Supplementary information for

System understanding shapes insights for eco-design

a comparison of four temporal perspectives

Authors: Thomas Arblaster*[1], Jeroen Guinée[1], Carlos Felipe Blanco Rocha[1,2], Ivana Burzic[3], Claudia Pretschuh[3], Fruela Pérez Sánchez[4], Nils Thonemann[1]

[1] Institute of Environmental Sciences (CML), Leiden University, Leiden, the Netherlands

[2] Circularity and Sustainability Impact Group, Netherlands Organization for Applied Scientific Research (TNO), Utrecht, the Netherlands

[3] Wood K Plus – Kompetenzzentrum Holz GmbH, Linz, Austria

[4] Instituto Tecnológico del Embalaje, Transporte y Logística (ITENE), Valencia, Spain

* **Corresponding author.** Email: t.p.s.arblaster@cml.leidenuniv.nl

Contents

List of Figures

List of Tables

S1 Supplementary materials and methods

S1.1 On the case of vehicle eco-design

Passenger vehicles impact a variety of environmental areas of concern. Historically, this is first and foremost tied to the use of fossil fuels in internal combustion engines. Following the emergence of battery electric vehicles, the impacts associated with driving a given distance are reduced considerably, now following to a larger degree from the production and manufacturing of the vehicle itself (Negri & Bieker, 2025). Thereby, the focus of vehicle eco-design increasingly shifts towards strategies like the use of bio-based materials or the implementation of resource life extension strategies.

Vehicles are long-lived products. According to Held et al. (2021), the average lifetime of passenger cars varies across European countries from 8.0 to 35.1 years, with a mean of 21.8 years. These lifetimes influence the use-phase impact. Consider, for example, the electricity mix powering electrified vehicles changing from one moment to another, or internal combustion engines consuming an evolving share of biofuels from year to year. Vehicle lifespan also has clear implications for the relevance of resource life extension strategies, affecting how materials accumulate in society, and subsequently, what secondary materials become available when (cf. Sazdovski et al., 2022).

The temporal characteristic of these considerations illustrate why a case study in vehicle eco-design could reasonably demonstrate the added value of time-explicit LCA. The case study we examine here concerns interior trims of light-duty passenger cars. The aim is to assess how the lifecycle impacts of such components could be reduced, focusing on the possible substitution of the incumbent talc-polypropylene compound with a novel softwood-polypropylene compound. This case study is detailed further in S1.1.2, following an overview of previous work in this context in S1.1.1.

S1.1.1 Previous work

There are several cradle-to-grave studies relevant to the case assessed here (i.e., relating to polymer composite use in vehicles), such as the work of Corbière-Nicollier et al. (2001), Luz et al. (2010), Roy et al. (2020), Tadele et al. (2020), and Zah et al. (2007). Typically, such studies include a representation of in-use mass-induced energy use to reflect component mass. Cradle-to-gate studies are sometimes framed as enabling an application-agnostic assessment (see, e.g., Hesser, 2015; Xu et al., 2008), although these of course also neglect end-of-life processes.

Previous studies are generally positive about the potential of bio-based fillers to reduce the impacts associated with their application in road transport, especially when this enables lightweighting (see Luz et al., 2010; Roy et al., 2020; Tadele et al., 2020; Wasti et al., 2024). Another recurring motif is an impact reduction realised by reducing the required volume of matrix material (see Corbière-Nicollier et al., 2001; Xu et al., 2008; Zah et al., 2007). Naturally, it is acknowledged that these factors are subject to the performance requirements of the application at hand.

Several prospective elements can be found in the literature, particularly when it comes to imagining the commercial deployment of a new material (see Corbière-Nicollier et al., 2001; Hesser, 2015), large-scale recycling (something which Civancik-Uslu et al. (2018) highlight in their review), or other novel technologies (see Zah et al., 2007). However, each of these studies develops temporally static product systems. We have found no previous studies in this area which explicitly reflect a change in the system over time – although there are such examples regarding other case studies, such as Abu-Ghaida et al. (2025), de Zilva et al. (2026), and Šimaitis et al. (2023).

S1.1.2 Vehicle component case study

In the automotive sector, polypropylene thermoplastic compounds are mainly used for interior, exterior, and under-the-bonnet applications. While mostly primary material is used, this can already be combined with some post-consumer recycled polypropylene, which is derived from household waste. We assume these materials as developed and produced in Europe (Borealis AG, 2014).

For interior applications, polypropylene reinforced for 13% (by mass) with a talc filler is currently a common compound. The processing is carried out via standard injection moulding, a high-volume manufacturing process, to obtain formed parts like dashboards or interior trims.

Research has identified fillers derived biological sources (such as wood, natural fibres, or cellulose) which have similar reinforcing effects to talc when compounded with polypropylene (see, e.g., Sobczak et al., 2012). Such compounds are referred to as biocomposites. Whether the component performs its functions well can be determined by conducting a variety of technical tests. The filler content of the compound also affects this performance to some degree. Here, we assume the in-service functionality of each of the assessed components to be equivalent. Following this assumption, the functional unit of the assessment is defined, as detailed in 2.2.1.

In addition, due to the low material density of wood and cellulose, they contribute to the lightweighting of automotive parts, which is a key eco-design approach of the sector. Wood fibres, as derived in Europe as by-products from the woodworking industry, are suitable for this approach and have been developed for selected automotive components (Josef Rettenmaier & Söhne GmbH + Co. KG, n.d.). There are clear environmental risks associated with talc mining (Pavolová et al., 2026). The respiratory toxicity of talc and its accessory minerals is an additional risk for people who work with the material (Johnson, 2021). The investigation of wood fibres as an alternative to talc could be step to mitigating such impacts on humans and the environment.

Here, we treat this case study in a way that reflects the early stages of the innovation process. Not all data necessary to construct a detailed lifecycle have been collected, so the use of LCA at this stage is to form an initial understanding of the product systems. This informs how to refine the data collection and can reveal novel avenues of investigation. The comparative performance of alternatives is relevant, but subject to much uncertainty.

At the stage of the development process considered here, the assumption is made that the novel component has the same volume as the incumbent reference component – thereby making use of the same injection moulds which are already in use. Furthermore, in the development of this novel compound, the desired quantity of filler content is to be determined. To this end, an analysis is performed across a range of possible values for this filler content, still assuming a constant component volume. As the filler has a higher density than the matrix material, a lower filler content results in a lower component mass. Thereby, not only the processes concerning the composition of the compound directly are affected (e.g., compounding, incineration), but also its use, through a change in mass-induced energy use. At this stage, we therefore consider a range of filler mass fraction from 10% to 20%. A value of 14% is used in the generation of results which do not consider a range for the filler content of softwood-polypropylene. The talc-polypropylene compound has a filler content of 13% by mass.

S1.2 Inventory analysis

S1.2.1 Foreground and background temporality

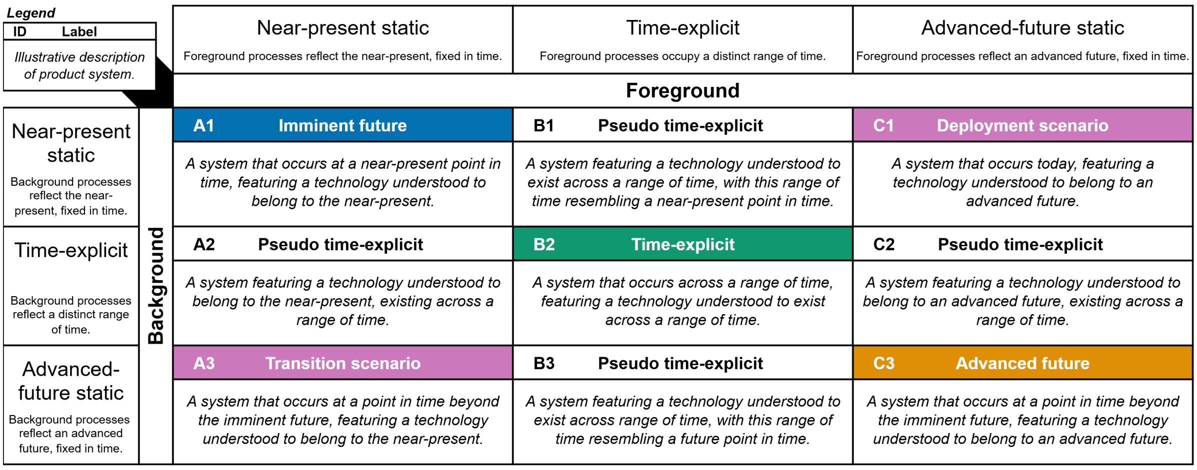
Here, we conceptually distinguish the specification of an activity to a particular time into either 'foreground' temporality or 'background' temporality. This follows from the established distinction between foreground and background processes, understood here as the distinction between the use of generic databases (the background) and the creation of case-specific processes (the foreground) (Guinée et al., 2002). There is no computational distinction between the background and foreground, but due to the differences in practise (how data is collected, reported, used, etc.), we consider it useful to define and treat these terms separately.

We distinguish between the temporal differentiation of the foreground and that of the background. Arvidsson et al. (2018) highlight the value of adapting a background database in order to align it with the study's temporal scope, but at the time this was uncommon or limited to select interventions (e.g., switching some fossil-based power to a renewable source). In recent years, the use of prospective background databases has become more common, with tools such as premise (Sacchi et al., 2022) and fossil-free ecoinvent (Schlesier et al., 2026) (see S1.2.6). Van der Hulst et al. (2020) use the term 'external developments' for this aspect of assessing a technology's future environmental performance (p. 1239). By instead referring to this as 'background temporality', we stress that we define this aspect solely with respect to the (modified) background database(s), regardless of whether this reflects developments 'internal' to the technology at hand.

Foreground temporality, then, is any other way in which an activity is temporally differentiated (see 2.2.3). The consideration of future changes specific to the technology at hand is most prevalent here, as is often the uncertainty in the exploration of possible futures (see, e.g., Blanco et al., 2025).

In 2.1, we only discuss temporal set-ups which are aligned in their perspective on the foreground and background. These static perspectives align with A1 and C3 of Figure S1. Both mosaic time-explicit and metabolic time-explicit perspectives can be understood as falling under B2: both represent temporal evolution in the foreground and background of the system across a range of time.

The distinction between mosaic and metabolic approaches cannot be defined as a particular degree



Legend

ID Label

Illustrative description of product system.

Background \ Foreground	Near-present static Foreground processes reflect the near-present, fixed in time.	Time-explicit Foreground processes occupy a distinct range of time.	Advanced-future static Foreground processes reflect an advanced future, fixed in time.
Near-present static Background processes reflect the near-present, fixed in time.	**A1 Imminent future** *A system that occurs at a near-present point in time, featuring a technology understood to belong to the near-present.*	**B1 Pseudo time-explicit** *A system featuring a technology understood to exist across a range of time, with this range of time resembling a near-present point in time.*	**C1 Deployment scenario** *A system that occurs today, featuring a technology understood to belong to an advanced future.*
Time-explicit Background processes reflect a distinct range of time.	**A2 Pseudo time-explicit** *A system featuring a technology understood to belong to the near-present, existing across a range of time.*	**B2 Time-explicit** *A system that occurs across a range of time, featuring a technology understood to exist across a range of time.*	**C2 Pseudo time-explicit** *A system featuring a technology understood to belong to an advanced future, existing across a range of time.*
Advanced-future static Background processes reflect an advanced future, fixed in time.	**A3 Transition scenario** *A system that occurs at a point in time beyond the imminent future, featuring a technology understood to belong to the near-present.*	**B3 Pseudo time-explicit** *A system featuring a technology understood to exist across range of time, with this range of time resembling a future point in time.*	**C3 Advanced future** *A system that occurs at a point in time beyond the imminent future, featuring a technology understood to belong to an advanced future.*

Figure S1: Overview of temporal set-ups obtained when combining three temporal perspectives (near-present static, time-explicit, and advanced-future static) across the foreground and background of a product system. Each set-up is given a label (e.g., 'Imminent future') and a unique identifier (e.g., 'A1'), as well as an illustrative description of a product system which embodies this representation of time in an application of LCA. These illustrative descriptions reflect the use of LCA to assess a specific technology, with foreground processes being those 'internal' to this technology (cf. van der Hulst et al., 2020), but the same general differentiation can be made for other applications and other understandings of foreground and background. Note that we consider the 'imminent future' as being included in the 'near present'.

of temporal evolution or whether or not temporal differentiation takes place. As explained in 2.1.2, these labels reflect how the system is modelled to represent particular conceptions of temporal evolution. This dimension is not considered in Figure S1.

Besides the four temporal set-ups of the main text, we present results for additional set-ups in S2.3. These provide further insight on how time-related choices affect the results and how results compare in the absence of particular tools (e.g., without prospective background databases). The application of these (additional) temporal set-ups is explained further in S1.2.2.

S1.2.2 Product system diagrams

As described in 2.1, metabolic time-explicit LCA leads to product systems of which a given element (e.g., use of the product) cannot be described by a single unit process in a time-explicit technology matrix. In Figures S2 and S3, we illustrate this by depicting the product system in terms of its connected elements while also indicating how some of these connecting flows do not have a set value, but one which depends on time at which this element occurs. As the affected elements occur across a range of time, these changing values lead to the necessity to represent them as discrete instances when performing LCA calculations.

The representation in Figures S2 and S3 is not entire accurate for the use phase ('Use of … interior trim'), as this process has a duration. The lightweighting that occurs in a particular year (Figure S3F), for example, is not that of a single unit process as it would occur in a disaggregated technology matrix, but the sum of all use processes which occur in that year. This reflects the way in which we implement this disaggregation, by modelling the duration of use not as many individual processes which have the appropriate duration (as would occur in a technology matrix as described above), but instead reflecting this duration by modelling a sequence of aggregated processes which each represent a duration of one year. This 'building block' approach is documented in our code repository (Arblaster et al., 2026). This is why panels C, D, E, and F in Figures S2 and S3 are shown with respect to the reference flow: these processes only come into being once they are appropriately scaled to the reference flow of the analysis.

A mosaic time-explicit approach treats these flows differently (see Figure 2), while treating other time-dependent processes as depicted here.

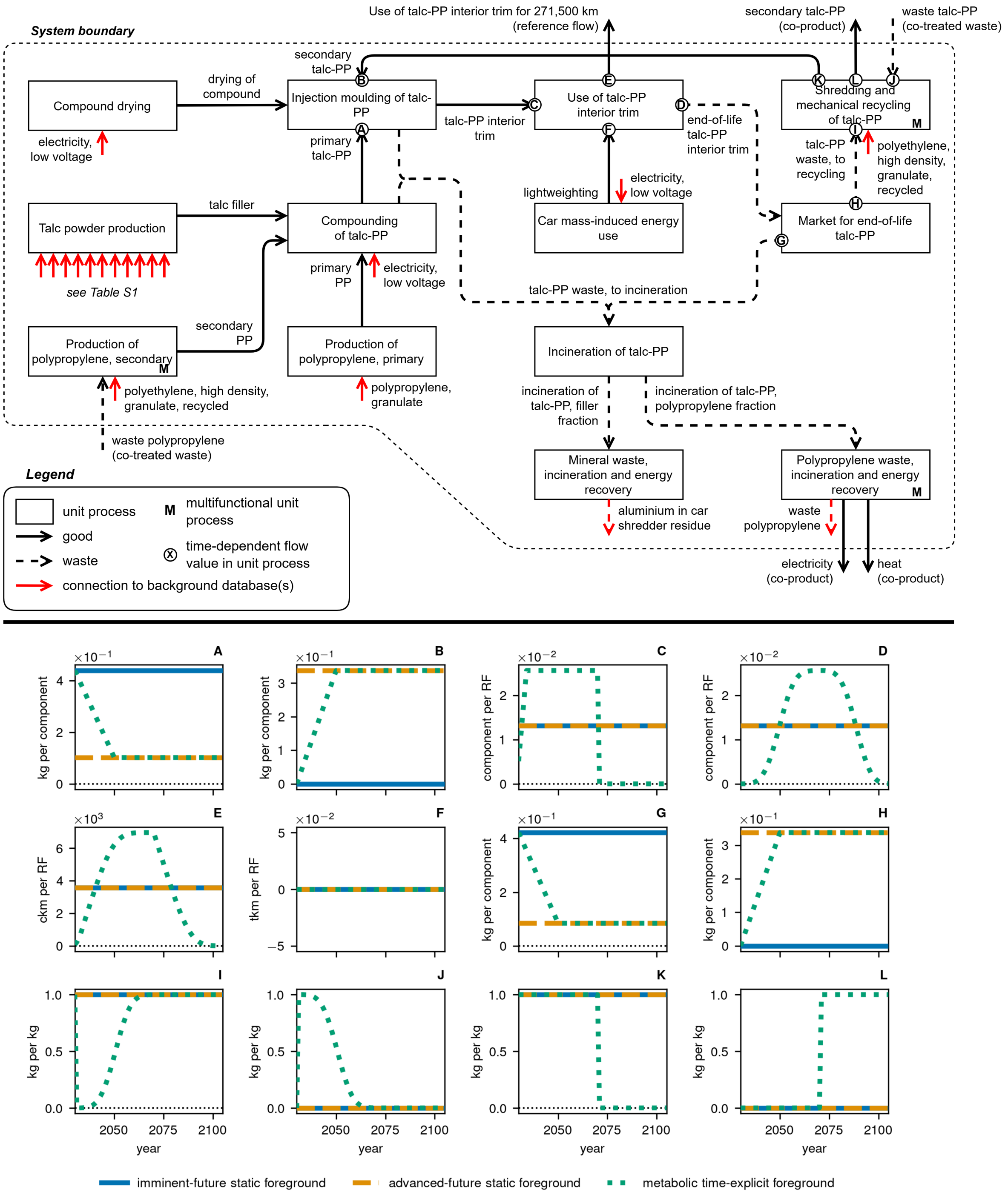


Figure S2: The diagram in the top half of the figure (above the bold horizontal line) represents the talc-polypropylene alternative. Temporally resolved economic flows are marked with a circle containing a letter. These letters each represent one of the panels in the bottom half of the figure (below the bold horizontal line). These panels depict the value of this economic flow in the labelled unit process according to a foreground system following each of the steady-state perspectives (imminent-future and advanced-future), as well as the metabolic time-explicit perspective. See S1.2.1 for the implementation of foreground vs. background temporality. Values are sometimes shown as they occur when scaled to the reference flow (RF) and otherwise with respect to another value, as indicated by the y-axis labels and explained further in S1.2.2.

This discretisation is also used to operationalise temporal set-ups that are temporally static or ‘pseudo

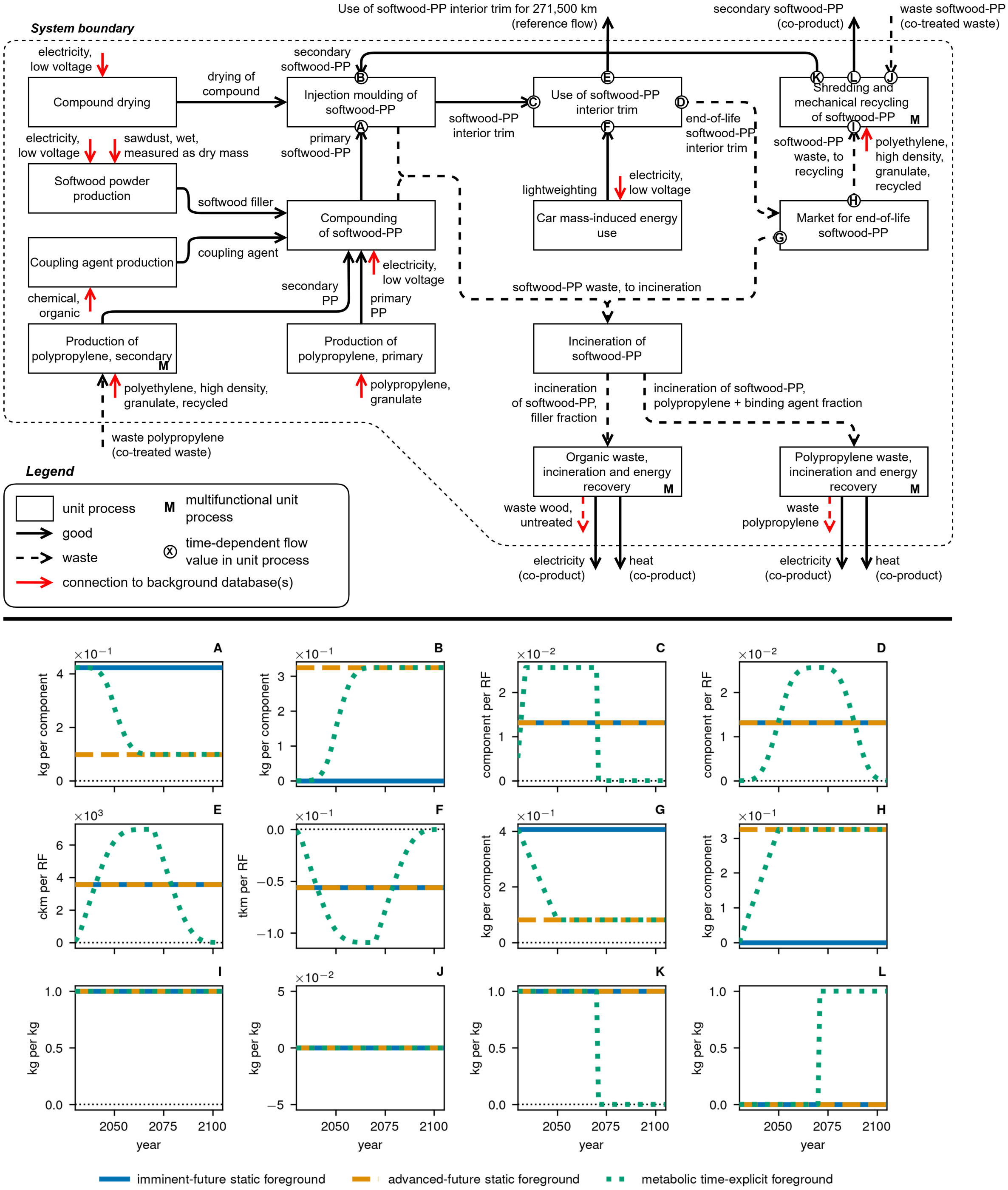


Figure S3: Product system diagram representing the softwood-polypropylene alternative (above bold horizontal line), with time-resolved value of economic flows (below bold horizontal line), using the same format as Figure S2.

time-explicit' (see S1.2.1). We model a temporally static foreground not as a single instance, but as many identical scaled-down copies spread over a timeline. This is also depicted in Figures S2 and S3. Similarly, a temporally static background is realised by connecting each timestep in the foreground timeline to the same background database. As a result, set-ups labelled B1 and B3 in Figure S1 create a changing timeline of foreground process, but connect each timestep to a single background database, while set-ups A2 and C2 use the same foreground data for each timestep, but still connect these to a variety of

Table S1: Input of economic flows for the production of 1 kg talc powder, based on the reporting of Tadele et al. (2020). All unit process data can also be obtained from our code repository.

Flow name	Process name	Amount	Unit
diesel, burned in building machine	diesel, burned in building machine	1.74×10^{-1}	megajoule
cement mortar	market for cement mortar	7.50×10^{-2}	kilogram
explosive, tovex	market for explosive, tovex	2.10×10^{-4}	kilogram
gravel, round	market for gravel, round	4.10×10^{-4}	kilogram
lubricating oil	market for lubricating oil	5.20×10^{-5}	kilogram
nylon 6-6	market for nylon 6-6	5.60×10^{-4}	kilogram
petrol, unleaded	market for petrol, unleaded	1.55×10^{-4}	kilogram
reinforcing steel	market for reinforcing steel	9.50×10^{-4}	kilogram
water, deionised	market for water, deionised	6.40×10^{-1}	kilogram
electricity, high voltage	market group for electricity, high voltage	1.57×10^{-1}	kilowatt hour
sawlog and veneer log, softwood, measured as solid wood under bark	softwood forestry, spruce, sustainable forest management	3.00×10^{-5}	cubic metre

background databases. Results for some of these hybrid set-ups are presented and discussed in S2.3.

The time-dependent flows relating to waste, recycling, and the share of secondary materials (panels A, B, and G through K in Figures S2 and S3) are all direct features of the coupled dynamic system (see S1.2.4). The disaggregation of whether secondary material comes from waste inside or outside of the system (panels I and J) and whether the recycled material is used within the system or not (panels K and L) is relevant to fully understanding the multifunctionality of the system (see S1.2.5). For both alternatives, when secondary material is used within the system, it is also used in the production of subsequent components (value of one in panel K). However, once no more components are manufactured, the secondary material can only leave the system boundary. Therefore, the value of panel K shifts to zero, with panel L seeing an accompanying shift to a value of one. For the softwood-polypropylene alternative, no outside waste is used (hence, panel I always shows a value of one and panel J a value of zero), while the talc-polypropylene alternative does have access to external waste sources.

S1.2.3 Time-independent foreground inventories

Here we describe aspects of the foreground inventories which stay static of time. These inventories form the product systems, as illustrated in Figure S2 and Figure S3 for the talc-polypropylene and softwood-polypropylene systems, respectively. However, depending on the temporal set-up, various time-dependent changes are made to how these static inventories contribute to the overall product system.

Based on industry estimates, we assume that the production of primary compound uses a matrix material that is 70% primary polypropylene and 30% secondary polypropylene. These are respectively modelled with the 'polypropylene production, granulate' and 'polyethylene, high density, granulate, recycled' background processes.

The production of the talc filler is based on the reporting of Tadele et al. (2020), who adapt the data of Badino et al. (1995) for use with ecoinvent (see Table S1). Note that the cradle-to-gate impacts following from this unit process show considerable deviations from more recent industry reporting (e.g., Imerys, 2023), indicating that further refinement of this data is warranted in order to achieve a detailed assessment.

Here, we assume that a ‘market for sawdust, wet, measured as dry mass’ background process is representative of the softwood filler precursor. This unit process represents the collection and transport of sawdust from a variety of sources, primarily as a co-product of sawing. The partitioning of the sawing processes attributes most of its burdens to the production of sawnwood, ostensibly following economic allocation. This multifunctionality could also be resolved differently (cf. S1.2.5), although we do not quantify this here. A relatively uniform filler material is obtained from the heterogeneous sawdust. We model this process as an electricity input of 0.4 kWh per kg; no material losses are considered here. Unlike talc-polypropylene, the formulation of softwood-polypropylene uses a small amount (3% by mass) of coupling agent, for example, polypropylene functionalised with maleic anhydride. We model this with a generic ‘market for chemical, organic’ background process, under the assumption that its small mass share translates to a low relevance to this early-stage assessment.

The filler content is as described in S1.1.2: 13% and 14% by mass for talc-polypropylene and softwood-polypropylene respectively. These ingredients are then compounded and dried, which is stylised as electricity consumption: 0.2 kWh per kg for compounding, with an additional 0.25 and 0.5 kWh per kg for drying talc-polypropylene and softwood-polypropylene, respectively. We assume the material loss here to be negligibly small. Next, components are manufactured with injection moulding, modelled as the consumption of electricity (5 kWh per 1 kg component) and a material efficiency (4% mass loss). For these steps, demands for transport, machinery, additional consumables, etc. are cut off.

The representation of the service life includes the effect of lightweighting on energy consumption. Given the prominence of battery-electric vehicles as a solution to decarbonising road transport (see, e.g., European Commission, 2020), we make the simplification that the components at hand are exclusively used in battery-electric vehicles. We assume that a reduction in mass of 100 kg results in a reduction in energy consumption of 0.6 kWh for every 100 km driven. This is in line with the ranges reported by Weiss et al. (2020) and Geyer and Malen (2020) for this same metric. We model this energy saving in the lighter of the two alternatives as negative energy consumption: the softwood-polypropylene system subtracts 0.06 kWh for every tonne-kilometre reduction realised when compared to the mass of talc-polypropylene (note that use intensity and lifetime are assumed to be equal for both alternatives, as described in 2.2.1). Of course, there is no such thing as ‘negative energy’ and this approach neglects the rebound effect associated with efficiency improvements (see Font Vivanco et al., 2015), but such a representation of lightweighting is typical for vehicle eco-design (Koffler & Rohde-Brandenburger, 2010).

The use of secondary polypropylene – and sometimes, the recycling of end-of-life compound (see S1.2.4) – is based on mechanical recycling. The non-functional flows of mechanical recycling are here modelled as the ecoinvent process ‘market for polyethylene, high density, granulate, recycled’. Because of the cut-off system model used, this process incurs all the demands of obtaining secondary granulate from post-consumer waste. However, it’s not an exact fit for what it represents here: besides representing high-density polyethylene instead of polypropylene, its upstream processes reflect that some of the material collected for recycling ends up being incinerated instead. This second aspect is appropriate when it represents secondary polypropylene entering the product system, but not when it represents the closed-loop recycling of compound, as the incineration involved here is already accounted for separately (see S1.2.4). This therefore leads to the double counting of some incineration, which is most notable for temporal set-ups which feature high quantities of secondary material.

Aside from end-of-life recycling, all waste generated from a foreground process is incinerated, recovering some heat and electricity. For convenience, the incineration of composite compounds is virtually split into the incineration of its constituent materials. The energy recovered from each material is listed

Table S2: Overview of energy recovered from waste incineration, based on Haupt et al. (2018), using the proxy materials given here. In each case, we consider the archetype of municipal solid waste incineration 'combined heat and power, low efficiency'.

Incinerated material	Proxy material	Electricity recovery [kWh per kg]	Heat recovery [MJ per kg]
polypropylene	mixed plastics	1.45	8.31
coupling agent	mixed plastics	1.45	8.31
softwood filler	cardboard	7.50×10^{-1}	4.30
talc filler	aluminum, ferrous metals and tinplate	0	0

in Table S2 (Haupt et al., 2018). Each incineration process is multifunctional (see S1.2.5), except for the incineration of talc, from which no energy is recovered. Background processes are used to model the non-functional flows of emissions (see Figure S2 and Figure S3).

The investigation into the mass fraction of softwood (from 10 to 20%) is enabled by parameterising each foreground activity which is affected by this variable: the quantities of filler and matrix materials required for compounding, the mass flows involved in the injection moulding of one component, the quantity of incinerated materials, and of course the in-use mass-induced energy consumption. The results of this exercise are presented in S2.2.

S1.2.4 Dynamic modelling

Here, we precisely describe the creation of time-dependent inventories, which were introduced in 2.2.3 and illustrated in S1.2.2.

The share of end-of-life compound turned into secondary compound through mechanical recycling increases linearly over time, as described with Equation 1. For both talc-polypropylene and softwood-polypropylene, we assume that this goes from 0% in 2030 ($y_0 = 0$; $t_0 = 2030$) to 80% in 2050 ($y_1 = 0.8$; $t_1 = 2050$). In our modelling, this value is not how much of the material is destined for recycling, but how much becomes secondary material. Therefore, the flow of end-of-life waste to incineration includes all of the material not recovered as secondary material, whether it has been collected for recycling or not. This leads to some contradictions in the recycling inventories used, as described in S1.2.3.

$$f(t) = \begin{cases} y_0, & t \leq t_0 \\ y_0 + (y_1 - y_0)\frac{t-t_0}{t_1-t_0}, & t_0 < t < t_1 \\ y_1, & t \geq t_1 \end{cases} \tag{1}$$

As described in 2.2.1 and 2.2.3, a dynamic stock-and-flow model is created to determine the provision of the metabolic time-explicit functional unit and its associated flows. The governing logic of this model is commonly used for dynamic material flow analysis (Müller et al., 2014) and is described by Equations 2, 3, and 4. This is a flow-driven model, meaning that it starts from a definition of the inflow (Equation 2).

The inflow (depicted in Figure 2 and panel C of Figures S2 and S3) also follows Equation 1, with the first non-zero inflow in 2030 ($y_0 = 0$; $t_0 = 2029$) and a stabilisation of the inflow by 2034 ($y_1 = 1$; $t_1 = 2034$). At this stage, the value at which the inflow stabilises is arbitrary, as it is later rescaled to align with the demand sum of the functional unit. Furthermore, the inflow stops in 2070 ($t_{\text{last inflow}} = 2070$), as motivated in 2.2.1.

$$Inflow(t) = \begin{cases} f(t), & t \leq t_{\text{last inflow}} \\ 0, & t > t_{\text{last inflow}} \end{cases} \tag{2}$$

Based on the inflow, the value of the stock at each timestep is determined, following Equation 3. The survival function (the complementary cumulative distribution function) used here is that of a Weibull distribution with shape parameter 3.5 and mean lifetime 18.1, as based on Held et al. (2021) (see 2.2.1).

$$Stock(t) = \sum_{c=t_0}^{t} Inflow(c) Survival(t - c + 0.5) \tag{3}$$

According to Equation 3, the stock is the sum of the surviving share of each cohort up to that point. Here, the survival function does use the cohort age $t - c$, as would be typical for a consistent treatment of the (in this case, yearly) timestep, but uses $t - c + 0.5$. This reflects the inflow being understood as occurring at the start of the year, while the stock is simulated at the year's midpoint. Because we use Equation 3 to calculate the use provided across a year, this shift avoids overestimating the surviving fraction of each cohort and is required to ensure that the use provided by a cohort is consistent with the mean lifetime of 18.1 years.

Following from the inflow and stock, the outflow is determined by mass balance (Equation 4). Because the stock of year t is understood as being evaluated in the year's midpoint, the same can be said of the outflow of year t.

$$Outflow(t) = Stock(t-1) + Inflow(t) - Stock(t) \tag{4}$$

For the talc-polypropylene product system, we assume that use of secondary compound increases at the same rate as the end-of-life recycling rate does, operating on the assumption that there are plenty of acceptable sources of talc-polypropylene waste which can make use of this expanding recycling infrastructure. Therefore, panels B and H show the same behaviour in Figure S2.

However, this assumption does not hold for the novel softwood-polypropylene compound: we expect no notable sources of softwood-polypropylene waste until after this material has been introduced as modelled here. Put differently, while the share of softwood-polypropylene waste which is recycled increases at the same pace as talc-polypropylene waste (see panel H in Figure S2 and Figure S3), secondary softwood-polypropylene can only be used after components of this material reach end-of-life. This puts a constraint on the use of the secondary material (compare panel B in Figure S2 to panel B in Figure S3). This constraint is described by Equation 5:

$$g(t) = f(t) Outflow(t-1) \frac{\eta}{Inflow(t)} \tag{5}$$

The share of end-of-life recycling ($f(t)$, obtained via Equation 1) is multiplied by the total end-of-life material created in the previous year ($Outflow(t-1)$) – which is then multiplied by the material efficiency of manufacturing (η) over the input of material to the use process ($Inflow(t)$) in order to obtain what share of compound used is secondary ($g(t)$). As described in 2.2.1, only the use process is assumed to have a duration. Therefore, the year following use sees end-of-life material immediately being recycled for subsequent use in that same year.

Table S3: Economic values assigned to waste incineration and compound recycling processes. Following the assumption that the creation of secondary material is much more valuable than treating waste by recycling, the allocation factors of these functions are 1 and 0, respectively.

Function	Economic value	Unit
treating waste by incineration	1.50×10^{-1}	EUR per kilogram
electricity generated from waste incineration	8.00×10^{-2}	EUR per kilowatt hour
heat generated from waste incineration	2.00×10^{-2}	EUR per megajoule
treating waste by recycling	~ 0	EUR per kilogram
creating secondary material from recycling	$>> 0$	EUR per kilogram

S1.2.5 Multifunctionality

In our application of LCA, we construct product systems which each have a reference flow. This reference flow describes one way of satisfying the functional unit of the assessment. The functions required to enable this reference flow are followed upstream (in the case of goods) and downstream (in the case of wastes), resulting in an extensive web of processes. However, in doing so, we encounter activities which provide more than one function. When the provision of these functions by such a multifunctional activity is not proportionally equivalent to the need for these functions by the product system at hand, the excess functions must somehow be reckoned with (Guinée et al., 2002).

A relevant example: when energy or materials are recovered from an end-of-life product, value is created in surplus of the product's service-life utility. A straightforward solution is to model a closed loop, consuming the surplus function (or functions) within the same product system. When this solution is undesirable, the surplus function crosses the system boundary. Several approaches exist to account for such multifunctionality, but as these approaches are context-dependent, applying them prospectively brings additional hurdles.

Here, we consider two approaches to resolving such multifunctionality. First, partitioning multifunctional processes following an economic rationale (economic allocation), thereby allocating the excess functions away from the product system at hand, and secondly, applying substitution logic to balance each excess function with some equivalent.

The first approach involves assigning an economic value to each function of a multifunctional processes. The market value of energy recovered from waste incineration and the respective value of treating a waste is hard to determine with precision, as the value of these functions varies from place to place and across time. Here, we consider the costs presented in Table S3, which reflect ballpark figures for wholesale energy prices and the gate fees of municipal solid waste incineration. In tandem with the energy recovery values of Table S2, this leads to the incineration of polypropylene being partitioned into shares of 27% for electricity recovery, 38% for heat recovery, and 35% for waste treatment. For economic allocation, only this last fraction of the multifunctional process remains within the system boundary. For softwood, the share of waste treatment is 58% and, naturally, for talc itself this is 100%, as no energy is recovered (see Table S2).

For recycling processes, we simplify the partitioning to the cut-off logic, i.e., all burdens are assigned to the production of a secondary material (allocation factor of one) and no burdens are assigned to the treatment of a waste (allocation factor of zero). This reflects the assumption that the value of recycling is much greater than the value of treating waste alone. This approach is also applied in the ecoinvent system model we use (see S1.2.6). Often, the product systems constructed here keep both of these functions within the system boundary (see S1.2.2). This reduces the relevance of this assumption.

Unless specified otherwise, all results are calculated using the above partitioning approach. As sensitivity analysis, we apply the Circular Footprint Formula (CFF), which incorporates substitution. This is part of the Product Environmental Footprint (PEF) method recommended by the European Commission (2021). As reported by Zampori and Pant (2019), energy recovered via incineration is accounted for as fully substituting an equivalent energy source. The generation of heat and electricity thereby lead to the subtraction of the same amount of heat and electricity, as generated in the respective European market mixes for that year. This is only applied to foreground incineration processes (indicated in Figures S2 and S3), not to energy recovery occurring in the background.

The substitution logic used for material recycling is less straightforward. Following Equation 6 – adapted from Zampori and Pant (2019, p. 66) – the use of secondary material that was recycled in another product system (share R_1, recycled with the process $E_{\text{previous recycling}}$) is accounted for with an allocation factor A (the default factor is 0.5, which we also use here) while additionally gaining a share of the burden associated with the production of an equivalent primary material (E_v). This burden is scaled by $1-A$ and $Q_{s_{in}}/Q_p$, which represents the difference in quality between the secondary material and its equivalent primary material – here, this ratio is 0.9, as recommended for mechanical recycling of plastics. When the product system creates additional secondary material (share R_2, recycled with the process $E_{\text{EoL recycling}}$), a similar shift of burdens occurs with respect to E_v, this time leading to a credit (i.e., negative impact) for the product system at hand. Again, we set the quality ratio used here to 0.9.

$$E_{\text{material recycling}} = R_1 \left(AE_{\text{previous recycling}} + (1-A)\, E_v \frac{Q_{s_{\text{in}}}}{Q_p} \right) + (1-A)R_2 \left(E_{\text{EoL recycling}} - E_v \frac{Q_{s_{\text{out}}}}{Q_p} \right) \quad (6)$$

This adding or subtracting of a fraction of E_v can be imagined as the burden of primary production being shifted in part from the product system which used this material initially to the product system which uses it subsequently. However, with time-explicit LCA, recycling considers E_v as the production of primary material when recycling occurs – not as when the material being recycled would have been produced. This makes the implications of substitution and the CFF even less straightforward than their static application, which we touch on in 4.2.2 and S2.4.

Zampori and Pant (2019) also suggest that the E_v process may not be the production of primary material alone, but some representative mix of primary and secondary materials. Here, we only consider full primary production for E_v, i.e., the substitution of primary compound is always made entirely from primary matrix material, primary filler, etc., rather than some mix of primary and secondary sources. This makes our implementation of the CFF less representative of its intended use, but we do not consider this to have relevant consequences for our assessment.

S1.2.6 Background databases

As described in 2.2.2, we represent 2030 using the ecoinvent database (Wernet et al., 2016). This is ecoinvent v3.10 following the ‘Allocation cut-off by Classification’ system model.

This system model generally uses an attributional approach to constructing inventories and resolving multifunctionality. Generally, this aligns with how we think of product systems here, where the provision of a certain function requires that other functions are consumed. An alternative to such a functional web is to conduct consequential LCA (CLCA), which understands the product system as cascading consequences, originating from the reference flow as choice to be evaluated. While this approach is not necessarily incompatible with the concepts we discuss here – our case study can also be understood as concerning a central choice (to advance one alternative or another) – our understanding of the lifecycle as entangled in society's metabolism aligns more closely with attributional modelling.

We do not use this database as-is, but apply changes to certain coke production processes, using a change file provided by ecoinvent (2025). This is necessary because of errors in v3.10 which heavily affect toxicity-related impact categories. This was fixed with the release of v3.10.1. Because our workflow had already been set up based on v3.10, we applied the aforementioned change file instead of redoing this for v3.10.1. The exact steps taken are documented in the code repository (Arblaster et al., 2026).

Past 2030, we use version 2.2.7 of premise (Sacchi et al., 2022) to generate prospective background databases, as introduced in 2.2.3. This package alters the ecoinvent database in line with the outputs of an integrated assessment model (IAM) and various descriptions of future markets and technical processes. However, not all sectors of the economy are altered with the same level of detail, with most transformations being related to energy systems. And beyond its technical details, there are important limitations to consider when generating and using such databases, as discussed by de Bortoli et al. (2025).

We consider two IAMs which each model two scenarios. From REMIND (Baumstark et al., 2021), we consider SSP2-PkBudg500 and SSP2-PkBudg1150. SSP refers to the Shared Social Pathways, with SSP2 being 'middle-of-the-road' – i.e., socio-economic trends do not deviate much from past data. PkBudg500 and PkBudg1150 refer to the ambition of climate policy imposed by the IAM, respectively leading to a global mean surface temperature increase by 2100 of under 1.5°C and under 2.0°C when compared to pre-industrial conditions. We also consider two equivalent scenarios from IMAGE (Stehfest et al., 2014), labelled SSP2-RCP19 and SSP2-RCP26. Again, these follow middle-of-the-road socio-economic trends, with climate policy imposed to limit temperature increase to respectively 1.5 and 2.0°C.

Background databases are generated for five-year intervals, from 2035 to 2080. The use of these five-year intervals for time-explicit LCA is explained in 2.2.1. In short, demands of the foreground on the background are split between their nearest years for which a database exists. This also applies to the first interval, from 2030-2035. After 2080, the closest year for which a database exists indefinitely continues to be 2080.

All results presented in the main text are based on REMIND following SSP2-PkBudg1150. Other pathways are only considered as sensitivity analysis (see S2.4).

S1.2.7 Contribution analysis

To understand how different aspects of the product system come together to form its environmental impacts, we examine the route by which impacts travel from the process causing them to the reference flow. We do this by turning the overarching product system into a set of connected subsystems. Van der Meide et al. (2025) calls this approach 'partial-LCA life cycle stage contribution analysis' (p. 2756).

The definition of lifecycle stages is presented in Figure S4. The same labels are used to indicate the contributions of these stages elsewhere, such as in Figure 6.

Because of our building-block approach to time-explicit LCA (see S1.2.2), the product system is already constructed from partial LCA results. To apply contribution analysis, the only extra step required is to group these partial results into the appropriate lifecycle stage (Arblaster et al., 2026).

By defining these subsystems (or stages), it becomes clear how different parts of the foreground system relate to its environmental impacts. We choose this approach because it is particularly relevant for eco-design: the designer can enable interventions to directly affect various aspects modelled in the foreground system, but has limited control of the background.

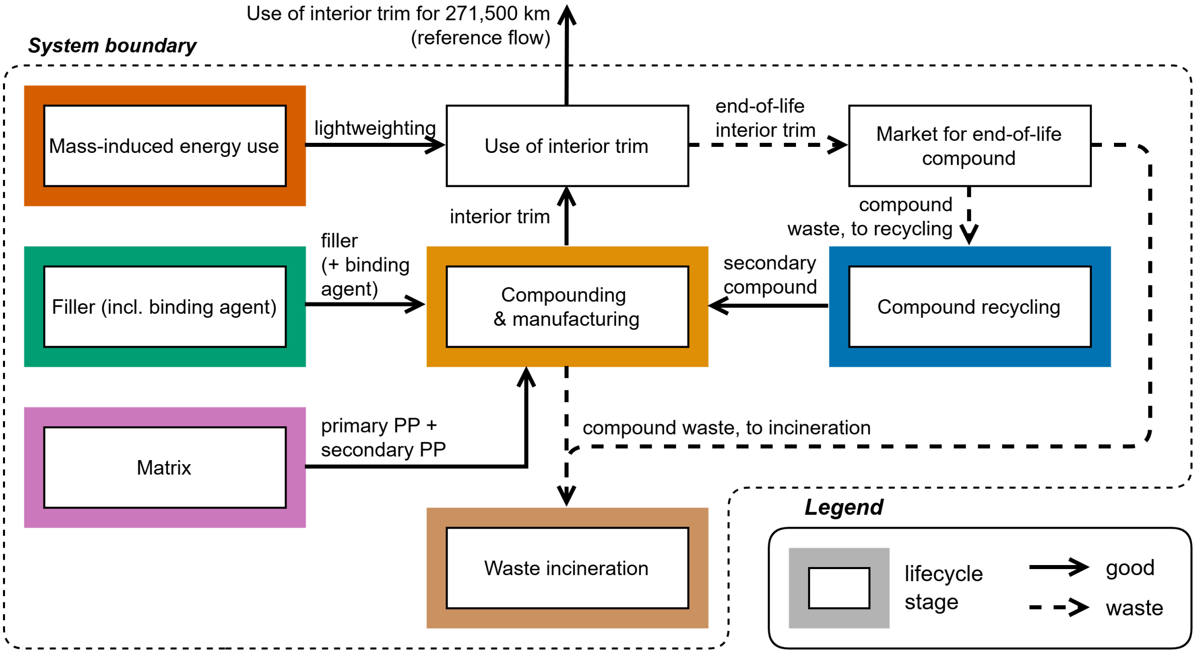


Figure S4: Illustration of lifecycle stages for contribution analysis. The product system is split into multiple connected subsystems. Each subsystem represents a lifecycle stage which contributes to the reference flow. Note that use and the division of end-of-life waste have no further impacts beyond those occurring in connected subsystems and are therefore themselves not part of any lifecycle stage.

S2 Supplementary results and discussion

S2.1 Contribution analyses

Contribution analyses are performed following S1.2.7. For each alternative, these contributions are shown in Figures S5 (talc-polypropylene) and S6 (softwood-polypropylene) following the four temporal set-ups presented in the main text.

By definition, the sum of all contributions within an impact category is 1. With the representation of lightweighting as a negative energy consumption, the softwood-polypropylene system has a negative contribution (Figure S6). This leads to the sum of positive contributions exceeding 1 by the additive inverse of the negative contribution.

This means that the value of a contribution is no longer directly comparable between alternatives and across impact categories. For example, in Figure S6A, ‘compounding & manufacturing’ has a contribution of almost 100% to ‘ionising radiation: human health’, while its relative contribution to the lifecycle (e.g., with respect to ‘matrix’) is not dissimilar to the one shown in Figure S5A for the same impact category.

S2.2 Investigation of filler mass fraction of softwood-polypropylene

As introduced in S1.1.2, we also assess one of the degrees of freedom available to designers in this case, being the filler content of novel softwood-polypropylene compound. This is illustrated in Figure S7 in terms of the relative change in impact realised by the softwood-polypropylene system with respect to the talc-polypropylene system. In this figure, a slope towards the bottom left indicates that a lower filler content is preferred, while a slope towards the bottom right indicates that a higher filler content is preferred. For most impact categories, a lower filler content is preferred, regardless of temporal set-

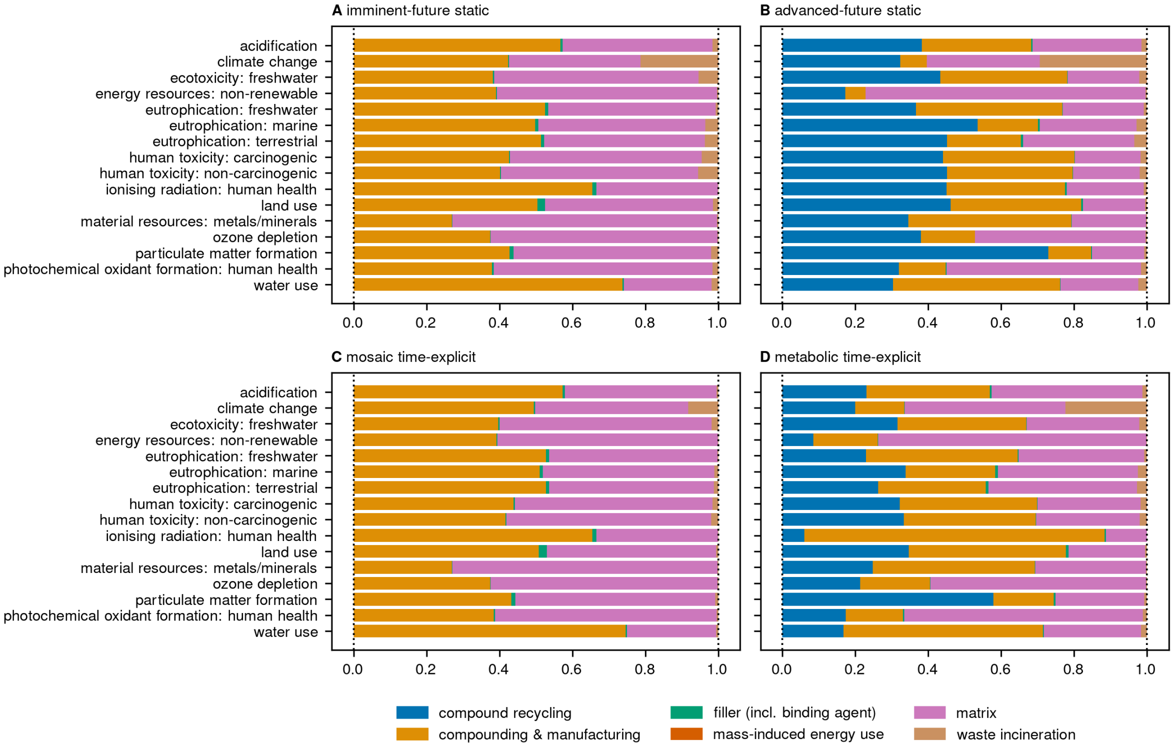


Figure S5: Contribution analysis of the talc-polypropylene alternative across each impact category assessed. Each panel represents a separate temporal set-up. Contributions are shown in relation to lifecycle stages of the foreground system, as described in S1.2.7. Note that the contribution of mass-induced energy use is zero for the talc-polypropylene alternative, as explained in S1.2.3.

up. A few impact categories show a preference for a high filler content, but only for specific temporal set-ups. For example, climate change (Figure S7B) shows a clear preference for a higher filler content under advanced-future static and metabolic time-explicit perspectives.

The mostly consistent preference for a low filler content within each impact category indicates a limited influence of the temporal perspective. However, the actual impact (see Figure 6) and the relative effect of a change in filler content (represented by a change in gradient in Figure S7) are affected. This means that the temporal set-ups might lead to different conclusions, depending on how impact categories are weighed against each other.

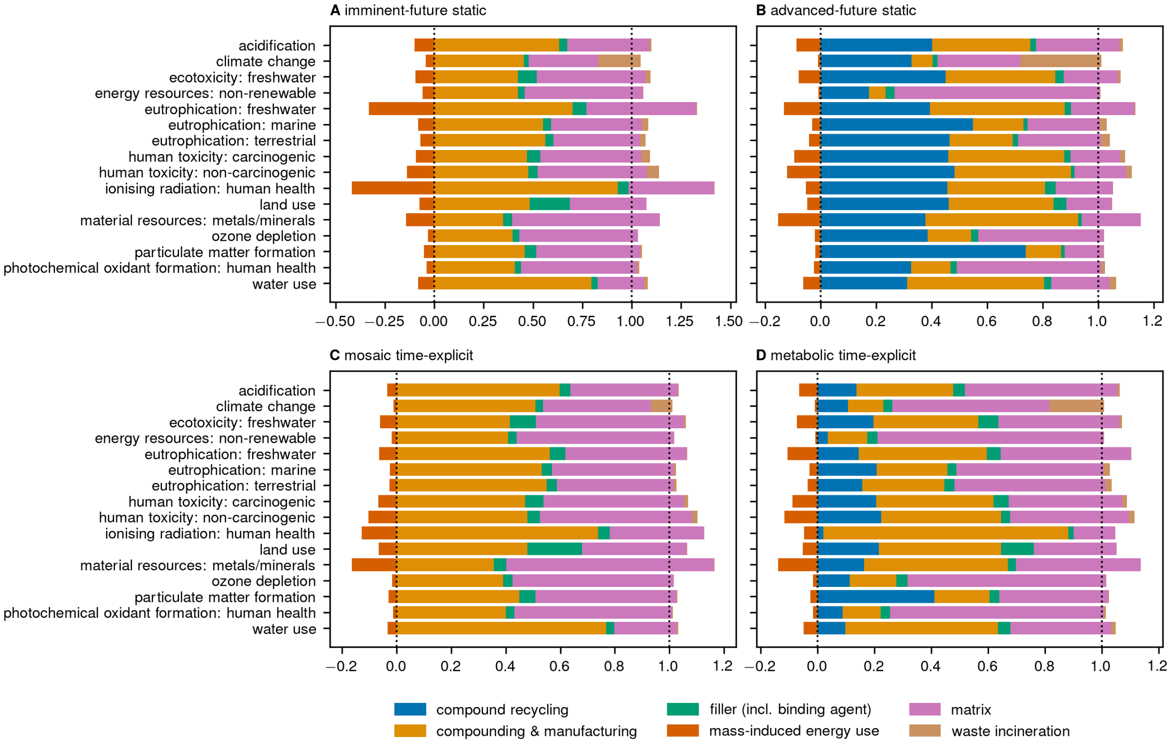


Figure S6: Contribution analysis of the softwood-polypropylene alternative across each impact category assessed. Each panel represents a separate temporal set-up. Contributions are shown in relation to lifecycle stages of the foreground system, as described in S1.2.7. Because mass-induced energy use has a negative contribution, the y-axis values differ from panel to panel.

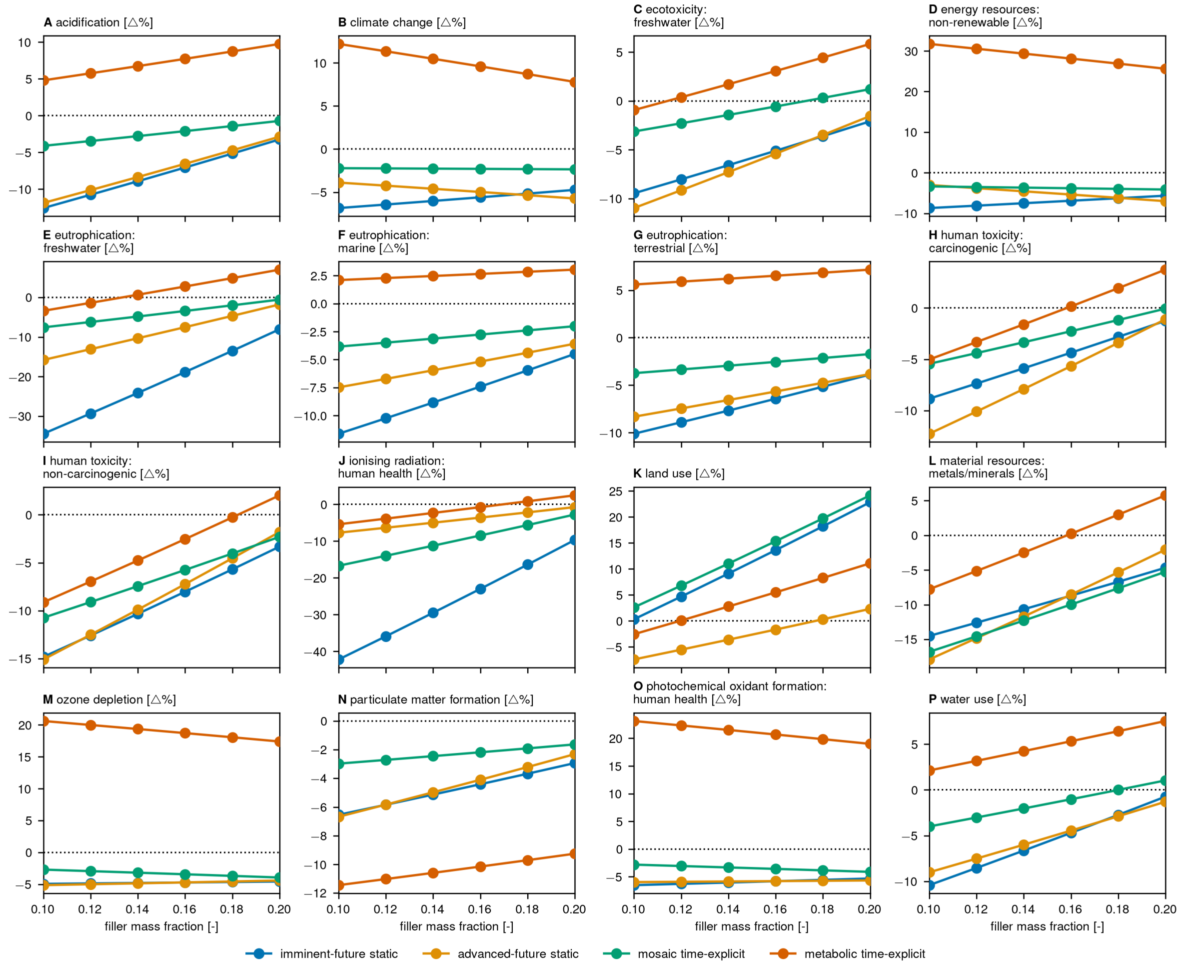


Figure S7: Representation of how the filler mass fraction of the softwood-polypropylene compound affects its comparison to the talc-polypropylene compound for each impact category assessed. For each indicated filler content, this comparison is calculated by dividing the impact of the softwood-polypropylene system by that of the talc-polypropylene system and subtracting this ratio from one, with the result being shown here as percentage values. As such, a negative value indicates that the softwood-polypropylene system has a lower impact, while a positive value indicates that the talc-polypropylene system has a lower impact. Each panel depicts results for four temporal set-ups.

S2.3 Extended overview of temporal set-ups

In the main text, we present four temporal set-ups (see 2.1). However, these do not cover the full breadth of approaches to representing temporal evolution which a practitioner might take. In S1.2.1, we explain how temporal set-ups include both foreground and background temporality, and that the understanding of temporal evolution across these two dimensions can differ. Figure S1 labels set-ups with mixed perspectives (such as a imminent-future foreground with an advanced-future background) and provides illustrative descriptions of a system aligned with each set-up.

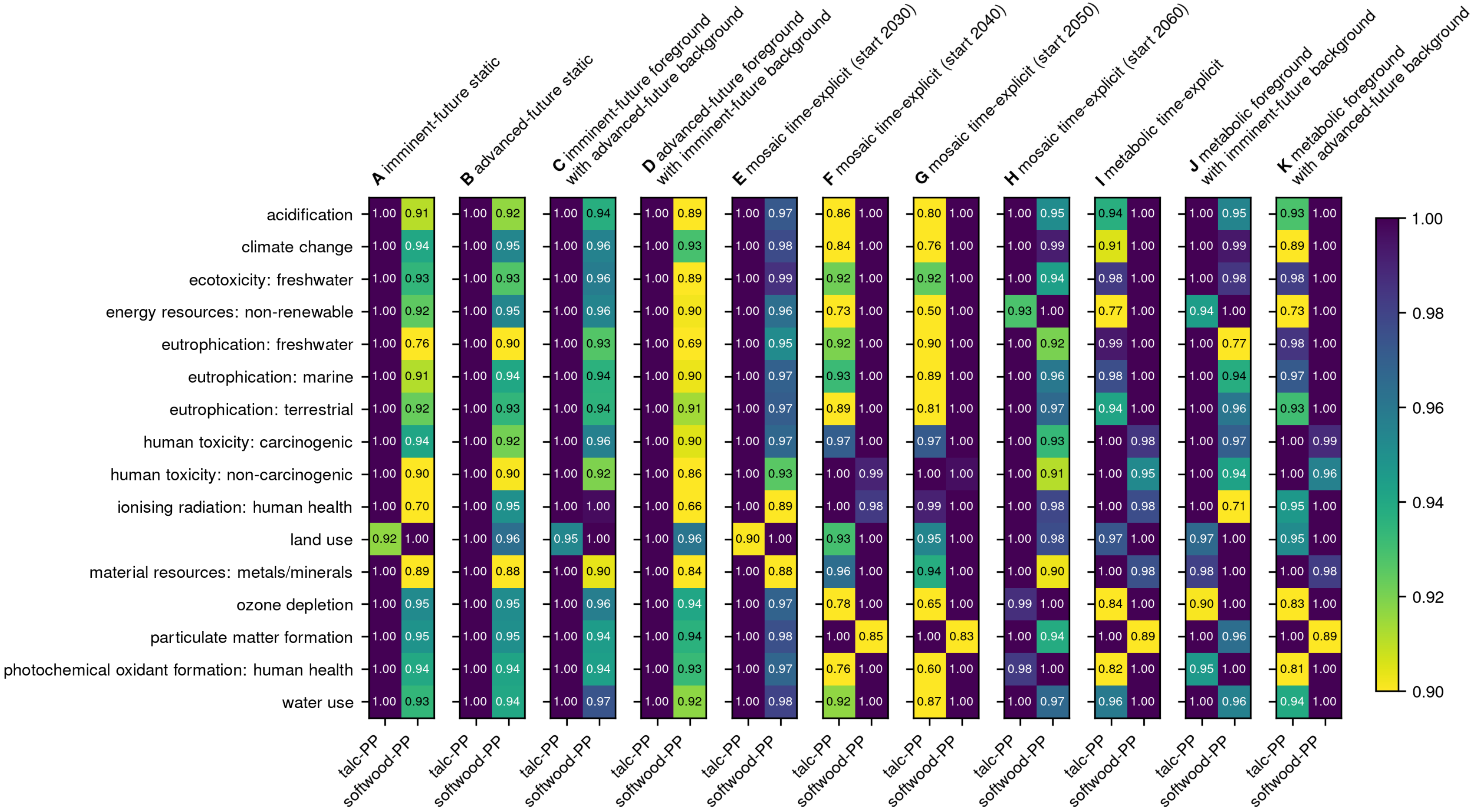
These descriptions concern how the system's relationship with time is understood. Similar diversity can be found for the case study considered here. We illustrate results for a variety of approaches in Figure S8 (comparable in format to Figure 5) and in Figure S9 (comparable in format to Figure 6).

For example, combining an imminent-future state of the foreground with an advanced-future state of the background (Figure S8C, labelled 'A3 Transition scenario' in Figure S1) represents a scenario where the automotive sector has not advanced much in its circularity, but other aspects of the economy have transformed considerably. This does not change the relative preference between alternatives, but Figure S9 shows how much of the impact change associated with temporal evolution can be attributed to the foreground (i.e., circularity of automotive plastics) and how much of it can be attributed to the background (e.g., the energy transition – see S1.2.6).

On the other hand, practical limitations faced by the practitioner might make it appealing to only represent temporal evolution in the foreground, with the background necessarily reflecting the near-present. This way, an impression can still be obtained of an advanced-future system (Figure S8D, equivalent to 'C1 Deployment scenario') or even of certain metabolic characteristics of a time-explicit system – likely reduced to a steady-state computational structure (Figure S8J, equivalent to 'B1 Pseudo time-explicit'). However, this considerably limits the degree to which potential impact reduction is represented (see Figure S9).

In an extension of this idea, foreground processes representing a metabolic understanding of the system can be coupled to a static advanced-future background database (Figure S8K, equivalent to 'B3 Pseudo time-explicit'). For this case study, this leads to results that are very similar to the fully realised metabolic time-explicit approach in both the comparison of alternatives (compare panels H and K in Figure S8) and the magnitude of impacts (see Figure S9). This leads us to the conclusion that a static computational structure can confer metabolic insights, at least to some degree. This is discussed further in 4.3.

Furthermore, while the mosaic time-explicit set-up has so far been restricted to a starting year of 2030, we now also consider 2040, 2050, and 2060 as starting years (respectively panels F, G, and H in Figure S8). When starting in 2060, the results are similar to those of the advanced-future static set-up, which models a steady-state in 2070. Results again have a similar magnitude when starting in 2040 or 2050 (see Figure S9), however, the comparison of alternatives is quite different here (compare panels F and G to the other panels in Figure S8). This is because these start years occur when the talc-polypropylene alternative requires less primary material than the softwood-polypropylene alternative. While the period of comparatively larger impacts from softwood-polypropylene is hidden when cradle-to-gate processes are isolated to 2030 or 2070, evaluating components as produced in 2040 or 2050 represents nothing but this period – obscuring that this is a temporary effect.



	A imminent-future static		B advanced-future static		C imminent-future foreground with advanced-future background		D advanced-future foreground with imminent-future background		E mosaic time-explicit (start 2030)		F mosaic time-explicit (start 2040)		G mosaic time-explicit (start 2050)		H mosaic time-explicit (start 2060)		I metabolic time-explicit		J metabolic foreground with imminent-future background		K metabolic foreground with advanced-future background	
	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP	talc-PP	softwood-PP
acidification	1.00	0.91	1.00	0.92	1.00	0.94	1.00	0.89	1.00	0.97	0.86	1.00	0.80	1.00	1.00	0.95	0.94	1.00	1.00	0.95	0.93	1.00
climate change	1.00	0.94	1.00	0.95	1.00	0.96	1.00	0.93	1.00	0.98	0.84	1.00	0.76	1.00	1.00	0.99	0.91	1.00	1.00	0.99	0.89	1.00
ecotoxicity: freshwater	1.00	0.93	1.00	0.93	1.00	0.96	1.00	0.89	1.00	0.99	0.92	1.00	0.92	1.00	1.00	0.94	0.98	1.00	1.00	0.98	0.98	1.00
energy resources: non-renewable	1.00	0.92	1.00	0.95	1.00	0.96	1.00	0.90	1.00	0.96	0.73	1.00	0.50	1.00	0.93	1.00	0.77	1.00	0.94	1.00	0.73	1.00
eutrophication: freshwater	1.00	0.76	1.00	0.90	1.00	0.93	1.00	0.69	1.00	0.95	0.92	1.00	0.90	1.00	1.00	0.92	0.99	1.00	1.00	0.77	0.98	1.00
eutrophication: marine	1.00	0.91	1.00	0.94	1.00	0.94	1.00	0.90	1.00	0.97	0.93	1.00	0.89	1.00	1.00	0.96	0.98	1.00	1.00	0.94	0.97	1.00
eutrophication: terrestrial	1.00	0.92	1.00	0.93	1.00	0.94	1.00	0.91	1.00	0.97	0.89	1.00	0.81	1.00	1.00	0.97	0.94	1.00	1.00	0.96	0.93	1.00
human toxicity: carcinogenic	1.00	0.94	1.00	0.92	1.00	0.96	1.00	0.90	1.00	0.97	0.97	1.00	0.97	1.00	1.00	0.93	1.00	0.98	1.00	0.97	1.00	0.99
human toxicity: non-carcinogenic	1.00	0.90	1.00	0.90	1.00	0.92	1.00	0.86	1.00	0.93	1.00	0.99	1.00	1.00	1.00	0.91	1.00	0.95	1.00	0.94	1.00	0.96
ionising radiation: human health	1.00	0.70	1.00	0.95	1.00	1.00	1.00	0.66	1.00	0.89	1.00	0.98	0.99	1.00	1.00	0.98	1.00	0.98	1.00	0.71	0.95	1.00
land use	0.92	1.00	1.00	0.96	0.95	1.00	1.00	0.96	0.90	1.00	0.93	1.00	0.95	1.00	1.00	0.98	0.97	1.00	0.97	1.00	0.95	1.00
material resources: metals/minerals	1.00	0.89	1.00	0.88	1.00	0.90	1.00	0.84	1.00	0.88	0.96	1.00	0.94	1.00	1.00	0.90	1.00	0.98	0.98	1.00	1.00	0.98
ozone depletion	1.00	0.95	1.00	0.95	1.00	0.96	1.00	0.94	1.00	0.97	0.78	1.00	0.65	1.00	0.99	1.00	0.84	1.00	0.90	1.00	0.83	1.00
particulate matter formation	1.00	0.95	1.00	0.95	1.00	0.94	1.00	0.94	1.00	0.98	1.00	0.85	1.00	0.83	1.00	0.94	1.00	0.89	1.00	0.96	1.00	0.89
photochemical oxidant formation: human health	1.00	0.94	1.00	0.94	1.00	0.94	1.00	0.93	1.00	0.97	0.76	1.00	0.60	1.00	0.98	1.00	0.82	1.00	0.95	1.00	0.81	1.00
water use	1.00	0.93	1.00	0.94	1.00	0.97	1.00	0.92	1.00	0.98	0.92	1.00	0.87	1.00	1.00	0.97	0.96	1.00	1.00	0.96	0.94	1.00

Figure S8: Results for each impact category assessed, where each panel presents results for a different temporal set-up. Within each panel, the two alternatives are compared: talc-polypropylene on the left and softwood-polypropylene on the right. The format of the figure is similar to Figure 5, with the exception that additional temporal set-ups are presented: mosaic time-explicit results for starting years other than 2030, as well as results of mixed temporality (see S1.2.1).

Figure S9: Comparison of results of various temporal set-ups, where each panel represents one of the impact category assessed. Contributions are shown in relation to lifecycle stages of the foreground system, as described in S1.2.7. Because mass-induced energy use has a negative contribution, the sum of each bar is indicated with a diamond (talcum-polypropylene) or circle (softwood-polypropylene). The format of the figure is similar to Figure 6, with the exception that additional temporal set-ups are presented: mosaic time-explicit results for starting years other than 2030, as well as results of mixed temporality (see S1.2.1).

S2.4 Sensitivity analyses

We explore the sensitivity of our conclusions to two of the modelling choices made: how multifunctionality is resolved (see S1.2.5) and what external scenario is used when generating background databases (see S1.2.6).

All results outside of this section resolve multifunctionality by partitioning, following economic allocation. Here, we compare this to another way of resolving multifunctionality, by following the Circular Footprint Formula (CFF). This approach is based on a substitution logic, as explained in S1.2.5.

Across these two approaches, the comparative performance of alternatives remains similar across temporal set-ups. It is only for the metabolic time-explicit set-up that using the CFF leads to a switch in which alternative has a lower impact on what impact category: while economic allocation (Figure S10D.A) suggests a lower impact of talc-polypropylene on ecotoxicity: freshwater, eutrophication: freshwater, eutrophication: marine, and water use, softwood-polypropylene has a lower impact following the CFF (Figure S10D.B).

Generally, the CFF appears to be favourable to softwood-polypropylene. This is in part because, for certain impact categories, the credits obtained from energy recovery turn waste incineration into a net negative impact for the system (see panels B.E-H, C.E-H, and D.E-H in Figure S11). Softwood-polypropylene features comparatively more incineration with the metabolic approach, but the softwood filler itself is at an advantage here over the talc filler (see Table S2).

These credits can also lead to the mosaic time-explicit results being higher than the imminent-future static results (e.g., Figure S11B.E-H), as the advanced-future substitution flows have a (much) lower impact compared to their imminent-future equivalent. Because of this effect, Šimaitis et al. (2023) consider credits modelled as occurring in the imminent future to be misleading, and suggest focusing on cradle-to-gate processes instead. It is true that cradle-to-gate processes should not be neglected, but it's also true that design for a circular economy requires ways of evaluating the success of circular practises (see 4.2). Broadly, we can state that the way in which multifunctionality is resolved does not affect the broader conclusions we draw here.

As explained in S1.2.6, we generate prospective background databases for years beyond 2030. The results in the main text do so following an SSP2-PkBudg1150 scenario of the IAM REMIND. Here, we consider the three other pathways that were mentioned in S1.2.6: REMIND following SSP2-PkBudget500 and IMAGE following both SSP2-RCP19 and SSP2-RCP26.

Results for all four pathways are presented in Figure S11. Note that 2030 is still represented with the unchanged ecoinvent database, meaning that the imminent-future static perspective is entirely unaffected by the choice of IAM pathway. For the other temporal perspectives, there are a few noteworthy deviations. There are some changes to particular contributions (e.g., compare 'compounding & manufacturing' for the advanced-future static perspective in panels B.A and B.C of Figure S11). This also leads to differences to the scale of impact, as can be noticed for land use (Figure S11D.A-H).

Overall, these changes are minor when compared to the changes observed across temporal perspectives (both in the main text as in S2.3). It is clear that the choice of background scenario does influence the results of an assessment, but for our purposes, this influence is not relevant to the broader research question at hand.

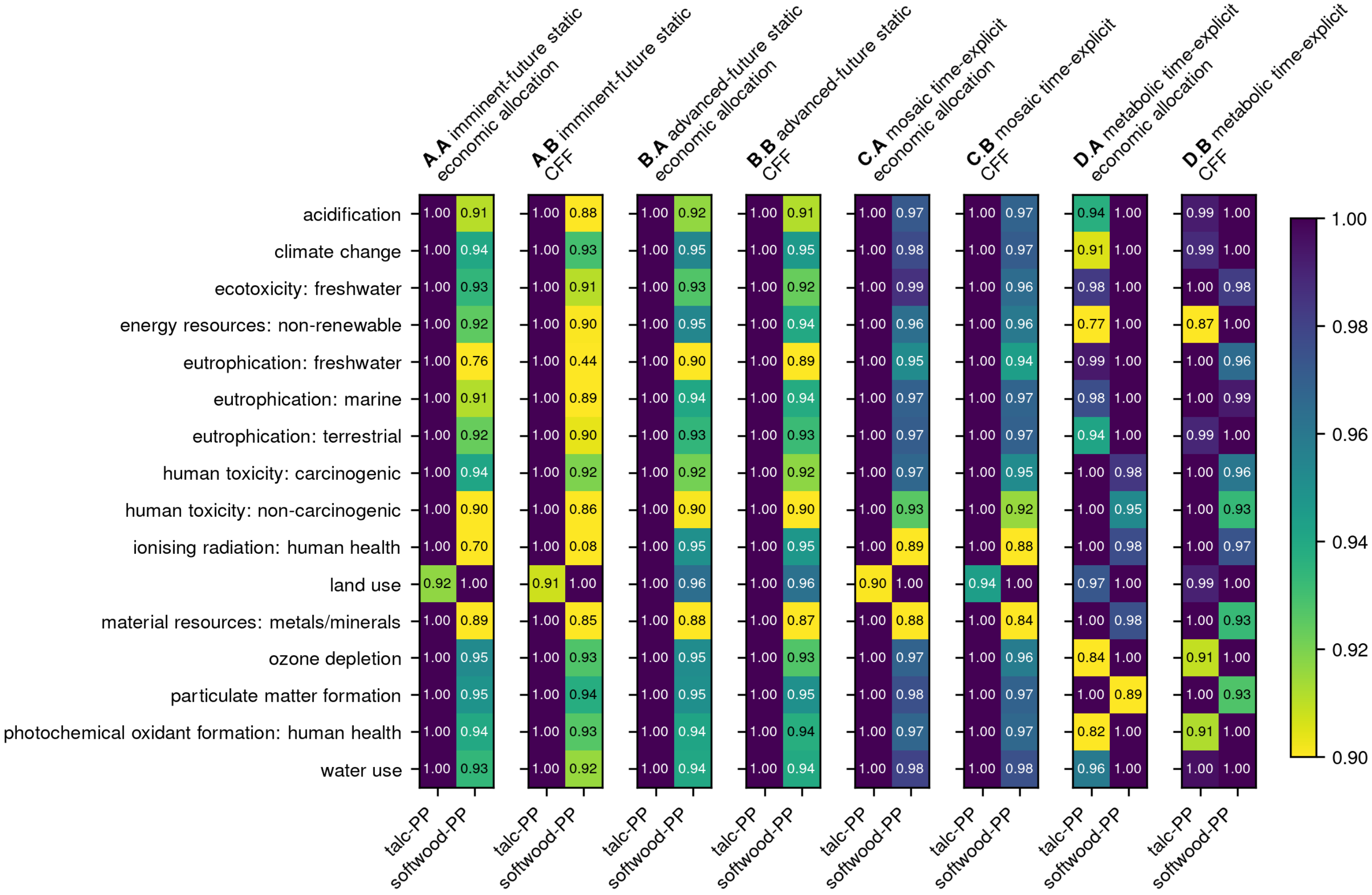


Figure S10: Results for each impact category assessed, where each panel presents results for a different temporal set-up, to explore sensitivity to the chosen method of resolving multifunctionality. Within each panel, the two alternatives are compared: talc-polypropylene on the left and softwood-polypropylene on the right. The format of the figure is similar to Figure 5, with each temporal perspective presented once for each of the two approaches to resolving multifunctionality. Composite panel labels indicate this combination, with the first letter representing the temporal perspective and the second letter the multifunctionality approach (i.e., imminent-future static results are labelled A.*X*, while results using the CFF are all labelled *X*.B).

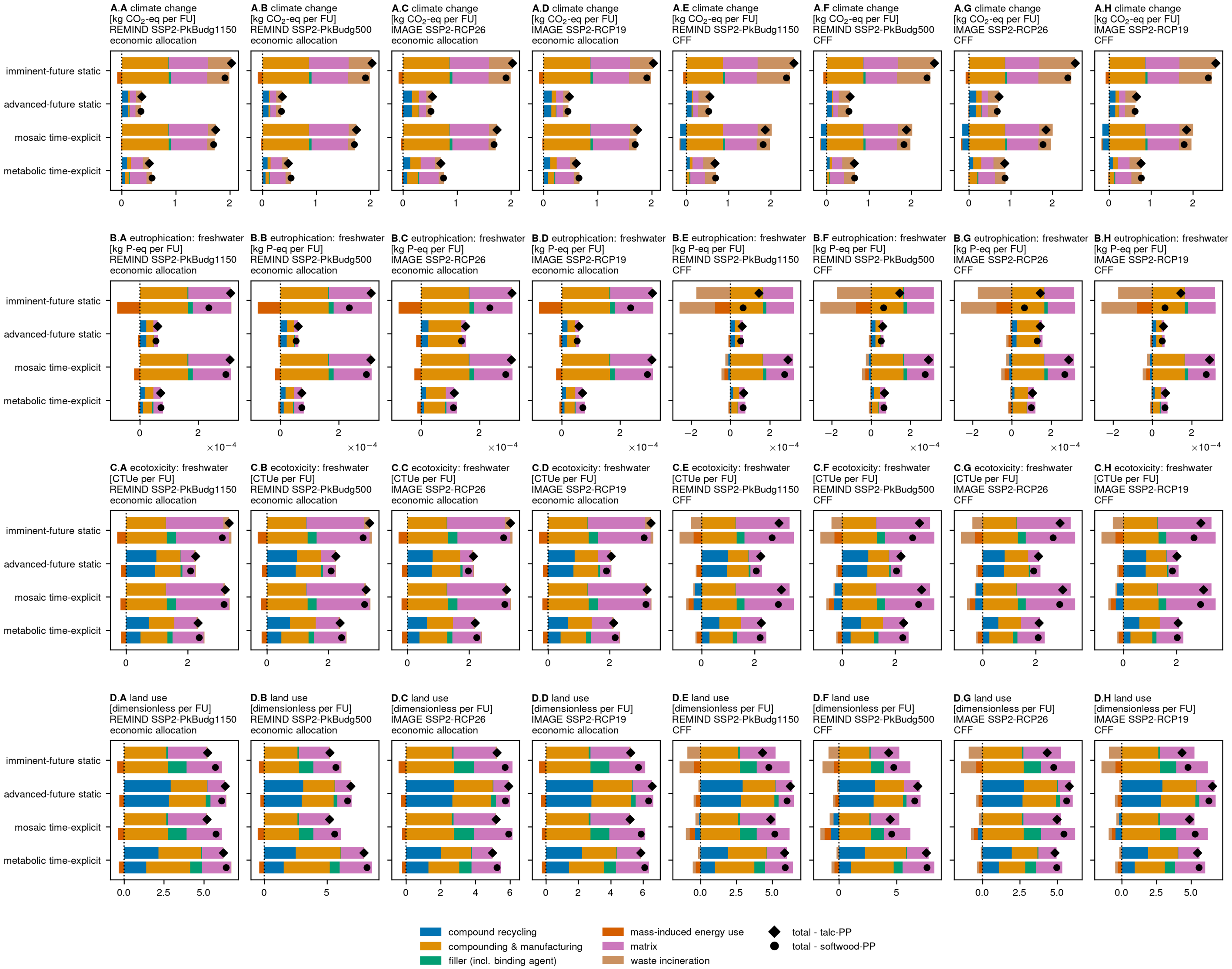


Figure S11: Comparison of results of various temporal set-ups to explore sensitivity to modelling choices. The format of the figure is similar to Figure 6, with each impact category being presented eight times, for each combination of the four background pathways and the two approaches to resolving multifunctionality. Composite panel labels indicate this combination, with the first letter representing the impact category and the second letter the combination of modelling choices (i.e., climate change results are all labelled A.*X*, while results using the CFF and IMAGE SSP2-RCP19 are all labelled *X*.D).

Supplementary references

Abu-Ghaida, H., Hollberg, A., Ritzen, M., Wöhler, A., & Lizin, S. (2025). Assessing the environmental benefits of design for disassembly in buildings with a time-resolved prospective LCA approach. *The International Journal of Life Cycle Assessment*, *30*(9), 1913–1929. https://doi.org/10.1007/s11367-025-02526-8

Arblaster, T. P. S., Guinée, J., Blanco Rocha, C. F., Burzic, I., Pretschuh, C., Pérez Sánchez, F., & Thonemann, N. (2026). *arblastertps/vehicle_timex* (Version v0.1.0). https://doi.org/10.5281/zenodo.19663639

Arvidsson, R., Tillman, A.-M., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D., & Molander, S. (2018). Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *Journal of Industrial Ecology*, *22*(6), 1286–1294. https://doi.org/10.1111/jiec.12690

Badino, V., Baldo, G. L., Fornaro, M., & Salvaia, E. (1995). Ecobalance of talc mineral production. *III International Congress "Energy, Environment and Technological Innovation"*, *2*, 125–131.

Baumstark, L., Bauer, N., Benke, F., Bertram, C., Bi, S., Gong, C. C., Dietrich, J. P., Dirnaichner, A., Giannousakis, A., Hilaire, J., Klein, D., Koch, J., Leimbach, M., Levesque, A., Madeddu, S., Malik, A., Merfort, A., Merfort, L., Odenweller, A., … Luderer, G. (2021). REMIND2.1: Transformation and innovation dynamics of the energy-economic system within climate and sustainability limits. *Geoscientific Model Development*, *14*(10), 6571–6603. https://doi.org/10.5194/gmd-14-6571-2021

Blanco, C. F., Behrens, P., Vijver, M., Peijnenburg, W., Quik, J., & Cucurachi, S. (2025). A framework for guiding safe and sustainable-by-design innovation. *Journal of Industrial Ecology*, *29*(1), 47–65. https://doi.org/10.1111/jiec.13609

Borealis AG. (2014, May 26). *Borealis launches polypropylene compound portfolio with post-consumer recycled content*. Retrieved April 16, 2026, from https://www.borealisgroup.com/news/borealis-launches-polypropylene-compound-portfolio-with-post-consumer-recycled-content

Civancik-Uslu, D., Ferrer, L., Puig, R., & Fullana-i-Palmer, P. (2018). Are functional fillers improving environmental behavior of plastics? A review on LCA studies. *Science of The Total Environment*, *626*, 927–940. https://doi.org/10.1016/j.scitotenv.2018.01.149

Corbière-Nicollier, T., Gfeller Laban, B., Lundquist, L., Leterrier, Y., Månson, J. .-. E., & Jolliet, O. (2001). Life cycle assessment of biofibres replacing glass fibres as reinforcement in plastics. *Resources, Conservation and Recycling*, *33*(4), 267–287. https://doi.org/10.1016/S0921-3449(01)00089-1

de Bortoli, A., Chanel, A., Chabas, C., Greffe, T., & Louineau, E. (2025). More rationality and inclusivity are imperative in reference transition scenarios based on IAMs and shared socioeconomic pathways - recommendations for prospective LCA. *Renewable and Sustainable Energy Reviews*, *222*, 115924. https://doi.org/10.1016/j.rser.2025.115924

de Zilva, D. B. K., Fishman, T., Tukker, A., & Hu, M. (2026). Deconstructing value in solar panel reuse with time-explicit life cycle assessment and costing. *Resources, Conservation and Recycling*, *226*, 108699. https://doi.org/10.1016/j.resconrec.2025.108699

ecoinvent. (2025). *Ecoinvent Version 3.10*. ecoinvent support. Retrieved April 8, 2026, from https://support.ecoinvent.org/ecoinvent-version-3.10

European Commission. (2020). Sustainable and Smart Mobility Strategy – putting European transport on track for the future. Retrieved January 15, 2026, from https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:52020DC0789

European Commission. (2021). Commission Recommendation (EU) 2021/2279 of 15 December 2021 on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32021H2279

Font Vivanco, D., Kemp, R., & van der Voet, E. (2015). The relativity of eco-innovation: Environmental rebound effects from past transport innovations in Europe. *Journal of Cleaner Production*, *101*, 71–85. https://doi.org/10.1016/j.jclepro.2015.04.019

Geyer, R., & Malen, D. E. (2020). Parsimonious powertrain modeling for environmental vehicle assessments: Part 2—electric vehicles. *The International Journal of Life Cycle Assessment*, *25*(8), 1576–1585. https://doi.org/10.1007/s11367-020-01775-z

Guinée, J. B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Sleeswijk, A. W., Suh, S., de Haes, H. A. U., de Bruijn, H., van Duin, R., Huijbregts, M. A. J., Lindeijer, E., Roorda, A. A. H., van der Ven, B. L., & Weidema, B. P. (2002). *Handbook on Life Cycle Assessment: Operational guide to the ISO standards* (J. B. Guinée, Ed.). Kluwer Academic Publishers.

Haupt, M., Kägi, T., & Hellweg, S. (2018). Life cycle inventories of waste management processes. *Data in Brief*, *19*, 1441–1457. https://doi.org/10.1016/j.dib.2018.05.067

Held, M., Rosat, N., Georges, G., Pengg, H., & Boulouchos, K. (2021). Lifespans of passenger cars in europe: Empirical modelling of fleet turnover dynamics. *European Transport Research Review*, *13*(1), 9. https://doi.org/10.1186/s12544-020-00464-0

Hesser, F. (2015). Environmental advantage by choice: Ex-ante LCA for a new Kraft pulp fibre reinforced polypropylene composite in comparison to reference materials. *Composites Part B: Engineering*, *79*, 197–203. https://doi.org/10.1016/j.compositesb.2015.04.038

Imerys. (2023, June 27). *Jetfine 3 CA - Talc: Cradle-to-gate ecoprofile data sheet* (Version 03).

Johnson, N. F. (2021). Inhalation Toxicity of Talc. *Journal of Aerosol Medicine and Pulmonary Drug Delivery*, *34*(2), 79–107. https://doi.org/10.1089/jamp.2020.1609

Josef Rettenmaier & Söhne GmbH + Co. KG. (n.d.). ARBOCEL® for plastics and elastomers. Retrieved April 16, 2026, from https://www.jrs.de/jrs-wAssets/docs/chemistry/lefleat/automotive-components-2412.pdf

Koffler, C., & Rohde-Brandenburger, K. (2010). On the calculation of fuel savings through lightweight design in automotive life cycle assessments. *The International Journal of Life Cycle Assessment*, *15*(1), 128–135. https://doi.org/10.1007/s11367-009-0127-z

Luz, S. M., Caldeira-Pires, A., & Ferrão, P. M. (2010). Environmental benefits of substituting talc by sugarcane bagasse fibers as reinforcement in polypropylene composites: Ecodesign and LCA as strategy for automotive components. *Resources, Conservation and Recycling*, *54*(12), 1135–1144. https://doi.org/10.1016/j.resconrec.2010.03.009

Müller, E., Hilty, L. M., Widmer, R., Schluep, M., & Faulstich, M. (2014). Modeling Metal Stocks and Flows: A Review of Dynamic Material Flow Analysis Methods. *Environmental Science & Technology*, *48*(4), 2102–2113. https://doi.org/10.1021/es403506a

Negri, M., & Bieker, G. (2025). *Life-cycle greenhouse gas emissions from passenger cars in the European Union*. International Council on Clean Transportation. https://theicct.org/publication/electric-cars-life-cycle-analysis-emissions-europe-jul25/

Pavolová, H., Kaňuchová, M., Bakalár, T., Kozáková, Ľ., & Nartowska, E. (2026). Environmental Risks of Talc Mining. *Applied Sciences*, *16*(5), 2317. https://doi.org/10.3390/app16052317

Roy, P., Defersha, F., Rodriguez-Uribe, A., Misra, M., & Mohanty, A. K. (2020). Evaluation of the life cycle of an automotive component produced from biocomposite. *Journal of Cleaner Production*, *273*, 123051. https://doi.org/10.1016/j.jclepro.2020.123051

Sacchi, R., Terlouw, T., Siala, K., Dirnaichner, A., Bauer, C., Cox, B., Mutel, C., Daioglou, V., & Luderer, G. (2022). PRospective EnvironMental Impact asSEment (premise): A streamlined approach to producing databases for prospective life cycle assessment using integrated assessment models. *Renewable and Sustainable Energy Reviews*, *160*, 112311. https://doi.org/10.1016/j.rser.2022.112311

Sazdovski, I., Bojovic, D., Batlle-Bayer, L., Aldaco, R., Margallo, M., & Fullana-i-Palmer, P. (2022). Circular economy of packaging and relativity of time in packaging life cycle. *Resources, Conservation and Recycling*, *184*, 106393. https://doi.org/10.1016/j.resconrec.2022.106393

Schlesier, H., Gómez-Camacho, C. E., Hischier, R., & Desing, H. (2026). Fossil-free ecoinvent: A parametric model for generating defossilized life cycle databases. *Resources, Conservation and Recycling*, *226*, 108693. https://doi.org/10.1016/j.resconrec.2025.108693

Šimaitis, J., Allen, S., & Vagg, C. (2023). Are future recycling benefits misleading? Prospective life cycle assessment of lithium-ion batteries. *Journal of Industrial Ecology*, *27*(5), 1291–1303. https://doi.org/10.1111/jiec.13413

Sobczak, L., Lang, R. W., & Haider, A. (2012). Polypropylene composites with natural fibers and wood – General mechanical property profiles. *Composites Science and Technology*, *72*(5), 550–557. https://doi.org/10.1016/j.compscitech.2011.12.013

Stehfest, E., van Vuuren, D. P., Kram, T., & Bouwman, L. (2014). *Integrated Assessment of Global Environmental Change with IMAGE 3.0. Model description and policy applications* (No. 735). Netherlands Environmental Assessment Agency (PBL). The Hague, the Netherlands.

Tadele, D., Roy, P., Defersha, F., Misra, M., & Mohanty, A. K. (2020). A comparative life-cycle assessment of talc- and biochar-reinforced composites for lightweight automotive parts. *Clean Technologies and Environmental Policy*, *22*(3), 639–649. https://doi.org/10.1007/s10098-019-01807-9

van der Hulst, M. K., Huijbregts, M. A. J., van Loon, N., Theelen, M., Kootstra, L., Bergesen, J. D., & Hauck, M. (2020). A systematic approach to assess the environmental impact of emerging technologies: A case study for the GHG footprint of CIGS solar photovoltaic laminate. *Journal of Industrial Ecology*, *24*(6), 1234–1249. https://doi.org/10.1111/jiec.13027

van der Meide, M., Heijungs, R., Guinée, J., Hu, M., & Steubing, B. (2025). Contribution analysis in LCA: An overview of approaches and when to apply them. *The International Journal of Life Cycle Assessment*. https://doi.org/10.1007/s11367-025-02487-y

Wasti, S., Kamath, D., Armstrong, K., Clarkson, C., Tekinalp, H., Ozcan, S., & Vaidya, U. (2024). Life cycle assessment of coir fiber-reinforced composites for automotive applications. *Journal of Cleaner Production*, *485*, 144368. https://doi.org/10.1016/j.jclepro.2024.144368

Weiss, M., Cloos, K. C., & Helmers, E. (2020). Energy efficiency trade-offs in small to large electric vehicles. *Environmental Sciences Europe*, *32*, 46. https://doi.org/10.1186/s12302-020-00307-8

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): Overview and methodology. *The International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Xu, X., Jayaraman, K., Morin, C., & Pecqueux, N. (2008). Life cycle assessment of wood-fibre-reinforced polypropylene composites. *Journal of Materials Processing Technology*, *198*(1), 168–177. https://doi.org/10.1016/j.jmatprotec.2007.06.087

Zah, R., Hischier, R., Leão, A., & Braun, I. (2007). Curauá fibers in the automobile industry – a sustainability assessment. *Journal of Cleaner Production*, *15*(11–12), 1032–1040. https://doi.org/10.1016/j.jclepro.2006.05.036

Zampori, L., & Pant, R. (2019). *Suggestions for updating the Product Environmental Footprint (PEF) method* (JRC115959). Publications Office of the European Union. Luxembourg. https://doi.org/10.2760/424613