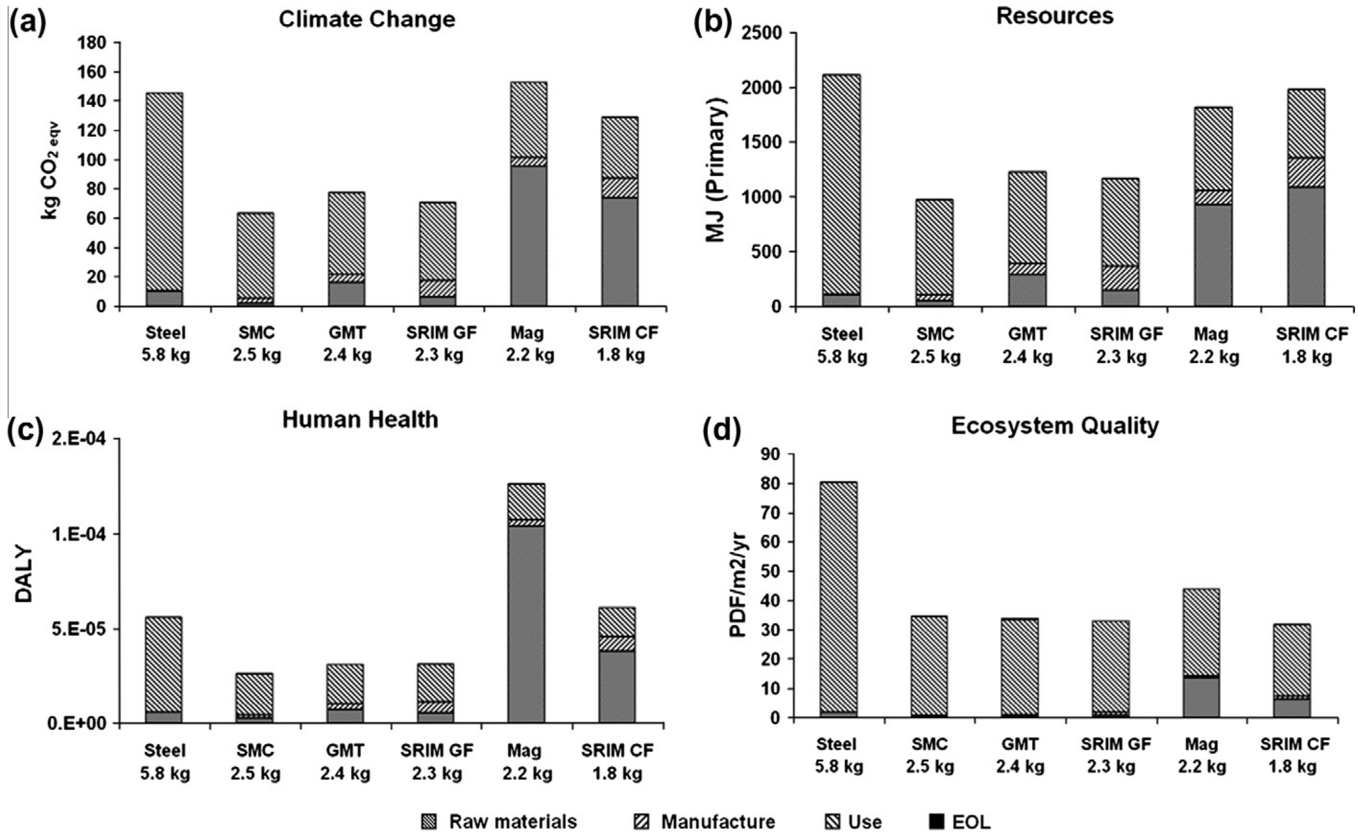


**Fig. 11.** (a–d) Life cycle assessment results for each material variant communicated through four endpoint categories related to climate change, resources, ecosystem quality and human health. Impacts are represented by the vertical scale, materials are identified on the horizontal axis with corresponding weights. Each material is represented by a divided bar showing the contribution of each lifecycle phase to environmental burdens.

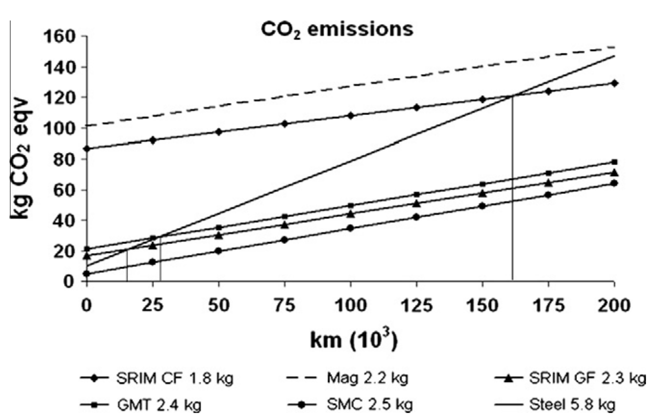


**Fig. 12.** Environmental breakeven analysis. Emissions in $CO_2$ eqv and vehicle distance are shown on the vertical and horizontal axes respectively.

generation during magnesium production contributes to 95% of these emissions. For the carbon fibre SRIM component (Fig. 13c) the production of the carbon fibres was responsible for approximately 82% of emissions with the next largest contribution coming from the preforming operation during the production process. This large contribution is the result of the energy intensive nature of the carbon fibre production process. The combined materials and manufacture phases of the steel and SMC components were approximately 90% and 95% lower in emissions respectively than for the magnesium component. Material production dominated emissions for the steel scenario, carbon fibre SRIM and magnesium components. Materials accounted for only 34% of emissions for the SMC component raising the relevance of transportation and manufacturing processes.

## 5. Discussion

In this work, manufacturing focused life cycle cost modelling was used in parallel with life cycle assessment to analyse the economic and environmental effects of substituting steel with lighter weight materials in an automotive application.

For this application, all lightweight components were found to be more costly to produce than their steel counterpart. The longer production cycle times and material costs were found to have the greatest influence on the production cost. In addition, component costs increased with weight saving potential with the lightest weight components incurring the highest costs. The higher component costs offset some of the benefits obtained through light weighting during the vehicle use phase to some extent, thus influencing the cost breakeven distance for each scenario. However, all lightweight scenarios still achieved total cost reduction over the complete life cycle. Surprisingly, the SMC component, which was also the heaviest, achieved the lowest life cycle cost and earliest break even of all the lightweight scenarios. The low manufacturing cost meant that advantages were seen early on in the vehicle life, benefiting the consumer with reduced running costs with a limited penalty to the manufacturer. On the other hand the lowest weight carbon fibre SRIM component achieved the lowest use phase cost, but incurred the highest component cost. The overall cost benefit of using such a material therefore became negligible as the benefits occurred towards the end of the vehicle's life. Higher and lower fuel cost scenarios reduced and increased breakeven distances

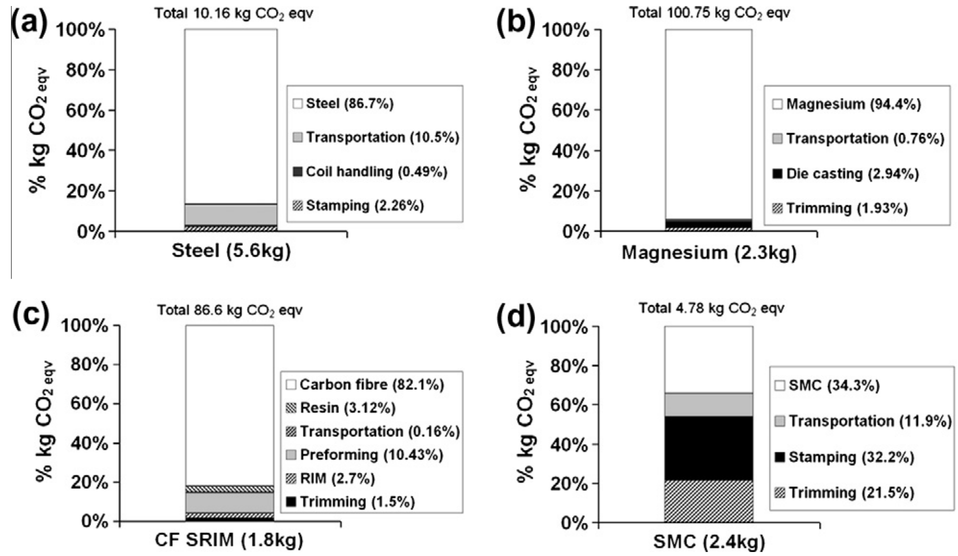


**Fig. 13.** (a–d) Contributions of materials and processes to $CO_2$ equivalent emissions from component manufacture.

respectively, with the latter pushing the breakeven point of the carbon fibre beyond the vehicle's life.

In the previous article by Song et al. [28], the life cycle energy content of a glass reinforced polyester composite was analysed. They concluded that the use phase is dominant in transport applications, therefore lighter weight materials should be favoured to minimise life cycle energy use. However, the analysis of several materials in this study has shown that weight saving is not a reliable single indicator of improved environmental performance. The environmental performance of the lightweight components were strongly influenced by the materials employed. The use of higher performance materials resulted in increased environmental burdens coming from the earlier life cycle phases for all of the components with the exception of SMC. The increased emissions from these early phases limited the effectiveness of light weighting to varying degrees. For instance, environmental impacts from the materials and manufacturing phases of the magnesium component were so large that they outweighed savings obtained from light weighting, resulting in a net increase in environmental impacts. This effect was also seen for the carbon fibre SRIM component, where only a 13% reduction of life cycle emissions were obtained despite superior light weighting capability.

The main contributions to environmental burdens for the composite components were from the production of raw materials (resins and fibres), which was particularly high for carbon fibres due to the energy requirements for their production [55]. The manufacturing or conversion phases were also more energy intensive than for steel due to the heat requirements of the curing and forming processes together with longer cycle times, thus resulting in greater environmental impacts and costs. Substituting steel for any of the lighter weight materials would result in the manufacturing process becoming more dominant in the product life cycle, emphasising the need for reliable LCI production data to better support such studies. Environmental burdens associated with the end of life of the composite components were very small and contributed to less than 1% of the life cycle. Incineration with energy reclamation was assumed for the disposal of the composites, thus enabling an energy credit to be allocated to the product system. Although incineration is a valid disposal route for ASR today; the proportion of a vehicle, which may be disposed of in this way in the future, is limited by the EU ELV directive [57]. As a consequence the directive effectively blocks the widespread implementation of composites in automotive applications unless valid recycling routes are implemented.

The approach undertaken in this study has been effective in quantifying the environmental impacts and costs related to each life cycle phase of a component in an automotive application. The shifting of impacts and costs between life cycle phases was identified as an important effect, thus highlighting the need for a life cycle approach during materials assessment. The possibility to assess impacts across multiple impact categories was also considered advantageous as it enabled shifts between categories to also be observed and enabled the total impact of the product to be studied rather than its effect in a specific area such as climate change.

The focus of the study was to determine the effects of material substitution at the component level. This level of study was considered appropriate as substitution in the automotive sector is more likely to occur gradually with specific materials being chosen for specific applications and benefits measured. However, the methodology is not restricted to this level and may be applied to larger assembly systems and complete vehicles if the appropriate data are available. This higher-level approach would be recommended to determine the net effects of larger scale materials substitution as detail may be lost through scaling results to other larger components.

Although the methodology is relatively well established, the quality of results obtained from such studies relies strongly upon the input data available and it is essential to ensure that data are current and representative of technologies assessed, or results could be strongly influenced. For instance, inventory data for magnesium production in the Ecoinvent database consider electrolysis production in Norway using Sulphur Hexafluoride ($SF_6$) as a cover gas for the process [27]. However, since the generation of this data, production of magnesium in Norway has ceased and China now dominates world production with the Pigeon process, using sulphur for melt protection. Data for Chinese production were sourced from literature to provide a more relevant picture of the environmental impacts today. However if Ecoinvent data were used, the results would be very different, showing 160% more $CO_2$ eqv emissions for the same component.

The same applies to data for composites. Here three sources of inventory data were considered for the manufacture of carbon fibres: existing data available in the Simapro software from the

IDEMAT LCI data base, more recent data used by Duflou et al. [23] in 2009, and data from the Japan Carbon Fibre Manufacturers Association (JCMA) [46]. The three sources provided different emission quantities for the production of 1 kg of carbon fibre, which were 12 kg 47 kg and 63 kg of $CO_2$ eqv respectively. The data used in this study were from the Japan Carbon Fibre Manufacturers Association (JCMA) as it was generated recently by industry [48]. Data used by Duflou et al. [23] employed a power mix that was not representative of the locations of European carbon fibre manufacture, while the IDEMAT LCI data were based upon publications produced in 1991 and 1995 and considered out of date. In this study the break-even of the carbon fibre component occurred towards the end of the vehicle's life at 162,000 km, while Duflou et al. found the figure to be 132,000 km [23]. The different datasets used between these two studies can explain the differing environmental breakeven points and highlight the need for more reliable and well documented data for carbon fibre manufacture. Although databases such as Ecoinvent are considered reliable sources for environmental LCI, they contain little data related to composite materials and processes, thus LCA practitioners are forced to obtain data from other sources which can lead to inconsistencies between such studies. Further work from the composites industry is needed to provide reliable and well documented LCI data which are key to ensuring comparable and reliable representation of composites in future studies, particularly as environmental performance becomes more relevant.

## 6. Conclusions

The results of this study have indicated that automotive component weight reduction will not always lead to improved environmental performance. Materials offering higher weight savings have been shown to give limited or negative benefits over their life cycle due to increased environmental burdens associated with their production.

In this study an SMC automotive component out performed other lighter weight materials in terms of life cycle cost and environmental performance. Although it did not achieve the same levels of weight reduction as other higher performance materials, the lower impacts and costs associated with its materials and manufacturing phases meant that it still outperformed the alternatives over the whole life cycle. Composites have been shown to display clear advantages in reducing impacts from vehicles, despite not being recycled, as the benefits of light weighting still outweigh any potential benefits from recycling.

The current requirement for automotive manufacturers is to reduce use phase emissions and to increase recycling at the end of life. This study has identified that these two priorities may not be sufficient by themselves to build a strategy for more environmentally acceptable transportation and has highlighted that an overall vision of the whole life cycle is important to build up such a strategy.

The combination of LCA with LCC has also been shown to be a worthwhile approach for comparing different materials in terms of cost and environmental impact to support materials selection and enabling the best trade off between cost and environment. Economic and environmental hotspots may be identified within single products, enabling focused improvement strategies. The effectiveness of such an approach is reliant upon data availability and it is imperative that reliable inventory data concerning materials and manufacturing processes, particularly for the composite materials industry, become more available.

## Acknowledgments

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## Appendix A. Manufacturing plant diagrams and tables of costs

### A.1. Steel scenario

See Fig. A1 and Table A1

### A.2. Magnesium

See Fig. A2 and Table A2

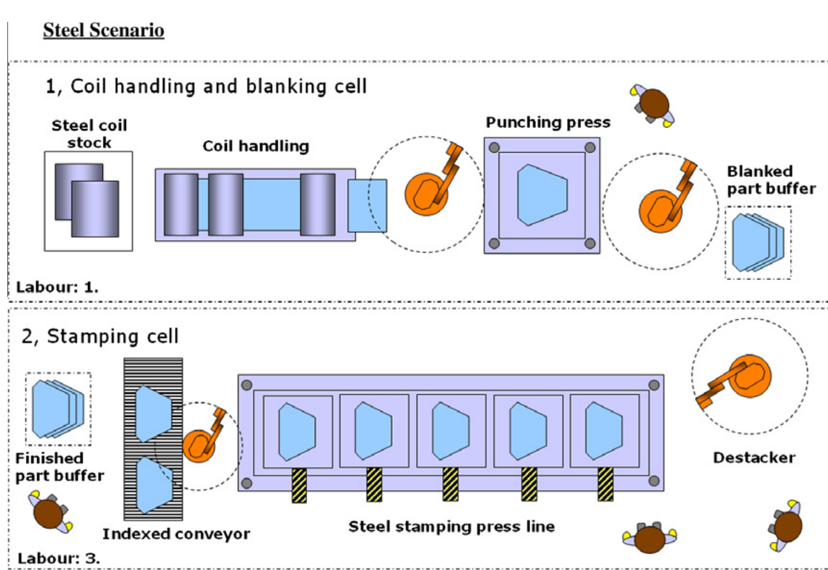


**Fig. A1.** The manufacturing facility considered consists of two cells comprising of a coil handling/blanking cell and a transfer pressing cell. Coils arriving at the manufacturing facility are transferred to the handling/blanking cell where material is de-coiled and cut, the cut material is then transferred to a blanking press. Once formed, the blanks are removed from the press by robot and placed in a buffer. A de-stacker is then used to feed the press line in the second cell, where a series of forming dies complete the part shape. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

**Table A1**
Manufacturing cell costs, amortization types and material costs used for cost analysis of the steel component.

| Item | Cost (k€) | Cycle time (s) | Amortisation |
|---|---|---|---|
| Cell 1 | 3750 [1,2] | 2 | Utilisation |
| Cell 2 | 9350 [1,2] | 2 | Utilisation |
| Steel coil | 0.63 (€/kg) | – | – |
| Steel scrap (new) | 0.16 (€/kg) | – | – |

**Table A2**
Manufacturing cell costs, amortization types and material costs used for cost analysis of the magnesium component.

| Item | Cost (k€) | Cycle time (s) | Amortisation |
|---|---|---|---|
| Cell 1 | 3750 [1,2] | 110 | Utilisation |
| Cell 2 | 325 [1,2] | 65 | Utilisation |
| Cell 3 | 415 [1,2] | 65 | Utilisation |
| Magnesium alloy | 6 (€/kg) | – | – |

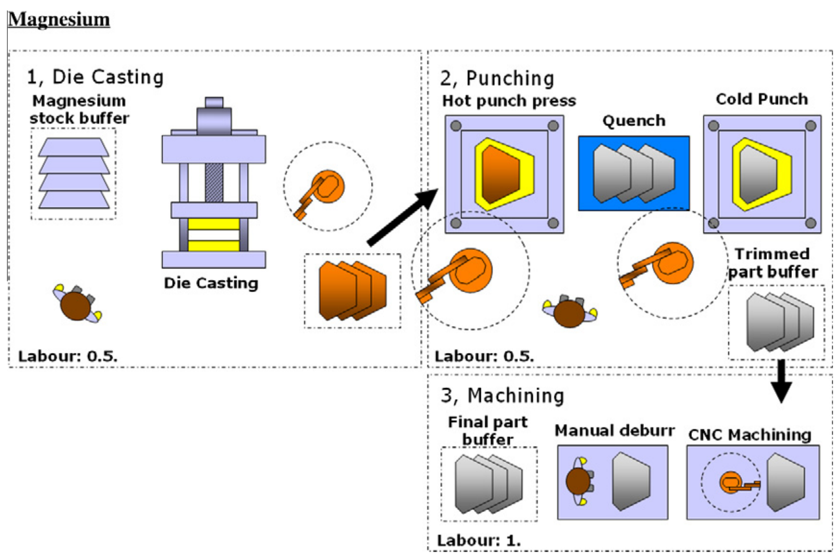


**Fig. A2.** Magnesium ingots are received and enter the casting cycle by a machine fed system where they are melted and injected under high pressure, which is maintained until the magnesium has solidified. After solidification the die is opened and the casting is removed by robot and placed in a buffer. While still hot the component is then punched to remove gates, sprues and flash, then quenched. A robot then moves the part to a cold press where the remainder of the casting excess is removed. Finally a robot transfers the components to the second cell where a final Computer Numerically Controlled (CNC) machining operation takes place. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

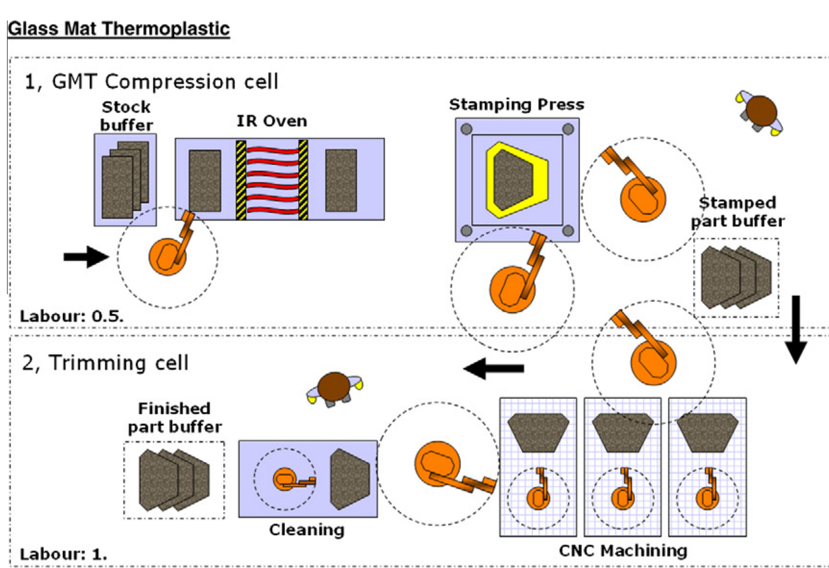


**Fig. A3.** The GMT is supplied as a raw material in rigid partially consolidated sheets. Within the first cell the sheet material is transferred by robot to an infrared preheating oven, once at the required temperature the blanks are transferred again to a moulding cell where the blank is automatically placed into a compression mould and pressed. When sufficiently cooled to retain structural integrity the mould is opened and the component is transferred by robot to an automated trimming cell. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

**Table A3**
Manufacturing cell costs, amortization types and material costs used for cost analysis of the GMT component.

| Item | Cost (k€) | Cycle time (s) | Amortisation |
|---|---|---|---|
| Cell 1 | 1554 [1,2] | 60 | Utilisation |
| Cell 2 | 1003 [1,2] | 60 | Utilisation |
| GMT | 5.50 (€/kg) | – | – |

**Table A4**
Manufacturing cell costs, amortization types and material costs used for cost analysis of the SMC component.

| Item | Cost (k€) | Cycle time (s) | Amortisation |
|---|---|---|---|
| Cell 1 | 698 [1,2] | 70 | Utilisation |
| Cell 2 | 993 [1,2] | 70 | Utilisation |
| SMC | 1.60 (€/kg) | – | – |

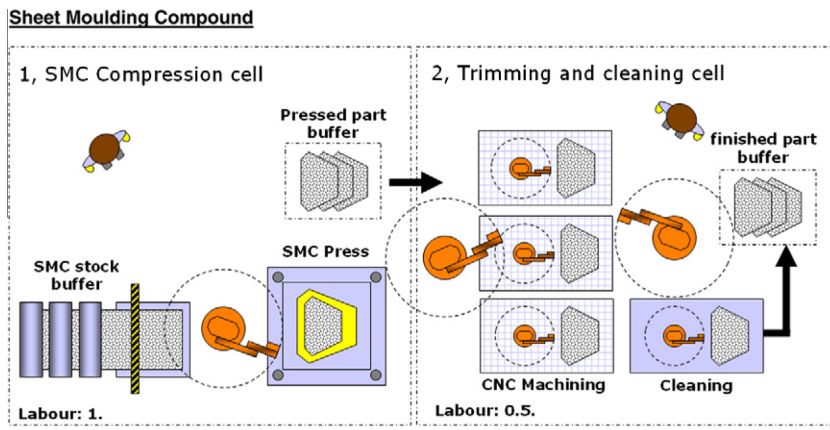


**Fig. A4.** Once inside the facility the uncured material enters the first cell where it is cut into blanks of the required charge quantity. A robot then transfers the charge to a heated steel tool which is closed by a hydraulic press and remains closed until curing is complete. When the tool is opened the cured component is transferred by robot to the second cell where an automated process removes excess trim, CNC machines are employed to carry out final machining operations. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

## A.3. Glass Mat Thermoplastic

See Fig. A3 and Table A3

## A.4. Sheet Moulding Compound

See Fig. A4 and Table A4

**Table A5**
Manufacturing cell costs, amortisation types and material costs used for cost analysis of the SRIM components.

| Item | Cost (k€) | Cycle time (s) | Amortisation |
|---|---|---|---|
| Cell 1 | 1033 [1,2] | 210 | Utilisation |
| Cell 2 | 9350 [1,2] | 120 | Utilisation |
| Cell 3 | 9350 [1,2] | 175 | Utilisation |
| Resin | 4.00 (€/kg) | – | – |
| Carbon fibre | 12.00 (€/kg) | – | – |
| Glass fibre | 1.6 (€/kg) | – | – |

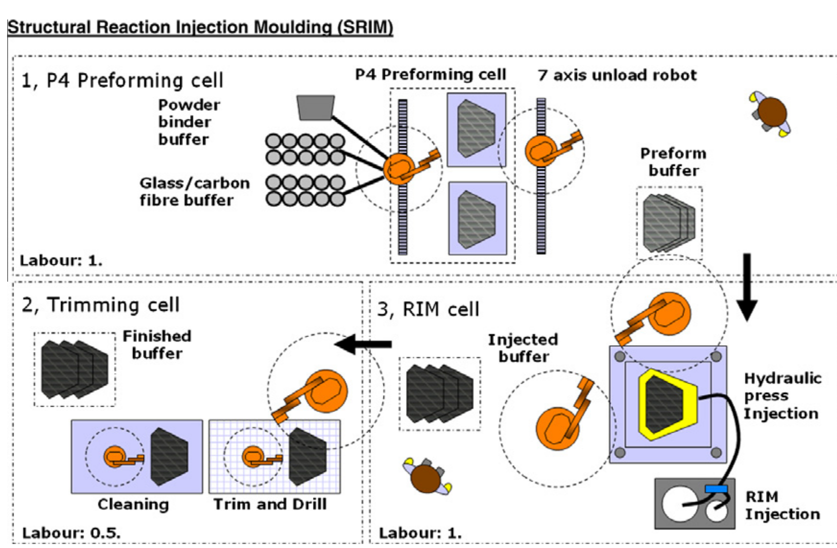


**Fig. A5.** Within the first production cell a dry fibre preform is produced with the P4 process, this process utilises a dry roving type material, which is fed through a robot mounted chopper gun and deposited with a binder into a heated mould. The preform is then transferred to an injection cell by robot where it is placed in an injection mould; polyurethane resin is then delivered at high speed. Once cured, the press is opened; a robot then transfers the component to a trimming cell where it is de-flashed and trimmed. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

### A.5. Structural Reaction Injection Moulding (SRIM)

See Fig. A5 and Table A5

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